


ENVIRONMENTAL ASSESSMENT METHODOLOGY FOR PVIS

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This manual is freely available in the webpage of the project and in zenodo platform (in SolaQua community).

For more details, please visit www.sol-aqua.eu

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Acronyms

ALCA	Attributional Life Cycle Assessment
ALO	Agricultural Land Occupation
AoP	Areas of Protection
BOD	Biological Oxygen Demand
CC	Climate Change
CF	Characterization Factor
CLCA	Consequential Life Cycle Assessment
COD	Chemical Oxygen Demand
CU	Cereal Unit
DALY	Disability-Adjusted Life Years
ED	Ecosystem Diversity
EU	European Union
FD	Fossil resource Depletion
FE	Freshwater Eutrophication
FET	Freshwater Ecotoxicity
FU	Functional Unit
GWP	Global Warming Potential
HH	Human Health
HT	Human Toxicity
IR	Ionising Radiation

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ISINPA	Irrigators, SMEs, Investors and Public Authorities
KEMT	Key Enabling Materials and Tools
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LCT	Life Cycle Thinking
LUC	Land Use Change
ME	Marine Eutrophication
MET	Marine Ecotoxicity
MIR	Maximum Incremental Reactivity
MRD	Mineral Resource Depletion
NF	Normalisation Factor
NLT	Natural Land Transformation
OD	Ozone Depletion
PAF	Potentially Affected Fraction of species
PDF	Potentially Disappeared Fraction of species
PMF	Particulate Matter Formation
POF	Photochemical Oxidation Formation
QALY	Quality-Adjusted Life Years
RA	Resource Availability
RE	Renewable Energy
SI	Solar Irrigation
SME	Small and Medium Enterprise
STLCA	Spatialized Territorial Life Cycle Assessment
TA	Terrestrial Acidification
TET	Terrestrial Ecotoxicity
TLCA	Territorial Life Cycle Assessment
ULO	Urban Land Occupation
WD	Water Depletion
WF	Water Footprint
WP	Work Package

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Summary

Life cycle assessment (LCA) is a powerful tool that may be used to analyse the potential environmental impacts related to agricultural sector. Specifically, LCA is based on an iterative approach that is well suited to the multi-functionality of the agricultural activity. The management of a cropping system should ensure food and energy security and protect natural resources. However, the food and energy requirement due to the continuous growing of world population might lead to an overexploitation of natural resources and adoption of intensive crop production practices. An increase in renewable energy use and the application of principles and technologies of conservation and precision agriculture might help to achieve a sustainable intensification of crop production. In order to face this challenge and specifically foster water and energy saving, the SolAqua project proposes to combine the solar energy for irrigation water pumping and high-efficiency water management techniques. On the other hand, installing a solar irrigation (SI) has environmental impacts which should be evaluated. The report highlights that LCA is a tool suitable for assessing the environmental burden of an SI system and useful to support both individual agricultural activity and the planning of a possible large-scale introduction of SI systems, thus adapting to the needs of the different ISINPA. For this purpose, the document describes in detail the different LCA phases (i.e., Goal and scope definition, Inventory analysis, Impact assessment, and Interpretation) by focusing on how adapting the LCA tool to the specificities of a cropping systems and an agricultural process (e.g., solar irrigation). These aspects are essential to achieve reliable results such as to draw sound conclusions, and thus to make useful recommendations. A glossary synthetically shows the meaning of the terms most frequently used in the LCA method. Finally, a case study describes an application of the LCA method to a horticultural cropping system.

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1. Introduction

1.1. SolAqua in a nutshell

SolAqua's overall objective is to increase the share of **renewable energy (RE)** consumption in Europe by facilitating the market uptake of **solar irrigation (SI)** in the farming sector. SI is based on a combination of **photovoltaic (PV)** technology, hydraulic engineering, and high-efficiency water management techniques to optimize irrigated farming.

The consortium of SolAqua, which represents more than 70% of European irrigators, is aware of the potential of SI to decisively improve the sustainability of farming and rural communities in Europe. Nevertheless, to fulfil this potential, it is necessary to overcome the existing barriers to the market uptake of SI. To do this, SolAqua will accelerate the clean energy transition in European agriculture by facilitating the development of a well-functioning market for SI. This will be done by producing and exploiting a set of 7 **Key Enabling Materials and Tools (KEMT)** and by creating awareness, skills, action, engagement, and commitment (ASAEC) opportunities among more than 150,000 farmers, 70 local SMEs, and 40 Public Administrations in Europe and beyond.

The execution of SolAqua will result not only in a reduction of the cost of SI for farmers but also in the availability of effective standards for consumers and environmental protection, more efficient policies and supporting schemes, and new business opportunities for SMEs. Furthermore, to exploit the project's results and to trigger the SI market, SolAqua will facilitate a joint promotion of more than 100 MW of reliable and affordable SI led by the end-users themselves: the farmers.

To achieve the overall objective of increasing the share of RE in the European farming sector by facilitating SI market uptake, SolAqua has established the following 5 specific objectives:

1. **Produce and disseminate a set of 7 KEMT**, designed to solve technical, economic, and legal issues which are acting as barriers for the market uptake of SI.
2. **Produce awareness and skills of SI among the target groups in six countries** (France, Italy, Spain, Romania, Portugal, and Morocco). At least 150,000 potential end-users will be reached, 70 SMEs will be trained, and 38 Public Authorities will be able to produce more informed policies and supporting schemes.
3. **Trigger the European SI market by facilitating a joint promotion of at least 100 MW of SI**, exploiting SolAqua's KEMT and led by the target audiences engaged in SI because of the project's dissemination and communication actions.
4. **Increase the effectiveness of public supporting schemes for on-farm investments for the promotion of SI**: SolAqua will produce a new European Agrarian Fund for Rural Development (EAFDR) financial instrument that will be implemented in 3 European regions and will support more than 40 MW of new SI capacity.
5. **Facilitate market uptake of reliable and affordable SI in markets outside the EU** that will result not only in increased cooperation but also in business opportunities for European SME´s and investors.

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1.2. Foreword

The application potential of life cycle assessment (LCA) was widely highlighted in several production sectors over time and as reported by the European Commission in the Communication on Integrated Product Policy (COM (2003) 302), the LCA methodology may be considered "*the best framework for assessing the potential environmental impacts of products currently available*".

In order to meet the food requirements of a world constantly growing population, agriculture should increase the crop production level through the adoption of high doses of inputs and intensifying the use of agricultural practices. Consequently, the levels of upstream processes such as water mobilization and downstream processes such as transportation and product processing should also increase to ensure food safety. On the other hand, this strategy may lead both to an overexploitation of natural resources (water and soil) and an overconsumption of energy (high reliance on fossil fuels and non-renewable energy sources) resulting in serious impacts on production, ecosystem services, and more generally on environment (pollution).

Agricultural activity might ensure food and energy security without jeopardizing the availability of natural resources, for instance by adopting principles and technologies of the conservation agriculture and the precision one, and fostering the use of renewable energy sources. Nevertheless, the agricultural production value chain should be assessed to better understand the degree of achieved sustainability (e.g., from environmental perspective), hotspots, and potential improvements to be made.

In the light of the above, the combination of the solar energy for irrigation water pumping and high-efficiency water management techniques, as proposed by the SolAqua project, might foster both water and energy saving. However, an assessment of the environmental impacts of installing a solar irrigation should be made in order to identify strengths and weaknesses useful to support both individual agricultural activity and the planning of a possible large-scale introduction of solar irrigation. The objective of this report is to describe a methodology suitable for assessing the environmental burden of a SI throughout its life cycle.

The report consists of six sections and one glossary. The first section provides a general description of the LCA procedure focusing on its usefulness for agricultural sector. The sections 2-5 regard the different LCA phases (i.e., Goal and scope definition, Inventory analysis, Impact assessment, and Interpretation) and depict in detail which aspects of an agricultural activity should be included in a LCA study and how adapting the LCA method to the specificities of an agricultural process (e.g. solar irrigation) in order to obtain reliable results and to draw robust conclusions such as to enable to make useful recommendations. Finally, a glossary synthetically reports the meaning of the main terms used to the LCA procedure.

1.3. Life cycle assessment

The activity is aimed to develop a methodological tool suitable for evaluating the environmental impact of installing an SI system in line with the main SolAqua objective, namely to increase the share of renewable energy (RE) consumption in Europe by facilitating the market uptake of SI. This system may provide energy for irrigation with 0 emissions and at a cost of up to 70% lower than existing fossil-fuel based solutions. Therefore, the evaluation of environmental performance due to SI may facilitate the decision-making process regarding SI project planning

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from various stakeholders (i.e., Irrigators, SMEs, Investors, Public Authorities (ISINPA) as reported by the SolAqua project) and at different spatial scale.

As reported by ISO 14044 standard (2006), life cycle assessment (LCA) "*addresses the environmental aspects and potential environmental impacts (e.g. use of resources and the environmental consequences of releases) throughout a product's life cycle from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal (i.e. cradle-to-grave)*".

In other words, it is an analytical tool for evaluating the potential environmental burden due to a product or a service aimed to identify possible hotspots throughout the life cycle and improvements in terms of environmental impact reduction, well-balanced management of natural resources, and production inputs (Baumann and Tillman, 2004).

As observed by Brankatschk (2019), the agricultural stage necessary to obtain a certain agricultural raw material such as wheat grains, is affected both by geographical variations (e.g., climatic conditions, soil properties, water availability) and by a great deal of various agricultural practices (e.g., fertilization strategies, soil management, farming systems). This variability is responsible for different environmental impacts related to an agricultural raw material obtained by different processes. Therefore, agricultural processes would require more attention in order to better understand and improve their environmental contributions.

The supply, demand, and consumption of agricultural products, the economic dynamics, and trade-offs triggered by agricultural production systems may have direct and indirect impacts on globally human health, well-being, and land use. In this context, the LCA tool allows to compare products and systems using their potential environmental profile in order to improve efficiency, reduce waste, and prevent products and activities potentially harmful to society and ecosystem (Sieverding et al., 2020).

Therefore, the LCA procedure may be useful to support decision-making of different stakeholders such as producers who want to improve the environmental profile of a certain production system, consumers who prefer eco-friendly product, and policy makers to provide them useful information to plan longer-term strategies (Cellura et al., 2012).

Although the multi-functionality of crop systems (i.e., the capacity to provide food and to maintain ecosystem services) and the heterogeneity of agro-ecosystems may be considered constraints for the LCA application, the use of this methodological tool may enable to highlight the contribution of the agricultural sector to greenhouse gas (GHG) production and to maintain ecosystem services through assessing of the environmental performance basically due to the agricultural practices adopted over the life cycle of a certain product (Recanati et al., 2018). Furthermore, agriculture and climate change are characterized by critical and controversial cause-effect linkages. These linkages may in turn affect the environmental, economic and social spheres and make it difficult to exclude farming from strategies to combat climate change (Solinas et al., 2021). On the one hand, in 2016, agriculture produced 431 $MtCO_2$ equivalents (CO_2e) of greenhouse gas emissions in the European Union - 28 (EU-28) + Iceland (ISL). Specifically, methane (CH_4), nitrogen dioxide (N_2O) and carbon dioxide (CO_2) emitted by agriculture corresponded to 47.5%, 72.2%, and 0.3% of the total EU-28 + ISL emissions, respectively (EEA 2018). On the other hand, agricultural management practices aimed at enhancing soil carbon stocks might play a key role in mitigating climate change (Söderström et al., 2014). Moreover, soil organic carbon (SOC) sequestration may be considered one of the most cost-effective options for counteracting the effects of climate change (Nayak et al., 2019).

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In addition, LCA is widely used to evaluate environmental burdens related to crop systems under different agro-ecological conditions and carry out comparisons both among agricultural products and among different managements of the agricultural inputs and practices (Goglio et al., 2012; Fazio and Monti, 2011; Brentrup et al., 2004).

LCA was formalized by the International Organization for Standardization (ISO) that provided rules and requirements for conducting LCA study by the ISO 14040/44 standards. The former summarizes the main components that should characterize a LCA analysis and the latter provides the guidelines to properly implement the LCA method. According to the ISO standard, the LCA is based on an iterative approach as highlighted by the four phases that compose it (Figure 1). Indeed, the LCA phases are closely linked with each other since the single phase takes into account the results of the others. The iterative approach within and between the phases influences the comprehensiveness and consistency of the results (ISO 14040, 2006).

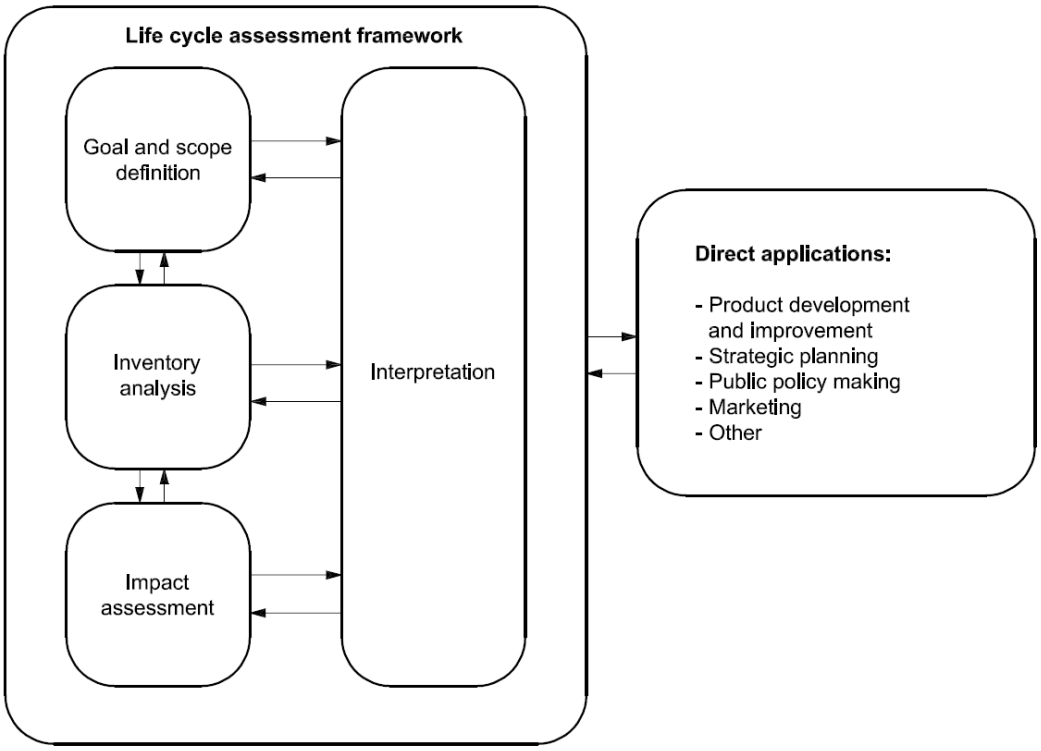

Figure 1 - Phases of a LCA (ISO 14040, 2006)

1.3.1 Life cycle analysis of agricultural products

Generally, the supply chain of agrifood sector is a sequence of activities including, water mobilization, farming (i.e. land cultivation and production of crops), processing/production, testing, packaging, warehousing, transportation, distribution, and marketing that are characterised by different flow types, that is physical material and product flows, financial flows, information flows, process flows, and energy and natural resources' flows (Tsolakis et al., 2014). Although sustainability and environmental burden evaluation are generally focused on agricultural activity, efforts were made to develop more systemic approaches (e.g., life cycle

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method) including stages of food processing, food retailing, and transportation in the assessment frameworks of food supply chains (Iakovou et al., 2014).

Dijkman et al. (2018) have described six stages in the life cycle of agricultural products (Figure 2). Even though, many LCA studies deal with the first two stages (generally from cradle-to-farm gate) in which the most environmental effects generally are caused by animal husbandry and manure handling, production and use of fertilisers, and the consumption of fuel for farm machinery.

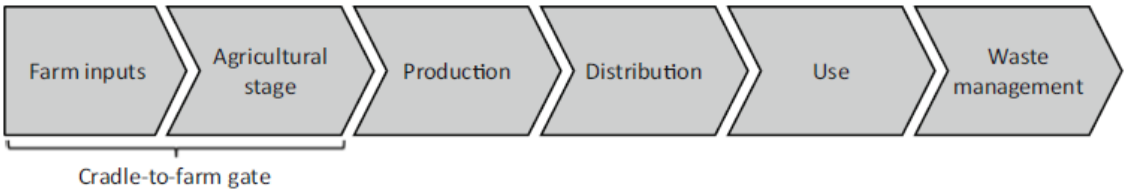

Figure 2 - Stages in LCA of an agricultural product (Dijkman et al., 2018)

In short, the first stage, includes the production and transportation of inputs (e.g., agrichemicals, machines, building elements, seeds and energy carriers - e.g., fuel and electricity to the farm -), which have a wide geographic scope. Water mobilization might also be included in this stage by considering its importance in the management of irrigated crop systems, although Dijkman et al. (2018) do not refer to it. Agrichemical inputs are pesticides (e.g., insecticides, fungicides, herbicides) used during weeding activity to defend crops from pests, diseases, or undesired plant growth, and fertilisers. Their production process may be harmful for the site of their production. Indeed, the fertiliser industry is responsible for GHG emissions, mainly carbon dioxide, nitrous oxides, and phosphorous that is obtained from phosphate rock may contribute to overexploitation of a non-renewable resource. In animal husbandry, the production and transport of animal feed may induce emissions of N_2O and CO_2, mostly due to fertiliser production and fuel use. Since farm equipment is used for a longer period, the impacts of the production and disposal of the farm equipment would have to be allocated over different product systems, but generally its impact is relatively small. The environmental contribution of infrastructures may increase in case of protected crops (i.e., greenhouses).

Agricultural stage (the second one) includes all inputs used to produce the food product, such as seeds, fertilisers, pesticides, water and energy in case of crops, and breeding of animals in case of meat or dairy products. Nevertheless, the positive effects on the crop growth and yield, the use of agrichemicals may cause emissions from the field and the buildings where animals are. Pesticides may be harmful for non-target species during or after application and bioaccumulation in harvested parts of the crops, potentially may cause ecotoxicity and human toxicity. Since pesticides are used in the agricultural field, the disposal phase is usually not considered in LCA analysis, although pesticide residues may be in packaging materials, in the sprayer and in the water used to clean the sprayer after distribution. Therefore, the disposal and following fate of residues should be taken into account in LCA. Fertilisers provide nutrients to crops, namely nitrogen (N), phosphorous (P) and potassium (K), but their use may have impacts owed to field emissions into soil, water, and air. More particularly, fertiliser distribution may cause emissions of ammonia (NH_3), nitrous oxide (N_2O) and nitrogen oxides (NO_x) to air, which in turn may result in acidification, climate change and eutrophication. Furthermore, N emissions in the form of nitrate (NO_3^-) and P loss in the form of phosphate (PO_4^{3-}) and particulate P via erosion, may have impacts in terms of eutrophication of water bodies. As regards animal

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husbandry, the main emissions are due to enteric fermentation and manure handling which contribute to climate change, eutrophication and acidification.

Fuel consumption is essential for many on-farm mechanical operations, such as ploughing, distribution of fertilisers and pesticides, roughage production, harvesting, heating of greenhouses, transport of the product, etc. However, agricultural fuel is generally from fossil origin, and thus water pumping and mechanical operations use non-renewable sources and have effects on climate change. Moreover, the use of machinery may damage soil quality via compaction and erosion making soil less productive. In this condition, soil needs additional inputs such as fertilisers to maintain food production steady in the short term. Soil ecological functions, such as buffering and filtering of toxic chemicals, water retention and soil-biota, may be affected as well. Accordingly, unsustainable farming practices may lead to irreversible soil degradation in the long term and a land use change (from natural land to cultivated land) has to occur to ensure food production, often at the expense of natural vegetation. Irrigation may induce serious impact because, particularly in arid regions and for water-intensive-crops (e.g., almonds, rice), the water consumption may transport from field salts, toxic and nutrient-rich pollutants, that might affect aquifers and surface water bodies downstream. In addition, water withdrawal might requires a wide use of non-renewable energy resources to be transported to field and/or to pressurize water with consequent environmental impacts.

Processing represents any step to transform the raw farm product into a (packaged) food one, being characterised by different logistic steps whose importance in terms of environmental burden resulting from food processing change a lot on the basis of the foodstuff under consideration.

Each food life cycle contains a distribution stage which regards (often refrigerated) transportation to the warehouse and to the retailer, sorting fruits, conditioning packaging and cool storage for good maintenance of food properties. All these processes may have negative effects on environment because they may require the use of non-renewable energy, such as fuel use in transportation and electricity in cool storage. The distribution phase may highly contribute to total impact in fresh products' life cycles.

The use of food refers to food transport from the retailer to the point of consumption and to energy use for cooking and storing it. This stage might include, on the basis of the LCA study, private households or restaurants and conventional kitchens. Although many studies highlighted that the use stage shows negative effects due to food storage or preparation, with differences in terms of energy use depending on the place where food are consumed, this step generally show a low contribution to total impacts in most categories. The LCA procedure currently began to consider exposure of humans to pesticide residues in food during consumption that might cause human toxicity impacts, even though most countries have regulations to limit human exposure to pesticides within certain safe levels.

The disposal stage of food product regards both the management of food generated along the entire life cycle and the treatment of human excretion resulting from food consumption. Since the food sector is wasteful, the reduction of waste in the use stage may be an effective way to lower food environmental burden. Several studies have highlighted that options such as landfilling, incineration, composting, digestion to produce biogas and conversion to animal feed, do not lead to the same impacts. For instance, different GHGs are emitted from wastewater treatment and following sludge disposal. Depending on the wastewater treatment facility, N and P emissions to surface water may contribute to the eutrophication potential.

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2. Goal and scope definition

The definition of both goal and scope has an essential role in the LCA application since the next steps depend on the statements made in this phase. According to the ISO 14044 standard (2006), the goal should be clear about the following aspects: "*the intended application; the reasons for carrying out the study; the intended audience, i.e. to whom the results of the study are intended to be communicated; whether the results are intended to be used in comparative assertions intended to be disclosed to the public.*" As observed by Baumann and Tillman (2004), it is not complicated to identify a reason to implement a LCA study and its audience, but it is not uncommon that the initial goal definition is too generic and unclear. On the contrary, the statement of an accurate goal is necessary to make significant methodological choices in the next phases. In other words, a general assertion (e.g., we want to know the environmental benefits and weaknesses of a product) should be replaced by a more detailed statement (also expressed as question e.g., which are the processes that more contribute to the environmental burden due to a certain product?) in order to define properly the goal of a LCA analysis.

The identification of the target audience during the goal definition may make it easier to find the most relevant review needs and the suitable form and technical level for the communication of the LCA results (EC - JRC, 2010a). Since the LCA results may be used by various audience categories (e.g., decision-makers, scientists, public authorities, and consumers) characterized by different backgrounds and expectations, the mode of result communication should be tailored to their needs considering the most significant messages to show and the audience (Sala and Andreasson, 2018).

The scope definition better outlines the main framework including various topics that will be detailed in the next phases, such as product system to be studied, functional unit, system boundary, allocation procedures, methodology used to assess the life cycle impacts and impact categories, data quality requirements, assumptions and limitations under which the analysis will be performed. Since the goal and scope definition do not concern the data collection or the result calculation, no detailed information on this subject has to be provided in this phase (Heijungs and Guinée, 2012). In the light of the above, the goal and scope definition in the agricultural sector might be, for instance, to explore the environmental performance due to different intensities in crop production (Brentrup et al., 2004).

2.1 SolAqua goal and scope definition

The SolAqua project is focused on the agricultural sector which in turn, is based on irrigation water management and the development and management of cropping systems. Therefore, the definition of SolAqua goal and scope might concern the mobilization and the application of solar irrigation to certain cropping systems in order to assess the environmental impacts due to the innovative distribution of irrigation water compared to the whole environmental burden arisen from the agricultural management of cropping system under consideration.

LCA might be applied to two different spatial scales: local (farm) and regional scale (irrigation scheme) to provide a methodological tool useful to assess the environmental performance of SI systems both at individual and collective project level.

The LCA analysis also might aim to highlight the potential implications due to the installation of the SI system on allocation of natural resources and planning of crop systems at both spatial

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scales. For instance, the LCA results might underline that the introduction of SI system at farm level might lead to an optimization of water use and irrigation practice with a consequent energy cost saving for water pumping and distribution, and concurrently fostering the product diversification and the adoption of innovative agricultural techniques. The increased availability of water resources provided by SI system might offer the opportunity for farmer to choose crops with a high water requirement whose cultivation would thus be difficult to implement without the support of irrigation practice or crops whose yield might be optimized through supplemental irrigation.

At regional level, the assessment of environmental performance related to SI system might provide useful information in support of decision-making regarding the water distribution by the public service. In regions where access to water is difficult because of it is too expensive, the increased availability of irrigation water provided by SI system and the consequent reduction or zeroing of energy cost might allow administrator and policy-maker to develop facilitation measures for access to water and its distribution. This opportunity might also enable farmers to enhance the supply of certain products with positive effects on agricultural sector, and thus on socio-economic development both at local and regional level.

Furthermore, the potential improvement in the use of water resource and energy saving ensured by SI at individual farm level, would allow the whole agricultural sector of a given area to strengthen its contribution in terms of adaptation measures to climate change.

Under the SolAqua project, the LCA methodology might be applied to some cropping systems considered among the most relevant and representative of Europe and specifically, of the Mediterranean area. A short description is reported in section 2.1.2.

2.1.1 LCA application potential within policy decision making

The design and application of new technologies are strictly dependent on the approval of the policy makers. It is not always easy for the researchers responsible for these technologies to obtain a positive feedback from the policy makers towards innovative technologies and their impacts (Prasad et al., 2020). However, the development and implementation of policy strategies aimed to support agricultural sustainability and food security should not disregard scientific evidence such as information provided by the LCA application which has obtained growing consideration by policy makers as a useful tool for identifying and orienting interventions to reduce the environmental load of agri-food systems, establishing the goals, and monitoring the effects of policies (Gava et al., 2019). Pombo et al. (2019) highlight that public policy development results from both several factors which affect decision making and various stakeholders characterized by different values, perceptions, and preferences. Furthermore, the application of life cycle concepts and tools might lead scientific and policy-making communities to the achievement of an appropriate balance among economic, environmental, and social points of view by leaving a fragmented overview of issues in favour of a more holistic decision making. From environmental perspective, life cycle thinking (LCT) and LCA might play a key role in the evaluation of a system sustainability since they may provide essential information for decision-making in a comprehensive and holistic manner in both business and policy contexts, even though many enhancements might be necessary to obtain the best effects from the LCA methodology in the case of a change of assessment scale (i.e., from product (micro) to the system (meso-macro)) (Reale et al., 2017).

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The LCA tool may be useful for the legislative policy process by facilitating the problem identification and supporting policy evaluation and implementation. Specifically, the use of LCT approach may allow to change stakeholders' problem perception as well as educate people with respect to the potential effects resulting from a policy decision or when encouraging sustainable lifestyles (Pombo et al., 2019). According to Thabrew et al. (2009), the stakeholders' involvement in environment decision making in order to have an essential and responsible role in the development process (i.e., stakeholders understand the issues and the different options to face them) should meet the following requirements: "*1) conducting context-specific analyses including upstream and downstream effects of direct activities in order to establish broad levels of environmental quality and development, and (2) making information accessible such that the interpretation of the information by different stakeholders from different sectors and disciplines are understood to the greatest extent by all stakeholders*".

The LCT approach may provide reliable information on environmental, social and economic effects to decision-makers in order to enable the inclusion of the sustainability concept in decision making processes and may be applied by decision makers in both the public and private sector to support the development of policies and products (UNEP/SETAC, 2012). For many years, the action for environmental safeguard was focused on the reduction of the impacts caused by production processes, treatment of waste and effluent streams. However, the reduction of air and water pollution resulting from a certain activity inevitably do not minimize neither the negative environmental effects due to the consumption of resources and materials nor account for the so-called "shifting of burdens", namely to resolve a problem while generating another one since in this case solutions might be counter-productive (e.g., from one stage in the life cycle to another, from one region to another, from one generation to the next or amongst different types of impacts (EC, 2010; EC - JRC, 2011b). For instance, businesses has often adopted strategy for reducing environmental effects without considering the supply chain on the whole or the use and end-of-life processes of products. Government measures have often concerned certain country or region neglecting negative and/or positive environmental consequences that might occur in other geographical locations. In both contexts, the lack of the adequate attention for the entire life cycle of goods and services may worse environmental impacts because of an inappropriate resource use resulting in poorer financial performance and reputation damages for the of the parties involved (EC, 2010).

These situations, especially shifting of burdens, might be avoided by using LCT, namely a conceptual approach aimed to identify options of environmental enhancements and to reduce the effects of goods and services (i.e., products) at all phases over the life cycle, from raw extraction and transformation, manufacturing, product distribution, use and fate at end-of-life (Figure 3).

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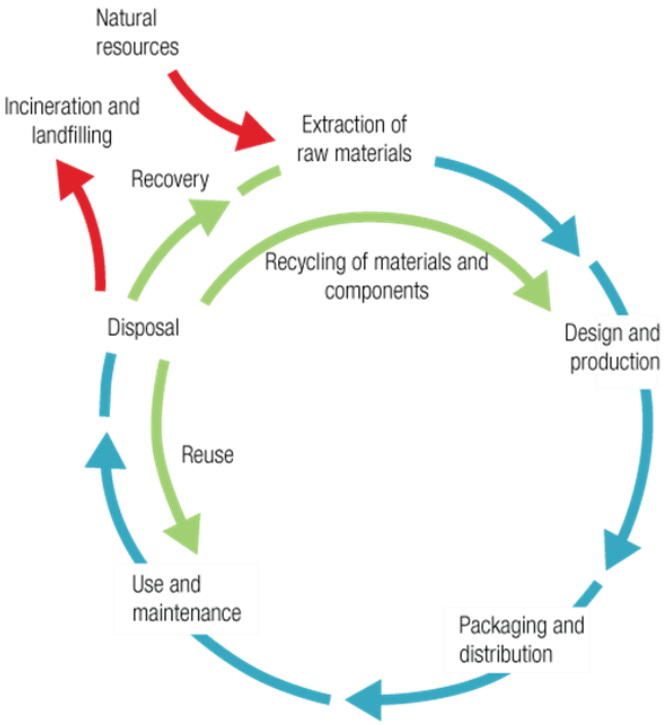

Figure 3 - A product lifecycle (UNEP) / SETAC, 2012).

In other words, LCT helps to prevent improving production technologies and, at the same time, to cause waste-related impacts, reducing emissions of greenhouse gases while increasing land use or acid rain, or reducing emissions in one country while increasing them in another. Furthermore, this approach requires more from the policy developer or environmental manager since it adopts a broader perspective which may generate important benefits from the advancement of knowledge, for instance through identifying process efficiencies or good management practices (EC - JRC, 2011b).

The LCA method enables to express in quantitative terms the environmental impacts of a product throughout its life cycle (EC, 2010). In other words, it translates the LCT principles, making it operational, in a quantitative framework which evaluates the environmental profile related to any goods or services in terms of emission release and resource depletion. Consequently, LCT includes the LCA method which may be considered a useful tool to contribute effectively and efficiently to make more sustainable worldwide consumptions and productions (EC - JRC, 2011b). The multi-criteria approach characterizing LCA makes it particularly suited to capture a wide variety of stressors and effects related to human health, ecosystem quality and resources. Therefore, in a policy context, the application of a life cycle methodology facilitates the identification of priorities in a clearer and inclusively way and the assessment of trade-offs by enabling to address policies more effectively so that the maximum benefit may be achieved in relation to the efforts expended (Sala et al., 2016).

As reported by EU-JRC (2010a), the potential of LCA use as support of decision making is mainly established during the goal definition when the decision-context is identified. The ILCD handbook (EC-JRC 2010a), provided a detailed description of the three decision-contexts in which LCA may be applied. They basically differ "*regarding the question whether the LCI/LCA study is to be used to support a decision on the analysed system (e.g. product or strategy),… and… whether the study is interested in interactions of the depicted system with other systems*

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(e.g. recycling credits) or not". Generally, Situation A concerns "*decision support typically at the level of products, but also single process steps, sites/companies and other systems, with no or exclusively small-scale consequences in the background system or on other systems*". Specifically, "*the consequences of the analysed decision alone are too small to overcome thresholds and trigger structural changes of installed capacity elsewhere via market mechanisms*". Situation B covers "*decision support for strategies with large-scale consequences in the background system or other systems. The analysed decision alone is large enough to result via market mechanisms in structural changes of installed capacity in at least one process outside the foreground system of the analysed system*". Finally, Situation C refers to "*a retrospective accounting / documentation of what has happened (or will happen based on extrapolating forecasting), with no interest in any additional consequences that the analysed system may have in the background system or on other systems*" (Figure 4).

<table>
<tr><td rowspan="4">Decision support?</td><td rowspan="3">Yes</td><td colspan="2">Kind of process-changes in background system / other systems</td></tr>
<tr><td>None or small-scale</td><td>Large-scale</td></tr>
<tr><td>Situation A
"Micro-level decision support"</td><td>Situation B
"Meso/macro-level decision support"</td></tr>
<tr><td>No</td><td colspan="2">Situation C
"Accounting"
(with C1: including interactions with other systems, C2: excluding interactions with other systems)</td></tr>
</table>

Figure 4 - Decision contexts in which LCA might be used (EC-JRC 2010a)

The LCA information may be differently used to support decision-making on the basis of the level of decision-context (Table 1).

Table 1 - Examples of LCA use according to the micro or the macro level of decision-context (Sala et al., 2016)

Level of perspective	Possible applications of the life cycle information
Micro	Identification of Key Environmental Performance Indicators (KEPI) of a product group for Ecodesign / simplified LCA
	Hotspot and weak point analysis of a specific product
	Ecodesign, design for recycling
	Comparison of environmental profile of specific goods or services
	Benchmarking of specific products against the product group's average
	Development of life cycle based Type I Ecolabel criteria
Macro	Forecasting and analysis of the environmental impact of pervasive technologies, raw material strategies, etc. and related policy development
	Basket-of-products (or -product groups) type of studies
	Identifying product groups with the largest environmental impact/improvement potential
	Monitoring environmental impacts of a nation, industry sector product group, or product

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Gava et al. (2019) underlined that a product system may be divided into foreground and background subsystems based on the direct or not influence of the decision maker on the processes type and mode of operation related to the product system under consideration. Furthermore, they specified that the LCA tools, from conceptual point of view, may be divided into retrospective and prospective. The former or LCA of the account type refers to the production of a certain product by considering materials and energy flows, before or after impact mitigation interventions, and the associated environmental impacts, whereas prospective or change-oriented LCA approach includes in the analysis the most important environmental flows in the production of a specific product to highlight the effects of potential impact mitigation interventions (e.g., future policy), that is the consequences of substitutions in the considered material and energy flows.

The conceptual difference between retrospective and prospective LCA also characterize two further modelling LCA approach, namely attributional and consequential LCA (see section 3 for more details). The former basically provides a static representation of the effects caused by the processes included in the system under consideration, whereas the latter is aimed to quantify market-mediated effects of decisions (i.e., strategies to reduce the environmental impact of productive activities) regarding the a certain system on other systems. The application potential of both LCA models in the policy context in terms of comparative appropriateness and complementary use is being debated within the scientific community.

Seidel (2016) analysed potentials and limits associated with the LCA use in public policy development. The information provided by the LCA tool might be considered appropriate for a good public environmental policy even though decision-makers may not be completely comfortable with LCA since its incorporating into decision-making process may be challenging because of nature of outcomes based on a range of indicators not always directly comparable, and from which might not be possible draw clear conclusions. In other words, the LCA method may be considered a valuable support since able to provide quantitative information on which develops public policy aimed to reduce environmental effects, identify chances for environmental enhancements and evaluate environmental trade-offs among possible alternatives. Furthermore, the application of life cycle concepts and tools may connect scientific and policy-making communities in a common effort aimed to achieve a balance between economic and environmental sphere, by moving from fragmented approaches towards more holistic decision-making.

Sala et al. (2016) described in detail the potential role that the LCA procedure might have if integrated in the policy cycle by highlighting that in each phase of the policy development (i.e., from problem identification up to policy evaluation) LCA might be used in supporting the identification of possible solutions to essential questions (Table 2).

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Table 2 - Main phases of the policy cycle and the potential application of LCA to support them (Sala et al., 2016)

Steps in the policy cycle	Related possible questions in the impact assessment	Description	Current and possible use of LCA
Policy anticipation and problem definition	What is the problem and why is it a problem?	Identification of emerging issues	LCA studies in scientific and grey literature, reporting "warnings" to be taken into account
Policy formulation	What are the various ways to achieve the objectives?	Definition of policy options	Policy options may: be based on LCA results (e.g. addressing a specific life cycle stage or relevant environmental impact, leading to impacts) to identify specific "hot spots" include some requirements based on LCA indicators (e.g. a life cycle based calculation) use LCA for identifying key elements to be monitored over time and, possibly, be standardize use LCA results to set a target
Policy impact assessment	What are their economic, social and environmental impacts and who will be affected? How do the different option s compare in terms of their benefits and costs?	Comparison of options	Supporting the comprehensive and systematic assessment of environmental aspects, and even beyond environmental aspects if including LCC (Life Cycle Costing) and SLCA (Social Life Cycle Assessment). LCA may spot impacts related to a number of different impact categories and may help avoiding shifting burden from one stage in the life cycle to another Complementary to risk assessment
Policy implementation		Country level implementation Compliance checks	If LCA indicators are used as requirements of the policy option, LCA studies will be needed
Policy evaluation	How will monitoring and retrospective evaluation be organized?	Effectiveness of the policy Evaluation of the need to revise (or phase out) the policy	Use of LCA for assessing the benefit of the policy (at macro scale) including systemic aspects Need of modifying/ repealing a legislation

Notwithstanding the application potential of LCA, according to Seidel (2016) its benefits on public policy on the basis of literature, are up to now limited because of some barriers such as:
"*1. Decision-makers lack the background or technical literacy to interpret and incorporate the results of the LCA;*
2. Technical results are not presented in a way that can be positively utilized by decision-makers;
3. Decision-makers have a lack of trust of LCA results or the overall process;
4. Clear or consistent results may be lacking as outcomes of the LCA;
5. LCA results are not seen as neutral;

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6. Governments lack a framework for integrating LCA information into the decision-making process;
7. Government agencies bring specific interests to the process, potentially limiting the scope based on internal focus and knowledge;
8. Comprehensive public LCAs require considerable resources to complete;
9. Complete and accurate inventory data may be difficult to find."

Furthermore, Seidel (2016) underlined that some of these limitations suggest that the success of LCA within public policy context is not owed to a weakness of the LCA tool, but rather the procedure in which LCA is involved. In order to effectively incorporate LCA within policy decision-making process, and consequently overcome barriers, a winning option should be to adopt a multidisciplinary approach including stakeholders and policy decision makers in a collaborative process also by encouraging the spread of LCT approach among policy-makers. In other words, moving towards a more open and qualitative approach focused on communication and transparency might facilitate the overcoming of barriers.

In this regard, Seidel (2016) referred to some recommendations such as:
"*1. Involve decision-makers and other stakeholders actively, wholly and genuinely throughout the LCA process;*
2. Translate values and limitations of LCA concepts and methodologies into language decision-makers understand;
3. Provide case studies of successful applications of LCA in public policy to give confidence to its use within the public policy arena;
4. Present assumptions and uncertainties transparently within the process, and actively involve stakeholders in all discussions regarding these factors;
5. Ensure the project team represents the full range of stakeholders affected by the policy, and vested interests are balanced."

As regards agro-food systems, Gava et al. (2019) highlighted that LCA application within the decision-making process in agricultural policy is not so widespread compared to other economic sectors despite the growing interest by governments and the research community aimed to develop policy strategies in support of the sustainability of agricultural sector and food security. Information provided by LCA studies may play an essential role for agro-food policies concerning cross-border pollution, transaction costs following the adoption of environmental standards, adoption of less polluting practices and/or technologies, and business-to-consumer information asymmetry.

The same authors provided an overview of the potential use of LCA outcomes in order to strengthen the environmental impact mitigation strategies in the agro-food systems by considering that the scientific literature suggests to face the need of reducing the environmental effects due to agro-food systems through the public support. In brief, this strategy should be addressed to innovation (practices, technologies), to enhance effectively the resource handling on firm (supply-side), or to help the change in food consumption patterns with indirect consequences on food supply (demand-side), or to actions concerning both the supply and demand-sides that require the collaboration of supply chain stakeholders (system level interventions).
Information provided by LCA studies on product impacts may be useful for establishing the legal limits on emissions (e.g., carbon tax) or entry levels to access public tenders (e.g., green public procurement). Furthermore, the policies that might make the most the LCA method in the agro-food sector regard cross-border pollution, transaction costs associated with the application of

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environmental standards, the adoption of less polluting practices or technologies, and the reduction of information asymmetry business-to-consumer.

Gava et al. (2019) also highlighted that by considering the growing consumer sensitivity towards the environmental impacts of agri-food products, LCA may also facilitate the adoption of environmental certifications and labelling by the agribusiness in order to encourage the product positioning on the domestic and export markets. Although consumers may be willing to pay a price for environmental label on food products, the labels' effectiveness in orienting consumer decisions does not depend only on the available impact information, but also the lack of knowledge about the evaluations behind the label. For this reason, label design is essential to clarify information and reduce consumer confusion. Consistent and complete and cohesive carbon labelling policy in combination with dedicated social learning campaigns, may help to achieve policy goals. Furthermore, policy makers should monitor the spread of sustainability labels, to provide a valuable support for the development of certification schemes. However, more empirical evidences need to be found on the possible combination among environmental, economic, and social impacts caused by certification adoption in order to evaluate the extent to which the value added associated with certification may be distributed among supply chain stakeholders and to verify if farmers are incentivized to modify their production practices and technologies or are "forced" by the market.

2.1.2 Cropping systems

Drawing on national statistics (FAOSTAT, 2017), wheat represents nearly half (48%) of the total cropped area in the Mediterranean region, followed by olive trees (11%), citrus (5%), vineyards (4%) and sunflower (3%). These proportions, of course differ between individual countries depending on physical (local soil and agro-climatic) and agro-economic conditions.

In this context, the range of representative cropping systems within the area of SolAqua project includes annual forage systems (e.g. silage maize - Italian ryegrass rotation, or winter cereal grasses for grazing and hay), permanent meadows, and alfalfa; open-field horticultural crops (e.g., globe artichoke, processing tomato, potato), tree crops (olive orchards and vineyards), and winter cereal crops for grain (mainly wheat) grown in monoculture or in rotation with cash crops (sugar beet and sunflower).

In the irrigated arable lands of the Mediterranean basin, Italian ryegrass for hay production and spring sowing maize for silage production are the most diffuse crops for livestock feeding. Italian ryegrass is cultivated from mid-October to mid-May to produce hay and is occasionally irrigated in case of drought, while the growing season of silage maize is from the end of May till September and is always irrigated. Permanent meadows, which are mainly found on the less-intensive farms, are generally cultivated without irrigation in the hills and with surface irrigation in the plains. When irrigated, permanent meadows provide an average of 13 Mg DM ha^{-1}, well distributed over 5-6 cuts, while 2-3 cuts are common in non-irrigated meadows with an average production of 5-6 Mg DM ha^{-1}, which is concentrated in the spring. Alfalfa represents an important resource for dairy cattle farms because of its productivity of feed protein per unit area, which is the highest among forage and grain legumes. Alfalfa is grown for about 4 years and then the land is ploughed during the summer and prepared for sowing in the autumn. The most common rotation is made up of alfalfa for three to five years followed by a winter cereal (wheat or barley) or Italian ryegrass. In Italy, 4 or more cuttings of alfalfa (up to 6 or 7 under irrigation) may be harvested annually. Alfalfa yield varies according to the age of the crop and the availability of water. Notwithstanding its resistance to drought, due to a deep root system,

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in the Southern part of Mediterranean basin, where average annual precipitation is below 500 mm, is always irrigated. Hay crops are very common in the rainfed area and are based on oats and/or Italian ryegrass annual species, sown in October-November, grazed until mid-February and mowed in May. Cool-season annual grasses such as ryegrass, rye, wheat, and oats produce some forage in late fall, very little from late December through mid-February, and a large production peak in spring. From 75 to 90% of annual clover production, depending on species, occurs in March, April, and May. If grown in an annual rainfall of less than 600 mm, irrigation will be necessary to obtain good annual ryegrass yields. The other cool-season annual grasses (rye, wheat, barley, or oat) should not be grazed shorter than 8 cm to maintain forage growth.

Globe artichoke is a perennial rosette plant grown as annual crop in some Mediterranean regions, for its large fleshy heads. It is an important vegetable crop in Mediterranean countries occupying 93,000 ha, about 70% of the total world cropped area. The globe artichoke growth cycle maybe poliannual or annual according to the varietal-type, the suitability of the growing area, the adopted agronomic technique, and the market requirement. Indeed, the early flowering artichoke is grown with the forced technique, namely by forcing the vegetative organs (offshoots) to start growing at a time when (the summer one) they should be dormant according to their natural growing cycle. The primary purpose of adopting the forcing technique is to meet the market requirement for early productions, also ensuring to farmers higher incomes.

The wheat-growing area within the Mediterranean Basin represents 27% of the arable land, and the region represents 60% of the world's growing area for durum wheat. Cereals, as durum wheat and barley are grown predominantly under rainfed conditions. Sowing occurs in the late autumn and grain harvest early in the following summer. Since rainfall is very erratic, and dry spells usually coincide with the most sensitive growth stages of wheat and food legumes in the dry Mediterranean region, there is a need for better management of available water through the conjunctive use of rain and irrigation water. Supplemental irrigation maybe an option to minimize the variations in yields due to aberrant weather conditions and to stabilize production at an economically acceptable level.

In recent years many olive orchards, which are a major crop in the Mediterranean orchard systems, have been converted into intensive high tree density (1975 trees ha^{-1}) or super-intensive hedgerow systems (2200 trees ha^{-1}), with very high plant density and irrigation. Therefore, the accurate estimation of crop water requirements, i.e., its space-time variability at field level is an increasingly important issue for optimizing water management. Despite their traditional and economic importance in the region, olive orchards and vineyards account for only 6% and 2% of water demand, respectively, reflecting the fact that only 8% of olive trees and 23% of vineyards are currently irrigated. However, because of the majority of the industry's vineyards are located in semi-arid climate (with average annual rainfall around 400 mm) supplemental irrigation should be adopted. Supplemental irrigation would help ensure the industry's forward momentum by offering growers a sustainable tool that would save both money and water without sacrificing the quality of the wine grapes.

In Morocco some fruits crops and vegetables like carrot, onion and melon and water melon are known to have a high consumption of water and inputs and have a huge impact on the environment.

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2.2 Functional unit

In the LCA methodology, functional unit (FU) is a key factor that should be clearly defined because it represents a measure of the function of the considered system (i.e., what a product or a process is designed to do) and it provides a reference to which all inputs and outputs may be related enabling the comparison of different systems in terms of environmental impacts (ISO 14044, 2006). Therefore, its definition is closely linked to the goal and scope of the study. The choice of functional unit may have significant effects on the LCA results and may become complex especially if the multi-functionality of the agricultural system is considered. According to Nemecek et al. (2011), three main functions may be identified in an agricultural system: i) the land management function refers to the land cultivation in order to minimize the environmental impact per area and time unit; ii) the financial function is related to the main goal of an agricultural activity, namely farmer's income (therefore, it is aimed to reduce the environmental impact per currency and to optimize eco-efficiency); iii) the productive function regards the production process of food, feed or biomass for other uses (e.g., bioenergy, renewable materials) ensuring the reduction of the environmental impact per product unit. Different functional units may be used to represent or express the function of the considered agricultural system appropriately, namely in a quantitative manner.

FU may be classified according to four categories: i) mass of product, (ii) land area, (iii) energy use, and (iv) economic value. According to the description reported by Cerruti et al. (2011), in mass based functional unit, the environmental burden may be evaluated on the basis of a specific quantity of product produced or effect per product quantity (e.g., per ton). Land based functional unit considers the assessment of environmental impact depending on the amount of land area used to obtain a certain product or impact per land area (i.e. hectare). In energy based unit, the calculation of environmental impact is related to the quantity of chemical energy contained in the final product or to the impact per unit energy associated with final product (e.g., calories stored in the harvested crop). The last functional unit (i.e., economic value-based) is related to the economic value of the considered product and it is used to optimize the economic eco-efficiency of the system under consideration.

2.3 System boundary

According to ISO 14044 standard (2006), "*the system boundary determines which unit processes shall be included within the LCA. The selection of the system boundary shall be consistent with the goal of the study. The criteria used in establishing the system boundary shall be identified and explained. Decisions shall be made regarding which unit processes to include in the study and the level of detail to which these unit processes shall be studied. It is helpful to describe the system using a process flow diagram showing the unit processes and their inter-relationships. Each of the unit processes should be initially described to define*

– where the unit process begins, in terms of the receipt of raw materials or intermediate products,

– the nature of the transformations and operations that occur as part of the unit process, and

– where the unit process ends, in terms of the destination of the intermediate or final products.

Ideally, the product system should be modelled in such a manner that inputs and outputs at its boundary are elementary and product flows. It is an iterative process to identify the inputs and outputs that should be traced to the environment, i.e. to identify which unit processes producing the inputs (or which unit processes receiving the outputs) should be included in the product system under study".

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It might be useful to define foreground and background systems in developing the system boundary since both of them might be an effective support in the choice on what kind of data and information should be collected. The former is the set of processes directly affected by the study, providing a FU specified in the definition of goal and scope. The latter is energy and material supply to the foreground system, using as reference a homogeneous market so that not to enable the identification of each plants and operations (Azapagic and Clift, 1999). Generally, site specific data are used for foreground system whereas secondary data from databases, public references, and estimated data by simulation models are collected for background system (Li et al., 2014). The fact that, as previously reported by ISO 14044 (2006), the setting of system boundary is an iterative process, is most likely due to the numerous processes and thus, the great deal of data involved in the product under consideration. Furthermore, the selection of the system boundary concerning a certain product might be considered an arbitrary step since various persons may set different system boundaries for the same product basing on their personal experience, so neglecting some important processes (Li et al., 2014).

There are three main types of system boundaries in the LCA methodology: i) boundary between the product system and the environment context; ii) boundary between the processes that are significant and insignificant to the product system (cut off); iii) boundary between the analysed product system and other product system (allocation) (Pathak et al., 2013).

Therefore, a LCA analysis may consider all processes that characterize a product, namely from raw material acquisition through to end-of-life and disposal, the so-called from cradle to grave approach. However, it is possible to use different approaches that exclude certain phases of production process (Figure 5).

Each type might be considered a module for use in other LCA studies – considering that for unit process, product and waste flows, the system boundary should be between the modelled process and the rest of technosphere (i.e., man-made environment) and thus is included in inventory as well as for elementary flows (i.e., substance or energy) that leave the process towards ecosphere (i.e., natural environment) or that enter from there and cross the system boundary (EC - JRC, 2010a). Consequently, cradle-to-gate and gate-to-gate approaches may be considered incomplete assessment of a product life cycle. In fact, the first obtains the product but does not arrive to its life cycle end (e.g., from resource extraction to the factory gate) neglecting the use and disposal phase of the product under consideration. The second approach is related to a single process.

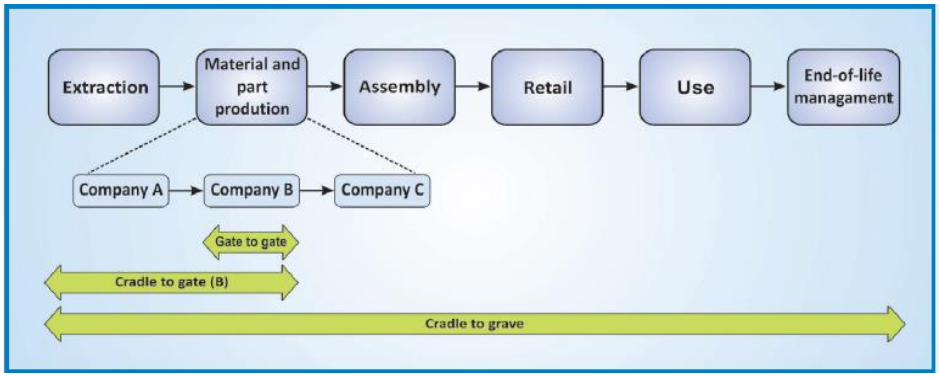

Figure 5 - Different phases in LCA procedure (EC - JRC, 2010a)

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In the agricultural sector, the identification of a frontier between ecosphere and technosphere might be complicated since resources arisen from natural environment and the biotic ones resulting from human activity are key factors within a cropping system. As reported by Crenna et al. (2018), the boundary between ecosphere and technosphere might be identified considering natural biotic resource as biotic resources extracted from natural environment and used for human interventions and biotic resources produced by human interventions (e.g., crops from agriculture) (Figure 6).

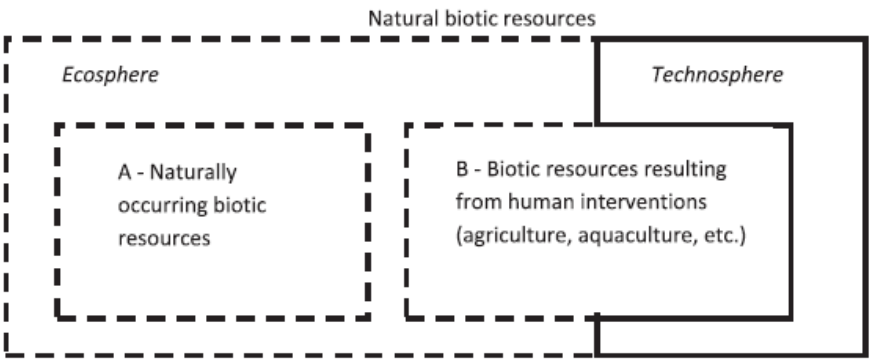

Figure 6 - System boundary for natural biotic resources Box A, biotic resources from natural environment; B, biotic resources from human activity (Crenna et al., 2018)

Specifically, in human-made systems in which the land portion is changed from natural to human-made context and is used for non-productive land (e.g., urbanization) or a productive land such as agriculture, livestock, wood production, the biomass yield may be considered a product arisen from a human-made system (i.e., technosphere) and not from natural environment (Alvarenga et al., 2013).

Generally, the boundary of an agricultural system may ideally stop when a certain product cross the farm gate without including in environmental computing of agricultural system, the impacts arisen from post-harvest operations, such as emissions due to transport and processing of a product and the environmental burden due to the end use of the product and its by-products (Cooper et al., 2011). However, the choice of system boundary size may have a significant effect on the LCA results, since the comparisons of environmental impact due to a certain agricultural product considered under different conditions may be made it more difficult if the system boundary dimensions are different (Roer et al., 2102). Therefore, the agricultural system boundaries may range from cradle-to-field gate, -to-farm-gate (when the transport operations are considered), and -to-grave (when the LCA is applied to the entire supply-chain of an agricultural product including, for instance manufacturing, packaging, use, recycling and disposal of a product). Specifically, field-gate production considers production to field edge (e.g., corn in the combine) whereas farm-gate gets to where product leaves the farm (e.g., corn after drying and storage) (Caffrey and Veal, 2013).

Given that a SI system is situated at a farm, its LCA might be indifferently performed considering a cradle-to-field or -farm gate approach (Figure 7).

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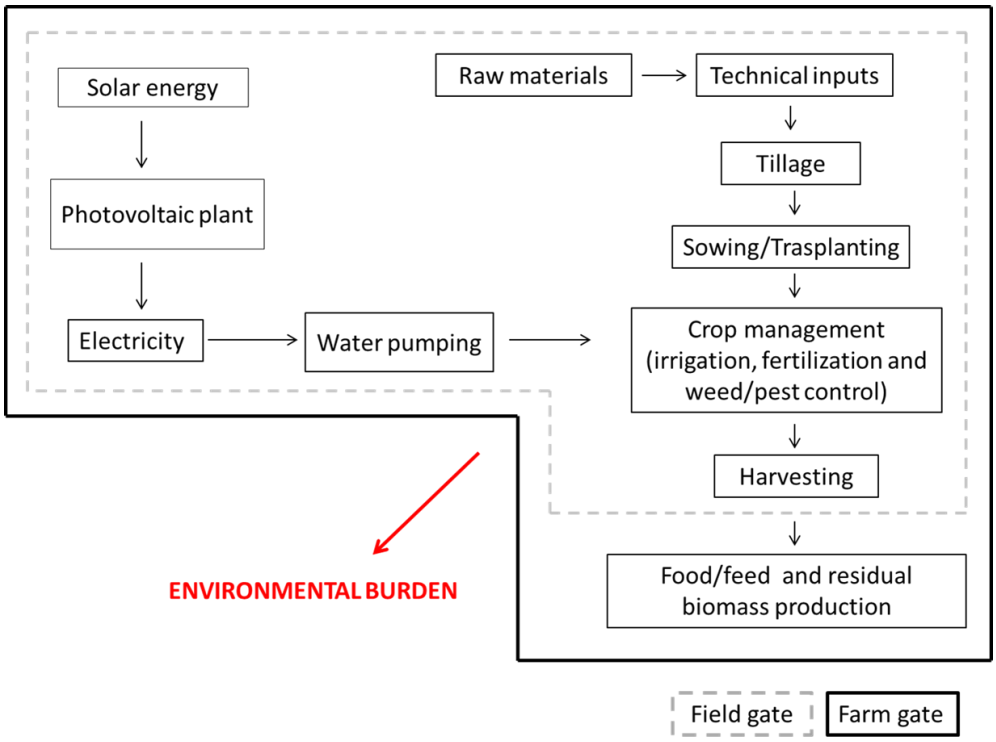

Figure 7 - Flow chart of the SI system boundaries

2.4 Allocation

Considering a farm as a whole, it is easy to find various products which may annually change depending on crop rotations and yield variations (Caffrey and Veal, 2013). In order to handle this condition and the resulting environmental impacts it is necessary to apply an allocation procedure. Its choice is one of the most controversial methodological aspect in the LCA community since it may considerably influence the final LCA results (Chen et al., 2010; Boschiero et al. 2015). As reported by ISO 14044 standard (2006), allocation is defined as "*partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems*". Specifically, allocation is to associate the environmental burden (e.g., resource depletion, emissions to air and water and solid waste) to various functions provided by a multi-function systems such as multiple-input systems (waste treatment processes), multiple-output systems (co-production), and multiple-use or "cascaded use" systems ("open-loop recycling") (Suh et al., 2010) (Figure 8a, b, and c).

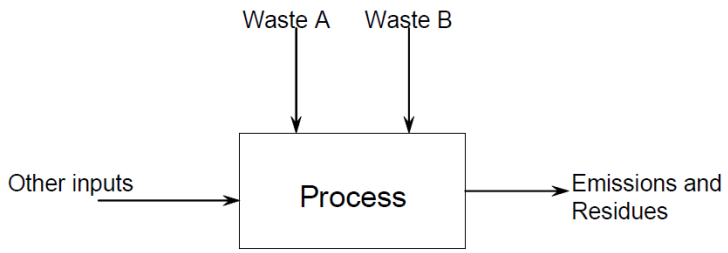

Figure 8a - Multi-input allocation system (Audsley et al., 2003)

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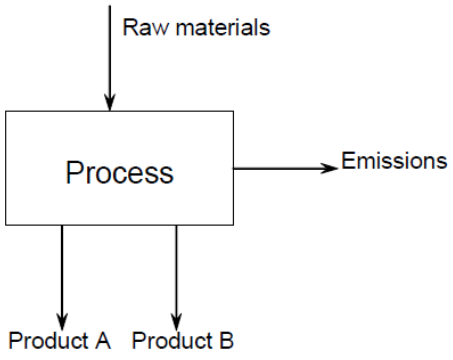

Figure 8b - Multi-output allocation system (Audsley et al., 2003)

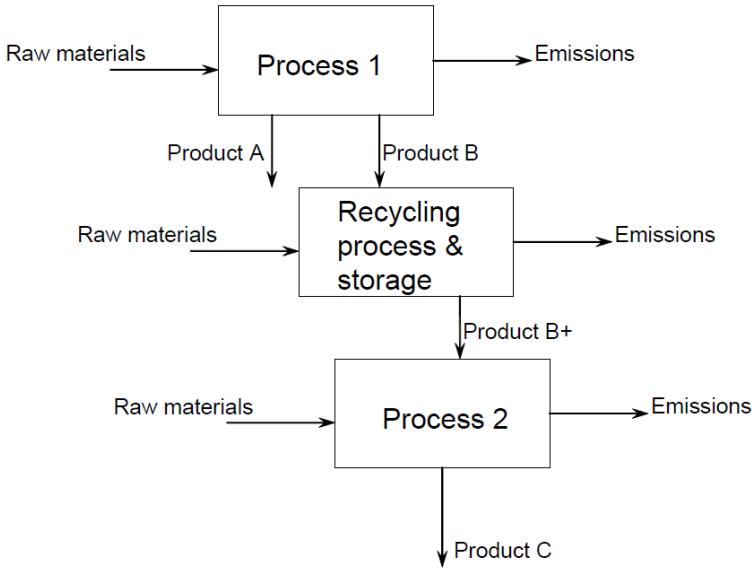

Figure 8c - Open loop recycling (Audsley et al., 2003)

In the first two systems, the allocation procedure shows similar problems since more than one input and output may characterized a multiple-input and -output system, respectively. As reported by Azapagic and Clift (1999), the allocation is mainly to find a procedure to associate the environmental burdens to different inputs in case of multiple-input system considering that they have different properties which contribute to the whole environmental burden from the system. Similarly, in multiple-output or co-product systems the allocation should associate with each products only the environmental burdens of which is responsible. This problem gets more complicated than the two previous situations in multiple-use or “cascaded use” systems as a products may be reprocessed and reused in different systems.

In the case of reuse and recycling, two different situations may occur: 1) the closed-loop recycling in which an output can go back into the same system from which it was produced avoiding an allocation problem because the use of a secondary element replaces the use of a primary one; 2) the open-loop recycling in which an output can become an input into a different

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system or process entailing an allocation problems as the recycle becomes an input to a different system producing a new product or products (ISO 14044, 2006).

Allocation procedure is hard to apply to agricultural activity since it is frequently characterized by closely inter-linking sub-systems of activities (Audsley et al., 2003). Specifically, allocation is to distribute benefits and burdens of a certain agricultural production among different products such as co-product, by-product and waste that may have a different weight within the allocation procedure. Co -products may share a part of burden or benefit arisen from allocated production, by-products may not greatly affect the allocation procedure because of their poor value, and wastes may be considered a burden for production regarding their disposal (Sieverding et al., 2020).

The ISO 14044 standard develops a hierarchy for the methodological choice on the basis of implementing a co-product allocation within a LCA analysis. Specifically, the procedure consists of three different steps: "*Step 1: Wherever possible, allocation should be avoided by 1) dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes, or 2) expanding the product system to include the additional functions related to the co-products,……b) Step 2: Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying physical relationships between them; i.e. they should reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system. c) Step 3: Where physical relationship alone cannot be established or used as the basis for allocation, the inputs should be allocated between the products and functions in a way that reflects other relationships between them. For example, input and output data might be allocated between co-products in proportion to the economic value of the products*".

Although the first option suggests by the ISO standard is to avoid allocation, the other options are not devoid of drawbacks. Since the system separation or the system expansion in LCA analysis might need a great deal of additional data - often hard to find - the development of new sub-processes or marginal systems and the use of these complex models might be less clear and accurate (Mackenzie et al., 2017). If allocation may not be prevented, physical relationship such as mass or energy may be used as basis for supporting the allocation of process flows among the different outputs. Economic value can be used as basis for allocation with caution, since the variability of co-product price might affect the results even in absence of change in the physical relationships among co-products (Sauer, 2012). The availability of different allocation procedure seems to suggest that its choice should be made on a case by case basis. Aylott et al., (2011) describe the use of different allocation procedures in a crop system based on wheat cultivation where the straw (i.e., co-product) is used for energy purpose. The use of substitution credits entails that the environmental burden arisen from grain is reduced by subtracting the credits received when co-products (i.e., energy from straw) substitute for the main products, namely a credit is allocated back to the grain since fossil fuel emissions are avoided. Furthermore, the environmental burdens may be assigned to product and co-product applying percentage allocation criteria. This means that emissions arising from wheat cultivation might be divided between the straw and grain on the basis of their: (i) relative energy contents (energy content allocation); (ii) relative masses (mass allocation); and (iii) relative market prices (price allocation). The application of these two methodologies (i.e., substitution credits and percentage allocation) may lead to difference results since GHG emissions related to additional processing of co-products are considered by substitution credits unlike percentage-based procedure.

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2.4.1 Crop rotation

Generally, the LCA application to agricultural activity is mainly focused on the environmental performance of a single crop or its (co-)product(s) neglecting the interactions among crops and throughout years within crop rotation (Costa et al., 2020). However, interactions among crops play influence the impact resulting from changes in cropping systems on their agro-economic and environmental performance, since fertilization, nitrogen mineralization, nitrate leaching, GHG emissions, pests and weeds, and final crop yield are affected both by the management of the single crops and by long-term processes on which crop sequence have an effect (Reckling et al., 2016.). Therefore, separating a single crop and its environmental burden from the cropping systems to which it belongs represents a difficult task or maybe even impossible, especially considering that its performance is greatly influenced by the previous crop in the cropping system, and also long-term effects caused by crop management on soil organic matter, nitrogen availability, weed population, and biodiversity (Goglio et al., 2018).

A common approach to include the crop rotation impacts in LCA procedure has not yet been achieved, even though the effects of crop rotation may significantly influence the whole LCA result of each single crop involved in a rotation system (Peter et al., 2017). It seems sufficiently clear that any effort to better catch the effects of crops and cropping systems should take into account the potential complexity that may characterise a cropping system. Indeed, the constructing of new cropping systems may be deemed an essential step in order to best use the multi-functionality of agriculture, namely incorporating biodiversity into agricultural systems and exploiting ecosystem services might both enhance the sustainable management of natural resource used in agriculture and meet the need to intensify agricultural production (Malézieux, 2012). These aspects may be included in the definition of cropping system that, as observed by Eckersten (2017), generally refers to a set of management procedures applied to a certain uniformly treated area, such as a field, part of a field or several associated fields and even non-cultivated areas associated with the fields. Moreover, a cropping system may include crops, crop sequence and crop and soil management (e.g., tillage, sowing, fertilisation, plant protection and harvest) for each crop in the crop sequence. A cropping system may be defined on the basis of time and space, and also considering within its boundaries the factors that affect it (inputs; e.g. climate) and factors generated by cropping systems that have effects on the surroundings (outputs, e.g. crop yield and GHG emissions).

Goglio et al. (2018) have proposed some approaches to estimate cropping system effects via LCA method, although none of them is completely exhaustive and precise. In short, two LCA applications may be used depending on the study purpose, namely product LCA that is aimed to evaluating the environmental performance of a product obtained by cropping systems and system LCA focused on assessing and comparing production systems (e.g., cropping systems) uses LCA to better understand the environmental mechanisms responsible for environmental profile. The cropping system approach proposed for system LCA, was used, for instance, to compare organic with integrated cropping systems. This approach considers a cropping system as a whole system producing various products. This entails that when the functional unit is represented by ton of yield, cereal units, GJ of energy output, or agricultural income, the result obtained for the entire system provide an assessment of the whole environmental impact of the system. In the cropping system approach, the interaction of crop management with soil and climate conditions is considered. Moreover, it is used to perform site-specific assessment, especially of complex systems (e.g., agroforestry and intercropping systems), but not provide

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findings for single crops and products. Concerning product LCA, environmental burden may evaluating on the basis of allocation approaches, crop-by-crop approach, and combined approaches. Allocation approach is used when the environmental load may be allocated to a crop depending on a generic criterion allowing to maintain the integrity of the cropping system. It may be applied to agroforestry and intercropping systems, although allocation should be avoided when possible as recommended by ISO standards 14040/14044 (2006), for instance for a specific crop management practice easily associated with a crop (e.g., seeding). Allocation based on cereal unit refers to the biophysical cereal unit (CU) which has the advantage of keeping cropping system integrity while is used for product LCA. On the other hand, cereal units refer to feed and food crops and the allocation of the impacts does not show the time dynamics of cropping systems, even though this aspect is considered.

Brankatschk and Finkbeiner (2014) have emphasized that the CU may be deemed an appropriate unit for the description of agricultural products and may be used as the basis for an agriculture-specific allocation approach in LCA. Indeed, allocation issue may often occur in agricultural LCAs, for instance when the wheat production is split into grains and straw, and during milling process wheat grains are split into flour, bran, and middlings. Accordingly, whenever allocation may be responsible for uncertainty that may propagate and potentially amplify whenever allocation is applied to different steps of the life cycle of a product. As reported by Brankatschk and Finkbeiner (2014), the Cereal Unit was developed by German agricultural authorities and scientists decades ago and continuously updated and used in the German agricultural statistic, in order to provide a support for handling the allocation problems for agricultural LCA studies. CU is mainly based on the nutritional value for livestock products and co-products occurring during their processing. To determine the CU, the specifically aggregated metabolizable energy content is calculated for each feed material and normalized using barley as a reference (1 kg barley = 12.56 MJ specifically aggregated metabolizable energy). In this way, Cereal Unit conversion factors have been computed for a wide number of agricultural products, also including vegetable products not used as livestock feed. To include these crops in the CU systems, a conversion factor is calculated breaking into relation the yield of the reference culture (i.e., barley) to the yield of the specialty crop, namely Cereal Unit conversion factor of specialty crop (CU/kg) is equal to yield of reference culture (CU/ha) divided by yield of specialty crop (kg/ha). This means that the CU conversion factor equal to 1.04 for wheat grain is 4% higher value resulting from 1 kg of wheat compared to 1 kg of barley (Table 3).

Table 3 - Example of steps for the CU allocation procedure applied to wheat harvesting (extracted from Brankatschk and Finkbeiner, 2014)

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Step no.	Short description	Example 1
1	Identify products and co-products	Process: Wheat harvesting; Product and co-product: wheat grains and wheat straw
2	Identify mass proportions	1 kg wheat plant → 0.56 kg wheat grains and 0.44 kg wheat straw
3	Identify Cereal Unit conversion factors for products and co-products	1 kg wheat grains = 1.04 Cereal Units (CU); 1 kg wheat straw = 0.43 CU
4	Convert product- and co-product-streams into Cereal Unit	0.56 kg * 1.04 CU/kg = 0.58 CU wheat grains; 0.44 kg * 0.43 CU/kg = 0.19 CU wheat straw
5	Calculate allocation ratio	0.58 CU/(0.58 + 0.19) CU equals 75% wheat grains 0.19 CU/(0.58 + 0.19) CU equals 25% wheat straw

The CU allocation was tested and compared with most common allocation approaches (i.e., mass, energy and economic allocations) for wheat, barley, soybean, rapeseed, sugar beet and sunflower showing no significant differences. Although this approach is developed in German context, other countries and international institutions (e.g., Food and Agriculture Organization of the United Nations (FAO)) have been developed similar procedures which making the CU approach easily adaptable to other countries and applicable to feed and food cropping systems. The availability of some region specific background information necessary for CU calculation via agricultural statistical reports and publications or directly obtained via expert consultation, makes this approach devoid of geographical constraints.

Within a cropping system, crop rotation may have important effects concerning the maintenance of soil health and ecosystem services, and at the same time ensuring good crop production levels. As reported by Zegada-Lizarazu and Monti (2011), the benefits due to crop rotation specifically regards: i) improvement of soil structure and maintenance of long-term productivity and organic matter; ii) increase in biodiversity and less monotony of the landscape; iii) reduction of erosion because of a longer period of land cover; iv) less use of agricultural inputs such as agrochemicals and synthetic fertilisers; v) enhancement of soil fertility and higher yields; vi) production diversification with greater market opportunities and lower economic and climatic risks; and vii) time-diluted farming activities. Crop residues remaining on the field may provide an important contribution in terms of benefits resulting from crop-rotation, although the occurrence of crop residues is not limited to crop rotations since they may also occur in monoculture (Brankatschk and Finkbeiner, 2015). It is clear that crop residues and thus crop rotation have environmental effects which should be included within the LCA procedure.

Brankatschk and Finkbeiner (2015) have proposed a new approach to include the crop rotation effects into the life cycle inventory. The crop rotation approach is based on the adaptation of the system boundary to the level of crop rotation during the data collection for the life cycle inventory and on the allocation of inputs to their respective outputs. Basically, the total aim of the LCA analysis of a product/crop, functional unit, and reference flow are not modified whereas the system boundary of LCI expands, but not the boundary of the entire LCA study. This approach may be applied according to six steps:

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"1. The crop-rotation system is identified, in which the agricultural crop studied, e.g. wheat, is grown. The system boundary of the LCI (not that of the entire LCA study) is defined, including all elements around this crop-rotation system. The definitions of the functional unit and reference flow according to ISO 14040 (e.g. production of 1 t of wheat grain) thus remain unaffected. It is relevant at this stage to consider the entire rotation in which the wheat is grown.
2. The agronomic inputs (seed, diesel fuel, energy, agrochemicals, fertiliser, etc.) of the entire crop rotation cycle including all crops grown in the crop rotation are quantified.
3. All outputs (including products, by-products, waste, leachate, emissions) of this crop rotation leaving the agricultural field are considered and quantified (i.e. tonnages of each individual product, such as wheat grain and the other products and co-products produced within the same crop rotation).
4. All from agricultural outputs for each crop in the rotation are converted mass or volume into Cereal Units.
5. Allocation factors are calculated for each individual agricultural output of the entire crop rotation using the amounts given in Cereal Units. Calculation check: the sum of all allocation factors must equal 100%.
6. Using the allocation factors calculated in the previous step, the sum of each agricultural input (seed, diesel fuel, energy, agrochemicals, fertiliser, etc.) is allocated among all individual agricultural outputs".

Therefore, the use of this approach allows to allocate specific quantities of inputs and emissions to each single output. The other LCA phases do not change when life cycle impact assessment is performed using the results determined via this approach.

Goglio et al. (2018) propose for product LCA other two approaches: crop-by-crop and combined approaches. The crop-by-crop approach is focused on the single crop considered as a separate entity from the previous and subsequent crops. It is the most commonly used for product LCA concerning both food and non-food crops. This approach is suitable to product LCA because neither need for a high level of agricultural expertise since only data on the considered crop have to collected, nor no specific knowledge of cropping systems is required. On the other hand, the approach considers temporal impacts of crop management only when they occur throughout the crop season making it inaccurate. Combined approaches and double approach adopted for crop rotations establish a temporal limit among various crops and enable the user to consider potential cropping system impacts occurring in the short term. Nevertheless, they include neither long-term crop management nor crop effects. Moreover, they need of more data and expertise on cropping system and more time to be adopted, although they are not suitable for intercropping and agroforestry systems.

Combined approach was developed by Nemecek et al. (2015) considering 64 crop rotations located in different regions of France. This approach is focused on the crop combination defined as the inventory of a certain crop, with a clear preceding crop and eventually including a catch crop. In this specific study, soft winter wheat was estimated after cereals, pea or rape seed or sunflower as preceding crops. The LCA computed on the basis of crop combinations may be deemed an efficient way to examine a wide number of crop rotations because the preceding crop is considered as part of the cropping system. The management of each crop is affected by the crop rotation but, even though crop rotations involve a great deal of resources, the differences in terms of environmental impacts for the same crop grown after different preceding crops may be significant (Nemecek et al., 2015).

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This approach may catch and reveal the cause-effect relationship of the cropping system features, although it needs expertise, data, and time to be adopted (Goglio et al., 2018).

The model for integrative life cycle assessment in agriculture (MiLA) was developed by Peter et al., (2017) to evaluate GHG emissions and cumulative energy demand (CED) of agricultural cropping systems including energy crops. Concerning crop rotations, the MiLA tool adopts a double approach considering nutrient cycling carryover from one crop to the next crops and using different functional unit to estimate various products. Evaluation of emissions and CED is performed depending on the number of crops following in the crop rotation for which the nutrients applied via fertilisers or green manuring crops are available. Therefore, environmental burden resulting from crop management (e.g., fertiliser application) are evaluated dividing according to the specified number of crops, including the crop where the fertiliser was applied. For cover crops used for green manuring, environmental load from the entire cultivation process are divided according to the number of crops that benefit from the nutrients supplied. Furthermore, the introduction of crop rotations within a LCA study relating to a certain crop may complicate the analysis and thus the assessment, because of the various products of the crops cultivated in the rotation. MiLA proposes a solution by adopting different functional units which enable to estimate the entire crop rotation without adopting a further allocation method. Nevertheless this tool is suitable for product-specific LCA studies and provide useful information on the interactions among crops, it still characterised by uncertainty regarding the approach to modelling crop rotation effects and does not consider environmental indicators useful to provide an assessment of environmental profile more comprehensive (e.g., biodiversity, erosion potential, eutrophication as well as economic and social indicators) (Peter et al., 2017).

In view of the above, Goglio et al. (2018) provide some recommendations concerning the choice of approach to be taken depending on the LCA type (i.e., system LCA and product LCA). Generally, the approach selected should be consistent with the purpose of the LCA analysis and problems associated with the approach adopted should be argue by the LCA practitioner in line with the ISO 14040/14044 standards (2006). Specifically, the cropping system approach is suggested for system LCA and it may be combined with one of the other approaches in order to attribute crop rotation impacts to single crops. Concerning product LCA, the issue related to the environmental load of cropping system due to a specific crop should be completely associated with the corresponding crop. For example, the emissions of machinery release during seeding should be attributed to the corresponding crop. When an impact need to be attributed to a crop, but it is difficult to separate it from the whole cropping system, combined approaches using specific criteria should be adopted. For example, this approach may be applied in case of estimating soil P loss in relation to eutrophication potential for a cropping system including wheat, clover, and potatoes. If specific criteria may not selected and a combined approach adopted, allocation approaches using generic attribution criteria should be applied. Also, a dual approach should be used (e.g., cropping system approach with one of the approaches specific for product LCA) where possible.

2.5 Cut-off criteria

These rules, that according to ISO 14040 standard (2006) represent "*the amount of material or energy flow or the level of environmental significance associated with unit processes or product system to be excluded from a study*", enable to better handle inventory and to make calculation procedure more efficient. As not all processes and elementary flows are quantitatively significant for a LCA analysis, these data may be cut-off allowing to focus on the collection of

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better-quality data for the significant processes and elementary flows (EC - JRC, 2011). Therefore, the cut-off criteria do not aim to hide data and they should clearly be described in the final report. As observed by Baumann and Tillman (2004), the term "cut-off" may be subject to different interpretation: i) if the environmental burden related to a certain phase of the life cycle is negligible compared to the rest of the life cycle, the relevance might be a significant criterion to identify what include, specifically in change-oriented LCA analysis; ii) the system boundary based on from cradle-to gate approach, which not includes waste treatment, might be considered as a cut-off criterion; and iii) the part of life cycle excluded because of lack of time, data, or financial resources might be solved including a further inflow from another system in the inventory or assessing the missing data. Nevertheless, the application of mass, energy, and environmental importance as cut-off criteria in order to exclude not relevant inputs or outputs from analysis should characterize the system boundary definition, besides the identification of which steps (i.e., cradle-to-grave, etc.) and process to include in a LCA study (Ling-Chin et al., 2016). According to ISO 14044 standard (2006), the exclusive use of mass-based criteria to support the initial identification of which inputs should not be considered in a study, on the contrary might entail the omission of significant inputs. Therefore, energy and environmental relevance should also be applied as cut-off rules in this phase. As regards the above criteria, ISO 14044 standard (2006) specified that: "*a) Mass: an appropriate decision, when using mass as a criterion, would require the inclusion in the study of all inputs that cumulatively contribute more than a defined percentage to the mass input of the product system being modelled. b) Energy: similarly, an appropriate decision, when using energy as a criterion, would require the inclusion in the study of those inputs that cumulatively contribute more than a defined percentage of the product system's energy inputs. c) Environmental significance: decisions on cut-off criteria should be made to include inputs that contribute more than an additional defined amount of the estimated quantity of individual data of the product system that are specially selected because of environmental relevance*".

The application of cut-off criteria makes it possible to better manage simplified inventories and to consider only the inputs responsible for an environmental load lower than a certain considered mass percentage. Specifically, these rules represent the quantitative definition of the system boundary; this may be quantified on the basis of a percentage of environmental impacts (e.g. 95 % means a cutting off about 5 % of the total environmental impact (or of a certain impact category)) and taking into account that the cut-off should not be too big in order to avoid having incomplete data and to increase of result uncertainty (EC - JRC, 2011).

2.6 Impact categories and characterization models

During the phase of goal and scope definition, it is necessary to determine both which impact categories and characterization model will be used to perform the assessment of environmental effects occurred throughout a product life cycle under consideration. In other words, this phase establishes which environmental impacts should be considered and the procedure to quantify them on the basis of category indicators and characterization models.

As reported by Baumann and Tillman (2004), the selection of impact categories should consider the following aspects: i) Completeness, it refers to the fact that impact categories should cover both the main environmental issues and those of specific interest for the LCA analysis; ii) Practicality, the set of selected categories should not include too many; iii) Independence, namely impact category should be independent to avoid double counting; iv) Possibility to integrate, this means the opportunity to connect the LCI result parameters in order to select

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impact categories and characterization methods; v) Environmental relevance, namely indicators arisen from characterization methods should have an environmental relevance to the impact category and safeguard subjects; vi) Scientific method, this means that the characterization methods should be scientifically valid.

The choice of impact categories should be made depending on the the goal of the study as well as the choice of category indicators and characterisation models that will be used in the impact assessment methodology.

As reported by ISO 14040 standard (2006) impact category, category indicator, and characterization factor represent: "*class representing environmental issues of concern to which LCI results may be assigned*", "*quantifiable representation of an impact category*", and "*factor derived from a characterization model which is applied to convert an assigned life cycle inventory analysis result to the common unit of the category indicator*", respectively. The characterization models quantify the environmental burden - represented by the category indicator - related to a certain substance, showing the environmental mechanism namely, as reported by ISO 14044 standard (2006), "*system of physical, chemical and biological processes for a given impact category, linking the life cycle inventory analysis results to category indicators and to category endpoints*". Specifically, the main function of characterization models is to link the results of life cycle inventory and the final impact categories (i.e., category endpoints) since characterization models provide characterization factors which quantify the contribution of an input or output flow in reference to a specific category indicator (Figure 7).

The Figure 9 also represents the logical framework used in the next phase of LCA methodology, namely impact assessment occurring throughout the life cycle product under consideration (see section 4).

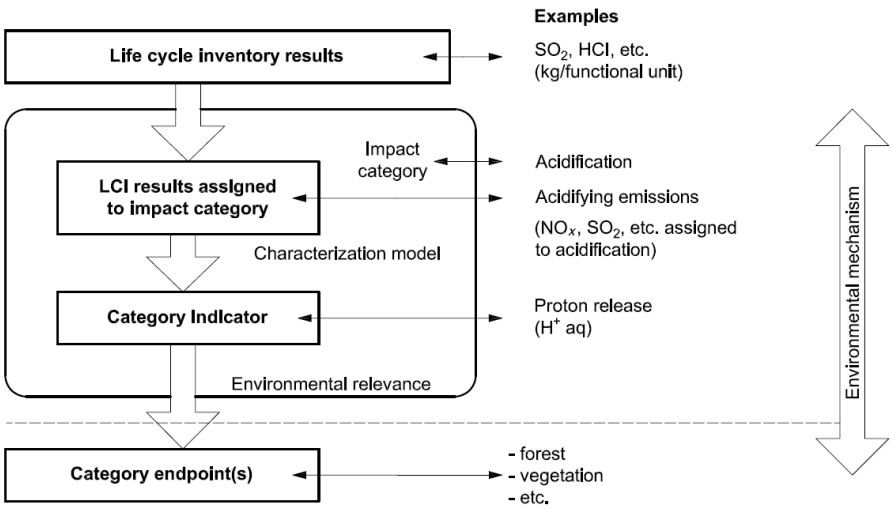

Figure 9 - Link among Impact categories, Characterization models, and Category indicators (ISO 14044, 2006)

A more in depth description of some terms reported in Figure 7 is provided in Table 4.

Table 4 - Meaning of some terms used in the phase of life cycle impact assessment (ISO 14044, 2006)

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Term	Example
Impact category	Climate change
LCI results	Amount of a greenhouse gas per functional unit
Characterization model	Baseline model of 100 years of the Intergovernmental Panel on Climate Change
Category indicator	Infrared radiative forcing (W/m^2)
Characterization factor	Global warming potential (GWP_{100}) for each greenhouse gas (kg CO_2-equivalents/kg gas)
Category indicator result	Kilograms of CO_2-equivalents per functional unit
Category endpoints	Coral reefs, forests, crops
Environmental relevance	Infrared radiative forcing is a proxy for potential effects on the climate, depending on the integrated atmospheric heat adsorption caused by emissions and the distribution over time of the heat absorption

Since different characterization models were developed, a set of criteria was established to apply to all impact categories taking into account of scientific quality and stakeholder acceptance, in order to make them as objective as possible and minimize the risk of biasing results of a model at the expense of another one. (Hauschild et al., 2013). This set consists of 5 scientific criteria which assesses: i) Completeness of scope; ii) Environmental relevance; iii) Scientific robustness and certainty; iv) Documentation, transparency and reproducibility; and v) Applicability. Stakeholder acceptance criterion concerns the "degree of stakeholder acceptance and suitability for communication in a business and policy context" (EC - JRC, 2010b).

2.7 Data quality requirements

Data quality is a relevant aspect within LCA method since it may improve the reliability of results and make it easier their interpretation depending on the quality level of data collected to build life cycle inventory. Latter often includes a great deal of data from various sources characterized by a different level of accuracy which might lead to inconsistent results and uncertain interpretation and thus, it compromises the reliability of inventory as an effective tool for decision-makers (De Smet and Stalmans, 1996). In order to meet data quality requirements, ISO 14044 standard (2006) indicates a list of elements which should be considered and they have been sorted based on relevance, reliability and accessibility criteria (Baumann and Tillman, 2004).

ISO 14044 standard (2006) reports that "*The data quality requirements should address the following: a) time-related coverage: age of data and the minimum length of time over which data should be collected; b) geographical coverage: geographical area from which data for unit processes should be collected to satisfy the goal of the study; c) technology coverage: specific technology or technology mix; d) precision: measure of the variability of the data values for each data expressed (e.g. variance); e) completeness: percentage of flow that is measured or estimated; f) representativeness: qualitative assessment of the degree to which the data set reflects the true population of interest (i.e. geographical coverage, time period and technology coverage); g) consistency: qualitative assessment of whether the study methodology is applied uniformly to the various components of the analysis; h) reproducibility: qualitative assessment of the extent to which information about the methodology and data values would allow an*

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independent practitioner to reproduce the results reported in the study; i) sources of the data; j) uncertainty of the information (e.g. data, models and assumptions)".

According to Baumann and Tillman (2004), relevance should be encompassed time-related coverage, geographical coverage, technology coverage, completeness, and representativeness considering relevance as "*the extent to which the data used represent what it is supposed to represent*". Reliability, which covers precision and consistency, regards the quantitative accuracy and uncertainty of data, although the reliability of data is also related to the consistency with which data was collected and documented besides to the competence of person or organization responsible of this activity. Furthermore, data are more reliable whether they may be reviewed. The data review may be simplified if data are supported by a transparent documentation, which may be useful also to support the reproducibility of the results. Finally, accessibility encompasses reproducibility and consistency of data.

The management of data quality and its assessment is an issue discussed at length within LCA community in order to find a procedure that enhances the result credibility. The quality of different databases may be connected to the goals regarding data quality through a set of indicators based on use of a pedigree matrix, aimed to specify the data quality depending on the way it is use in the analysis. Furthermore, these indicators may provide further information on uncertainty of single data and thus, on the one of the inventory results by improving their quality level (Weidema and Wesnaes, 1996). Although different attempts was made in order to provide guidelines for data quality assessment, additional efforts would be useful to refine existing data quality systems and approaches specifically regarding interoperability of LCA data, since data quality scores are generally not translatable from one methodology to another one. An additional improvement of data quality might be provide by developing and using of data quality indicators able to capture further dimensions of LCA data because the updated pedigree matrix does not include all dimensions of LCA data and models. Also, the automation of data quality score basing on metadata extraction might reduce subjectivity in the production on data quality judgements (Edelen and Ingwersen, 2018).

2.8 Assumptions and limitations

The ISO 14040 standard (2006) reports that assumptions and limitations should concern and be described in the goal and scope definition, although the ISO 14044 standard (2006) use both terms widely to describe LCA method as a whole and underline that assumptions and limitations *"…shall be transparent and presented in sufficient detail to allow the reader to comprehend the complexities and trade-offs inherent in the LCA*". As reported by Baumann and Tillman (2004), their description should be done likely for major assumptions and not for assumptions on single data sets or the like (e.g., in case of a system expansion), while limitations may arise from decision taken in the scope phase (e.g., the analysis may cover a certain geographical area or a specific period). Other limitations might concern the unavailability of some data which, according to their relevance, might considerably affect the impact assessment phase.

Generally speaking, the main limitation of LCA application to agricultural sector is basically due to the inherent variability that characterizes the primary production compared to industrial activity. The yield of crops, in addition to depending on the physiological processes of the crop, is highly sensitive to external factors and their variations such as the pedo-climatic conditions whose changes may not always be easily manageable or predictable.

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3. Inventory analysis

The life cycle inventory (LCI) development and elaboration concern the compilation and quantification of inputs and outputs characterizing a certain production system that in turn, are the basic information for the assessment of life cycle impacts (Vieira et al., 2016). Specifically, LCI consists in collecting and quantifying of inputs and outputs concerning the function or product generated by a process within the system boundary and related to the functional unit. It represents the most resource consuming in terms of time and work, even though it may be made it easier by the availability of good databases (Roy et al., 2009). Since inventory describes flows from and to the environment which are used for the impact assessment and interpretation LCA phases, it is essential that it is methodologically sound, complete, and unbiased (Sauer, 2012).

LCI is developed on the basis of the unit process that is the "*smallest element considered in the life cycle inventory analysis for which input and output data are quantified*" (ISO 14040, 2006). It may be considered a black box, that is the building element of a LCA analysis which transforms a set of inputs (e.g., products, waste for treatment, and natural resources) in a set of outputs (i.e., again products, waste for treatment, and residuals to the environment (i.e., pollutants to air, water, and soil, waste heat, and noise). Therefore, a product is not detrimental to environment as such, unlike processes include in the product realisation (Heijungs, and Guinée, 2012) (Figure 10).

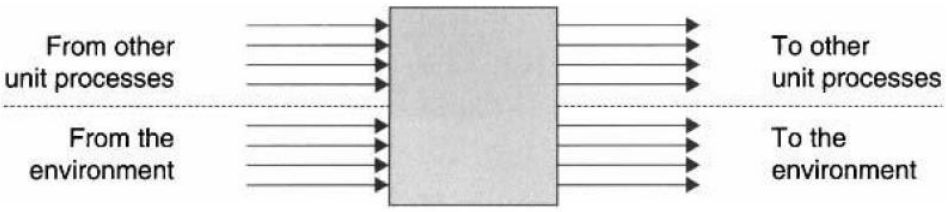

Figure 10 - Framework of a unit process (grey rectangle). It is characterized by inputs (left-hand side) and outputs (right-hand side) from and to other unit process (top arrows) and from and to the environment (bottom arrows) (Heijungs, and Guinée, 2012).

As reported by Baumann and Tillman (2004), different phases are included in LCI, that is: i) the construction of a flow chart representative of the system under consideration; ii) data collection concerning all inputs and outputs characterizing the system; iii) calculation of environmental burden (e.g., resource use and pollutant emissions) in relation to the selected functional unit.

The same authors above mentioned also underline that an inventory analysis is aimed basically to develop a flow model of the system under consideration, which may be considered an incomplete mass and energy balance in which only the environmental significant flows are included. Furthermore, LCI model is generally static, namely time is not considered as a variable and is represented as a flow chart – in which the activities are included, as well as the linear flows existing between them, depending on the system boundary developed in the goal and scope definition – (see section 2.3, Figure 6).

The criteria for modelling LCI and the method approaches used in the system modelling should be chosen consistent with goal and scope definition in addition to other principles such as reproducibility and robustness, practical feasibility, since this aspect has implication on the

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choice of inventory data that are to be collected (EC - JRC, 2010a). Although two logical approaches may be used to model a LCI that is, attributional and consequential LCA (ALCA and CLCA, respectively), ALCA is more used than CLCA for historical and practical reasons (Ronzon et al., 2017). As reported by EC - JRC (2010a), the ALCA "*depicts the potential environmental impacts that can be attributed to a system (e.g. a product) over its life cycle, i.e. upstream along the supply-chain and downstream following the system's use and end-of-life value chain. Attributional modelling makes use of historical, fact-based, measureable data of known (or at least know-able) uncertainty, and includes all the processes that are identified to relevantly contribute to the system being studied. In attributional modelling the system is hence modelled as it is or was (or is forecasted to be)*". In contrast to ALCA, CLCA "*aims at identifying the consequences that a decision in the foreground system has for other processes and systems of the economy, both in the analysed system's background system and on other systems. It models the analysed system around these consequences. The consequential life cycle model is hence not reflecting the actual (or forecasted) specific or average supply-chain, but a hypothetic generic supply-chain is modelled that is prognostizised along market-mechanisms, and potentially including political interactions and consumer behaviour changes*".

Specifically, the attributional LCI modelling principle tries to answer the question: "how are things (pollutants, resources, and exchanges among processes) flowing within the chosen temporal window?" whereas consequential LCI tries to answer the question: "how will flows change in response to decisions?". Furthermore, LCI may be prospective attributional ("how will things be flowing in the future?"), prospective consequential ("how will a future decision change flows?"), retrospective attributional ("how were things flowing in the past?") and retrospective consequential ("how did a prior decision change the flows?") (Curran et al., 2005). The use of two LCA approaches was debated in the LCA community, specifically concerning the data choice, that is between marginal and average data. In this regard, Ekvall et al. (2016) highlight that ALCA should be based on average data that represent the actual physical flows whereas marginal data should be used for CLCA, namely when they are important to model consequences, even though marginal data refers to changes that are small enough to be considered infinitesimal. Specifically, average data represent the average environmental load for obtaining a unit of goods and/or services in the system under consideration. On the other hand, marginal data refer to the impacts due to a small change in the output of goods and/or services from a system on the environmental load of the system. Generally, ALCA is not based on marginal data while average data showing the actual physical flows should be used. On the contrary, CLCA adopted marginal data when they are important for evaluating the implications (Finnveden et al. (2009). In short, ALCA may be considered static, context independent, and average, whereas CLCA ideally is dynamic, context specific, and marginal since the former is aimed to describe the operation mode of a statistic system regardless of economic or policy context, and the latter assesses how flows to and from environment would change as result of certain decisions (Plevin et al., 2013).

As regards data collection, it is considered the most time-consuming activity of LCA procedure, since no one may be an expert on all various technical aspects that might be included in a certain system and it is often difficult to have direct access to important data sources (Baumann and Tillman, 2004). Since a great amount of process and data may be necessary to inventory construction, the sources for the data and information should be clearly identified and documented in order to meet quality data requirements and thus the study goal.

The data used to create the inventory may be both quantitative and qualitative; the former generally concern the inputs of raw materials, the use of energy, products, any information

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useful for performing the allocation and environmental emissions into air, water and soil; the latter may refer to the description of the process technology, how and when the emissions were detected, the geographic location of the process and where the inflows come from and where the outflows are directed (Baumann and Tillman, 2004).

Two type of data source may be mentioned: i) primary data that come directly from the source (e.g., producers of goods and operators of processes and services) may include: interviews, questionnaires or surveys, bookkeeping or enterprise resource planning system, data collection tools (online, offline), and on-site measurements; ii) secondary data come from reports found in: databases, statistics, and open literature (Curran, 2012).

The inventory calculation, that is conducted subsequently to the construction of the flowchart and the data collection, consists in the following steps: i) normalizing data for all activities included in the system boundaries, that is inputs and output of a certain activity must be related to one of the products (i.e., to its specific unit such as 1 kg or 1 ton of product) and not to the functional unit. In practice, the normalization of data for each activity may concern the conversion of units that, although it may be considered a simple operation, actually it might lead to mistakes especially when dealing with a great quantity of data; ii) determining the flows linking the activities and the flows exiting the system boundaries, that is to calculate the amount of raw material and products involved in a specific process in relation to a quantitative reference flow (i.e. functional unit); iii) summing up the resource use and emissions to the environment related to the entire system under consideration; iv) documenting the calculation as well as possible (Baumann and Tillman, 2004).

The above procedure may be applied to an unit process in order to obtain an unit process dataset that results from quantifying of inputs and outputs related to a quantitative reference flow characterizing a specific process. It is possible to develop an aggregated process dataset from the collection of similar unit process or from other aggregated datasets (Curran, 2012).

3.1 Agricultural inventory

In the light of the above, the construction of an agricultural inventory should take into account all resources that are needed for and all emissions that are released by a certain crop system in relation to the functional unit (Brentrup et al., 2004). Specifically, an agricultural system should be described both in terms of the flow of energy and material entering the system and of the flow out of the system in the form of emissions into the environment, considering that the latter are highly dependent on climate, soil type, agricultural practices, all of which are easily subject to variation (Audsley et al., 2003). The inventory compilation related to a crop system might be supported by using of a specific questionnaire possibly fulfilled through direct interviews in order to facilitate the collection of reliable data and information (Mourad et al., 2007). Generally, the use of natural resources and agricultural inputs (e.g., fertilisers, pesticides, seeds, water use), implementation of crop management operations (e.g., tillage, sowing, irrigation, fertilization, weed control, harvesting) and crop production level are included in the inventory. Furthermore, the main features of the machineries used during field operations, (e.g., power, weight, fuel), energy consumption for the implementation of crop operation (e.g., irrigation), and the main field emissions arisen from mineral and organic fertilisers and pesticides should be also considered along with the release due to production of technical inputs (Audsley et al., 2003). Additional general information on farm, such as total area, effectively cultivated area, density of plants, crop and harvest practice management, cultivated varieties, type of soil,

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climatic conditions (e.g., temperature and rainfall) may be included in the inventory (Tables 5 and 6) (Mourad et al., 2007).

Table 5 - Example of an agricultural inventory (Mourad et al., 2007)

Parameters	Unit	Average (kg/t)	Average (kg/ha)
Input			
Energy			
Total	MJ		
for cultivation	MJ		
for processing	MJ		
for transport of the chemicals	MJ		
Other Resources			
Water for product processing	kg		
Fertilizers			
Total (actives and fillers)	kg		
N, P, K (macronutrients)	kg		
B, Cu, Fe, Mn, S, Zn (micronutrients)	kg		
Pesticides			
Total (actives and fillers)	kg		
Fungicide	kg		
Herbicide	kg		
Inseticide	kg		
Bactericide	kg		
Acaricide	kg		
Acaricide/ Inseticide	kg		
Correctives			
Total (actives and fillers)	kg		
Ca, Mg	kg		
Land Use			
Land use	ha.yr		
Output			
Organic residue used as fertilizer	kg		
Waste water	kg		

Table 6 - Example of agricultural data and information to include in an inventory (Mourad et al., 2007)

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Parameters	Unit	Average
Productive plants	Plants	
Density	Plants / ha	
Yield	Ton / ha	
Pluviometric index	Mm / year	
Solar radiation index	MJ / m^2.day	
Labour	labour hours / ton	
Plant varieties		
Soil classification		
Crop rotation	Comments about tillage sequences, period, etc.	
Main agricultural management practices	Comments about seeding type, soil preparation, soil covering, irrigation system, manual or mechanical harvesting, characteristics of the processing, etc.	

Although an inventory should contain data and information such as to allow an exhaustive description of crop system under consideration and consistent with the goal of LCA study, data and information might not be completely available. As reported in the description of goal and scope definition (particularly in the sections concerning cut-off criteria, assumptions and limitations, and data quality requirements), it is essential to clearly state the lack of certain data and information and their potential consequences for results in order to ensure the reliability and robustness of a LCA study.

The publication of a LCA study containing confidential data should not be disclosed in order to avoid the dissemination of sensitive information. However, the unavailability of certain LCA study might make it difficult the decision-making aimed to identify the best possible solution relative to a specific system. For this reason, it might be useful to disclosure average data instead of single one. For instance, Table 3 and 4 do not show the exact chemicals and commercial names that may be made available only for internal purposes (Mourad et al., 2007). The potential dissemination of a LCA study and the result management may be clarified during the goal and scope definition (see section 2).

The level of GHGs emitted during crop growing season may be deemed a key factor to assess the environmental burden related to the agricultural management of a certain crop system. Since the GHG release is mainly dependent on the degree of agricultural intensification, on what type of agricultural practice are adopted, and on level of resource consumption and of technical inputs, the identification, and the computing of GHG fluxes is a crucial operation in a LCA study.

A detailed description of how identifying, computing, and reporting GHG emissions to obtain a comprehensive agricultural inventory is reported hereafter, using as reference some information contained in the GHG Protocol. This protocol was developed by the World Resources Institute (WRI) and the World Business Council for Sustainable Development (WBCSD) in 1998 using the life cycle and attributional approaches to GHG computing. It is aimed to develop accounting and reporting standards for GHG emissions, that are specifically designed for different private and public sector activities such as agriculture, and to reduce the potential negative effects of climate change on natural resources (WRI and WBCSD, 2011a). The agricultural guidelines developed by WRI and WBCSD (2011b) resulted from the verification that agriculture is responsible for the production of GHG emissions and thus, it contributes to climate change. Therefore, a GHG emissions inventory is an essential tool in order to better understand the GHG fluxes due to a production activity and to develop effective adaptation and mitigation

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measures to tackle climate change. According to WRI and WBCSD (2011b), the main objective of these guidelines is to enhance the GHG accounting and reporting in order to obtain a clear inventory such as to meet the decision-making needs and to improve the management of agricultural GHG fluxes.

Several GHG sources that may be associated with agriculture (e.g., fuel consumption, soils, and manure management), may differently influence GHG flux in qualitative terms and consequently, to be essential for the construction of inventory, including computation, reporting, and performing the quality control of GHG flux data. Since agriculture is based on biological systems whose GHG emissions or removal generally are characterized by much more complicated mechanism than the emissions arisen from the mechanical equipment used in a farm, the GHG Protocol distinguishes between two types of emission sources: mechanical (i.e., equipment or machinery operated on farms) and non-mechanical (e.g., soil amendment and management, crop residue burning, and land use change) emission sources, (Figure 11).

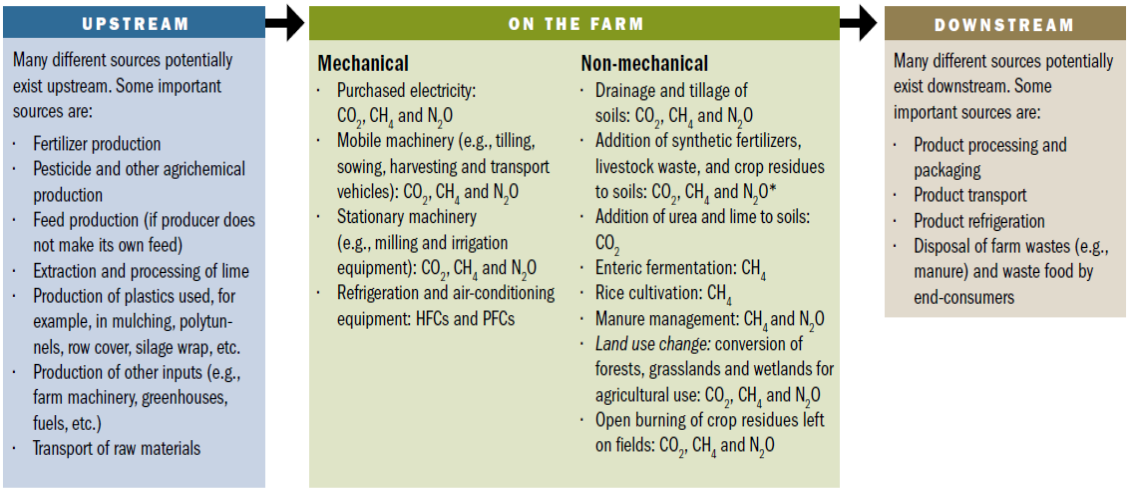

Figure 11 - Some of the most relevant emission sources associated with agriculture (Russell, 2011)

In the setting up of inventory, particular attention should be paid to both emission sources, also considering that at farm scale, the contribution of the GHG sources may deeply differ depending on the type of farm (e.g., crop or livestock), management practices (e.g., concerning land and waste), and natural factors (e.g., original land cover farm topography and hydrology, soil microbial density and ecology, soil temperature, moisture, organic content, and composition). It is difficult to precisely predict the relative magnitude of different sources for a certain farm, although some general information may be provided (Table 7).

Table 7 - Example of contribution of various sources to overall GHG fluxes due to different type of farming system (WRI and WBCSD, 2011b)

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Emission source	Type of system				
	Sheep	Beef	Dairy (pasture)	Arable crop	Horticulture
Enteric fermentation					
Deposition or application of fertilizer and/or wastes to soils					
Crop residue burning					
Manure management					
Fuel use					
Soil CO_2					

Key:

	Small contribution
	Medium contribution
	Large contribution

Among the non-mechanical sources, the GHG Protocol highlights the importance of including emissions from soil management (i.e., N_2O emissions and carbon pools) within GHG inventory because of the several agricultural operations in which may be involved and from which may be originated.

N_2O emissions due to an agricultural activity depends on the N availability. The increase in N supply caused by the distribution of fertilisers or animal wastes to soil, or arising from the storage and manure treatment encourage denitrification and nitrification processes which are responsible for the production of N_2O emissions. The latter may directly happen from the site of manure storage, or fertiliser application, or they may indirectly happen via leaching and volatilization. The volatilized N is finally deposited on soil or on surface water body such as lakes, in which N_2O emissions occur. The leached N may also cause N_2O emissions from the groundwater below the farm and from ditches, rivers, estuaries, etc. that may accumulate the leachate.

Arable land may be considered a C pool which may act both as source and sink of CO_2. Specifically, an agricultural system may contain C in different pools from which the sequestered C may potentially be emitted to the atmosphere in the form of CO_2. These pools are: above-ground and below-ground biomass (e.g., trees, crops, and roots), dead organic matter (DOM) in or on soil (i.e., decaying wood and leaf litter), soil organic matter (i.e., non-living biomass that is too fine to be recognized as dead organic matter), and harvested products. Furthermore, variations in C stocks may last decades before reaching an equilibrium as a result of a change in farm management or land use.

The agricultural guidance identifies two emission sources: direct and indirect that, according to WRI and WBCSD (2011b), refer to "*emissions from sources that are owned or controlled by the reporting company*" and to "*emissions from sources that are owned or controlled by another company, but are nonetheless a consequence of the activities of the reporting company*", respectively. The GHG Protocol provides for the setting up of the organizational and operational boundaries to make it easier the identification of emission sources should be included in the inventory. Specifically, organizational boundaries determines "*the operations owned or controlled by the reporting company*….." and operational boundaries determine "*the direct and indirect emissions associated with operations owned or controlled by the reporting company*".

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All emissions are categorized in three different scopes aimed to make it easier and more precise the computation of GHG emissions. Specifically, Scope 1 refers to "direct GHG emissions from sources owned or controlled by the reporting company" specifically it includes direct emissions arisen from mobile equipment, stationary combustion, and refrigeration and air-conditioning systems. Scope 2 concerns "*indirect emissions from purchased electricity, steam, and other energy sources*" such as emissions from the purchased electricity. Finally, Scope 3, that is "*indirect emissions other than those covered in scope 2*", for instance emissions from production of agrichemicals and purchased feed (Table 8).

The GHG Protocol also provides information on the step necessary for the accounting and reporting of GHG emissions. In this regard, we refer to section 4 in which it is reported a detailed description of the emission computing and of the whole environmental burden associated with an agricultural activity according to LCA methodology.

Table 8 - Example of setting operational boundaries (Russell, 2011)

A producer owns or controls all of the sources occurring on its farm and sells its produce to a food processing company.		
Emission source (example)	As accounted by the:	
	Producer	Food processor
Non-mechanical sources (e.g., enteric fermentation, soils, manure management, and land-use change)	Scope 1	Scope 3
Mechanical sources (excluding purchased electricity)	Scope 1	Scope 3
Electricity purchased by the producer for use in agricultural operations	Scope 2	Scope 3
Agrichemical production	Scope 3	Scope 3
Product processing	Scope 3	Scope 1 or 2

As regards the SolAqua project, the photovoltaic plant might be considered a component of cropping system, since it represents the energy source of the irrigation system. As reported by Parvez Mahmud et al. (2018) the inventory relating to a photovoltaic system should include data and information on input resources (e.g., raw materials and energies) and output emissions per unit process. Specifically, energy and material flows should be describe throughout the life cycle of photovoltaic system depending on the availability of information in terms of raw-material extraction from mines, raw-material transportation to the plant location for material manufacturing, transportation of the produced materials to the solar plant area, installation and operation of the plant, and finally the end-of-life waste management.

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4. Impact assessment

According to ISO 14040 standard (2006), Life cycle impact assessment (LCIA) "*.....is aimed at evaluating the significance of potential environmental impacts using the LCI results. In general, this process involves associating inventory data with specific environmental impact categories and category indicators, thereby attempting to understand these impacts. The LCIA phase also provides information for the life cycle interpretation phase. The impact assessment may include the iterative process of reviewing the goal and scope of the LCA study to determine if the objectives of the study have been met, or to modify the goal and scope if the assessment indicates that they cannot be achieved*".

Specifically, this phase may be considered the translation of the inventory data into numerical values called environmental indicators whose function is to provide information on environmental burden of a product or a system (Greenhut et al., 2013). LCIA is aimed to analyse and evaluate the extent and relevance of the potential environmental impacts caused by a product or a system over its life cycle (Heijungs and Guinée, 2012). It is essential to perform a LCA study, especially since it allows to improve the readability of results and thus, making them more comprehensible and easier to disclose. The inventory result may include from 50 to 200 parameters or more, making it difficult to handle. LCIA enable to reduce the parameter number to about 15 by grouping the environmental burdens resulted from inventory into environmental impact categories, or even down to one by weighting of the impact categories (Bauman and Tillman, 2004). LCIA is aimed to evaluate potential human health and environmental impacts of the natural resources and environmental releases detected throughout the inventory (Margni and Curran, 2012).

In other words, this phase allows to link LCI results (i.e., emissions and extractions) to their potential environmental damages on the basis of impact pathways which may be considered connected environmental processes, and thus they represent the casual chain of effects arisen from an emission or extraction (Jolliet et al., 2004).

The complexity of an environmental problem may make it difficult to describe and hence, to understand the environmental impacts of emission and resource use quantitatively, for instance via the three environmental categories resource use, human health and ecological consequences. A cause-effect chain may depict this complexity since, for instance, it enables to highlight how the emissions of the pollutants are linked to their effects (Figure 12). A primary effect of a certain pollutant may be responsible for various secondary effects or vice versa as well as to cause potential feedback effects (Baumann and Tillman, 2004).

Specifically, the effect of a stressor may be evaluated via many cause-effect chain models, which may be considered both independent and interactive, and are aimed to show the linkages between a midpoint indicator, that is related to the stressor, and an endpoint indicator that is connected to the midpoint one. For instance, through a cause-effect chain the effect of CO_2 emissions (i.e., stressor) may be evaluated relative to radiative forcing (i.e., a midpoint indicator) and human health (i.e., an endpoint indicator which depends on global warming). In other words, a midpoint indicator refers to a specific ecological issue (e.g., radiative forcing), whereas an endpoint indicator stands for primary dimensions of well-being (e.g., human health) (Othoniel et al., 2019) (see section 4.1 for more detailed information on midpoint and endpoint indicators).

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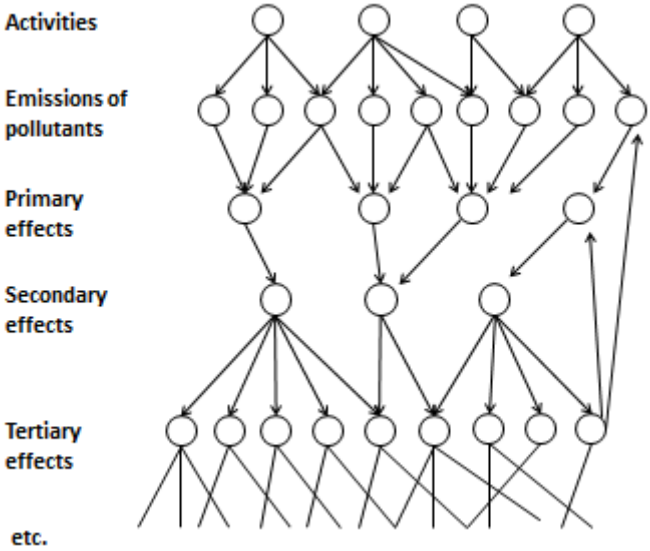

Figure 12 - Cause-effect chain of environmental impacts (Baumann and Tillman, 2004)

In the light of the above, it seems clear that cause-effect chains enables to present environmental impacts at different levels. Therefore, an environmental burden described as low-order load means that the potential effect of a pollutant are considered rather than the actual one neglecting the pollutant fate. On the other hand, this procedure is useful to avoid the trouble concerning the addition of environmental problems located in various geographical sites in a life cycle. Consequently, global impacts are better addressed than local ones (Baumann and Tillman, 2004).

The areas of protection on which commonly environmental burdens is assessed, are: i) natural resources; ii) natural environment; iii) human health; and man-made environment (Margni and Curran, 2012). They may be considered a cluster of category endpoints of recognizable value to society (Guinée, 2015). The use of the man-made environment as an endpoint indicator has been suggested considering that polluting impacts may affect not only the natural environment, but also society (e.g. acid rain could cause corrosion on steel structures and, consequently damage to cultural heritage) (Baumann and Tillman, 2004).

To generate LCI data and to perform the assessment of environmental burdens, specialized software tools (e.g., GaBI, SimaPro, Umberto, and OpenLCA) make available several impact assessment methodologies and databases. The wide availability of datasets within LCA software tools may greatly reduce the time and effort required to perform a LCA study, although the practitioners should be aware of the differences in various databases and the necessity to review them and to make adjustments in order to ensure consistency for the system under consideration (Sauer, 2012). Similar considerations should be also to be taken into account to impact assessment methodologies and toolset which, thus should be chosen so as to meet the goals of study. In fact, the slight differences detectable in the impact assessment methodologies and software tools might influence results (Sieverding et al., 2020).

According to ISO 14040/14044 standards (2006), LCIA consists of mandatory and optional elements. The former includes: selection of impact categories, category indicators and characterization models; classification; and characterization. The latter covers: normalization, grouping, and weighting (Figure 13).

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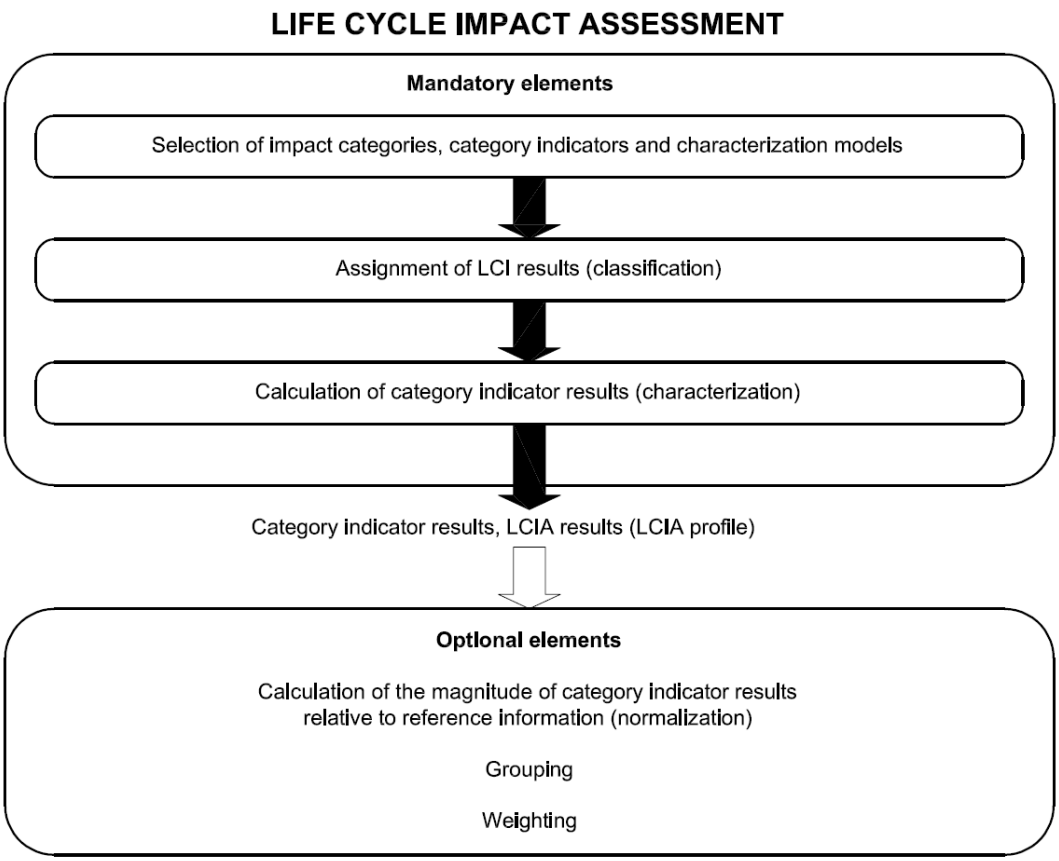

Figure 13 - Elements of the LCIA phase (ISO 14040 standard, 2006)

Selection of impact categories, category indicators and characterization model regard the choice both of impacts depending on the goal of the study and a method for all considered impact category (Hauschild and Huijbregts, 2015). Specifically, the selection of impact categories must be exhaustive, that is all important environmental issues associated with the system under consideration must be included unless a limitation was indicated in the goal definition. The exclusion of important impacts should be well documented and taken into account in the interpretation of results (EC - JRC, 2010a).

Classification is the process in which LCI data are sorted and assigned to different impact categories. Specifically, the elementary flows included in the inventory (e.g., resource consumption and emissions into air or water) are associated with impact categories selected in the previous phase, depending on their capacity to contribute to various environmental problems (Hauschild and Huijbregts, 2015).

Characterization is the assessment of impact, namely the evaluation of the intensity of each impacts related to inventory data previously associated with different environmental categories making possible the LCI output comparison within each category (Roy, et al., 2009). In other words, in this phase the magnitude of the environmental impact category is determined (Baumann and Tillman, 2004).

Normalization relates the value of environmental indicators to reference values to better understand the relative magnitude of each indicator used to evaluate the environmental

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performance of a certain system. Normalization changes an indicator result by dividing it by a selected reference value (e.g. the global, regional, national or local value for the respective impact categories) (ISO 14044, 2006). In this phase, the characterised impact value is connected to a reference, such as the impacts produced by one person throughout one year in a certain region, making it easier comparisons among categories and/or Areas of Protection (EC - JRC, 2010b).

The weighting phase enables to convert indicator results by using numerical factors (so-called weighting factors) based on value-choices in order to express the magnitude assigned to each impact category and to compare of indicator results across impact categories (Hauschild and Huijbregts, 2015). Specifically, weighting may ease decision making when trade-offs among impact category results do not enable to find the best option among the alternatives or an improvement. The weights may be considered an evaluation of the relative importance of an environmental impact, on the basis of value choices, which represent preferences of, for instance people, experts or organisations, regarding time (present vs future impacts), geography (local vs global), urgency, political agendas or cost (Pizzol et al., 2017).

Grouping (and also sorting) is aimed to reduce the number of impact categories and ranking them on the basis of their importance (Margni and Curran, 2012). In this phase, the values resulting from weighting may be aggregated across to impact or damage categories to a single score in order to facilitate the interpretation (Hauschild and Huijbregts, 2015).

As regards impact assessment, it is important to underline that the impacts evaluated via LCIA should be considered as impact potentials and not as actual impacts, nor overcoming of thresholds or safety margins, or risk, since they are: i) relative expressions of potential impacts associated with the life cycle of a reference flow which, in turn, is connected to a functional unit; ii) based on inventory data that often refer to different spatial and temporal scales; and iii) based on impact assessment data which may lack information regarding the specific environmental context, such as the simultaneous exposure to substances coming from other product systems (Rosenbaum et al., 2018).

A detailed description of each LCIA element is reported below.

4.1 Selection of impact categories, category indicators and characterization models

According to ISO 14040 standard (2006), impact category is the "*class representing environmental issues of concern to which life cycle inventory analysis results may be assigned*" (e.g., climate change) and category indicator is the "*quantifiable representation of an impact category*" (e.g., radiative forcing). An impact category and a category indicator are quantitatively connected via characterization model which thus, may be considered a quantitative representation of the impact associated to the elementary flows with respect to a certain category indicator. In other words, characterization model quantifies the contribution provided by both inflows and outflows of materials, resources, substances, etc. on a specific category indicator, namely what is known as characterization factor (e.g., global warming potential). Therefore, to facilitate the collection of information on the important elementary flows within the inventory, impact categories should be chosen on the basis of what reported in the goal and scope definition phase about the objective of LCA study, namely before the inventory construction so as to ensure that the inventory results may meet targets regarding impact

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assessment. In other words, this phase is aimed to meet the following question: "*Which impacts do I need to assess?*" (Rosenbaum et al., 2018).

The selection of impact categories should followed different criteria: i) completeness, namely the set of impact categories should encompass environmental problems both of general interest and of specific interest for a certain LCA study; ii) practicality, that is too many categories should not be included in the list; iii) independence, in order to avoid double counting; iv) possibility to integrate in the LCA calculations, this means that the linkage between LCI result parameters and the selected impact categories should be feasible; v) environmental relevance, it refers to the environmental relevance of category indicators with respect to impact categories and protection areas; vi) scientific method, this aspect concerns the validity of characterization methods (Baumann and Tillman, 2004).

Impact categories and their indicators may be arranged according to two different level along the cause-effect chain: at midpoint and endpoint level (Margni and Curran, 2012). Both of them refer to cause-effect chain, although in different points. The midpoint method analyses the impact in the cause-effect chain prior to the endpoint one is reached, whereas the latter is focused on environmental impacts at the end of the cause-effect chain. Specifically, the midpoint or problem oriented approach converts impacts in environmental subjects (e.g. climate change and human toxicity), whereas the endpoint or damage-oriented approach converts environmental impacts in environmental issues of interest (e.g., human health, natural environment and natural resources) (Chatzisymeon et al., 2017) (Figure 14).

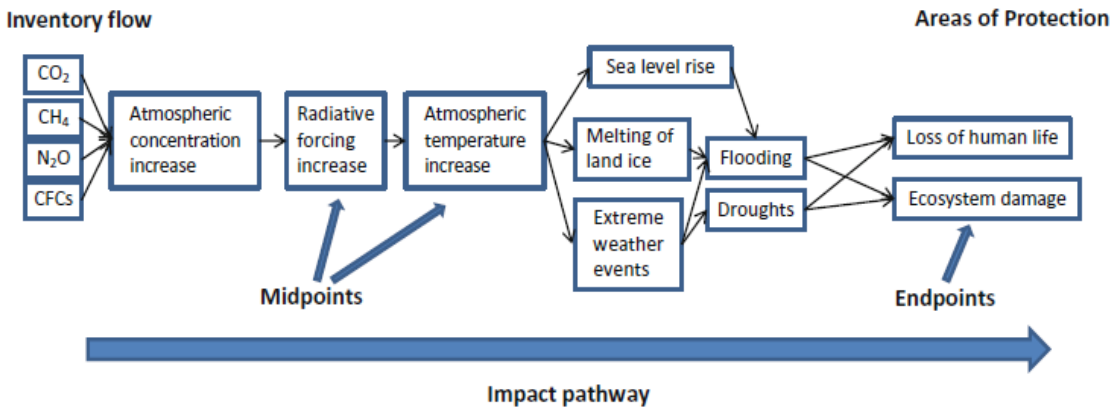

Figure 14 - Connection of elementary flows for global warming from inventory to the areas of protection (Hauschild and Huijbregts, 2015)

Generally, midpoint indicators are characterized by a lower uncertainty since they are a small part of environmental pathway under consideration, whereas endpoint indicators may have a significant level of uncertainty but for this much simpler to understand and interpret by decision makers (Goedkoop et al., 2016). Several midpoint indicators may contribute to a relatively small number of endpoint indicators which are representative of the areas of protections (also reported to as damage or severity), namely human health, natural resources, and ecosystem quality (Rosenbaum et al., 2018). Furthermore, the same set of impact categories as midpoint indicators may be associated with more than one endpoint indicators (e.g., climate change usually has one midpoint indicator, but two endpoint indicators, that is human health and ecosystem quality).

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From a practical point of view, several ready-made LCIA methods are available for practitioner who does not need to deepen the different impact assessment steps since the environmental information on pollutants and resources is aggregated to a characterization indicator or a single number (i.e., an index) inside the ready-made LCIA methods (Baumann and Tillman, 2004).

As reported by Rosenbaum et al. (2018) numerous category indicators associated with certain characterization models are combined into predefined methods, such as ReCiPe, CML, TRACI, EDIP, LIME, IMPACT 2002+, etc., and included in LCA software. Moreover, considering the wide availability of LCIA methods and indicators and that ISO 14040/14044 do not provide guidance on which LCIA method should be used, the choice of the assessment procedure should be made on the basis of criteria and questions aimed to facilitate the systematically identification of the most appropriate method. For instance, some questions might concern:

a) selection of impact categories or environmental issues depending on specific needs;
b) region in which the life cycle under consideration occurs;
c) choice between midpoints, endpoint or both;
d) choice of the elementary flows to characterise;
e) availability of documentation about the selected method;
f) ease of disclosing the results;
g) need to perform normalisation and if yes, the selection of reference system since in most cases it would be best to avoid mixing characterisation and normalisation factors from different LCIA methods because of the difference in characterisation modelling, units, numerical values, etc.;
h) resources and data availability to use a regionalised methodology, and consequently obtaining more precise results.

Baumann and Tillman (2004) have highlighted that characterisation methods are based on scientific methods, coming from environmental chemistry, toxicology, ecology, etc. for describing an environmental burden whose complexity may lead to certain impact categories which may have several alternative characterisation models. Furthermore, the characterisation methods used for several emission-caused impacts (e.g., acidification, eutrophication, and global warming) are more developed than others related to, for instance, resource use, land use, and toxic substances resulting in an assessment in which some environmental impacts are accentuated at the expense of others.

Over time, different set of impact categories were developed aimed to perform a LCIA considering both midpoint and endpoint approaches. Diversity and overlap in impact categories that characterise these lists might be due to a semantic question although they cannot be completely explained only by the name of impact category, but they may depend on the specificity of environmental problem to whom are addressed (Guinée, 2015). Therefore, the set impact categories may result from information included in the inventory analysis and used to specify environmental impacts considered important in the goal and scope definition (Baumann and Tillman, 2004).

As reported by Rosenbaum et al. (2018), typical and *new* midpoint categories, and respective sub-categories are:

a) Climate change;
b) Stratospheric ozone depletion;
c) Acidification (terrestrial, freshwater);
d) Eutrophication (terrestrial, freshwater, marine);
e) Photochemical ozone formation;

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f) Ecotoxicity (terrestrial, freshwater, marine);
g) Human toxicity (cancer, non-cancer);
h) Particulate matter formation;
i) Ionising radiation (human health, aquatic and terrestrial ecosystems);
j) Land Use (biotic productivity, aquifer recharge, carbon sequestration, albedo, erosion, mechanical and chemical filtration capacity, biodiversity);
k) Water use (human health, aquatic ecosystems, terrestrial ecosystems, ecosystem services);
l) Abiotic resource use (fossil and mineral);
m) *Biotic resource use* (e.g. fishing or wood logging);
n) *Noise*;
o) *Pathogens*.

A brief description of the main impact categories included in the most commonly used assessment methods is reported below.

4.1.1 Climate change

According to IPCC (2018) "*Climate change refers to a change in the state of the climate that can be identified (e.g., by using statistical tests) by changes in the mean and/or the variability of its properties and that persists for an extended period, typically decades or longer. Climate change may be due to natural internal processes or external forcings such as modulations of the solar cycles, volcanic eruptions and persistent anthropogenic changes in the composition of the atmosphere or in land use*".

As observed by Levasseur (2015), greenhouse gases are able to absorb radiation at certain infrared wavelengths coming from the Earth's surface and by clouds, causing a net warming effect called the greenhouse effect. Furthermore, human activities (e.g., deforestation) may responsible for the change in the fraction of solar radiation which is reflected by Earth's surface (i.e., terrestrial albedo), producing a net warming or cooling effect. These climate forcing elements may affect radiative forcing, namely the perturbation of the Earth's energy balance resulting in an increase in the temperature of atmosphere and oceans along with changes in rainfalls, extreme meteorological events, and increase in the sea level. These climate perturbations may have negative effect on agricultural production and foster desertification, ice cover reduction, flooding, etc. also resulting in damages to two protection areas, that is human health and ecosystem quality.

GHG emissions are the only elementary flows included in the present LCIA methodologies. Carbon dioxide is considered the main responsible for climate change; its primary source is fossil fuel combustion and the effects of land use change on plant and soil carbon. Other climate-change emissions are: nitrous oxide (N_2O) which comes from agricultural activity, especially from fertilization and land use; and methane (CH_4) which is mainly produced by agriculture (i.e., ruminant animals, rice cultivation, and biomass burning), with a lower contribution from industrial activity (e.g., landfill, natural gas processing) (Levasseur, 2015).

The potential contribution of a substance to climate change impact category is represented by global warming potential (GWP) which expressed by the ratio between the increased infrared absorption of which is responsible and the increased infrared absorption produced by 1 kg of CO_2 used as reference substance (Baumann and Tillman, 2004). The computing of GWP is made for all GHGs and considering three time horizons (i.e., 20, 100 and 500 years) (Levasseur, 2015).

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The GWP values used in LCA have been calculated by Intergovernmental Panel on Climate Change (IPCC), that consider the above three time horizon since GHGs have different life span in the atmosphere (Baumann and Tillman, 2004). EC - JRC, (2011) highlights that all LCIA methodologies include an impact category "Climate Change" developed by IPCC which periodically publishes updates regarding midpoint characterization factors, even though not all methodologies use them. Some of LCIA methodologies calculates GWP values on the basis of 100-year time horizon because it is the reference adopted for the Kyoto Protocol. Others consider 500-year time frame in order to assess the most of the damage produced by the substances characterised by a log residence time in atmosphere whereas the 100 and 20-year time horizon catch partially the effect of substances with a long life span. Different LCIA methodologies also include characterization factors for more than one time horizon to enable the user to test the sensitivity of results related to a subjective choice (i.e., the time frame) (Levasseur, 2015).

4.1.2 Stratospheric ozone depletion

Ozone (O_3) is a natural component of the Earth's atmosphere and it is mostly found in the upper atmosphere, the stratosphere. Specifically, approximately 90% of the ozone is located in the lower to middle stratosphere, that is the band between about 15 and 35 km altitude (the so-called ozone layer), whereas the remaining 10% resides in the troposphere (Langematz, 2019).

The maintenance of ozone layer occurs via a complex sequence of chemical reactions (Baumann and Tillman, 2004). Indeed, ozone is a highly reactive substance whose presence in the stratosphere is due to a constant cycle of formation and degradation processes. The latter may happen chemically and by photo dissociation (Lane, 2015). Ozone is obtained from the absorption of ultra violet (UV) radiation by oxygen molecules and it is destroyed by UV radiation, visible light and specific substances which act as catalysts (e.g., H, OH, NO, Cl, and Br). Specifically, the higher the concentration of these catalysts the greater the rate of ozone breakdown over the time they are in the stratosphere. An increase of CO_2 emissions causes the contrary effect, namely high CO_2 emissions are responsible for a lower stratospheric temperatures and consequently, brake the ozone destroying reactions (Baumann and Tillman, 2004).

The ozone affecting substances encompasses halogen gases, such as chlorine and bromine, which are emitted at Earth's surface by human activities or natural process (troposphere) before they arrive at stratosphere where are converted from inactive reservoir gases ($ClONO_2$, $BrONO_2$, and hydrogen chloride HCl) to chlorine (Cl) and chlorine monoxide (ClO) radicals in the higher stratosphere where they are able to destroy ozone (Langematz, 2019). Similar to halogen gases, nitrous oxide (N_2O) is a long-lived stable substance in the troposphere whereas in the stratosphere, 90% of N_2O is convert to stable N_2. A portion of the remainder breaks down in NO radicals that trigger catalytic ozone destruction (Lane, 2015). Bacterial activity is responsible for N_2O production in natural and cultivated soils (Langematz, 2019). Unlike N_2O emissions, carbon dioxide and methane may have a mitigating effect on depletion of ozone layer, since the radiative properties of CO_2 and water vapour which reacts with CH_4 are able to reduce temperatures in the stratosphere and consequently slow down the rate of ozone destruction (Lane, 2015).

Depletion of the ozone layer may result in an increase of the transmission of UVB to the Earth's surface. Excessive exposure to UVB may be responsible for different damages to human health

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and ecosystems. As reported by Lane (2015), mixed results are detected regarding effects on terrestrial ecosystem. Some fields studies have depicted decreases in plant productivity due to extreme changes in UVB radiation in high latitude regions, whereas others have highlighted high UVB levels may positive effects on productivity by hindering herbivorous insect activity. The effect caused by changes in UVB radiation on agri-production systems is also somewhat unclear.

The ozone depletion potentials (ODP) used in LCA were developed by World Meteorological Organisation which is also involved in their update. ODP factors are ratios of the ozone change caused by an unit emission of a specific substance, compared with the ozone change due to an unit emission of CFC-11 (Lane, 2015). A theoretical steady-state model is used for the computation of the ODP of a substance because of the complexity and poor understanding of ozone chemistry (Baumann and Tillman, 2004). Steady-state ODP may be calculated using modelled predictions of ozone loss or using a semi-empirical procedure on the basis of observed data regarding the atmospheric performance of chlorine and bromine (Lane, 2015). ODPs are available for the most significant chlorinated and bromated substances, whereas they are missing for different substances which affect the stratospheric ozone depletion. Furthermore, the use of ODPs calculated for short time horizons might be appropriate since a steady-state is achieved in hundreds of years and the ozone depletion rate may be higher for several ozone depleting substances in the short term (Baumann and Tillman, 2004).

4.1.3 Toxicity

Toxicity is a complex impact category which is often divided into human toxicity and eco-toxicity (which includes aquatic toxicity (i.e., freshwater and marine toxicity) and terrestrial toxicity) because of the several types of impacts and likely a great amount of substances that includes. For instance, organic solvents, heavy metals and pesticides may cause various types of toxic effects, such as neurological damage, carcinogenic and mutagenic effects, whereas some toxic substances may spread such as pesticides utilised in agriculture, may reach water bodies damaging aquatic organisms or making drinking water unhealthy (Baumann and Tillman, 2004).

As reported by Brentrup et al. (2004), the potentially toxic emissions which may result from an agricultural activity are inorganic air pollutants (e.g., NH_3, SO_2, and NO_x), plant protection substances, and heavy metals. Arable farming systems depicted that not less than 70% of the SO_2, NO_x, NH_3, CO and particle emissions are produced during on-field operations (e.g., tractor use, fertiliser distribution) in spring and summer. Substances used for weeding may have a negative impact on terrestrial and aquatic ecosystems or even humans since a part of the distributed chemical compounds may undergo phenomena of wind drift, evaporation, leaching, and surface run-off. Furthermore, the application of mineral fertilisers (e.g., phosphate) and organic materials such as and slurry, sewage, sludge, or compost may release emissions of heavy metals to soil. In case of heavy-metals, contamination may change essentially on the basis of the origin of the raw material (e.g., P rock, industrial and household waste).

Toxicity, both towards humans and ecosystems is essentially characterised by the same driving factors, that is (1) emitted quantity (determined in the LCI), (2) mobility, (3) persistence, (4) exposure patterns and bioavailability (specific for ecotoxicity), and (5) toxicity. Specifically, toxicity is not the only parameter able to define the potential ecotoxic impact of a substance in the environment, since it has to reach the potential target organism. For instance, a chemical may be highly toxic, but never reach a target organism because of its short lifetime (i.e., rapid deterioration) or low mobility which allows to be reach a target organism remaining into soil or

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into a sediment, and thus contributing little to ecotoxic impact. Chemical emissions may damage terrestrial, freshwater, marine and aerial ecosystems on the basis of environmental context and the features of the released substance, causing increased mortality, reduced mobility, reduced growth or reproduction rate, mutations, behavioural changes, changes in biomass or photosynthesis, etc. (Rosenbaum et al., 2018).

Models and factors used to deal with toxicological effects within the LCA procedure should be based on the relative risk and connected consequences of chemicals released into the environment. Moreover, LCA characterisation models and factors related to toxic effects should be depend on models which consider chemical's fate in the environment, human exposure, and differences in toxicological response in terms of likelihood of effects and severity (EC - JRC, 2010b). According to Rosenbaum (2015), the LCIA characterisation factor for ecotoxicity consists of four parts, that is:

1. *"Fate modelling estimates the increase in concentration in a given medium due to an emission quantified in the life cycle inventory.*
2. *The exposure model quantifies the chemical's bioavailability in the different media by quantifying the bioavailable fraction of the total concentration.*
3. *The effect model relates the amount available to an effect on the ecosystem. This is typically considered a midpoint indicator in LCA, as no distinction between the severity of observed effects is made (e.g. a temporary/reversible decrease in mobility and death are given the same importance).*
4. *Finally, the severity (or damage)model translates the effects on the ecosystem into an ecosystem population (i.e. biodiversity) change integrated over time and space"*.

These components are included in the definition of the characterisation factor (CF) related to a specific substance and a specific emission compartment:

$$CF = FF \times XF \times EF \times SF$$

where FF is the fate factor, namely this model "*predicts the chemical behaviour/distribution in the environment accounting for multimedia (i.e. between environmental media and compartments) and spatial (i.e. between different zones but within the same compartment or medium)*"; XF the exposure factor, that is "*the contact between a target organism and a pollutant over an exposure boundary for a specific duration and frequency*"; EF the effect factor, which represents "*the fraction of species within an ecosystem that will be affected by chemical exposure*"; and SF the severity factor which is associated with a damage model. "*A damage model, incorporating the severity of the effect, goes even further along the cause-effect chain and quantifies how many species are disappearing from a given ecosystem*". Disappearance may be responsible for mortality, reduced proliferation, or migration, for example (Rosenbaum, 2015).

Human toxicity is basically characterised by the same driving factors of ecotoxicity, even though the specific mechanism and parameters are different, particularly regarding the exposure modelling in which several factors capturing human behaviour (e.g., dietary habits) affect human exposure pattern. Chemical exposure of humans may result from both emissions into environment which may be harmful for the entire population, but also from the numerous chemical components of products which are released during the production, use or end-of-life treatment and thus, may be detrimental for workers and consumers. Chemical emissions may be responsible for or contribute to a number of non-cancerous diseases as well as increase the

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risk of cancer from carcinogenic chemicals (Rosenbaum et al., 2018). The characterisation factor for human toxicity is substantially determined as for ecotoxicity, that is considering the environmental fate (F), exposure (X), dose-response (R) of a chemical for midpoint factors and additionally severity (S) for endpoint factors, according to the following equation:

$$CF = S \times R \times X \times F = S \times R \times iF$$

The fate factor connects the emission flow to the change in mass in the environment while the exposure factor relates the variation in mass in the environment to the change in intake rate. The dose-response slope is the likelihood of an additional effect per unit additional intake and the severity is the effect per case connected to mortality and morbidity. iF (i.e., intake Fraction) results from the combination of fate with exposure factor and shows the fraction of emission which is absorbed by the whole population. These parameter may change on the basis of location (e.g. habitat characteristics, local stressors, mixtures, background concentrations) and time (e.g. seasonal life stage sensitivity) (EC - JRC, 2010b).

According to Rosenbaum et al. (2018), the midpoint human toxicity characterisation factor (i.e., number of cases/$kg_{emitted}$) is "*the toxic impact on the global human population per mass unit emitted into the environment*" and the endpoint human health characterisation factor (i.e., DALY/$kg_{emitted}$) "*quantifies the impact on human health in the global population in Disability-Adjusted Life Years (DALY) per mass unit emitted into the environment*". DALY is a statistical measure which expresses the population life years lost or affected by disease.

4.1.4 Particulate Matter Formation

Particular matter (PM) may be deemed one of the most significant environmental stressor in terms of contribution to the global human disease burden and responsible for harmful effects on human health. PM may be differentiate on the basis of formation type (primary and secondary) and aerodynamic diameter (respirable, coarse, fine, and ultrafine). Primary PM refers to the direct emission of particles, such as from road transport, power plants or farming activities whereas secondary PM is related to organic and inorganic particles formed via reactions of precursor substances (e.g., nitrogen oxides (NO_x), sulphur oxides (SOx), ammonia (NH_3), semivolatile and volatile organic compounds (VOC). Secondary particles also refers to sulphate, nitrate and organic carbonaceous materials and may reach 50% of ambient PM concentrations. As regards aerodynamic diameter, respirable particles (PM_{10}) is characterised by a diameter less than 10 μm, coarse particles ($PM_{10-2.5}$) between 2.5 and 10 μm, fine particles ($PM_{2.5}$) less than 2.5 μm, and ultrafine particles (UFP) less than 100 nm (Rosenbaum et al., 2018). $PM_{2.5}$ is used as a reference to integrate its effects on human health into LCIA. In epidemiological studies, exposure to $PM_{2.5}$ is considered responsible for different harmful health effects and reduction in life expectancy due to chronic and acute respiratory and cardiovascular morbidity, chronic and acute mortality, lung cancer, diabetes, and adverse birth outcomes. Also, toxicological studies highlights that the exposure to $PM_{2.5}$ may have effects on key biological systems, although there are some evidence that not all particles are likely responsible for the same health effects (Fantke et al., 2015).

The determination of the characterization factor for particulate matter is very similar to that of the other toxicity categories and it is based on the same equation above reported. The combination of all factors from emissions to health impacts or damages provides the

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characterisation factor for PM formation with unit (disease cases/$kg_{emitted}$) at midpoint level and (DALY/$kg_{emitted}$) at endpoint level (Rosenbaum et al., 2018).

As reported by Humbert et al. (2015), in the impact assessment, the effects on human health resulting from airborne pollutants may be represented by various units, mainly "non-health" based indicators, such as the equivalent quantity of a reference substance (e.g., kg of 1,4-DCB-eq or kg of $PM_{2.5}$-eq) and health based indicators such as the number of cases of illness, the number of premature deaths, the reduction in life-expectancy expressed as quality-adjusted life years (QALY) or as disability-adjusted life years (DALYs). The latter considers many forms of burden (e.g., mortality and morbidity) using disability weights globally recognised and DALYs allows the comparison of damages results across impact categories within methods that use DALY for impacts on human health. Furthermore, this unit is especially suitable to describe PM impacts since data used to define dose-response is expressed as health outcomes. At midpoint level, results may be expressed in in kg $PM_{2.5}$-equivalents by dividing the DALY by the quantity of DALY per kg of $PM_{2.5}$ since much of the health effects are generally associated with the particles with diameter less than 2.5 µm.

4.1.5 Photochemical Ozone Formation

High concentration of ozone, especially near Earth's surface may be detrimental to human health, vegetation, and organic materials. It is a secondary pollutant whose formation is based on a sequence of photochemical reactions for which sunlight and heat are necessary (Ye et al., 2016). In different LCIA methods, this impact category is mentioned through several various names, such as (tropospheric) ozone formation, photochemical ozone formation or creation, photo-oxidant formation, photosmog or summer smog. Although these expressions show small differences, they all basically deal with the impact due to ozone and reactive oxygen compounds produced in the troposphere as secondary pollutants by the oxidation of the primary pollutants volatile organic compounds (VOC, namely organic compounds with a boiling point below 250 °C), or carbon monoxide in the presence of nitrogen oxides (NO_x) under the influence of light (Rosenbaum et al., 2018). In this context, the main interest of LCIA methods is the impact estimation related to NO_x and NMVOC emissions as ozone precursors in the troposphere. NMVOC refers to many substances or a group of substances, such as alkanes, alkenes, aromatics, aldehydes and alcohols, etc. Sometimes VOC is used as a synonym for NMVOC, although the latter does not include methane. NO_x is the sum of NO and NO_2 (Preiss, 2015).

Photo-oxidant formation may be considered a problematical impact category for many reasons: i) several VOCs are subjected to similar enough reactions in the atmosphere, although every species has its own reaction pathway; ii) ozone production due to a specific VOC depends on the chemical and meteorological conditions of the environment in which VOC emission occurs; iii) the same VOC may provide a high ozone production with high NO_x concentrations and a low ozone production with poor availability of NO_x; iv) atmospheric NO_x levels are affected by several factors, such as the emission patterns, the strength of dispersion and on chemical loss; v) the production of numerous radicals during VOCs photolytic degradation will accelerate the oxidation of other VOCs and consequently, foster the ozone production. Therefore high radiation intensity will boost the efficiency of VOCs to produce ozone. In addition, a wide spatial and temporal variability in the ozone creation potential of VOCs is expected (Labouze et al., 2004).

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As reported by Rosenbaum et al. (2018), the harmfulness of photochemical contaminants basically depends on their reactive nature which allow them to oxidise organic molecules. Human health is affected when ozone and other reactive oxygen compounds during their interaction are inhaled damaging tissue and causing respiratory diseases (e.g., throughout smog episodes, the concentrations of ozone and other photo-oxidants may reach extreme levels causing immediate damage to human health). Impacts on vegetation occur when the reactive compounds interact with surface of plants or enter plant leaves via stomata producing oxidative damage on photosynthetic organs, and leading to discolouration of the leaves followed by withering of the plant. Although plants show different sensitivity to ozone and other photo-oxidants depending on season and species, remarkable growth reductions are detected in regions with high ozone concentration throughout the growth season as well as an agricultural production loss of 10-15% was estimated for common crop plants. On the other hand, VOCs are emitted in large quantity from vegetation, especially forest, but unless a man-made change of the natural system influences the VOCs emissions, these will not be include in a LCI and thus, no assessment of their impacts will be made. Important sources of these contaminants are: road traffic, the use of organic solvents, and industrial processes for VOC emissions; various forms of incomplete combustion of fossil fuels or biomass in stationary systems for carbon monoxide; and combustion processes in transport, energy- and waste incineration systems for nitrogen oxides.

The development of characterisation factors within LCIA methods considers the complexity and the large number of substances NMVOC both at midpoint and endpoint level. For instance, the midpoint indicator may be the ozone creation or ozone exposure whereas the endpoint or damage indicators may be human health impacts and impacts on natural vegetation and crops (Preiss, 2015). It is possible to determine the characterisation factors for photochemical ozone formation via a simplification of characterisation modelling which may be achieved applying two approaches: 1) the non-linear and dynamic behaviour of the photochemical oxidation is simplify in a model in which one or more typical situations in terms of meteorology, atmospheric chemistry and concomitant emissions of other air pollutants have been shown. For each individual VOC, characterisation factors may then be presented for each situation or in the form of a weighted average across the situations. 2) The variation between individual VOCs is disregarded and only a few substance-specific characterisation factors are calculated. The first procedure is used in the models based on the POCP (Photochemical Ozone Creation Potential) or MIR (Maximum Incremental Reactivity) concept. The second procedure is used in regionally differentiated models which try to capture the non-linear nature of the ozone formation along with its spatially and temporally determined differences (EC - JRC, 2010b). According to Press (2015), incremental reactivity is "*the number of molecules of ozone formed per NMVOC carbon atom added to a certain atmospheric reaction mixture of NMVOC and NO_x. The peak IR value of an NMVOC is known as its maximum incremental reactivity (MIR)*" whereas the POCP indicates the potential capacity of an organic compound to create ozone in the troposphere. Specifically, the POCP of a VOC may be defined "*the ratio between the change in ozone concentration due to a change in the emission of that VOC and the change in the ozone concentration due to an equally relative change in the emission of ethylene*" (Labouze et al., 2004).

4.1.6 Acidification

This phenomenon mainly concerns soil and aquatic ecosystems in which a reduction of system's acid neutralizing capacity (ANC) occurs, namely a decrease in the amount of substances in a

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certain system able to neutralize hydrogen ions which reach the system. ANC may be reduced by: i) the availability of hydrogen ions which shift other cations which may move away from system by leaching; ii) uptake of cations by plant or biomass which is collected and removed from the system.

Specifically, hydrogen ions is important for acidification impacts within LCA procedure. Acidification is a natural process, although it may be exacerbated by man-made input of hydrogen ions to soil and vegetation. Acidification is basically caused by airborne emissions of gases which are responsible for hydrogen release following their degradation in the atmosphere or after deposition to soil, vegetation, or water. Deposition may increase throughout precipitation events during which the gases are dissolved in water and carried by rain which may be rather acidic, that is with pH values equal to 3-4 in case of strong air pollution (i.e., acid rain) (Rosenbaum et al., 2018).

Acidification in terrestrial and freshwater ecosystems, and a little less in (coastal) marine ecosystems is caused by the atmospheric deposition of inorganic substances on Earth's surface, mainly oxides of sulfur (e.g., SO_2) and nitrogen (e.g.,NO_x). The products obtained by their dissociation modify alkalinity, pH, and inorganic carbon storage in oceans. Although these compounds may be naturally produced in volcanic eruptions and emissions from oceans (e.g., volatile sulfur gases), most of them are originated by anthropogenic activities, such as the combustion of fossil fuels at power stations and industrial plants, vehicle exhausts, and agriculture (van Zelm et al., 2015). As regards agricultural activity, emissions of SO_2, NO_x and NH3 may occur throughout arable crop production. Specifically, the use of organic and synthetic fertilisers may cause relevant emissions of NH3 owed to volatilization which occur during and after distribution of fertilisers containing urea and ammonium (Brentrup et al., 2004).

The most vulnerable areas to terrestrial acidification are characterized by an unreactive geology, such as granite and a base-poor soil. The quantity and site of deposition are influenced by atmospheric climate conditions (e.g., wind, temperature, precipitation), chemical interactions with the atmosphere and topography (van Zelm et al., 2015). Soils with a high clay content are resistant to acidification because of their capacity to adsorb the protons on clay mineral surfaces and release of metal ions, whereas sandy soil are more susceptible to acidification (Rosenbaum et al., 2018).

Acidifying substances may be neutralized by a different buffer reactions in an ecosystem which are depending on the chemical condition of an aquatic or terrestrial system and may change globally (van Zelm et al., 2015). For instance, calcareous soils characterized by a high calcium carbonate content may be well buffered, namely they may oppose the pH variation by neutralising the hydrogen ions with the basic carbonate ions. Furthermore, the atmospheric residence time of acidifying substances is a few days because of their high water solubility. Thus, acidification may be considered a regional effect located near emission point. When the acidifying compounds reach the soil, protons are released and lower soil water pH causing the release of metal ions contained in the soil. Some of these metals may be toxic for plants and limit plant growth. Root and leaves are damaged and after prolonged exposure the plants may die as direct consequence of this condition or via diseases or parasites which take advantage of weakened condition of the plant (Rosenbaum et al., 2018).

Freshwater acidification is mostly due to protons arising from the mineralization of nitrogen as well as sulfur deposition. On the contrary, the pH reduction of the oceans is caused by the accelerated dissolution of CO_2 from the atmosphere which arises from an increase of CO_2 concentration in atmosphere owed to anthropogenic activities. A high quantity of the CO_2

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emitted into atmosphere from human activities dissolves into the oceans raising the hydrogen ion concentration in the ocean, and consequently, lowering ocean pH (van Zelm et al., 2015)

The acidification potential (AP) of a pollutant reflects the maximum acidification a substance may produce, namely its capacity to form hydrogen ions. This parameter has been used for the characterization modelling of this impact category in LCA (Baumann and Tillman, 2004).

The characterization factor (CF) for acidification may be expressed as a function of an atmospheric fate factor (FF), a receiving environment exposure factor (XF) and an effect factor (EF) according to the following equation:

$$CF_{i,x} = \sum_{j} (FF_{i,j,x} \cdot XF_{j,x} \cdot EF_{j})$$

where the atmospheric fate factor (FF) shows the source-receptor relationship, that is the atmospheric impact pathway from the emission location i of pollutant x to the corresponding deposition location in the receiving environment j. The receiving environment exposure factor (XF) evaluates the ability of the receiving environment to withstand acidic deposition due to buffer reactions. Finally, the effect factor assesses the effects or damage caused by the acid deposits to species, (e.g., changes in biodiversity). AP values are used to midpoint indicators for this impact category and SO_2 is commonly used as a reference substance and thus, has an AP equal to 1 (van Zelm et al., 2015). Specifically, AP is defined as the number of H^+ ions produced per kg substance relative to SO_2. Nevertheless, acidification may vary depending on site in which the acidifying contaminants are deposited or the buffering capacity of soil and water, climate conditions, and rate of harvesting. Different approaches are suggested in order to consider local differences, but so far few methods are easily applicable. Therefore, a simple solution is to neglect the impact due to acidification in non-sensitive regions. Some regionalized impact may be assessment by using the average acidification equivalents, for instance they are available for Europe and for three acidifying substances (i.e., SO_2, NO_x, and NH_3) (Baumann and Tillman, 2004).

Endpoint indicators relate the consequences of the acidifying emissions with ecosystem, for instance, in terms of biodiversity or plant productivity loss, or relative species richness (i.e., the change in species composition in a community) so as to provide useful information on the ecosystem quality (van Zelm et al., 2015).

4.1.7 Eutrophication

This process generally refers to an unwanted increase in biomass production in aquatic and terrestrial ecosystems due to a relevant availability of nutrients which cause a change in species composition (Brentrup et al., 2004). Eutrophication concerns not only the impacts of nutrients (e.g., nitrogen and phosphorous), but also effects due to degradable organic pollutant and sometimes waste heat since they may have impact on biological productivity. These organic pollutants show one aspect in common which is used for characterization modelling, that is they conduct to oxygen depletion. Release of degradable organic matter into water bodies are broken down by microbial community which use oxygen resulting in a consequent reduction of oxygen level in the water and harmful impacts on aquatic ecosystems. In other words, flow of nutrients

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as well as waste heat into water bodies may determine an increase in biological productivity and biomass formation, which in turn may cause a rise in oxygen depletion during the biomass decomposition (Baumann and Tillman, 2004).

In LCA context, this impact category appears with different names, such as nutrient enrichment, nutrification, and oxygen depletion being associated with the availability of nutrient in the aquatic and terrestrial ecosystems (EC - JRC, 2010b). In fact, eutrophication is a complex category because it is highly site-dependent and substances responsible for environmental burden may be released both to air and water, and impact may occur in various types of terrestrial and aquatic systems. The fate processes depend on characteristics of emitting source, environmental mean, and receiving environment. Furthermore, different nutrients may limit the growth in various ecosystems, but also some impacts that cause an increased growth, namely may be considered a positive impact rather than a negative one (Finnveden and Potting, 1999). Nutrients are a natural component of environment and essential for the existence life. The different species composition and productivity that characterise ecosystems express the availability of nutrients. Therefore, natural differences in the availability of nitrogen and phosphorus are one of the causes for the diversity of existing species and ecosystem variety. Ecosystems are dynamic, and if they undergo a change in terms of nutrient availability, they merely try to achieve a new balance with surrounding environment (Rosenbaum et al., 2018). For instance, once released to air, nitrogen may be deposited on water, soil or vegetation. As reported by Finnveden and Potting (1999), its fate may be different. Nitrogen may be fixed by biomass or leaches to surface water which depends on site specific aspects. If not immediately leached, it may be used by vegetation contributing to terrestrial eutrophication. The vegetation and thus, the nitrogen may leave ecosystem, for instance, as timber or it may degrade. In the latter case, nitrogen may be taken up again by growing plants and consequently contribute to eutrophication a second time. However, nitrogen may leach to surface water, immobilised in the soil, or denitrified and released as gas. When nitrogen is directly deposited on the water surface or via leaching from a terrestrial system, the water body may be phosphorous limited. In this context, nitrogen does not cause eutrophication, but may undergo denitrification and thus leave definitively the system, or it may be transported to another water surface which might be nitrogen limited. In this context, nitrogen may use by phytoplankton, thus causing eutrophication, or leave the system, for example together with fish. The phytoplankton degradation may make nitrogen available again which may be used for growing phytoplankton, be transported to other waters, be buried in sediments, or be nitrified and leave the system as nitrogen gas.

As regards terrestrial ecosystems, a high nutrient availability may cause a change of the species composition by fostering those species which may take advantage of high level of nutrients useful to a faster growth than more nutrient efficient plants. Consequently, this condition may change the plant community from a vegetation characterized by low availability of nutrients (for example heaths, dunes and raised bogs) to a plant community rich in nutrients due to their widespread dispersion. In other words, the primary impact on the plant community determines a secondary impact on other species in the terrestrial ecosystems. Terrestrial eutrophication is mainly due to deposition of airborne emissions of nitrogen compounds (e.g., NO_x, namely NO and NO_2) from combustion processes and ammonia from agricultural activity. The nitrogen compounds have a greater role in the terrestrial eutrophication than phosphorus-based ones (EC - JRC, 2010b).

Nitrogen reaches aquatic ecosystem from different sources, such as agricultural fertilisers and effluents from sewage works as well as part of the atmospheric emissions NO_x (Baumann and

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Tillman, 2004). Nitrate (NO_3^-) leaching is the main pathway for diffusion N emissions from soil to aquatic ecosystems, although a linear relationship between N-input to the soil (e.g., fertiliser) and nitrate content of ground and surface water has not been detected. Nitrate loss to ground water through leaching are highly dependent on agricultural management (e.g., fertilization rates, N removal with harvested crops), site-specific soil and climate conditions (e.g., field capacity, rate of drainage water). Consequently, NO_3 leaching loss from soil to groundwater is extremely variable and should be assessed without disregarding all significant parameter which influence the NO3- content in the soil and in groundwater (e.g., site-specific soil and climate characteristics (Brentrup et al., 2004). The phosphorous-based fertiliser is mostly insoluble and only a small fraction leave soil to reach a water body (Baumann and Tillman, 2004).

In aquatic systems, the increase in nutrients cause a change of species composition and an increase in biological productivity with several implications for the ecosystem: i) "*Species composition of the plant community changes to more nutrient-demanding species*"; ii) "*Algal blooms create shadowing, filtering the light penetrating into the water mass, changing life conditions from the macrophytes, which need the light for photosynthesis, and for predatory fish which need the light to see and catch their prey*"; and iii) "*Oxygen depletion near the bottom of the water body where dead algae deposit and degrade*". In aquatic system, one of the macronutrients is frequently a limiting factor for the growth of algae(EC - JRC, 2010b). Although limiting nutrients show seasonal variations, generally P is the limiting nutrient in freshwater systems whereas nitrogen is limiting in marine systems (Baumann and Tillman, 2004).

According to EC - JRC (2010b), the eutrophication characterization factor should be calculated including a fate factor on the basis of equation:

$$\mathrm{CF_{i,m,r}} = \mathrm{FF} \cdot \mathrm{EF} = \mathrm{f_{i,mr}} \cdot \beta_{\mathrm{dose-response}}$$

where: $F_{i,m,r}$ is the fate factor representing the transport of substance (i) in the media air or water (m) and the transfer to receiving environment (r). (dimensionless (kg/kg)). $\beta_{dose\text{-}response}$ is the effect factor expressing the response of the ecosystem to the change in nutrient status (e.g., Impact/kg N or P, or (-)). Furthermore, since emissions of organic matter may cause oxygen depletion by bacterial degradation, emissions of BOD (biological oxygen demand) or COD (chemical oxygen demand) are included in some LCIA methods as characterization factor, although their inclusion is not consistent with the impact pathway at midpoint level, but they may provide a contribution to some of the same damages.

In LCA, eutrophication is generally estimated using site-generic characterization factors, even though this impact category is strictly site-dependent (Henryson et al., 2018). In fact, the existence of various ecosystems in turn limited by different nutrients makes eutrophication highly variable geographically, and consequently more complicated the characterisation modelling. The easiest solution should be neglect the geographical variation. Therefore, eutrophication potential expresses the maximum eutrophying effect of a substance which supposes that airborne nutrients reach aquatic systems and includes emissions of N and P compounds to air and water and the emissions of organic matter. (Baumann and Tillman, 2004).

The computation of eutrophication potential is based on the proportions of nitrogen, phosphorous, carbon, and oxygen in the average chemical composition of aquatic organisms (i.e.,$C_{106}H_{263}O_{110}N_{16}P$) that is considered representative of average biomass. According to this

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approach, one mole of biomass requires 16 moles of N and 1 mole of P. When organic matter is emitted, its degradation may be quantified as chemical oxygen demand (COD). It is assumed that degradation of one mole of biomass requires 138 moles of O_2 (Guinée et al., 2002).

Midpoint LCIA methods usually propose units in P- and N-equivalents such as kg P-eq or kg PO_4^{3-}-eq and kg N-eq or kg NO_3^--eq. Specifically, it considers the number of moles of nitrogen or phosphorus which may be released into the environment from one mole of the substance emitted (Rosenbaum et al., 2018). In other words, the reference substance to determine eutrophication potential is PO_4^{3-} equivalent which is easily converted in other types of equivalent (i.e., NO_3^- and O_2-eq) on the basis the molar ratio of the chemical formulae. According to this approach, the characterisation factor for eutrophication does not consider what substance is the limiting factor in a particular location. Therefore, the obtained values represent global characterisation factors, because it is independent of local differences and it is unknown which environmental compartment (e.g., freshwater, salt water, groundwater or soil) an emitted substance will end up (Guinée et al., 2002).

For endpoint characterisation, many models use Potentially Disappeared Fraction of species (PDF) in (m^2 years) (Rosenbaum et al., 2018).

As regards waste heat, a specific method to assess its impact similar to procedure adopted for N and P compounds and organic pollution, has not yet been developed. Generally, waste heat is neglected in several LCA studies although it should be useful to assess its impact, especially when the release of waste heat is remarkable (e.g., discharge of waste heat from a nuclear plant to water). A simple option might be to include the quantity of waste heat (in MJ) released into aquatic ecosystems separately from the other sources of eutrophication (Baumann and Tillman, 2004).

4.1.8 Land use

This impact category refers to a series of consequences due to human land use, such as occupying, reshaping, and managing land (Brentrup et al., 2002). It is a relatively recent topic and still being discussed within the LCA context because of multi-functionality of land and as natural resource. Land use to support human activities as agriculture, forestry, mining, house-building, or industry may induce considerable impacts, especially on biodiversity and on soil quality as a key factor of essential life functions (e.g., cycling of nutrients, water and carbon, and the provision of habitat for both human and non-human life) (Milà i Canals et al., 2007). Soil as finite resource may contribute to environmental effects arising from its use. Soil loss regards not only an extremely low soil formation rate, but also qualitative aspects due to unsustainable management practices, particularly for high quality soils which are able to meet numerous purposes. Croplands, pastures, urban areas and other land-use-intensive activities that occurred over time worldwide at the expense of natural areas to meet the growing society's needs for food, fibre, living space and transport infrastructure frequently cause irreversible effects on ecosystems and human quality of life (e.g., deforestation releases carbon emissions to atmosphere, loss and change of habitats contribute to biodiversity decline). Furthermore, inappropriate management practices may alter soil physical properties, for instance the use of mechanical inputs in farming may cause soil compaction which may affect aquifer recharge and the soil capacity to remove contaminants and intensification of agricultural activity may exacerbate erosion process whereas in urban and industrial areas, soil may lose all its functions because it is replaced by concrete (Rosenbaum et al., 2018).

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In a LCA analysis, direct impacts due to land use, such as nitrate leaching or diffuse emissions from soil to air are included in certain impact categories. The impact category "land use" encompasses only the environmental effects caused by the land use itself, for instance owed to removal of landscape element (e.g., forest, hedges, pond, bushes), planting of monocultures (e.g., cereals, conifers) or artificial vegetation (e.g., gardens), or the sealing of surfaces (e.g., for buildings or roads) (Brentrup et al., 2002). Two actions relative to land use are generally considered in life cycle inventory and impact assessment, namely land transformation and land occupation. The former (also called land use change, LUC), refers to the conversion from one condition to another whereas the latter regards the use of a certain area for a specific purpose. In other words, land transformation results in the change of the properties of a certain area to make it suitable for a planned use (e.g., deforesting or draining land to establish arable fields). Land occupation refers to the maintenance of land properties on the basis of its intended productive purpose (e.g., arable land and the regrowth of forest is avoided on the arable land) (Koellner et al., 2013).

According to EC - JRC (2010b), the occupation impact (I_{occ}) might be calculated on the basis of the following equation:

$$I_{occ} = A * t_i * (Q_{pot} - Q_{act})/S_i$$

where A is the area occupied, t_i represents the time of occupation, Q_{pot} is the quality indicator for the reference situation, Q_{act} the quality indicator for present occupation, and S_i stands for the slope factor that reflects the duration of restoration.

The transformation impact (I_{trans}) may be calculated using the following formula:

$$I_{trans} = A * t_r * (Q_{pot} - Q_{act})/S_i$$

where t_r is the time of restoration.

The same human activity may be responsible for different land use impacts whose magnitude depends on climate, soil quality, topography and ecological quality which characterise a region. Therefore, land use category may be unlike other global impact categories (e.g., climate change is independent of the spatial source of emission) considered a local impact category. Specifically, the inventory should include information on the geographic location of the human activity and whose degree of detail is based on the goal and scope of the study. In the LCIA, characterisation factors should take into account the sensitivity of habitat to the impact under consideration. For instance, characterisation factor for erosion might include information on the soil depth relative to the specific location in which the activity occurs, because the impact of soil loss depends on the soil stock size (Rosenbaum et al., 2018). In order to assess the impacts due to the occupation and transformation aspects of land use both at midpoint level and damage level, appropriate quality indicators should be selected for the different impact pathways, such as biotic production potential, biodiversity and ecological soil quality (Milà i Canals et al., 2007). Specifically, all areas of protection undergo the effects of land use activities since land

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occupation and transformation may occur as physical (e.g, compaction, and erosion), chemical (e.g, pH, salts composition, and toxicity), or biological (e.g., vegetation cover and species composition) alterations of land, which are connected to some direct impacts. Such impacts may involve land-based processes (e.g., albedo, and water cycle) which may cause effects detected in the midpoint of the cause-effect change (e.g., biotic production), in turn related to all areas of protection (Milà i Canals and de Baan, 2015).

As regards impacts on biodiversity, the most commonly applied indicator uses as reference species richness, whereas damage on biodiversity is generally expressed in quantity of species biodiversity loss, in relative terms (potentially disappeared fraction of species times surface, PDF m2) or in absolute species loss. Although a series of indicators are available to assess impact associated with ecosystem services they are difficult to include in LCIA and further proposals are still under development (Rosenbaum et al., 2018). As reported by Milà i Canals and de Baan (2015), indicators to estimate land use impacts on biodiversity is mainly based on specie richness or on ecosystem metrics, whereas the land use impacts on ecosystem services are expressed through two types of indicators: pressure (describing land degradation processes, such as erosion and salinization) and state (describing overall quality and ecosystem productivity such as soil organic carbon and soil organic matter, and net primary productivity for biomass) indicators. Species richness is often used as indicator for biodiversity because of its ease of communication and inn terms of availability of data. Species richness is based on the considered taxonomic groups, for instance, insects show a high degree of diversity (ca. 60 %), followed by fungi (ca. 10 %), plants (ca. 2 %) and vertebrates (ca. 0.4 %; including mammals, birds, etc.). In order to lessen the bias due to the choice of taxonomic group used as indicator for biodiversity, the reduction of species richness initially was estimated in relative instead of absolute terms, based on the unit potentially disappeared fraction of species (PDF). Most of studies are focused on the reduction of vascular plant species richness, mainly using European, Asian, or global data because of relatively good data availability and the significant role of plants as primary producers in ecosystems. Other indicators suggested to estimate the biodiversity of land refer to vascular plant species richness of an ecosystem, ecosystem vulnerability (i.e., the relative number of species affected by a change in the ecosystem area), and ecosystem scarcity (i.e., the natural (global) scarcity of an ecosystem type).

According to Rosenbaum et al., (2018), there are LCA methods for some ecosystem services, such as: i) Biotic production potential, namely "*capacity of ecosystems to produce and sustain biomass on the long term. Available indicators are based on the soil organic matter (or carbon) content…, the biotic production…, and the human appropriation of the biotic production…*"; ii) Carbon sequestration potential, that is "*capacity of ecosystems to regulate climate by carbon uptake from the air*"; iii) Freshwater regulation potential refers to "*capacity of ecosystems to regulate peak flow and base flow of surface water*"; iv) Water purification potential, namely "*mechanical, physical and chemical capacity of ecosystems to absorb, bind or remove pollutants from water*"; v) Erosion regulation potential refers to "*capacity of ecosystems to stabilise soils and to prevent sediment accumulation downstream*"; vi) Desertification regulation potential, that is "*capacity of dry lands to resist irreversible degradation on the human time-frame*".

4.1.9 Water use

Water is an fundamental resource to maintain vital functions and it may be replaced by no other substance (Berger and Finkbeiner, 2010). Specifically, freshwater, if available in sufficient

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quantity and of good quality is essential to ecological and societal activities, such as food production, industrial activities, and human sanitary conditions (Jefferies et al., 2012).

Although it is not disappear as a renewable resource, its functions are strictly connected to its geographical and seasonal availability, especially since its displacement and storing may be difficult to implement and expensive (Rosenbaum et al., 2018).

The LCA approach generally deals with freshwater use neglecting the environmental burden due to the use of sea or brackish water. Furthermore, a distinction between degradative and consumptive use of water is made in order to better identify impacts. Degradative water use regards contamination of water bodies that occurs during its use, whereas consumptive use refers to water which is not released back to the original watershed, and thus it is not exploitable by downstream users. As a result, consumptive water use is considered as water consumption and encompasses water that is evaporated, transpired, included into products, and released to a different watershed or directly to saline water (Pfister, 2015). Actually, the availability of water globally would be adequate to meet ecosystem and human needs. Approximately 119,000 km^3 of water yearly reaches land in various forms of precipitation, of which 62% goes back directly to atmosphere through evaporation and plant transpiration. Human activities use about 3% of the remaining quantity (i.e., 38%). This percentage is divided between agricultural activity (2.1%), industrial sector (0.6%), and domestic use (0.3%) (Rosenbaum et al., 2018). Within the agricultural sector, water losses are due to water conveying and irrigation distribution (7%), field application (20%), farm distribution (8%), whereas 65% (the percentage depends on irrigation technique) is water used by crop (Chartzoulakis and Bertaki, 2015).

The issue related to water use is not only due to have enough water to meet human needs, but also the competition for water resource between human and ecosystem requirements. On the one hand, the increased human demand for water is caused by a continuous growing population and changing diets, and on the other hand, water availability is affected by climate change exacerbating droughts and flooding. These conditions contribute to increase the gap between the demand and availability in many regions densely populated in the world. Because the issue related to water is dependent on where and when water is available, as well as its quality, all these aspects should be taken into account within the LCA procedure, namely in the assessment of potential impacts of freshwater use on the environment and human health (Rosenbaum et al., 2018).

The impact assessment associated with water use should be based on the different contribution included in the inventory as flows, that is the water exchanges between environment and the processes under consideration. In other words, the water balance of each process should be considered as well as all inputs and outputs in order to determine water consumption (i.e., water flows from and to the environment, and flows of water within the technosphere (e.g., water content of products and tap and waste water flows). As regards agricultural activity, the process environment (i.e., agricultural land) should be included in water management in order to consider the water flows of natural water supply from soil (i.e., moisture) and precipitation. The precipitation flows is important in rainwater collection systems in industry or agriculture, whereas the evapotranspiration flow is man-induced evapotranspiration. It refers to flow from water inputs to the technosphere and is the main factor responsible for water consumption, namely the total amount of water consumed from water withdrawals. Salt water is generally disregarded, although may be useful if processes based on seawater cooling systems are involved. Water consumption of seawater is usually neglected in impact assessment owed to water use (Pfister, 2015).

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Impact assessment method for water use at the midpoint level are generally based on water scarcity index, that is the comparison between water used and renewable water availability and it expresses the level of competition between the various users (in theory human users and ecosystems) (Rosenbaum et al., 2018). Therefore, this index leads to a dimensionless characterisation factor (Pfister, 2015).

According to Pfister (2015), impacts due to water use may cause an ecosystem disruption and biodiversity loss. This impact may be estimated through the conversion of water consumption into land use equivalent of potentially disappeared fraction (PDF) of species on an area during a time. Damages to human health may be produced because of difficulty to access to safe water owed to lack of infrastructure and to water pollution. The estimation of damage may refers to the standard unit of disability adjusted life year (DALY) lost based on the relationship between DALY from malnutrition and malnourished people. Finally, water use and consumption may be responsible for overusing of groundwater stocks and lakes, potentially reducing the water availability for future generations. The quantification of damages at endpoint level generally refers to resource depletion, especially freshwater depletion whose estimation may be based on a corrected extraction rate in order to take into account regeneration of water reserve resources.

4.1.10 Resource depletion

This category regards one of the most debated topic within the LCA approach, namely how impact assessment related to resource depletion should be performed. Therefore, a wide range of methods are developed, although no consensus of any of the impact assessment methods is still achieved and research does not stop. Basically, the methods differ in the way resources are considered an environmental problem, namely some consider resource reduction as an environmental problem in itself; others deems as a societal issue, and hence outside the LCA scope; and others view the environmental burdens related to resource extraction (Baumann and Tillman, 2004). Specifically, the debate is focused on the lack of a scientifically appropriate method to determine characterisation factors due to some reasons: i) abiotic depletion is a critical point which passes through economic-environment system boundary, since the availability of resources depends on future technologies for extracting them; ii) various ways may be used to define the depletion issue, and each one may be explained from different points of view; iii) although various methods may quantify the depletion definition, no one may be empirically verified since all of them depend on the presumed availability of, and request, for resources in the future and on future technologies (van Oers and Guinée, 2016). According to van Oers (2002) depletion of resources may be defined as: "*abiotic resource depletion is the decrease of availability of the total reserve of potential functions of resources*" thereby emphasizing that from the functional perspective the human interest for abiotic resources , is not the resource itself, but to the potential it has to meet for humanity.

The characterising modelling shows differences on the basis of the resource type (Baumann and Tillman, 2004). As reported by Swart et al. (2015) resources may be divided into: i) Renewable and non-renewable: the former refers to "*…natural resources that can be replenished at approximately the same rate at which they are used (e.g. wind and solar energy)*".The latter "*…cannot be produced (or re-grown) at the same rate at which they are consumed (e.g., coal and natural gas)*"; ii) Biotic and abiotic: the first includes "*materials derived from presently living organisms…and they typically have an important role in maintaining ecosystem services and also intrinsic value (examples are tropical hardwood and ivory)*". "*…Abiotic resources are the product*

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of past biological processes (coal, oil and gas) or physical/chemical processes (deposits of metal ores)"; iii) Funds, flows and stocks: "*…stocks extraction inevitably leads to the depletion of the resource, i.e. reduction of the available amounts in nature, whereas funds may be depleted but also have a renewal rate which is high enough to allow the resource to recover*". *"…Flow resources though cannot be depleted. Their availability per unit time however is limited, and thus their extraction is marked by competition (e.g. wind energy)"*. In other words, the distinction in funds, flows and stocks is based on the capacity of the resource to be regenerated and the speed by which it may occur. Actually, stock resources may be considered non-renewable resources whereas fund and flow resources may both of them be related to renewable resources. Furthermore, resources may be categorized in exhaustible and inexhaustible, that is they may totally depleted or not, respectively (Rosenbaum et al., 2018).

It is clear that natural resources may be classified in several ways and the terms may be interchangeable or a type of natural resources may be included within another group. As above reported, for instance, deposits may be considered non-renewable resources (e.g., fossil fuels, minerals and clays) whereas flows being continuously regenerated (e.g., wind, rivers, and solar energy) may be deemed renewable resources (Baumann and Tillman, 2004).

Although resource depletion is generally described as a single impact category in the LCA approach, there are some methods which mix numerous problems and use numerous processes within a single impact category by determining an unclear situation. For instance, harvesting crops or wood may be considered a land-use problem, even though the extraction of funds, namely the reduction of the available quantity of standing trees would fall in resource depletion. In other words, it is not always simple identify which impact category a certain resource use should be associated with. The impact of renewable resource use such as wood and fish, may be expressed on the basis of weight, volume, and exergy, or considering the regeneration rate (EC - JRC, 2010b).

LCIA methods utilised for non-renewable resource depletion may be modelled on the basis of: i) energy or mass; ii) use of stock, namely on the ratio of use to deposits; iii) exergy consumption or entropy production; and iv) future consequences of present resource extraction (Swart et al., 2015). Specifically, methods to evaluate impacts due to resource depletion may be divide into three classes: i) resources accounting methods (RAM) which simplify impact assessment since they are based on grouping the resources into single score indicators, such as energy or mass; ii) midpoint resource depletion methods, which unlike the previous class, estimate resource depletion impacts depending on its use, as the use-to-availability ratio; iii) Endpoint resource depletion methods, which consider the effects of resource depletion, sometimes via backup technology, that is estimating the additional effort, in terms of energy or cost, required to extract less economically feasible resources (Alvarenga et al., 2016).

As reported by Guinée et al. (2002), an effective method to evaluate resource depletion impacts is based on ultimate reserves and extraction rate, since these parameters best may represent the gravity of resource depletion. In this method, the indicator is expressed in kg of a reference resource (i.e., antinomy) according to the following equation:

$$\text{abiotic depeltion} \sum_{i} \mathrm{ADP_i} \times \mathrm{m_i}$$

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where:

$$ADP_i = \frac{DR_i}{(R_i)^2} \times \frac{(R_{ref})^2}{DR_{ref}}$$

and ADP_i is Abiotic Depletion Potential of resource i (generally dimensionless); m_i is quantity of resource i extracted (kg); R_i is ultimate reserve of resource I (kg); DR_i is extraction rate of resource i (kg yr^{-1}); R_{ref} is ultimate reserve of the reference resource, antimony (kg); DR_{ref} is extraction rate of Rref (kg yr^{-1}).

The ADP method based on the quantity of a specific resource in the Earth's surface, that is the ultimate reserve, was suggested only for midpoint assessment whereas no suggestion was provided for endpoint LCIA methods (Alvarenga et al., 2016).

At endpoint level, the characterization factors for resource depletion are evaluated in terms of the future consequences of resource extraction. This approach is based on the fact that the extraction of a large amount of resources at present, will necessarily lead future generations to extract a less amount or a lower value resources. Therefore, further efforts will be need resulting in higher energy or costs which will lead to a greater impact of environment and economy. The endpoint indicator may be determined, for instance, via "willingness to pay" approach which expresses the future payment for extracting a resource, or using "surplus energy" concept of that represents the energy demand need for additional extractions of the resource in the future (EC - JRC, 2010b).

4.1.11 Areas of protection

In order to apply properly the impact assessment framework, the definitions of some terms and their logical connections should be taken into account to avoid double counting, for instance of impact categories which should not be confused with the human activities which cause the impacts. Impact category is an environmental issue to which may be associated the LCI results. Physical processes and variables such as extractions, emissions or other types of interaction between the product system and the environment, which are linked to a certain impact category, are defined the environmental mechanism of that impact category. Within and in relation to environmental mechanism it is possible to identify areas of protection, that is groups of endpoints which have some noticeable value for society, namely human health, natural resources, natural environment and man-made environment which are associated with specific value in society (e.g., the intrinsic value of human life, the intrinsic value of nature, and cultural values.) (Table 9) (Udo de Haes et al., 1999).

Table 9 - Areas of protection and corresponding main societal values (Udo de Haes et al., 1999)

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Areas of protection	Societal values
1. human health	Intrinsic value of human life, economic value
2. natural environment	Intrinsic value ofnature (ecosystems, species), economic value of life support functions
3. natural resources	Economic and intrinsic values
4. man-made environment	Cultural, economic and intrinsic values

An overview of the areas of protection (AoP) and related damage indicators most commonly used (i.e., human health, natural environment, and natural resources), are reported below on the basis of the most relevant information provided by EC - JRC (2010b).

The assessment of human health refers to quantification of changes in both mortality and morbidity which are related to good or services in an integrated way. The reference endpoint indicator is usually DALY (Disability Adjusted Life Years) which combines information on quality of life and life expectancy in one indicator, obtaining the (potential) number of healthy life years lost due to premature mortality or morbidity. Specifically, the DALY is calculated using equal weightings to the importance of 1 year of life lost for all ages and not discounting for future damages, according to the following equation:

$$\mathrm{DALY} = \mathrm{YLL} + \mathrm{YLL}$$

where YLL is years of life lost and YLD is years of life disabled.

In turn, the YLD is expressed as:

$$\mathrm{YLD} = \mathrm{w} \cdot \mathrm{D}$$

where w is the disability weight between 0 (complete health) and 1 (dead), and D is the duration of the disease.

Natural environment concerns the worldwide natural ecosystems in terms of their function and structure. The resource as ecosystem (i.e., biological renewable resources and managed, man-made ecosystems like plantations or agricultural fields) is included in the area of protection related to natural resources. The assessment of the “natural environment” area is focused on quantify the negative effects on the function and structure of natural ecosystems resulting from exposure to chemicals or physical interventions. The Potentially Disappeared Fraction of species (PDF) concept is generally used as endpoint indicator for this area of protection. This indicator may be considered as the fraction of species that has a high probability of no occurrence in a region due to unfavourable conditions. Eco-toxicity arisen from chemicals is assessed on the basis of results from laboratory test of the chemicals on organisms of different species. The sensitivity of a selection species considered as representative of the ecosystem, is estimated via a statistical sensitivity distribution curve. This curves are used to support the estimation of the fraction of the species in the ecosystem that is exposed above the level which affects them (the

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Potentially Affected Fraction of species, PAF) or above the threshold level where the species will disappear (PDF). The relationship between PAF and PDF "*is based on the assumption that the quality of the media (e.g. water) has a direct link with the biodiversity, i.e. that a species disappears when the chemical concentration in the ecosystem reaches a certain level, and reappears when the concentration, due to for example degradation, comes below that level again*".

Regarding the Area of Protection of "Natural Resources" no suggestion is provided at endpoint level, since it is difficult to find a clear distinction between human health' and natural environment. They are strictly linked; for instance, the extraction of resources (e.g., mineral deposits, fossil energy carriers, fish, trees, and water) may affect the environment. Furthermore, the extracting activity in itself (e.g., mining, forestry, fishery) may release toxic emissions, create noise, damage the landscape, all effects which might concerns other areas of protection. Natural Resources also regards the removal of resources from the environment (and their use) which induces a decrease in the availability of the total resource stock, whereas the availability of renewable resources (generally biotic) depends specifically on the time they take to regenerate relative to the time we take in consuming them. Therefore, the resource scarcity may be considered the rationale for this area since the economic system from which human welfare depends may be damaged as resource availability decrease. The characterisation models used in assessment of category indicators for natural resources have an anthropocentric approach since they usually consider the use value for humans and largely excluding its non-use and intrinsic value.

The choice of midpoint and endpoint indicators may result from a clear definition of the area of protection "Natural Resource" and for this purpose some points should be taken into account, such as "*Is the AoP for Natural Resources restricted to the role of resources for humans, or does it also include the role for ecosystems or parts of ecosystems?*"; "*Is the role of natural resources for humans restricted to its present uses, or should we also address future needs?*"; "*Are the resources we distinguish for human needs focused on essential functions (such as nourishment), or does it also include luxury items (such as using ivory for pianos)?*"; and "*To what extent do we need to address developments in population growth and affluence in the future? For instance, in assessing the future role of iron ore, what population size do we take into account, and do we assume that e.g. Africa's needs are similar to Europe's on a per capita basis?*".

These points may be used as support for define midpoint and endpoint indicator which enables to select or develop a characterisation model. In other words, these elements might make it easier the choice of the categories themselves, both the one of an aggregate indicator of resource depletion and a wide range of resource depletion indicators (e.g., for metals, fossils, water, fish, wood, land). The availability of numerous resources-related impact categories enables to include various important issues in the different resource groups. However, it is hard to determine which resource functions to maintain, since needs in the future are unknown or not recognized. A behavioural change may have multiple effects, for instance in terms of resource scarcity and price increase. A potential reduction of a certain resource demand may encourage the use of replacement resources, the development of new technologies and recycling techniques. Furthermore, this condition might also lead to further exploration and the discovery of new reserves. Finally, it will make non-economic reserves more profitable, maybe with more intense environmental consequences owed to an intensification of drilling, mining, and refining activities.

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As highlighted by Rosenbaum et al. (2018), there is not currently a common consensus on how to model damage in terms of resource depletion. Some suggestions consider the future costs for extraction of the resource due to present depletion. These costs might be expressed as energy or exergy use for future extraction (measured in MJ) or in monetary terms (measured in current currency like USD, Yen or Euro).

4.2 Classification

In this phase, the results of the inventory are attributed to impact categories based on the environmental problems to which it is known they may contribute (Margni and Curran, 2012). Specifically, the elementary flows included in the inventory (e.g., resource consumption and emissions into air or water) are associated with the impact categories selected in the previous phase, according to the capacity of each element, such as pollutants and resources use to provide a contribution to various critical environmental points (Figure 15) (Hauschild and Huijbregts, 2015). This phase is used for understanding which impact(s) each LCI result contribute to (Rosenbaum et al., 2018).

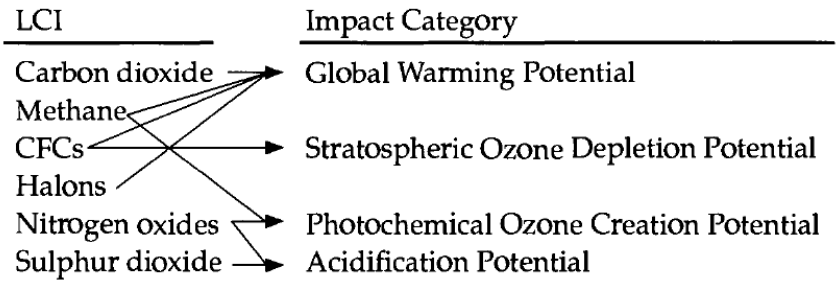

Figure 15 - Example of classification (Margni and Curran, 2012)

For instance, CO_2 emissions to air may be attributed to climate change and water consumption to the water use impact category, although the assignment is not without difficult since certain emissions may show more than one impact, basically according to two modes: i) in parallel, namely a substance may show numerous concurrent impacts (e.g., SO_2 may induce acidification and at the same time may be toxic if inhaled by humans); and ii) in series, namely a substance may have a negative effect which in turn may cause another adverse impact (e.g., SO_2 may induce acidification, which may mobilise heavy metals in soil which in turn may be toxic to ecosystem and humans) (Rosenbaum et al., 2018). In case of contribution to multiple impact categories, the responsible substance will be attributed to all categories on which shows an impact, thus without partitioning or allocating. For instance, NO_x may cause both photochemical ozone creation and acidification, therefore it is attributed to both categories in its totality (i.e., 100% to photochemical ozone creation and 100 % to acidification) (Margni and Curran, 2012). The multiple attribution is only made for effects that are independent on each other, as in the case of NO_x which may be simultaneously responsible for acidification, eutrophication and photo-oxidant formation. On the other hand, if the effects are dependent on each other, such as global warming and global warming which has effects on biodiversity, double attribution leads to double-counting (Baumann and Tillman, 2004).

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In addition to the aforementioned emissions with parallel and serial impacts (i.e., substance that may contribute to more than one impact category and have subsequent impacts, respectively), other two impact categories may be distinguished by Guinée (2015), namely emissions with indirect impacts and with combined impacts. The former regard emissions of substances with a primary impact that in turn may induce one or more secondary impacts, such as aluminium toxicity caused by acidification, or methane contributing to photo-oxidant formation, with the produced ozone may cause climate change, which in turn may contribute to stratospheric ozone depletion. The latter refers to emissions of substances characterised by a mutual effect on each other's ones, such as synergistic or antagonistic impacts of toxic substance mixes, or NO_x and VOC, both of them needed for photo-oxidant formation. The possible double counting or partitioning, or no allocation of impacts should depend on type of emission. As reported by Guinée (2015), in order to avoid double counting for emissions with parallel impacts, it is usually suggested by the literature to specify the emission contributions to relevant impact categories. Nevertheless, no guidelines provide information on how this activity might be performed. Generally, this task should be carried out when the contribution of a certain substance to one impact category considerably reduces its potential contribution to another. In case of lacking of clarity on how emissions with parallel impacts should be partitioned, they are often attributed in their totality to relevant impact categories. In general, the literature proposes for serial and indirect impacts to allocate emissions in their totality to relevant impact categories unless the corresponding characterisation factors are lacking (e.g., indirect GWP factors). Concerning emissions with combined impacts, the recommendations are focused on introducing assumptions relating to background concentrations of the other relevant substances. At present, this is only feasible for NOx as a precursor in photo-oxidant formation, but not for synergistic or antagonistic impacts of toxic substance mixes, because of lack of knowledge on this aspect.

Since the classification phase needs notable understanding and expert knowledge of environmental burdens, it is usually handled automatically by LCA software via expert-based and pre-programmed classification tables (Rosenbaum et al., 2018).

4.3 Characterisation

This phase consists of computing of category indicator results determining contributions from the environmental flows included in the inventory to the various impact categories. This task is generally performed automatically by LCA software. In other words, characterisation should meet the question: "*How much does each LCI result contribute?*" (Rosenbaum et al., 2018). The extent of environmental impacts are quantified applying equivalency factors (also known as category indicators for ISO standard, characterisation factors, potential, or simply equivalents) which are specified within the scope of modelling the cause-effect chain (Baumann and Tillman, 2004).

As reported by Rosenbaum et al. (2018), a characterisation factor (CF) is the quantitative contribution of one elementary flow to a certain environmental impact category. Its computing is performed applying (scientifically valid and quantitative) models of the environmental mechanism that represents the cause -effect chain of processes leading to effects on environment relating to all elementary flows which are responsible for this impact. Any characterisation model is based on the development of a model for the environmental mechanism which corresponds to a cause-effect chain. This consists of elementary flows, responsible for environmental processes, which may be basically differentiate on the basis of

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the direction of the relevant elementary flows between technosphere and ecosphere, that is: i) an emission into the environment (i.e., elementary flow from the technosphere to the ecosphere); ii) a resource extraction from the environment (i.e., elementary flow from the ecosphere to the technosphere). Rosenbaum et al. (2018) also underlines that, in both situations, the cause-effect chain may be divided in four steps which, all or only some, may characterise an emission into the environment (i.e., mainly emission into air, water or soil) and a resource extraction from the environment (e.g., minerals, crude oil, water or soil). The four phases refers to: i) fate, ii) exposure; iii) effects; and iv) damage. As regards an emission into the environment, fate refers to environmental processes responsible for transport, distribution and transformation of the emitted substance in the environment. Exposure regards contact of the substance from the environment to a sensitive target (e.g., animals, plants, whole ecosystems, and humans). Effects are the detected harmful consequences in the sensitive target subsequently exposure to a substance in a human population or a number of species. Damage refers to the intensity of detected effects through quantifying the reduction of biodiversity or human health of a affected population. The meaning of latter phase does not basically change regarding a resource extraction from the environment, whereas fate and exposure refers to a change. The former represents physical changes to local conditions in the environment (e.g., soil organic carbon content), and the latter regards a change in available quantity, quality or functionality of a resource and possible competition among users (humans or ecosystems).

In the light of the above, characterization factor may be determined according to the following equation:

$$CF = FF \times XF \times EF$$

where FF is the fate factor; XF is an exposure factor for the exposure of sensitive targets in the receiving environment; and EF is an effect factor expressing the effects of the exposure on the targets for the impact category. This generic relation has been used in most of the emission-related impact category, although contents, metrics, and meanings of the three factors change according to the impact category (Hauschild and Huijbregts, 2015).

The CF value is expressed by a metric common to all environmental contributions within a specific impact category and it may express the impact directly in absolute terms (e.g., number of disease cases/unit toxic emission) or indirectly through relating them to the impact of a reference elementary flow (e.g., CO_2-equivalents/unit emission of greenhouse gases) (Rosenbaum et al. 2018).

Impact (IS) scores are calculated according to the following equation:

$$IS_{j,i,k,l} = Q_{i,k,l} \cdot CF_{j,i,k,l}$$

where $IS_{j,i,k,l}$ is the contribution from elementary flow i extracted at location k or emitted to environmental compartment l at location k to the indicator score for impact category j; $Q_{i,k,l}$ is the quantity of elementary flow i extracted at location k or emitted to compartment l at location k (from the inventory); and $CF_{j,i,k,l}$ is the characterisation factor under impact category j for

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elementary flow i extracted at location k or emitted to compartment l at location k. The relation highlights two essential points of the characterisation phase: i) CF depends on intrinsic properties of the elementary flow and for some impact categories also on the location of emission or extraction of the flow; ii) the impact is proportional to the emitted quantity, and consequently the modelling of the impact from the product system is linear (Hauschild and Huijbregts, 2015).

The indicator result for each impact category is determined by adding the characterisation score of each environmental flow (Rosenbaum et al. 2018). The final score of the environmental burden relating to an impact category may calculate using the following equation:

$$IS_j = \sum_i \sum_k \sum_l IS_{j,i,k,l}$$

where all parameters are the same reported in the previous equation. Impact categories show a different site dependency, namely from no dependence for global impact categories as climate change and stratospheric ozone depletion to a high degree of sensitivity to local or regional conditions as for water use or acidification (Hauschild and Huijbregts, 2015).

Impact categories and corresponding indicators may be located at two levels along the cause-effect chain: at a midpoint and at an endpoint level (Margni and Curran, 2012). The characterisation phase performed at midpoint level relating to the elementary flows of an inventory results in a set of midpoint impact indicator scores which as a whole represent the characterised profile of the product system at midpoint level. This profile may be considered as the outcome of the LCIA step and also it may be used to perform the characterisation of impacts at endpoint level (Figure 16).

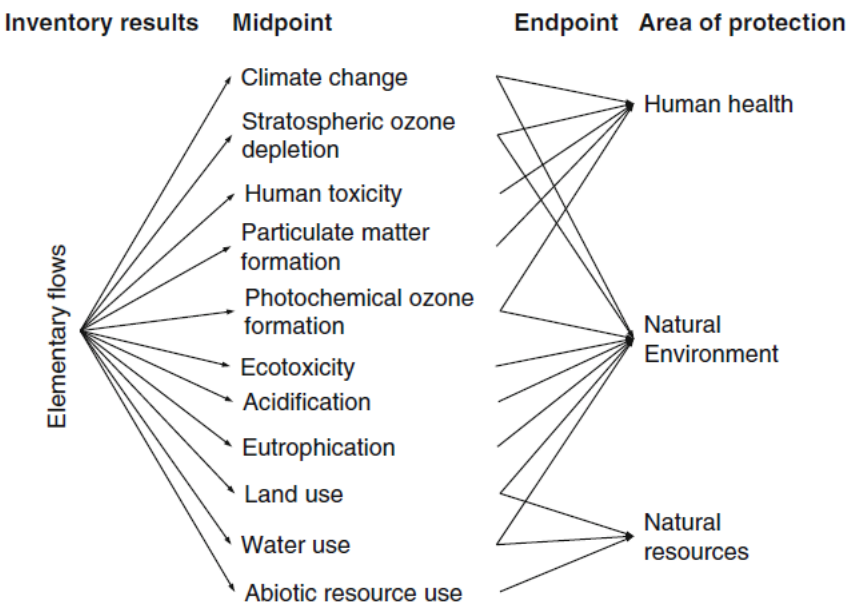

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Figure 16 - Connection of elementary flows to indicator results at midpoint level and endpoint level for 11 midpoint impact categories and 3 areas of protection in the characterisation phase (Hauschild and Huijbregts, 2015)

Characterisation at the endpoint level needs modelling of the entire impact pathway to the point in which impact occurs on the areas of protection, thus enabling cross-comparison of various impact categories within AoPs on a natural or social basis, and possibly considering substance-specific differences. Impact categories at the midpoint level are identified at the location in which a common process for a range of substance with a certain impact category occurs. For instance, "Global Warming" impact entails different phases, from the release of greenhouse gases to impacts on humans and ecosystems. The GHG may have an effect on the radiative forcing in a specific point along a certain pathway. In other words, GHG emissions show a pathway that is different before that point, but the same beyond that point (EC - JRC, 2010b). The passage from midpoint to endpoint indicator scores may occurs by using further midpoint-to-endpoint characterisation factors (sometimes also referred to as severity or damage characterisation factors) able to show the capacity of a change in the midpoint indicator such as to affect the endpoint indicator. In contrast to the midpoint characterisation factors which represent the properties of the elementary flow, and thus are elementary flow-specific, the midpoint-to-endpoint characterisation factors express the properties of the midpoint indicator, that means only one characterisation factor per midpoint impact category (Rosenbaum et al. 2018). In other words, the elementary flows will show different characterisation factors at midpoint level, but their mid-to-endpoint characterisation factor is the same (Hauschild and Huijbregts, 2015).

Endpoint indicators referring to the same AoP are characterised by a common unit and may be summed up to an aggregated impact score per AoP adopting equal or different weighting of each endpoint indicator (Rosenbaum et al. 2018).

4.4 Normalization

This phase allows to compare impacts associated with a certain product or system with reference values representing the impacts of a reference system (e.g., a different product, a region, a country, or the whole globe) (Crenna et al., 2019). The absolute values of environmental impacts as an assessment measure is hard to interpret unless it is considered under an appropriate environmental context. Indeed, normalization enables to translate abstract impact scores for each impact category into relative contributions of a product to a reference situation (Sleeswijk et al. 2008). Basically, the main purpose of this phase is to better understand the magnitude of the environmental impacts owed to the system under consideration (Baumann and Tillman, 2004). In other words, normalization representing impact results relating to environmental burdens of a reference system, may meet the following question: "*Is that much?*" (Rosenbaum et al., 2018). Furthermore, this step may support the search of inconsistencies, the communication of information on the relative importance of the indicator results, and the preparation for further phases, such as weighting, grouping, or interpretation (Dahlbo et al., 2013). The use of normalisation references (i.e., a common reference impact) enables to relate the relative magnitude of an impact with other impacts within the life cycle of a product via a common unit and a comparison may be made between the relative magnitude of the contribution of the impact in one impact category and the

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magnitude of the contribution in another one (Sala et al., 2015). For instance, normalization make possible to detect when acidification impact due to a certain product are high in relation to entire acidification impact in the region where the product is obtained and utilised, although this comparison might not be important if it is made between the impact per functional unit and the entire impact in the region. The importance may increase if the comparison occurs between the entire impact of the total use of the product and the entire impact in the region (Baumann and Tillman, 2004).

Generally, a region is selected as a reference of the background environmental load associated to all activities (e.g., economic and production activities) in the region under consideration. Also, normalisation values require to be regularly update, especially in order to ensure significance of the values (Sala et al., 2015). From this, it is easy to understand that the bias owed to the choice of normalisation references is the main criticality of this phase, since it might induce a change of the conclusions resulted from the LCI assessment (Pizzol et al., 2017).

In summary, normalisation is aimed to: i) strengthen the information on the relative magnitudes of the environmental impact potentials; ii) facilitate the performance of the subsequent phase, namely weighting; iii) support the check of the consistency and reliability of the results; and iv) facilitate the communicating results. The commonly used references are entire impacts per impact category per: i) geographical zone which may refer to the global, continental, national, regional, or local dimension; ii) inhabitant of a geographical zone (e.g., the "environmental space" occupied per average person); iii) industrial sector of a geographical zone (e.g. the "environmental space" occupied by a certain product system relative to similar industrial activities); and iv) baseline reference scenario, such as another product system which, for instance, may be expressed as the "environmental space" occupied by a specific product system relative to a similar reference system using best available technology). The first three reference systems reported above is also known as external normalisation, whereas the last reference system is referred to as internal normalisation (Rosenbaum et al., 2018). The normalisation phase may be performed according to two different approaches: external or internal depending on whether the chosen reference system is included within the system under consideration or independent from it (Norris, 2001).

As reported by Laurent and Hauschild (2015), the internal normalisation may be performed in several ways, by division such as division by baseline, division by maximum or division by sum. The first is based on the selection of a reference alternative or a baseline scenario among the ones included in the analysis and whose characterization impact indicator scores for each impact category are divided by the score of the selected reference alternative. Therefore, comparisons of the obtained ranking, relative to the reference alternative, may be conducted among impact categories (e.g., expressing the obtained ratios as percentages). The other options describe the normalisation reference as a function of the characterised impact indicator scores of the different alternatives (i.e., division by maximum and division by sum). In the "division by maximum" approach, the normalisation reference for every impact category corresponds to the maximum impact indicator score resulting from considered alternatives. Hence, all resulting scores range from 0 (i.e., the best scoring) to 1 (i.e., the worst scoring). The "division by sum" option, expresses the normalisation reference as the sum of the impact indicator scores for all considered alternatives within a certain impact category. As with the "division by maximum", the normalised scores range from 0 to 1, though the sum of normalised impact scores across the alternatives' scores is equal to 1 within every impact category. Laurent and Hauschild (2015) also highlight that, unlike the internal normalisation, the use of external one requires the availability of normalisation references, namely the environmental profile of an external reference system

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which is independent from the system under consideration, and frequently refers to a much wider scale. Specifically, the development of external normalisation references may be considered a LCIA of the inventory relative to a wide-scale system of which inputs and outputs in terms of resources and emissions throughout a certain period are inventoried and characterised. The scope of the normalisation reference should be taken into account what the reference system is the most relevant to compare the assessed system depending on the goals of the LCA analysis. Therefore, scoping should not disregard two points, that is the boundaries of the reference system and the period of time or reference duration of the considered activities. The system boundaries may be distinguished in generic system boundaries which is generally termed on the basis of geographical boundaries at a wide scale (i.e., the system is a region, a country, a continent or the entire world); and case-specific boundaries which is characterised by case-study-specific parameters, such as municipality in which the evaluated systems may be included (e.g., waste management), a company manufacturing the product under consideration, or an entire branch of manufacture and trade relating to the product (e.g., the textile industry). The choice of time duration (e.g., a month or a business year) depends on the context of the LCA analysis (i.e., goal and scope) and on the type of considered reference system. Frequently, a given year is taken as reference duration. This year is generally termed as the latest year for which reliable data are available for impact categories. Also, the same reference time or reference year should be used for all impact categories included in the set of normalisation references to avoid the bias that may potentially be introduced into the set of normalisation references.

According to Rosenbaum et al. (2018), normalisation is performed applying normalisation factors (NF) that should be computed using the same characterisation factors for the reference inventory as used for the inventory of the product system under consideration. The NF values are calculated by quantifying all environmental interventions E for all elementary flows i for the reference system and applying the characterisation factors CF per elementary flow i, respectively, for each impact category c as showed by the following equation:

$$NF_c = \left(\frac{\sum_i (CF_i \cdot E_i)}{P_r} \right)^{-1}$$

The sum of the product CF·E for all elementary flows represents the total impact of the reference system per impact category. Also, the normalisation reference is generally (but not obligatory) divided by the population P of the reference region r, in order to express the NF per average inhabitant of the reference region (per capita impacts or person equivalents). On the basis of the previous equation, a total impact of the reference system per impact category is computed determining one NF per impact category c.

In order to ensure consistency of result, the inventory data used to quantify a NF should represent a common reference year and duration of activity (generally one year) for all impact categories. The NF value obtained from this result is characterised by a unit expressing an impact per person and year, also referred to as person equivalent. Furthermore, a normalised impact score NS for a product system is computed according to the equation:

$$NS_c = IS_c \cdot NF_c$$

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where IS is the impact score for a certain product system and NF is normalisation factor per impact category c.

In order to collect inventory data for computing of normalisation, two different approaches may be used: production-based (or top-down) which refers to the interventions occurring in the reference region as result of the total activities in the region; and consumption-based (or bottom-up), which represents the interventions that are caused somewhere in the world resulting from the consumption occurring in the reference region (and hence they refer to the demand for industrial and other activities within and outside the reference region).

Laurent and Hauschild (2015) describe various limitations and disadvantages that may be encountered both in internal and external normalisation approaches. The use of internal normalisation may be characterised by: i) the risk of including a division-by-zero in the event that the baseline alternative shows no effect for some impact categories; ii) the application of the same set of weighting factors in the "division by sum" and "division by maximum" approaches, potentially may conduct to different ranking; and iii) the result obtained by the application of generic weights in any internal normalisation are insensitive to the magnitude of the impact indicator scores. In order to achieve robust results a recommendation is to use different approaches (e.g., "division by sum" and "division by maximum") and to analyse any possible inconsistency before making conclusions. As regards external normalisation, the uncertainties associated with the normalisation references, that is the degree in which the defined normalisation references represent the total environmental burden of the selected reference system, may be considered a limitation of the external normalisation. Basically, if the calculation of both characterised results and the normalisation references is performed using the same characterisation database, the main biases may be caused by: i) substance emissions or resource consumptions included in the inventory of the considered system but not in the normalisation inventory; ii) substance emissions or resource consumptions included in the normalisation inventory but not in the inventory of the considered system; iii) missing characterisation factors for substance emissions or resource consumptions. Generally, the combination of the uncertainties of the characterised scores (in the numerator), those relating to the normalisation references (in the denominator), and the potential biases between the two complicate the opportunity to make assumptions about the magnitude of the total uncertainty in the normalised score, since these uncertainties / biases are not additive and some of them may compensate each other or cancel out. Nevertheless, the use of normalisation in such situations may lessen the total uncertainty of the impact scores, and the normalised impact score may show a lower uncertainty than the characterised impact score one.

In the light of the above, the interpretation of the normalised LCA results should be done with caution, since on the one hand normalization enables to express the different impact potential on a common scale, on the other hand it may modify the results of LCA analysis, and thus the final conclusions. Normalisation does not allow to describe weight or importance of one impact category compared with the others. Also, it may be applied at endpoint level with the same purposes of normalisation at midpoint level and the computing of endpoint normalisation references is performed using the same procedure, merely applying combined midpoint and endpoint characterisation factors (Rosenbaum et al., 2018).

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4.5 Weighting

Weighting may be described as a qualitative or quantitative procedure in which the relative importance of an environmental impact is weighted in comparison to the impacts (Baumann and Tillman, 2004). This phase may only be performed following the normalization and enables to prioritise impact categories by using different or equal weights to every category indicator, basically meeting the question: "*Is it important?*" (Rosenbaum et al., 2018). Weighting represents the estimation of the various impacts, (e.g., global warming or acidification) on the basis of their potential to damage environment (Brentrup et al., 2004). The evaluation stars from the indicator results, generally normalised ones, for the different impact categories or damages which are multiplied by a specific weighting factor which is aimed to describe the relative importance of the various impact categories / category endpoints among each other (EC - JRC, 2010a). Therefore, weighting typically provides one final number, according to the following equation:

$$W = \sum_{c} WF_c \times I_c$$

where I_c represents the impact score (or normalized impact score) for impact category c; WF_c is the weighting factor for this impact category; and W the weighted result (Heijungs and Guinée, 2012). The weights should represent an estimation of the relative importance of impacts, according to certain value choices, expressing preferences of, for instance, people, experts or organisations, regarding time (i.e., present vs future impacts), geography (i.e., local *vs* global), urgency, political agendas or cost (Pizzol et al., 2017). It is difficult to develop weighting factors that describe the values of all stakeholders, thus the most important stakeholders might be taken as a reference, although it might not be possible to develop one set of weighting factors that may meet the preferences of considered stakeholders. Vice versa, numerous sets of weighting factors may be used, describing the preferences of the most important stakeholder groups. If the use of different set of weighting factors does not lead to the same final recommendations, additional prioritisation of the stakeholders would be useful, or the analysed product system (s) should be modified so that an unambiguous recommendation across the applied weighting sets may be attained (Rosenbaum et al., 2018). Considering that ethical and ideological are components of the weighting phase, likely a consensus on these value will never be achieved. For this reason, the resulting weighting factors should in principle be verified and documented and a sensitivity analysis to evaluate the consequences of the impact results and weighting methods should be conducted (Baumann and Tillman, 2004). Although weighting is not science-based and objective step but characterised by subjective choices of one person or a group of individuals, it may be useful for: i) aggregating impact scores into several or one single indicator; ii) comparing across impact categories; and iii) communicating results applying an underlying prioritisation of ethical values (Rosenbaum et al., 2018). Specifically, weighting may ease decision making when trade-offs between impact category results do not enable to select one preferable solution among the alternatives or an improvement among possible ones (Pizzol et al., 2017).

Itsubo (2015) describes various principles used to determine weighting factors both at midpoint and endpoint level. Midpoint methods are widely divided on the basis of the weighting principle adopted, namely a panel method and a distance-to-target method. In the panel methods,

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weighting factors are calculated considering the level of importance attributed to every environmental issue by sampled subjects or a panel of experts. On the other hand, the comparison among more than ten impact categories requires an excessive load on respondents, the statistical significance of weighting factors based on answers to questionnaires is not always analysed, and information provided to the respondents as the basis for weighting is limited, thus reducing the transparency of the factors. The distance-to-target method is characterised by the assumption that the importance of an environmental impact is expressed as the difference between the desired value and the actual value. This method allows to use desired value on the basis of information authorised by the national government, etc., instead of being determined subjectively by individuals, and thus obtaining generic weighting factors. However, the results may differ highly since they depends on how the desired value is obtained.

Endpoint methods consider normalised midpoint scores which are weighted among endpoints (type 1) and weighting obtained by multiplying the value per unit of damage to the endpoint by the result of endpoint characterisation (type 2). In the endpoint method (type 1), specialists, general consumers, etc. may provide the values of environmental impacts in questionnaires or through group discussions. Normalisation is based on a procedure that enables the sum of the weighting factors to be 1, and since the number of endpoints for weighting is limited to three (i.e., the areas of protection) the effort of respondents is relatively small. Furthermore, in this method weighting is performed for each of three perspectives or lines of environmental thought (hierarchist, egalitarian, and individualist, that is three cultural perspectives), allowing the practitioner to make analysis based on his or her own environmental thinking by determining the group to which subjects belong. However, the number of samples used to determine the weighting factors is small, and thus the weighting factors are little representative, making this method inappropriate for general use.

With endpoint method (type 2) the result is determined by computing the value per unit of damage to an endpoint and multiplying this value by the result of characterisation. Different methods adopt as reference the economic value per unit of endpoint and the results of assessment is obtained are expressed in economic metrics, namely in monetary values.

In other words, as reported by Baumann and Tillman (2004) values regarding environment may be described as the costs of different types of environmental damage or as the prices of different environmental goods, although for these goods there is no market, and thus no price. A price may be obtained by individuals' willingness-to-pay (e.g., they are asked how much they are willing to pay to avoid extinction of a species) or shown by their behaviour (e.g., the difference in price of similar houses near and far away from an airport detects the cost of noise). The economic approach considers the assumption that social values are based on the aggregation of individuals' preferences and that values of environmental qualities may be replaced by other goods.

Furthermore, the use of economic value facilitates the understanding and communication of the results and for instance, enables to be used these values for cost-effectiveness analysis. However, methods for translating environmental impacting resulting in health damage and ecosystem deterioration into economic values are still under development, and results may be highly biased or incomplete and their use may be unsuitable depending on how the information are disclosed. It is also emphasized that expressing of people's health etc. into economic value poses an ethical problem in the application of this method (Itsubo, 2015).

The results obtained by the weighting phase may be aggregated across the impact categories to a single score in order to make it easier the last LCA phase, namely interpretation (Hauschild and

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Huijbregts, 2015). The conversion of endpoint result in single scores is an issue strongly debated in the LCA community and is still deemed open, since, a subjective weighting is used for the aggregation of endpoint results in single score. The use of midpoint or endpoint indicators might be much more transparent than only showing single score indicators since it is still possible identify the best alternatives with respect to the different indicators and normalisation may highlight which environmental indicators the product under consideration contributes the most to, or the least. Therefore, the sole use of the single-score indicators might be inappropriate but combining endpoint or single-score indicators with midpoint indicators may facilitate the interpretation of the results and improve transparency if the contribution of the different midpoint categories to the endpoint or single score results is shown (Kägi et al., 2016).

4.6 Grouping

The last optional step of the LCA procedure consists in attributing the impact categories into one or many groups applying two methods: sorting and ranking (ISO 14044, 2006).

As reported by Rosenbaum et al. (2018), sorting midpoint impact categories is to assign of categories on a nominal basis (e.g., by characteristics such as emission-related and resource-related, or global, regional or local spatial scales). Ranking the impact categories occurs according to a set (subjective-based on ethical value-choices) hierarchy (e.g., high, medium or low priority). Furthermore, grouping may be useful for the presentation of LCI results, for instance LCI result parameters may be shown in the groups emissions to air, emissions to water, etc. (Baumann and Tillman, 2004). According to ISO 14044 standard (2006), "*Different individuals, organizations and societies may have different preferences; therefore it is possible that different parties will reach different ranking results based on the same indicator results or normalized indicator results*".

4.7 ReCiPe

It was developed in 2008 by RIVM (National Institute for Public Health and the Environment), CML (Centrum Milieukunde Leiden), PRé Consultants and the Radboud University Nijmegen on behalf of the Dutch Ministry of Infrastructure and the Environment (Goedkoop et al., 2013).

From methodological point of view, the ReCiPe method is the most developed method to support LCIA according to the literature and the European Commission (Mota et al., 2015). Furthermore, this method is well suited to evaluate the environmental performance associated both with life cycle of agri-food products and photovoltaic systems (Mahmoudi et al., 2020; Rashedi and Khanam, 2020; Shrestha et al., 2020; Agostini et al., 2021; Harun et al., 2021; Laca et al., 2021;).

The LCI assessment of a certain product may be implemented on the basis of two methodological approaches, namely the midpoint and the endpoint that provide environmental indicators at different levels (EC - JRC, 2011a). The midpoint is considered as a point on the cause-effect chain between stressors and endpoints; in contrast, the latter are physical elements that society establishes as worthy of protection (e.g., human health, ecosystem, and natural resources) (Bare and Gloria, 2008). On the basis of above, the main purpose of ReCiPe is to harmonized the existing midpoint and endpoint approaches in terms of modelling principles and choices but it provides results at both midpoint and endpoint level (Goedkoop et al., 2013) (Figure 17).

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Basically, the strength of this method is the ability to connect the midpoint and the endpoint levels converting the former into the latter through a set of endpoint characterization factors (Dong and Ng, 2014).

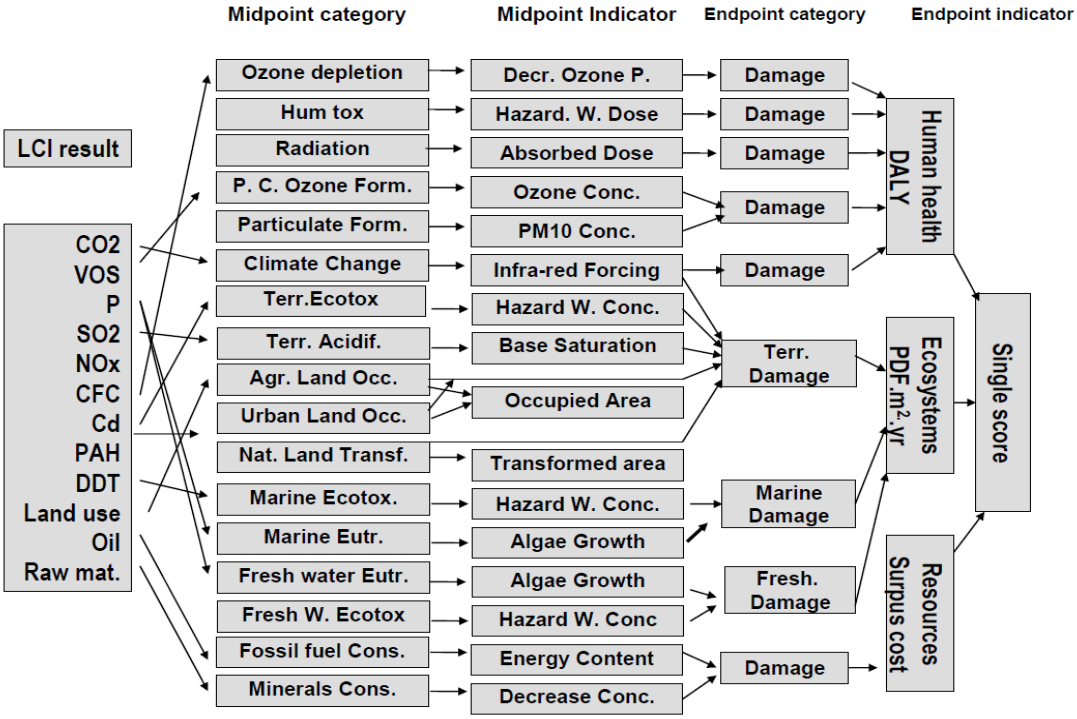

Figure 17 - Impact categories and pathways encompass by the ReCiPe methodology (EC - JRC, 2010c)

ReCiPe is organised on the basis of the main steps of LCIA procedure, namely characterization, damage assessment, normalization, weighting and single score. In short, with the characterisation, the only compulsory step according to ISO standards, the quantity of each substance involved in life cycle of a certain product and that contribute to one or more impact categories, are multiplied with a characterisation factor, which represents the relative contribution of the corresponding substance. Such contributions are summed up to obtain the characterisation result indicators at midpoint or endpoint level (Rashedi and Khanam (2020).

Specifically, the characterisation factor at midpoint level is determined according to the equation:

$$I_m = \sum_i Q_{mi}\, m_i$$

where I_m represents the indicator result for midpoint impact category m; Q_{mi} is the characterisation factor that relates any emission, extraction, or use of a substance i to midpoint impact category m; and m_i is the magnitude of emission, extraction, or use of the substance i (e.g., the mass of CO_2 released to air) (Table 10) (Goedkoop et al., 2013).

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As reported by Huijbregts et al. (2016), the endpoint characterisation factors included in ReCiPe regard three areas of protection: human health, ecosystem quality and resource scarcity. For each AoP, specific endpoint indicators are determined, namely DALYs (disability adjusted life years) associated with human health, represent the years that are lost or that a person is disabled due to a disease or accident. For ecosystem quality, the reference unit is the local species loss integrated over time (species year). Dollars ($) is used for resource scarcity and it represents the extra costs involved for future mineral and fossil resource extraction.

Endpoint characterisation factor (CFe) is directly obtained by the midpoint one (CFm) according to the following equation:

$$CFe_{x,c,a} = CFm_{x,c} \times F_{M\rightarrow,E,c,a}$$

where c represents the cultural perspective, a is the area of protection, x is the stressor of interest, and $F_{M\rightarrow,E,c,a}$ is the midpoint to endpoint conversion factor for cultural perspective c and area of protection a. The midpoint to endpoint factors are constant per impact category, since environmental mechanisms are considered identical for all stressors after the midpoint impact location on the cause-effect pathway (Huijbregts et al., 2016).

Table 10 - The main impact categories based on ReCiPe method (extracted from Goedkoop et al., 2013)

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Impact categories	Abbrev.	Indicator	Unit	Midpoint characterisation factor	Abbrev.	Unit
climate change	CC	infra-red radiative forcing	$W \times yr/m^2$	global warming potential	GWP	kg (CO_2 to air)
ozone depletion	OD	stratospheric ozone concentration	ppt[a]×yr	ozone depletion potential	ODP	kg (CFC[c]-11 to air)
terrestrial acidification	TA	base saturation	$yr \times m^2$	terrestrial acidification potential	TAP	kg (SO_2 to air)
freshwater eutrophication	FE	phosphorus concentration	$yr \times kg/m^3$	freshwater eutrophication potential	FEP	kg (P to freshwater)
marine eutrophication	ME	nitrogen concentration	$yr \times kg/m^3$	marine eutrophication potential	MEP	kg (N to freshwater)
human toxicity	HT	hazard-weighted dose	-	human toxicity potential	HTP	kg (14DCB[d] to urban air)
photochemical oxidant formation	POF	Photochemical ozone concentration	kg	photochemical oxidant formation potential	POFP	kg (NMVOC[e] to air)
particulate matter formation	PMF	PM10 intake	kg	particulate matter formation potential	PMFP	kg (PM10 to air)
terrestrial ecotoxicity	TET	hazard-weighted concentration	$m^2 \times yr$	terrestrial ecotoxicity potential	TETP	kg (14DCB to industrial soil)
freshwater ecotoxicity	FET	hazard-weighted concentration	$m^2 \times yr$	freshwater ecotoxicity potential	FETP	kg (14DCB to freshwater)
marine ecotoxicity	MET	hazard-weighted concentration	$m^2 \times yr$	marine ecotoxicity potential	METP	kg (14-DCB to marine water)
ionising radiation	IR	absorbed dose	man×Sv[b]	ionising radiation potential	IRP	kg (U^{235} to air)
agricultural land occupation	ALO	occupation	$m^2 \times yr$	agricultural land occupation potential	ALOP	$m^2 \times yr$ (agricultural land)
urban land occupation	ULO	occupation	$m^2 \times yr$	urban land occupation potential	ULOP	$m^2 \times yr$ (urban land)
natural land transformation	NLT	transformation	m^2	natural land transformation potential	NLTP	m^2 (natural land)
water depletion	WD	amount of water	m^3	water depletion potential	WDP	m^3 (water)
mineral resource depletion	MRD	grade decrease	kg^{-1}	mineral depletion potential	MDP	kg (Fe)
fossil resource depletion	FD	lower heating value	MJ	fossil depletion potential	FDP	kg (oil)

a: ppt refers to units of equivalent chlorine;
b: (man×Sv), Man Sievert (1 Sv = 1 J/kg body weight)
c: CFC, chlorofluorocarbon;
d: 14-DCB, 1,4 dichlorobenzenee;
e: NMVOC, Non Methane Volatile Organic Carbon compound.

In damage assessment phase, the mid-point impact indicators relating to a specific end-point/damage category are grouped to determine the end-point indicator result. Furthermore, different mid-point and end-point indicators are expressed in different units which make it difficult to compare the result of one mid-point or end-point indicator with another following characterization. Normalisation provides a solution for the unit incompatibility by dividing each indicator value with a set reference, but it is not representative of the relative importance of indicators. The order of magnitude is determined through the weighting phase in which a specific weighting factor is applied to all normalized values (Rashedi and Khanam, 2020).

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At this point, outputs (i.e., the weighted end-point scores) may be expressed in form of single score which may be obtained by the aggregation of weighted results (Itsubo, 2015). In other words, the weighting allows to express the importance of each damage category relative to each other so that they may then be summed up to get a single number for the total environmental impact (Solinas et al., 2015). Specifically, the single score (e.g., the so-called ecopoint) is a unit of environmental penalty, thus the higher ecopoints for a certain process, the higher will be the environmental impact (Monti et al., 2009).

In the normalisation phase, the European and global population are used as reference values (PRé, 2014).

The aggregation of eighteen impact categories into only the three damage categories mentioned above, facilitates results interpretation and communication to the detriment of their uncertainty. On the other hand, the environmental mechanism and damage models are sources of uncertainty since the different relationships have been modelled on the basis of the state-of-the-art knowledge of the environmental mechanisms which are characterised of a certain level of incompleteness and uncertainty (PRé, 2014). According to Goedkoop et al. (2013), the ReCiPe method assesses the environmental burden on the basis of three uncertainty perspectives, that is individualist (I), hierarchist (H), and egalitarian (E). All of them should not be considered representative of archetypes of human behaviour, but they are only used to group similar types of assumptions and choices. Specifically, "*perspective I is based on the short-term interest, impact types that are undisputed, technological optimism as regards human adaptation; perspective H is based on the most common policy principles with regards to time-frame and other issues; and perspective E is the most precautionary perspective, taking into account the longest time-frame, impact types that are not yet fully established but for which some indication is available, etc.*". A specific weighting set is available for each perspective. Generally, the hierarchist version of ReCiPe with average weighting is chosen as default and value choices made for this perspective are accepted from scientific and political point of view (PRé, 2014).

5. Interpretation

The final LCA phase relates the results of inventory and impact assessment to the goal and scope of the study in order to reach conclusions and recommendations (Heijungs and Guinée, 2012). Specifically, interpretation has two main purposes: i) throughout the iterative LCA steps and for each obtained result, this phase is useful to address the study towards improving the LCI model in order to meet the needs contained in the analysis goal; and ii) to achieve the final LCI model and results through the iterative LCA steps, and for comparative LCA studies, interpretation allows to draw strong conclusion and, often, recommendations (EC - JRC, 2010a). Therefore, the deliverable of this phase should consist of conclusions and recommendations which i) pay attention to the contents included in the goal definition and the limitations that the study purposes establish through the scope definition; and ii) consider the appropriateness of the functional unit and system boundaries. Moreover, the interpretation should depict the LCA conclusions in an understandable way and support the users to evaluate their robustness and potential weaknesses on the basis of any study limitations (Hauschild et al., 2018).

The ISO standard only describes the three steps through which interpretation should proceed, without specifying any methods for analysing the results (Figure 18) (Baumann and Tillman, 2004). In other words, although this step may be considered a key factor to support the decision

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making process, details on procedure and techniques to be used are not provide by ISO standards (Heijungs and Guinée, 2012).

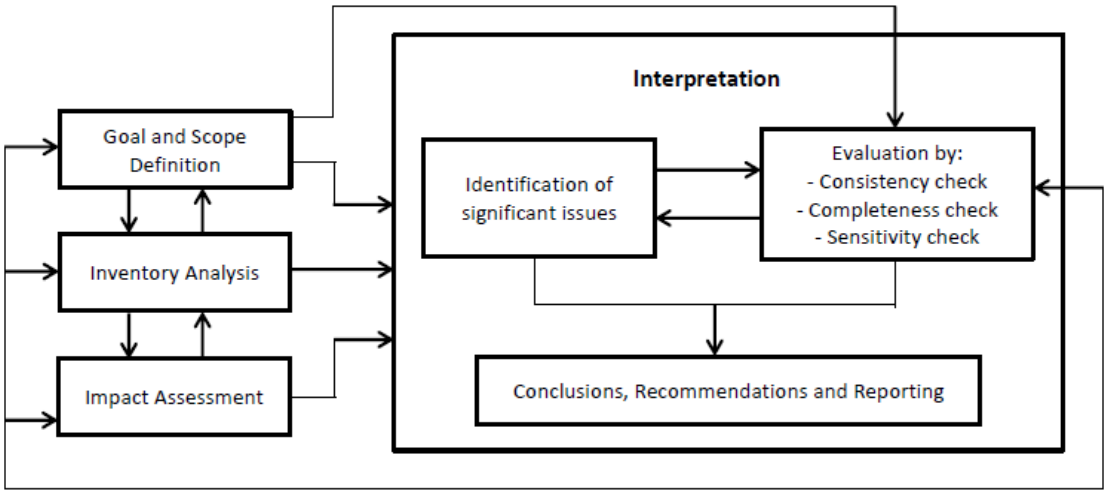

Figure 18 - Elements of the interpretation phase, their mutual relations, and the relations with the other phases of LCA (Curran, 2006)

Considering that the LCA methodology is becoming increasingly important for decision support in the policy context and consequently for entire sectors and society, a practical support on how to implement a comprehensive interpretation of LCI and LCIA results might be an essential tool to ensure robust conclusion and recommendations (Zampori et al., 2016).

According to ISO 14044 standard (2006), the LCA interpretation basically encompasses the following elements:
"- Identification of significant issues;
- An evaluation that considers completeness, sensitivity and consistency checks;
- Conclusions, limitations, and recommendations.
- Appropriateness of the definitions of the system functions, the functional unit and system boundary;
- Limitations identified by the data quality assessment and the sensitivity analysis".

Specifically, the first component of the life cycle interpretation phase, namely the identification of significant issues is aimed to determine the most environmentally significant issues represented by any problem that may potentially change the final LCA results. The significant issues may regard methodological choices and assumptions, inventory data, and/or characterisation, normalisation or weighting factors used in the LCIA phase (Hauschild et al., 2018).

According to Curran (2006), this phase may be considered as the support for the evaluation one because of the large amount of collected data. Therefore, before establishing which data and/or information of the LCI and LCIA phase more contribute to the results for each scenario, the other phases of LCA should be completely re-examined (e.g., study goals, ground rules, impact category weights, results, external involvement) to verify if the goal and scope of the LCA analysis have been met. If they have, the significant issue may be determined. They may include

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inventory parameters (e.g., energy use, emissions, waste), impact category indicators (e.g., resource use, emissions, waste), and contributions to LCI or LCIA results, such as individual unit processes or groups of processes (e.g., transportation, energy production). In order to identifying the environmental issues and their significance, ISO 14044 standard (2006) recommends different approach, that is: i) "*contribution analysis, in which the contribution of life cycle stages … or groups of processes … to the total result are examined by, for example, expressing the contribution as a percent of the total*"; ii) "*dominance analysis, in which, by means of statistical tools or other techniques such as quantitative or qualitative ranking (e.g. ABC analysis), remarkable or significant contributions are examined …*"; iii) "*influence analysis, in which the possibility of influencing the environmental issues is examined …*"; iv) "*anomaly assessment, in which, based on previous experience, unusual or surprising deviations from expected or normal results are observed. This allows a later check and guides improvement assessments …*".

The second component of interpretation provides the basis for the conclusion and recommendation drawn up in the last part of a LCA study. It is developed via an iterative interaction with the previous step in order to determine the reliability and strength of the results on the basis of the identification of the significant issues. As the first component, the evaluation refers to results from the inventory analysis and the impact assessment, to their consistency with the goal and scope of the study, and pay particular attention to the significant issues identified among methodological choices and data. The evaluation is essential to highlight the stability of the conclusions and recommendations, and thus it should be depicted in a clear and transparent way in order to facilitate as much as possible the understanding of the final result for the commissioner and user (Hauschild et al., 2018). In order to attain the evaluation purpose, this step involves a completeness check, a sensitivity analysis, and a consistency check (Figure 16).

Curran (2006), describes each activities emphasizing their importance to ensure the reliability of the LCA results. In short, the completeness check is aimed to guarantee that all important information and data for the interpretation are available and complete. Data for life cycle stages, different processes or unit operations, or any type of considered data (e.g., raw materials, energy, transportation, environmental release to air, land, or water) should be included in an appropriate checklist to show the significant area included in the results. The checklist enables to verify that all data are consistent with the system boundaries and consequently that results are representative of LCA goal and scope. If some data are missing, the gap should be filled, but they are not available, it is necessary to indicate the differences in the data with the final results and assess the impact to the comparison quantitatively (percent uncertainty) or qualitatively (i.e., the result of the reported alternative A may be higher since "X" is not included in its assessment). In other words, each LCA phase should be reviewed in order to fill gap in case of missing or incomplete information, and the unavailability of some data should be considered as a limitation in the conclusion of the study (Hauschild et al., 2018).

Sensitivity check may be useful to assess the reliability of the results detecting if the uncertainty in the significant issues found in the previous phase may influence the decision-maker's ability to draw comparative conclusions (Curran, 2006). This task is basically aimed to determine the most significant processes and elementary flows as the ones that contribute most to the total impacts due to a certain product system. Furthermore, it may be used as support of the iterative approach applied in LCA, particularly in the boundary setting for the product system, inventory data collection, and impact assessment. The results obtained by these sensitivity analyses are included in the sensitivity check of the interpretation phase (Hauschild et al., 2018).

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Curran (2006) specifies that a sensitivity check may be applied on the significant issues according to three techniques generally used for data quality analysis, that is: "*1. Contribution Analysis – Identifies the data that has the greatest contribution on the impact indicator results.*
2. Uncertainty Analysis – Describes the variability of the LCIA data to determine the significance of the impact indicator results.
3. Sensitivity Analysis – Measures the extent that changes in the LCI results and characterization models affect the impact indicator results".

Finally, the consistency check is aimed to examine if the assumptions, methods, and data, which have been involved in the study, are consistent with the goal and scope. When the analysis regards comparison among various product systems, the consistency check also analyses if allocation rules, system boundary setting, and impact assessment have been consistently applied to each compared product systems. In case of inconsistencies, their effect on the findings is assessed and took into account in the study conclusions (Hauschild et al., 2018).

As reported by Curran (2006), the last component of interpretation is to draw conclusions and recommendations on the basis of the LCIA results (not the LCI) to identify which product/process has the total least impact to human health and the environment, and/or to one or more specific areas of interest as stated in the goal and scope of the analysis. When a final score is computed, the recommendation might be to consider as the best alternative, the product/process with the lowest score, or to examine the reasons how the process might be changed to reduce the score. The conclusions and recommendations should be based on a balance between the potential human health and environmental impacts depending on the study goals and stakeholder concerns. It is essential to draw conclusions and provide recommendations taking into account only the facts. It might occur that is not be easy identify which product or process is better, because of uncertainties and limitations in the methods used to perform the LCA analysis or the availability of good data, time, or resources. In this case, the LCA findings are still valuable, since, via the conclusions and recommendations, they might provide useful information to decision-makers about human health and environmental pro and cons, understanding their impacts, for instance where they are occurring (locally, regionally, or globally), and the relative magnitude of each type of impact compared to the other alternatives included in the study.

Zampori et al. (2016) provide useful information on the focal points that should not be disregarded in order to provide a robust support to decision makers that may be ensured by a proper interpretation of LCA findings. In short, the robustness of conclusions strongly depends on the consistency of data and elementary flows included in the inventory, characterisation models, the appropriateness of the functional unit, the intended goal, the system boundaries and value choices, normalisation and weighting set which may produce different results. Therefore, it is recommended that the total uncertainty of a LCA analysis is evaluated and reported in the final report in order to draw conclusions in a more consistent and harmonized manner such as to allow users a better understanding of the results and their interpretation.

6. LCA application at regional scale

An agricultural activity may be considered sustainable from the environmental perspective whether the release of potentially harmful substances may be tolerated by the natural environment in the long term (Payraudeau and van der Werf, 2005). The environmental burden due to agricultural sector does not remain within the borders of the single farm. Sustainability of agricultural systems regards different aspects related to resilience (i.e., the capacity of

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systems to buffer shocks and stresses) and persistence (i.e., the capacity of systems to maintain in the long term), and encompasses numerous broader aspects of economic, social and environmental sphere (Pretty, 2008). Because of extent of agricultural territories and considering that numerous emissions and impacts caused by farming depend on their surroundings, the use of spatially explicit data to evaluate territorial environmental load carefully is necessary (Nitschelm et al., 2016).

Sustainability evaluation of products or technologies generally covers impacts occurring in three dimensions, namely the social, the environmental, and the economic ones for which the life cycle approach may be important to avoid problem shifting in the product system (Finnveden et al., 2009). However, ecosystem degradation and environmental change are characterised by numerous processes strictly connected to each other, and local ecological effects may be caused directly or indirectly by activities in or by other regions. In other words, in a global context in which natural capital may be a limiting factor for human development and sustainability and where an activity in a certain region may be concurrently responsible for an impact in another region and in its turn to be affected, the evaluation, determination, and modelling sustainability should go beyond the single region and to include the interregional component (Kissinger and Rees, 2010).

Despite LCA was originally designed as a spatially independent procedure, namely as a product-oriented approach, overt time different attempts were made to broaden the object of LCA analysis, including companies, consumer lifestyles or nations as a whole, with the purpose to support decision-making by identifying environmental hotspots, and thus to enhance the environmental effects of future policies (Loiseau, et al., 2018). Accordingly, it looks important to develop a method to evaluate environmental burden of activities within a territory ("territorial LCA") by considering their locations in a spatially explicit manner (i.e., "spatialized territorial LCA").

Agricultural territories have been examined spatially for years. On the other hand, several LCA analysis of agricultural products have been performed at field or farm level, whereas some studies have integrated regionalization in agricultural LCA (e.g., regional land-use impacts on biodiversity or on climate change, or potential desertification due to agricultural activities at the country level) emphasizing that the use of regionalized data and spatial differentiation might allow to obtain a more precise and comprehensive evaluation than the ones of conventional LCA analysis (Nitschelm et al., 2016). Indeed, environmental impact related to agricultural activity are too affected by technology and geographical location, consequently the regional characterization of environmental impacts combining midpoint and endpoint modelling might provide a more realistic and balanced description of agricultural practices and of the potential environmental burden ensuring a more accurate evaluation of product sustainability. Specifically, the use of site-generic characterisation factors data in the LCIA phase may determine results that are not representative of the impacts (i.e., in terms of the magnitude of impacts, the most relevant midpoint categories, and their relevance on endpoint level) highlighting, on the contrary, that more site-specific data relating to cropping practices and emission factors together with spatial differentiation and harmonized midpoint–endpoint analysis might increase the discriminating power of LCA (Canaj et al., 2020). However, the application of the LCA approach to an entire territory is not free of difficulties, particularly regarding the definition of the functional unit(s) of the system, the selection of appropriate boundaries, the modelling of the system, and the collection of required data to meet the goal of the study to avoid double counting and reduce efforts on data collection (Loiseau et al., 2012).

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Before performing a large-scale LCA study, it is necessary to define the term "territory and/or region" in order to apply the method and conduct all phases accurately. Payraudeau and van der Werf (2005) emphasize that farm is the main management unit of the agricultural system whereas a farming region is considered a geographic entity characterised by the activities and the social groups which occupy it and interact there. The geographical burdens of a farming region are thus, highly variable since they are influenced by the considered political, economic, social and environmental factors. For this reason, the regional scale analysis may enable to examine the relationships, communication and competition among farms highlighting the potentiality of the agricultural sector at this scale. Accordingly, the assessment of environmental burden associated with a farming region should not be obtained by the sum of the evaluations for each farm, since this geographical scale enables to identify the positive or negative impact of interactions among farms on the emissions of pollutants and on the consumption of resources in a certain area. These interactions might regard exchanges of services (e.g., field operations), exchanges of products (e.g., grain, straw, fodder, manure) or shared equipment for product transformation or waste treatment. On the contrary, the whole environmental load at farming region level is frequently assumed to be equal to the sum of the impacts for each farm on the basis of a system classification, which allows to extrapolate to the level of the farming region the results obtained at the farm scale (by assuming uniformity of farmer practices and production systems within a class). At regional scale, the environmental impacts of a certain crop may be evaluated by relating specific emissions, expressed as quantities per ha, to the total area of the crop under consideration.

In the light of the above, it seems clear that the implementation of a LCA study at regional scale is closely linked to the concept of territory which has been widely discussed within the scientific community over time. As reported by Nitschelm et al. (2016), territory may be considered "*a geographically contiguous area in which human activities occur that is managed by local stakeholders, whose representations (individual, ideological, and societal) of the territory influence their decisions*". Also, territory is different from region because of the inclusion of stakeholders that may face about common issues (e.g., environmental, economic, societal). Specifically, an agricultural territory refers to an area in which "*most land uses or economic activities are based on agriculture…*" and *"…stakeholders focus on questions such as the trade-off between agricultural production and the environment*". Therefore, territorial LCA is aimed to describe the potential environmental burden of scenarios considered in the decision-making process relating to land planning. Since regional characteristic are crucial to achieve this purpose, a spatial differentiation method is recommended in the performing of a territorial LCA (e.g., by using Geographic Information Systems (GIS) to geolocated processes).

As regards spatialized territorial LCA (STLCA), Nitschelm et al. (2016) emphasize that it allows to: i) provide more precise results than a territorial LCA (TLCA) without spatial differentiation; ii) facilitate the reduction of impacts within a territory by identifying which agricultural activities should be developed and their potential location; and iii) help avoid or minimize impacts due to inputs exchange from other territories. Specifically, STLCA may be deemed an useful tool for LCA study focused on agricultural territory since: i) changes in agricultural land-use may condition environmental effects of agriculture; ii) some effects may depend on the biophysical context in which harmful substances are emitted (e.g., weather, soil type); and iii) several decisions that condition environmental impacts are made at farm level, but may be affected by public policies at the territorial scale, for instance through subsidies or regulations.

Nevertheless, spatialized LCA is still under development since different issues are unsolved, such as how to connect the spatialized LCI to a spatially explicit characterization method and how to

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best map impacts to support decision making. In other words, the quantity of local data necessary for spatialized LCA studies may increase uncertainties in the LCI phase. However, studies have shown that use of generic CFs may cause errors in LCIA. Consequently, it looks necessary to consider spatial variability of impacts in order to reduce uncertainties on local impact results, particularly when evaluating environmental burden of a relatively large area such as a territory.

Loiseau, et al. (2018) have developed a framework to apply the LCA procedure to geographically and administratively defined systems. In short, according to this approach, the concept of territory is strictly linked to three dimensions: "*i) the material dimension of a geographic area defined by the physical properties that can be considered as opportunities or constraints for the development of human systems, ii) the organizational dimension defined by social and institutional actors structured within activities, organizations or jurisdictions that embody the strategies of territorial development, and iii) the identity dimension defined by the way social and institutional stakeholders think and implement a project for their territory*". On the basis of these three dimensions, a territory may be considered a multifunctional system that through the territorial functions provide goods and services according to the nature of the land and the manner it is used (i.e., from material functions such as provision of food or housing to intangible types, namely the landscape quality or cultural heritage). In the LCA context, two application may be distinguished: the first is focused on estimating specific sectors of activity that are situated in a given territory, such as agricultural systems, waste management systems, or water management systems in order to support decision-making by supplying information, determining hotspots, prioritizing actions and optimizing the studied systems. The second one describes the territory as the whole of its production and consumption activities with the purpose of estimating its eco-efficiency, namely the ratio of territorial services and the corresponding environmental impacts.

Basically, two types of territorial LCA applications may be identified: type A when the procedure refers to a specific activity or supply chain anchored in a certain territory and dependent on the geographical context. Type B is focused on evaluating all production and consumption activities situated in a specific territory to quantify its eco-efficiency, considering environmental forces included in trade flows with other territories. For both types, the aim is to support decision-making at subnational level (i.e., from city level to regional level), to elaborate environmentally friendly policies through making available of a territorial baseline and the comparison of spatial planning scenarios.

As regards the conventional LCA phases, territorial LCA of type A refers to the three territorial dimensions since they are deemed essential to determine the appropriate function of the system under consideration whereas no functional unit is need for LCA of type B. It rather requires defining and quantifying a set of land use functions regarding environmental impacts in order to provide two different findings useful for determining eco-efficiency ratios. Both types of TLCA show minor difference regarding the other LCA phases, although territorial contextualization should be included for both of them in LCI, LCIA and interpretation. This means that spatial variability for calculating representative inventories local environment when assessing site-dependent impacts should be taken into account.

Various aspects are still unsolved and should be improved within the TLCA application. For instance, the territorial physical properties in LCI and LCIA should be more systematically included by using of GIS tools. Indeed, the propagation of spatial information during the four LCA phases seems important to perform consistent territorial LCAs, although technical issues

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should be overcome for an effective coupling between LCA and GIS software. Furthermore, the multiscaling of a territory is another aspect that TLCA should not disregard, since a territory as an open system may be incorporated into other territories. Accordingly, spatial planning strategies implemented within its borders may highly affect other territories. To conduct a consistent assessment, these impacts should be taken into account, for instance, by adopting a consequential modelling approach. The necessity of territorial economic models is an important future research issue and should support the development of a solid methodology for territorial LCA.

Although different adaptations are needed to perform a TLCA compared to the conventional LCA framework, one of the major change regards the goal and scope definition phase (Table 11). Specifically, Loiseau et al. (2013) have provided detailed information on which aspects and how these should be modified to conduct a territorial LCA as accurately as possible. In short, the framework assumes that a territory, as has already been reported previously, is deemed a multifunctional, open dynamic, and complex system incorporated in a local context (geographical and societal). The multi-functionality may make it difficult the functional unit definition, the terms "open and dynamic" may make the boundary selection problematic. Furthermore, the complexity of the system may limit the data collection, and local context should not be disregarded in order to provide indicators for decision-making. Therefore, the main methodological advances regard the functional unit definition since in the TLCA the reference flow is defined a priori by the association of a territory and of a studied land planning scenario. This means that the starting point, for TLCA study, is the reference flow and no longer the functional unit. The functional unit results from the choice of the reference flow, i.e., the territory and its associated land use scenario. Consequently, the services provided by two land planning scenarios on a same territory are different and should be estimated together with the associated impacts.

In other words, unlike conventional LCA, two findings are obtained by this approach: not only environmental impacts, but also goods and services that are both related to the human activities situated in the territory under consideration. These results are expressed by two vectors (i.e., indicators): a vector of potential environmental impacts and a vector of land use functions provided by the territory (i.e., good and services) for different stakeholders (e.g., provision of work, recreation, and culture). This adapted framework might provide to land planning managers an useful tool for evaluating the environmental effects of plans or programs.

Table 11 - The main features of the two types of territorial LCA procedure (extracted from Loiseau et al., 2018)

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		Type A	Type B
Goal and scope definition	System	Specific activity sectors embedded in a given territory	All production and consumption activities located in a given territory and defined by a spatial planning scenario (total territory responsibility)
	Audience	Political and administrative decision-makers and all stakeholders who would like to manage and develop their territories	Political and administrative decision-makers and all stakeholders who would like to manage and develop their territories
	Application	Scenario comparison	Territorial diagnosis Spatial planning scenario comparisons
	Reasons	To assess the environmental performance of a territorial project or plan to support the local decision-making process	To assess the eco-efficiency of a territory as a whole to support the local decision-making process
	Decision context	Meso-level decision support	Accounting (eco-efficiency observatory) Meso-level decision support
	Functions	Function(s) provided by project or plan implemented in the territory Multifunctionality possible	Multifunctionality (economic, societal, envinromental functions)
	Functional unit	Functional unit(s) according to the territorial context	Consideration of functions and services provided by the territory. Evaluation of a set of territorial indicators, as an output of the approach
	Boundaries	From cradle to gate (e.g., agricultural production), gate to cradle, or cradle to grave depending on the study issue	From cradle to gate (i.e., for production activities such as agriculture) or grave (i.e., for consumption activities)
	Scenarios	Defined according to the territorial context	Spatial planning scenario defined by stakeholders/experts
	Allocation rules for foreground system	Substitutions determined according to the territorial context	No allocations
LCI	Data collecting	Process LCI based on site-specific data and spatial databases for the foreground systems and generic database for the background systems	Combining monetary and physical flows to compute hybrid LCI
	Mass and energy balance	Yes, considering all exchanges of environmental flows that take place between the studied activities in a territory	Not yet, because for now, intra-territorial flows are not considered
LCIA	Fate, exposure, and effect factors	When needed, should consider local conditions (climate, topology, receiving environment, fauna, flora, etc.) to adapt these factors to the territorial context	When needed, should consider local conditions (climate, topology, receiving environment, fauna, flora, etc.) to adapt these factors to the territorial context
Interpretation	Result presentation	Distinction of direct (in-site) and indirect (off-site) impacts Mapping the results	Distinction of direct (in-site) and indirect (off-site) impacts Mapping the results To date, a clear distinction between production and consumption contributions is needed to avoid any double counting of impacts

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Glossary

Allocation: process of assigning to each of the functions of a multiple-function system only those environmental burdens and impacts that each function generates (Azapagic and Clift, 1999).

Attributional LCA (ALCA): estimate of the pollution and resource flows within a chosen system attributed to the delivery of a specified amount of the functional unit (Thomassen et al., 2008).

Area of protection: cluster of category endpoints of recognisable value to society, viz. human health, natural resources, natural environment and man-made environment (Guinée et al., 2002).

Background system: set of processes aimed at supplying material or energy to the foreground system (Gava et al., 2019).

Cause-effect chain: environmental mechanism that depicts the relations between a midpoint indicator that depends on a stressor, and an endpoint indicator that depends on the midpoint indicator. For instance, the impact of CO2 emissions (stressor) is assessed on radiative forcing (midpoint) and human health (endpoint that depends on global warming) (Othoniel et al., 2019).

Characterisation: assessment of the magnitude of potential impacts on the environment (Margni and Curran, 2012).

Characterisation indicator: scientifically based indicators that are quantitative measures of environmental impact. Characterisation indicators may also called equivalents, potentials, category indicators or characterisation factors (Baumann and Tillman, 2004).

Characterisation model: mathematical model of the impact of environmental interventions with respect to a particular category indicator (Guinée et al., 2002).

Characterisation method: method for quantifying the impact of environmental interventions with respect to a particular impact category; it comprises a category indicator, a characterisation model and characterisation factors derived from the model (Guinée et al., 2002).

Classificiation: attributing of the inventory data to categories according to the impact to which they are known to relate (Margni and Curran, 2012).

Consequential LCA (CLCA): estimate how pollution and resource flows within a system change in response to a change in output of the functional unit (Thomassen et al., 2008).

Cradle-to-farm gate: LCA model which included the processes of a product's life cycle from raw material extraction to the point of material leaving the farm (Caffrey and Veal, 2013).

Cradle-to-field gate: LCA model which included the processes of a product's life cycle from raw material extraction to the point of field edge (Caffrey and Veal, 2013).

Cradle-to-grave: LCA model which covers all aspects of a product's life cycle (from raw material sourcing to disposal or consumption) (Sieverding et al., 2020).

Cradle-to-gate: LCA model which covers upstream part of the product life cycle (from raw material extraction to product at factory gate) (Baumann and Tillman, 2004).

Cut-off criteria: mode to simplify an LCA model based on the possibility to exclude negligible environmental impacts and upstream production minor components (Baumann and Tillman, 2004).

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Ecosphere system: natural resources used by human (Vadoudi et al., 2017).

Elementary flow: matter or energy entering or leaving the product system under study that has been extracted from the environment without previous human transformation (e.g. timber, water, iron ore, coal) or is emitted or discarded into the environment without subsequent human transformation (e.g. or noise emissions, wastes discarded in nature) (Guinée et al., 2002).

Endpoint level: translating of environmental impacts into issues of concern, such as human health, natural environment and natural resources and it considers environmental impact at the end of the cause-effect chain. It may also be called damage-oriented (Chatzisymeon, et al, 2017).

Environmental burden: substances and energy emitted to soil, air and water by a certain product during its life cycle Environmental burden may be also called environmental load (Solinas et al., 2015).

Environmental impact: consequence of an environmental intervention in the environment system (Guinée et al., 2002).

Environmental intervention: human intervention in the environment, either physical, chemical or biological; in particular resource extraction, emissions (incl. noise and heat) and land use; the term is thus broader than ('elementary flow') (Guinée et al., 2002).

Flowchart: graphic representation of the interlinked unit processes comprising the product system (Guinée et al., 2002).

Foreground system: collection of processes on which measures may be considered regarding their selection or mode of operation as a result of decisions based on the study (Tillman, 2000).

Functional unit: quantitative description of the performance of the product systems, for use as a reference unit (Weidema et al., 2004).

Gate-to-gate: partial LCA considering single processes of the entire production chain (Jiménez-González et al., 2000).

Goal definition: activity aimed to elaborate, define and describe the purpose of the study (Bjørn et al., 2018a).

Grouping: assignment of impact categories into one or more sets by sorting (on a nominal basis, like global/regional/local spatial scale) or ranking (in a given hierarchy, like high/medium/low priority) (ISO 14044, 2006).

Impact category: representation of specific environmental impacts due to emissions or resource use (Greenhut et al., 2013).

Life cycle impact assessment (LCIA): understanding and evaluating the magnitude and significance of the potential environmental impacts for a product system throughout the life cycle of the product (Heijungs and Guinée, 2012).

Life cycle inventory (LCI) analysis: quantification of exchanges between the processes of the product system and the environment (Hauschild, and Huijbregts, 2015.).

Life cycle interpretation: phase of the LCA procedure, in which the results of an LCI or an LCIA, or both, are summarized and discussed as a basis for conclusions, recommendations and decision-making in accordance with the goal and scope definition (ISO 14044, 2006).

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LCA model: description of material flows from raw material extraction from natural resources (i.e., cradle) through production and use to disposal (i.e., grave) (Baumann and Tillman, 2004).

LCA procedure: step-wise process of conducting al LCA study. The LCA procedure consists of the goal and scope definition, the inventory analysis, the impact assessment and the interpretation (Baumann and Tillman, 2004).

Life Cycle Assessment (LCA): objective process aimed to evaluate the environmental burdens associated with a product, process or activity by identifying and quantifying the energy and material uses and releases to the environment (Chau et al., 2015).

Life Cycle Thinking (LCT): a conceptual approach aimed to account for the environmental upstream and downstream benefits and trade-offs of a product (goods and services) considering all stages across the life cycle, from raw material extraction and conversion, through product manufacture, product distribution, use and fate at the end-of-life stage (EC-JRC, 2011b).

Midpoint level: translating of impacts into environmental themes, such as climate change and human toxicity and it considers the impact earlier in the cause-effect chain. It may also be called problem-oriented (Chatzisymeon, et al, 2017).

Normalisation: calculating the magnitude of category indicator results relative to certain reference information (ISO 14044, 2006).

Scope definition: activity aimed to determine what product systems have to be assessed and how this assessment should take place (Bjørn et al., 2018b).

System boundary: choice of processes to be included in the an LCA study (Li et al., 2014).

Technosphere system: global technology systems integrating all human activities (Vadoudi et al., 2017).

Weighting: attributing and possibly aggregating (normalised) indicator results for each impact category to numerical factors based on value-choices (e.g., monetary values, standards, expert panel) (Guinée et al., 2002).

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Appendix: Application of the LCA method to a globe artichoke SI system

Executive summary

The application potential of the LCA tool to evaluate environmental burden of a SI system may be fully described through a case study. Therefore, it was considered as an example the case of a horticultural cropping system based on the globe artichoke (*Cynara cardunculus* L. var. *scolymus*) cultivation at a farm with a SI irrigation system located in Sardinia (Italy).

The LCA analysis was performed according to the four phases developed by the International Organization for Standardization.

1. Goal and scope definition

The reference farm is specialized in the open field cultivation of the local globe artichoke (i.e, cv. Spinoso sardo) whose planting commonly starts from the end of June. The cultivation technique adopted by the farmer reflects the conventional agricultural management used for the globe artichoke in Sardinia, which is essentially based on the forcing technique. This practice consists in spurring the vegetative organs (i.e., offshoots) activity in summer when they are in a natural dormancy status, by means of irrigation. Since Sardinia is characterized by a dry-summer climate, the success of the globe artichoke cultivation subject to the forcing technique depends heavily on irrigation management. In order to avoid the head atrophy fostered by high temperatures along with low air humidity throughout the transition from vegetative phase to reproductive one, the farmer has adopted a precision irrigation technique. It is based on the canopy-cooling treatment able to induce a relevant decline in air or canopy temperature, thus overcoming the negative effects due to high temperatures on crop production capacity. The irrigation water distribution in the canopy-cooling treatment is powered by a photovoltaic system.

In order to plan new investments aimed to obtain an environmentally friendly final product, the farmer needs to know the degree of environmental sustainability achieved by the agricultural management used for the globe artichoke system starting from the impacts due to the SI system. In the farmers' opinion, the SI system is essential for the success of the crop since it has enabled the zeroing of the energy cost for irrigation, a more accurate water distribution, and an improvement of the production supply.

Therefore, the goal of this LCA analysis of which the farmer is the customer and to whom the outcomes will be communicated via a report, is to evaluate the relevance of the potential environmental burden due to the SI system compared to the one caused by the other agricultural practices and inputs used for the globe artichoke cultivation. In order to meet the goal, the LCA analysis was performed by using two functional units as a reference for the inventory building, namely hectare of cultivated land and ton of heads and residual biomass produced. The land- and product-based unit allow to better understand how to improve natural resource use and optimize crop production, respectively, on the basis of the environmental profile of the agricultural practices and production inputs. Since the analysis is focused on the agricultural management of globe artichoke, the boundary system of the LCA analysis includes the unit processes from cradle to field gate, that is from raw material and energy use by neglecting the transport operations of the final product.

The LCA analysis was performed by a specialized software (SimaPro 8.0.4.30) which includes several databases and impact assessment methodologies, which in turn make it easier the

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implementation of an LCA study by enabling to obtain exhaustive and reliable results. Environmental burden of the SI system and agricultural management of globe artichoke cultivation was evaluated by the ReCiPe method which provides impacts and damages associated with a wide variety of environmental subjects (e.g., climate change, eutrophication, and toxicity for humans and ecosystems, etc.) and three areas of protection (i.e., natural resource availability, human health, and ecosystem diversity).

In the farm under consideration, globe artichoke is cultivated mainly for fresh product by burying the biomass residues which in turn is a technical input available for the subsequent growing season. The lack of data with respect to the potential supply of organic matter and nutrients provided by residual biomass of globe artichoke to soil is a limitation of the present LCA study. Furthermore, the canopy-cooling treatment carried out by the SI system does not differentiate between heads and residual biomass, and thus environmental impacts arisen from SI system, as well as for the other agricultural inputs and practices, should be partitioned between heads and residual biomass. For this reason, an allocation method based on physical relationships (i.e., mass) between product (i.e., heads) and by-product (i.e., residual biomass) was used.

Some assumptions were made in the analysis, such as data and information based on scientific literature and provided by the farmer may be on the whole considered representative of globe artichoke cultivation in Sardinia and waste production and treatment is considered as a cut-off criterion since the farmer's requests are not focused on these aspects.

2. Inventory analysis

The agricultural management generally adopted by the farmer for the cultivation of globe artichoke was included in the inventory to support subsequent LCA phases (i.e., impact assessment and interpretation).

The inventory of the crop system was developed via a specific questionnaire which was filled out with the direct farmer's support. He provided the primary data regarding the technical inputs and structural components of the irrigation system, and the agricultural operations that have characterized the management of the globe artichoke system. This information was integrated with secondary data from scientific literature and the processes of the Ecoinvent3 database included in the SimaPro 8.0.4.30 software. These processes related to technical input production (e.g., fertilisers and pesticides), structural components of photovoltaic system and the implementation of mechanical operations such as tillage, and crop maintenance (e.g., fertilization, weeding, etc.), may also content information on the consumption of natural resources, raw material, fuels, and electricity, heat production and chemical emissions to the environment. Furthermore, fertiliser and pesticide emissions as well as loss of phosphorous were included in the corresponding Ecoinvent processes in order to consider in the impact assessment as much the crop system-specific data as possible.

3.Impact assessment

The comparison of the environmental performance between the SI system and the other agricultural practices and production inputs was performed on the basis of impact and damage categories.

The findings on land basis highlighted the positive environmental profile of the SI system in almost all impact categories (17 out of 18 total categories) except for water depletion category in which it was the worst practice because of the water volume distributed by irrigation. Although the SI system did not show the absolute best environmental performance (it was the

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best practice only in two impact categories, that is ozone depletion and terrestrial acidification) it showed much lower impacts than other production input use and agricultural practices in almost all categories. In fact, fertilisers were the most impacting practice in nine categories out of eighteen, such as climate change, eutrophication, human- and eco-toxicity. The second most negative factors were the mechanical operations, in particular tilling affected five impact categories, such as ozone and fossil depletion. The environmental performance of fertilisers and mechanical operations showed no changes as well as the SI system one with regard to the damage categories on production-based unit and for both production scenarios. In fact, N-fertiliser was the worst factor in the human health and ecosystem diversity categories but not in the resource availability one in which the most harmful factor was tilling. On the contrary, SI system showed a low environmental load in all three damage categories as well as occurred in most impact categories. Although the SI practice was little harmful along with hoeing and the pesticide use, it equally affected firstly human health and ecosystem diversity, and finally resource availability. These findings might be due to the production processes of the SI system structural components whose implementation most likely consumes natural resources and releases emissions in the environment.

4.Interpretation

The appropriateness of the definition of the functional unit and system boundary on the basis of the information provided by the farmer and consistent with his requests, the availability of site-specific data collected in the inventory and the databases included in the SimaPro software allowed to meet the goal of the LCA analysis. On the other hand, lack of the upstream processes related to the production of one fertiliser (i.e., NPK-fertiliser) would potentially have changed the final LCA results. Nevertheless, the analysis enabled to emphasize the negative environmental performance of fertilisers and some mechanical operations (e.g., tilling) and the positive one of the SI system within the agricultural management adopted for the artichoke cultivation. The result reliability was supported by an uncertainty analysis, which highlighted that the probability that the difference between SI and the other agricultural practices and inputs in terms of environmental damage may occur is correct 19 times out of 26, that is 73%.

Although the LCA analysis underlined that greater attention should be paid to the fertiliser use and mechanical operations, SI system should not be neglected being responsible for a low environmental load. The adoption of principles and technologies of conservation agriculture and precision might offer effective solution to foster an intensive and environmental sustainable agriculture. Conservation agriculture through the use of cover crops in intercropping may be a good strategy for farmer to diversify production, reduce the synthetic fertiliser application without compromising crop productivity and at the same time prevent environmental damage such as pollution. Despite the farmer has already adopted a precision technique (i.e., the SI system with cooling effect on globe artichoke), the use of new precision technologies such as remote sensing, sensors, and computers along with appropriate software (e.g., auto-steer on tractor) might improve the management of cropping system.

Finally, raw materials realized by recycling or into circular economy processes might be used for the structural components of SI systems in order to further reduce their environmental load.

Introduction

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This appendix provides a practical example of how to perform an LCA analysis of a SI system. Specifically, it focuses on the case of an open field globe artichoke (*Cynara cardunculus* L. var. *scolymus*) cultivation at a farm located in the district of Uri (40° N, 8° E, 150 m asl), in the north-west of Sardinia (Italy) (Figure 19).

Figure 19 - Experimental site of the MASLOWATEN project

The SI system under consideration was implemented as a part of a Horizon 2020 project (MASLOWATEN, MArket uptake of an innovative irrigation Solution based on LOW WATerEnergy Consumption, 2015-2018, Grant number 640771) whose main objective was to develop a new green product consisting of PV pumping systems for productive agriculture irrigation consuming zero conventional electricity and 30% less water.

Goal and scope definition

The farm under consideration is located in an area particularly suited to the cultivation of the local globe artichoke, (i.e, cv. Spinoso sardo). The globe artichoke growth cycle can be poliannual or annual depending on the varietal-type and its suitability to the climatic condition of the growing site, the adopted agricultural practices, and the market demand (La Malfa and Argento, 2009).

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The cultivation technique adopted by this farmer reflects the conventional agricultural management used for the globe artichoke in Sardinia where Spinoso sardo is commonly considered an annual crop system. Its life cycle is between the end of June to 20th July for planting; harvesting occurs from the early of October to January-February for fresh-product and until April for industrial processing. Furthermore, the availability of an early product obtained from the application of the forcing technique may enable farmers to be more competitive on the market. This practice consists in spurring the vegetative organs (i.e., offshoots) activity at a given time, namely in summer when they are in a natural dormancy status, by means of irrigation in order to reach field capacity into the soil layers explored by root system (Pisanu et al., 2009). Nevertheless, high temperatures along with low air humidity throughout the transition from vegetative phase to reproductive one following the first irrigation may lead to the head atrophy (Cantore et al., 2016). Specifically, this process is fostered by temperature higher than 24°C and by a calcium deficiency throughout the flowering phase (Bianco and Calabrese, 2009).

Since Sardinia is characterized by a typical Mediterranean climate, described by Belda et al. (2014) as a subtropical dry-summer climate, it is easily understandable that the success of the globe artichoke cultivation subject to the forcing technique depends heavily on irrigation management.

Furthermore, the intensification of extreme weather events such as heat waves may jeopardize the production potential of a crop and consequently the food security. Although the effect of high temperatures on crops are different according to the species, the plant response depends on its capacity to uptake water from the soil to meet the greater atmospheric water vapour demand. Indeed, its increase induces a greater transpiration by the leaf and a reduction of water supply which in turn, lowers stomatal conductance with a consequent higher leaf temperature and a decrease in photosynthesis (Hatfield and Prueger, 2015).

However, plant responses at morphological and physiological level along with management options (e.g., genetic improvements in combination with the proper cultural practices) may improve crop tolerance to abiotic stress (Fahad et al., 2017). As regards globe artichoke, farmers should face two important challenges: i) the sensitivity to heat stress, particularly throughout the early stages of growth and the hottest daytime hours; ii) the low efficiency of the irrigation water distribution during the summer season.

The adoption of precision irrigation technique, especially the water cooling treatment, may be a winning option to overcome the negative effects due to high temperatures on crop cycle and its production capacity. Furthermore, the potential application of evaporative cooling systems as countermeasure to heat stress were tested in several horticultural species by showing in all cases a relevant decline in air or canopy temperature (Caravia et al., 2017).

Deligios et al. (2019) showed within the MASLOWATEN project, that the canopy-cooling treatment applied to a globe artichoke system cultivated in the farm considered in the present LCA analysis may improve production crop capacity (i.e., + 60% of marketable heads) and water productivity (+ 36%). Specifically, an experiment was conducted during 2017-2018 growing season by comparing the conventional irrigation management with an innovative system characterized by canopy-cooling effect treatment located in two different sectors (2,500 m^2 for each sector) of the used land. The irrigation water distribution in the canopy-cooling treatment is powered by a photovoltaic system installed during the MASLOWATEN project. It provides electricity only used for globe artichoke irrigation and that is able to turn on the conventional

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grid system in case of photovoltaic energy is insufficient to meet the water requirements of the crop.

The combination of the solar energy for irrigation water pumping and high-efficiency water management techniques (i.e., canopy-cooling treatment) on the basis of what highlighted by the scientific evidence above reported and the farmer's statements allowed the zeroing of the energy cost for irrigation, a more accurate water distribution, and an improvement of the production supply. Specifically, the farmer underlined via direct interview that the application of SI system to the globe artichoke cultivation resulted in an increase in the competitiveness on the market and an optimization of water use in environmental limited conditions. In fact, the canopy-cooling treatment enabled to obtain an early production and quantitatively greater (i.e, a significantly higher head number per plant and a lower incidence of atrophic heads) than the production level achieved by the conventional treatment. Furthermore, the possibility of meeting, if necessary, the water demand of the globe artichoke in order to maintain the canopy temperature below the critical threshold for photosynthesis might ensure a better environmental-friendly management of water.

The farmer might consider the opportunity to make new investments (e.g., adoption of innovative agricultural practices to diversify production) as a consequence of the cost saving and the higher income, in order to strengthen the environmental sustainability related to the management of agricultural practices used for the globe artichoke cultivation. From the farmer's point of view, a final product obtained by environmentally friendly cultivation techniques, associated with an Ecolabel or an environmental product declarations, would be more competitive on the market. In fact, an environmental-certified product might meet the growing consumer sensitivity towards the environmental impacts of agro-food products, sensitivity supported by the fact that consumers would be willing to pay a higher price for environmentally friendly agricultural products. Therefore, to plan new investments the farmer needs to know the degree of environmental sustainability achieved by the agricultural management used for the globe artichoke system starting from the impacts due to the SI system.

In the light of the above, the goal of the present LCA analysis of which the farmer is the customer and to whom the outcomes will be communicated via this report is to evaluate the relevance of the potential environmental burden due to the SI system within the agricultural management adopted by the farmer for the globe artichoke cultivation. Specifically, to assess the importance of the SI system from environmental point of view, a comparison between the environmental impacts of SI system and the ones caused by the other agricultural practices (e.g, mechanical operations and use of technical inputs) will be performed.

The pursuit of the goal will enable the farmer to better understand the potential effects of the installing of the SI system, for instance, in terms of natural resources consumption and risk of toxicity for humans and ecosystems within handling of own agricultural activity. This information together with the impact results of the comparison between SI system and the other agricultural practices make it easier to identify "hot spots" of the adopted crop management and develop countermeasures. Although the farmer may use the outcomes of this study at his discretion, the analysis of the environmental load of the SI system may allow him to improve the environmental sustainability of globe artichoke production by better using natural resources and more accurately choosing agricultural practices and technical inputs.

In the present LCA study, two functional units were used: hectare of cultivated land and ton of heads and residual biomass produced. The first unit was chosen because it makes it easier the evaluation of the impacts on the land by highlighting the effects of different external input level

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in terms of potential impact change and also identifying the which impact categories are mostly affected by the agricultural activity (Goglio et al., 2012). Specifically, the land-based functional unit is useful to depict the potential consequences related to the intensity in the use of agricultural inputs in order to optimize processes whereas the production unit helps to identify the optimal level of crop production for instance limited to the intensity of a certain practice (Charles et al., 2006).

In the case of the crop system under consideration, the adoption of the two functional units may be considered the best choice according to the goal of the present analysis. In fact, the impacts due to agricultural practices such as SI system in reference to one hectare of land may provide useful information on potential adjustments to be made to improve the use of agricultural inputs and natural resources.

Furthermore, this functional unit enables to highlight the environmental implications of a certain agricultural management at farm and land scales. Indeed, set-up production inputs and agricultural land allocation play a strategic role in the choices of farmers and thus policy maker decisions that often are land-based. Indeed, cropping system planning is affected by policy guidance at a land scale which in turn should also be developed considering the environmental sustainability of the agricultural management of a crop system to minimize natural resource exploitation and to support farmers in maximizing yield. Using land as functional unit can also allow the identification of a trade-off between environmental burden and productivity that arise from one hectare of land, which might be an added value that enhances overall land management (Solinas et al., 2015).

However, the main objective of the farmer is generally to achieve the maximum yield. The evaluation of the environmental impacts depending on the intensity level with which an agricultural practice is used (e.g., SI system) may provide useful suggestions on what is the optimal level of yield that may be achieved by minimizing environmental burden. Therefore, the application of the mass-based functional unit is considered an added value for the present analysis since it enables to better set-up the agricultural inputs and natural resource use (e.g., irrigation water) on the basis of their environmental performance. Although the product functional unit provides no information on the cost-effectiveness of globe artichoke production, its adoption enables to better understand the environmental implications of the agricultural practices carried out during the crop cycle in referring to the two globe artichoke products (i.e., heads and residual biomass). In other words, the mass-based functional unit may be a useful support for the farmer with respect to the adoption and management of agricultural practices with reference to their environmental profile.

In the present LCA analysis, the adoption of functional units based on land and mass may contribute to obtaining more accurate and reliable results since both are well suited to the multi-functionality and complexity of the globe artichoke system fully in line with the goal of the study.

Since the analysis is focused on the environmental impacts of the SI system and the ones caused by the other agricultural practices throughout the crop production cycle, the adopted boundary system includes the unit processes from cradle to field gate that is from raw material and energy use to the product harvesting by neglecting the transport operations of the final products out of the farm. Hence, the LCA evaluation stopped at the edge of the cultivated field.

As mentioned above, the cultivation technique used by farmer may be considered representative of the agricultural management adopted for the Spinoso sardo artichoke varietal type in Sardinia, especially in the north Sardinia. Indeed, small differences are possible between

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the north and the south of island in terms of yield and agricultural management (e.g. planting and harvesting date) because of a crop performance is highly sensitive to external factors and their variations such as the edaphic-climatic conditions.

Some quantitative and qualitative information on agricultural management, crop performance, temporal coverage, and site location reported in Deligios et al. (2019) are used as reference in the present LCA analysis whereas others are directly provided by the farmer (e.g., agricultural practices and technical inputs). Specifically, the experiment conducted during 2017-2018 in the farm under consideration concerning effects of canopy-cooling treatment on the globe artichoke may be considered the starting point of the LCA study. The availability of a scientific manuscript focusing on the same context and horticultural species as the subject of this analysis strengthens precision, completeness, and representativeness of the data and reduce uncertainty of information. Further details necessary for the inventory building (e.g., description of the characteristics of mechanical operations and technical components of irrigation system) were provided by the farmer. Databases were used to collect information on background system such as raw material consumption and emissions related to the production of technical inputs and structural components.

The LCA analysis was performed by a specialized software (SimaPro 8.0.4.30) which is one of the most tool internationally used (Goedkoop et al., 2013a, 2016). It includes several databases and impact assessment methodologies which make it easier the implementation of an LCA study by enabling to obtain exhaustive and reliable results. Environmental burden of the SI system and agricultural management of globe artichoke cultivation was evaluated by the ReCiPe method which provides impacts associated with a wide variety of environmental subjects and areas of protection (Goedkoop et al., 2013b). Therefore, this method may fully meet the goal of the study and farmer's requests aimed to obtain all information useful to support possible changes/adjustments in terms of adopted technologies (i.e., SI system) and agricultural practice management on the whole. Specifically, the Recipe method was applied by considering an egalitarian perspective both at midpoint and endpoint level. It is the most precautionary perspective and considers the longest time-frame, impact types that are not yet fully established but for which some indication is available (Huijbregts et al., 2016). Therefore, this approach enables to evaluate effects on future generations.

It is important to note that the use of databases including information at global and national level and methodologies suitable for different contexts together with the site-specific data would enable other practitioners to reproduce the outcomes of the study. However, the crop performance is highly sensitive to variation of factors often hardly manageable or predictable such as edaphic and climatic ones. The influence of these factors on the crop system performance may in turn affect the agricultural practice management such as irrigation schedule and consequently the environmental burden of SI system. Therefore, considering the strategic role of temperature on the production capacity of the globe artichoke, temperature changes, especially during summer may be considered a limitation of the present LCA study. In the light of this aspect, it is an evidence the usefulness of performing an uncertainty analysis of the results in order to interpret them at their best.

Although the globe artichoke cultivation in Sardinia is mainly aimed at food supply chain through harvesting of fresh product (i.e., heads), it may be deemed a dual-purpose crop since at the end of crop cycle the biomass residues (i.e., leaves, stalks and unmarketable heads traditionally burned or buried) might be used as raw material in the bioenergy supply chain, thus representing a valuable opportunity for farmer to improve its income (De Menna et al., 2016).

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In the farm under consideration, globe artichoke is cultivated each year mainly for fresh product burying the biomass residues which in turn is a technical input available for the subsequent growing season.

The lack of data with respect to the potential supply of organic matter and nutrients provided by residual biomass of globe artichoke to soil is a limitation of the present LCA study. However, the canopy-cooling treatment carried out by SI system does not differentiate between head and residual biomass. In other words, both products share the same practice (i.e., canopy-cooling treatment) and process (SI system). Therefore, environmental impacts arisen from SI system, as well as for the other agricultural practices, should be partitioned between heads and residual biomass. For this reason, the present analysis applied an allocation method based on physical relationships (i.e., mass) between product and by-product. Furthermore, the allocation procedure may be useful to provide the farmer information in the case of he decides to cultivate the globe artichoke with a dual production purpose (i.e., food and energy).

The present analysis is based on the following assumptions: i) data and information based on scientific literature and provided by the farmer may be on the whole considered representative of globe artichoke cultivation in Sardinia; ii) waste production and treatment is considered as a cut-off criterion since the farmer's requests are not focused on these aspects, therefore the environmental impacts due to waste are not included in the evaluation; and iii) energy used to perform the canopy-cooling treatment by irrigation is provided by photovoltaic system; iv) optimal phytosanitary condition of propagation material used for planting has ensured the absence of failures or a so low percentage that it may be neglected.

Inventory analysis

Business as usual agricultural management adopted by the farmer over the life cycle of globe artichoke was included in the inventory to support subsequent LCA phases (i.e., impact assessment and interpretation) (Figure 20). The cultivation was handled on the basis of the traditional forcing techniques by including the forcing one, adopted in Sardinia for the globe artichoke. Specifically, before proceeding with the new planting, a tilling was performed to incorporate residues of previous crop system. Soil tillage was carried out by a two-furrow plough which was followed by an irrigation so that the soil water content achieved field capacity that is the optimal condition to foster the globe artichoke growth. Planting occurred in late June with 10-cm-long semi-dormant offshoots of Spinoso sardo artichoke varietal type by a transplanter coupled with a 4WD-58 kW tractor to ensure a 0.70 m row spacing necessary to achieve a plant density of 9,500 plants ha^{-1}.

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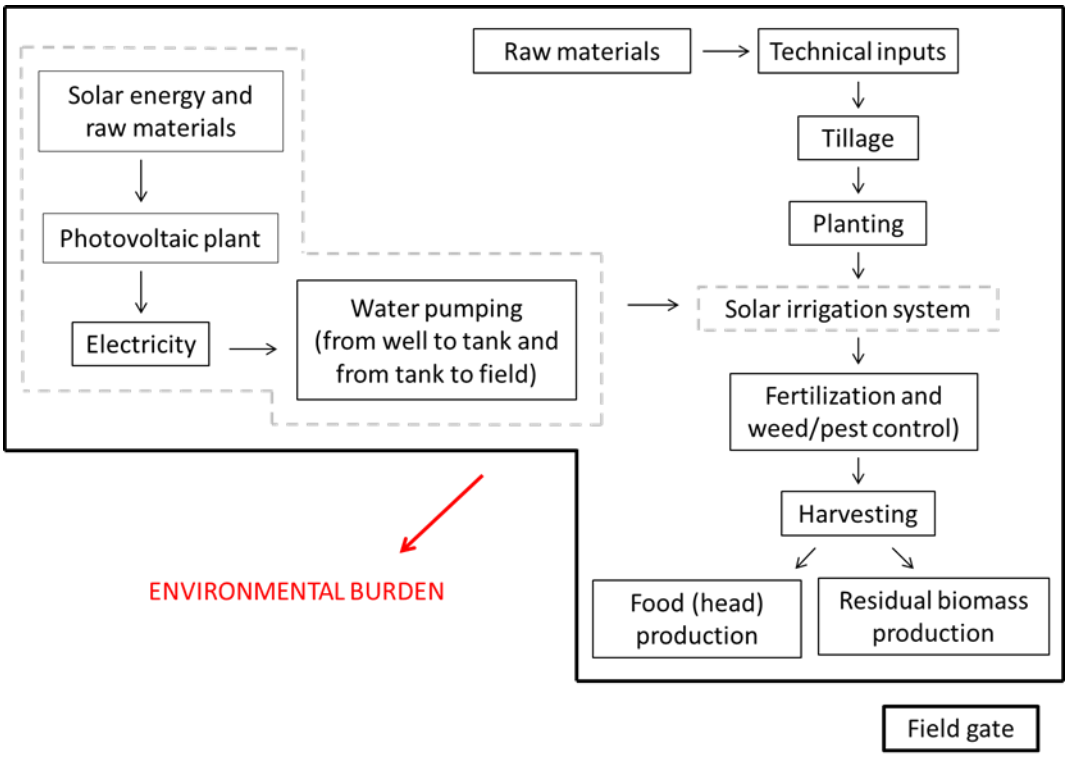

Figure 20 - Flow chart of analysed processes

The same tractor was coupled with a hoeing machine to incorporate weed residues following the herbicide treatment applied by means of irrigation over the globe artichoke growth. It was also used together with a spreader to distribute the fertiliser, which since in granular form was subsequently incorporated into the soil by a motor hoe. Total N, P, and K (150, 80, and 100 kg ha^{-1}, respectively) were applied in four split doses at planting, mid-September, late-November, and late-February The hand harvesting of globe artichoke heads was performed every 5 to 7 days from 29 September 2017 to 31 March 2018.

The irrigation water distribution in the canopy-cooling treatment is powered by a photovoltaic system (40 kWp) installed during the MASLOWATEN project. It is responsible for water withdrawal from two wells of approximately 90 meters deep both with two pumps (3 and 18.5 kW) and for water carrying from collection tank to field by means of a third pump (5.5 kW) The consumption of photovoltaic energy to ensure the operation of the low pressure micro-sprinkler irrigation system during the globe artichoke cultivation (135 kWh ha^{-1} yr^{-1}) was provided by the Universidad Politécnica de Madrid (UPM).

As reported by Deligios et al. (2019) the irrigation system and its schedule based on canopy-cooling treatment is characterized by the installation of a low pressure micro sprinkler system and by a sensor set located in different irrigated farm sectors identified on the basis of a soil spatial variability study and whose purpose was to monitor temperature and moisture conditions at three different depths (10, 20 and 30 cm). A self-operating central control unit is responsible for the system management since it communicates with a weather station, wireless transceiver modules for irrigation valves control and with soil moisture sensors located at the different depths in each sector. After planting, daily irrigation for soil moisture restoration began when soil moisture was less than 35% of the available water volume in the first 10 cm and it

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ended before the soil moisture value at 30 cm did not cause deep percolation due to the presence of over-irrigation condition.

Daily cooling irrigation started at 45 days after planting and ended at 105 days after planting and it occurred whenever the air temperature exceeded 25°C during daylight hours with an average daily duration of about 6 h. The handling of system was setting to enable a rotation of the cooling events among sectors and each cooling-event occurred in each sector with a 15-minute of duration. The duration of each cooling event was chosen considering the time it takes the plant to enter underworking pressure, about 1–2 min, and to ensure a homogenous leaf wetting. The choice was also made by assessing the time need by leaf surface to go from wet to dry and then back to wet.

The same authors also highlighted that the combination of evaporative cooling practice with precision irrigation technique may optimize the head yield in addition to result in a water saving (-34%) compared to conventional treatment based on drip irrigation system. Nevertheless, the season water volume used in the canopy-cooling treatment and considered in the evaluation of SI system impacts, was just below the one of the conventional irrigation (5,880 and 6,050 m^3 ha^{-1}, respectively).

The inventory of the crop system was filled out via a specific questionnaire with the direct farmer's support who provided the primary data useful to describe the technical inputs and structural components of the irrigation system, and the agricultural operations that have characterized the management of the globe artichoke cultivation (Table 12).

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Table 12 - The questionnaire used for inventory building

AGRICULTURAL INVENTORY	
General information	Farm name:
	Location:
	Crop system:
	Soil type:
	Soil pH:
	Cultivated area:
	Growing season:
	Sowing time:
	Harvesting time:
	Rotation:
	Crop residue incorporation:
	Seeding material:
	Plant density:
	Harvested product:
	Yield:
Crop operations	Tractor (brand, weight, model, year, power, lifetime):
	Agricultural machinery (brand, weight, model, year, power, lifetime):
	Type of operation (n. of operations/ha, fuel consumption, working time, working speed, working width):
Fertilizers	Type of fertilizer (synthetic or organic):
	Name:
	Application time:
	Incorporation:
	Quantity:
	Number of application/ha:
	N, P, K content:
	Dry matter (only for organic fertilizer):
Pesticides	Commercial name:
	Active principle:
	Quantity of active principle distributed in each category:
	Category (e.g., herbicide):
	Application time:
	Number of application/ha:

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Table 12 - The questionnaire used for inventory building

AGRICULTURAL INVENTORY	
Irrigation system	Water source (e.g., well):
	Well depth:
	Well flow:
	Pump from well to tank and from tank to field (number of pumps, for each pump: lifetime, power, materials, weight, flow rate):
	Tank (dimension, material and its quantity, usage time, lifetime, water capacity):
	Irrigation system type (e.g., sprinkler)
	Cabin (dimension, material and its quantity, usage time, lifetime):
	Valves (weight of one valve, material, number of valves, usage time, lifetime):
	Filters (weight of one filter, material, number of filters, usage time, lifetime):
	Sprinklers (quantity, material, weight of one sprinkler, usage time, lifetime):
	Rods (quantity, material, weight of one rod, usage time, lifetime):
	Energy consumption:
	Seasonal irrigation water volume:
	Irrigation water quantity for each event:
	Number of irrigation events:
	Irrigation interval:
	Water quantity drawn from well:
Photovoltaic system	Photovoltaic power:
	Area occupied by PV system:
	PV panel type:

Since the information were not exhaustive, they were integrated with secondary data from scientific literature and international databases, primarily the Ecoinvent3 database. In the present study, this database which is included in the SimaPro 8.0.4.30 software, was used in order to encompass processes regarding technical input production (e.g., fertilisers and pesticides,) structural components of photovoltaic system and the implementation of mechanical operations such as tillage and crop maintenance (e.g., fertilization, weeding, etc.), in the evaluation phase. Specifically, these processes which may refer to national, European or global context, may content information on the consumption of natural resources, raw material, fuels, and electricity as well as heat production and chemical emissions to the environment.

Specifically, the Ecoinvent database has enabled to include in the inventory processes useful to describe the structural components of the photovoltaic system. such as converter, battery, cable, breaker, power meter, inverter, and PV panel type.

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The nitrogen field emissions due to fertiliser application (i.e., ammonia (NH_3) and nitrous oxide (N_2O) in air and nitrate in water (NO_3^-) were evaluated using the Estimation of Fertilisers Emissions Software (EFE-So) (Fusi and Bacenetti, 2015) which in turn refers to the model developed by Bentrup et al. (2000). Specifically, this software allows to obtain more accurate emission values than other methods since it requires various site-specific data to contextualize the fertiliser application such as fertiliser type, soil characteristics, climate context (e.g., air temperature during distribution, summer and winter precipitation) as well as the N content in the harvested crop and its coproducts (Schmidt Rivera et al., 2017).

Losses of phosphorous were assessed on the basis of the Product Category Rules (PCR) developed to evaluate the environmental performance of arable crops and vegetables (EPD, 2020). Pesticide emissions in air, soil, and surface or ground waters were calculated by using according to the percentages provided by Schmidt Rivera et al. (2017).

These data as well as ones on technical inputs and mechanical operations were included in the corresponding Ecoinvent processes contained in the SimaPro software in order to consider in the impact assessment as much the crop system-specific data as possible.

Impact assessment

The evaluation of the environmental burden associated with a globe artichoke system characterized by the SI practice was carried out by the Recipe method. It may provide exhaustive results since it works at both midpoint and endpoint level. In the first case, a stressor (e.g., the effect of CO_2 emission) is converted in environmental subjects (e.g., climate change) and it considers the impact earlier in the cause-effect chain, namely prior to the endpoint. At this level that is at the end of the cause effect chain, an environmental impact is translated into issues of concern, such as human health, ecosystems and natural resources.

Therefore, two different sets of findings were calculated to provide detailed information on the environmental burdens caused by the agricultural practice management adopted for the globe artichoke cultivation. The first group consists in all impact categories at a midpoint level, which is the total amount of substance-equivalent released or resource-equivalent consumed. Both measures are classified into the environmental themes to which they potentially contribute. The latter group identifies the potential environmental damages derived from the emissions and resources depletion at the endpoint level, namely, certain vulnerable targets (e.g., human health, ecosystems and natural resources).

Considering that generally midpoint results are characterized by a lower uncertainty but provide a more detailed information since they refer to a small part of the analysed environmental pathway, the land-based functional unit was used for this evaluation. The results referred to one hectare of cultivated land enable to better highlight the environmental impacts caused by each agricultural practice with particular attention with the relevance of SI system compared to the effects of the other practices without distinction between head and residual biomass production. In other words, this functional unit allowed to perform an evaluation aimed to show the environmental impacts of each agricultural practices irrespective of crop production performance.

However, endpoint indicators even though characterized by a higher uncertainty level make it easier the understanding and interpretation of results by stakeholders also because they refer to few environmental issues (i.e., damage categories) and thus facilitate the result

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representation. Therefore, this evaluation was performed with respect to the mass-based functional unit and by taking into account the two different products obtained from the globe artichoke system through the allocation procedure. In other words, the assessment of damage categories enabled to provide the farmer findings that are easy and fast to read but reliable so as to meet the goal of the present LCA study and the farmer's requests strictly connected to optimize the production. The application of the Recipe methodology built on the harmonization of midpoint and endpoint approach enabled to obtain effective information to support the farmer's decisions.

Impact category assessment

The findings on land basis obtained by the characterization phase showed a high variability (from a little more 0.00004 in ozone depletion (OD) to more 8,500 in water depletion (WD)) although they are expressed on the basis of different substance equivalents (Figure 21 and Table 13). In other words, the WD category exceeds by approximately 200 billion times OD by highlighting an extremely different environmental behaviour of agricultural practices and technical inputs within impact categories (Figures 22, 23, and 24). In fact, solar irrigation was the main responsible for the high impact shown by WD whereas tilling had the worst impact on OD. Furthermore, all other agricultural practices and technical inputs except for solar irrigation showed a minimal effect on WD (the second highest value is 0.68 m^3 due to N-fertiliser) whereas the OD category was especially affected by two mechanical operations (i.e., ploughing and weeding, 61% and 50% of the tilling impact, respectively) and one technical input (i.e., N-fertiliser (54%)) (Figures 22 and 24).

The NPK-fertiliser and tilling played a relevant role in the climate change (CC) category with a contribution equal to 81% and 77%, respectively compared to the worst performance due to N-fertiliser (Figure 21 and 22). Ploughing, weeding, and planting did not provide a contribution greater than 50% (48%, 40%, and 33%, respectively). On the other hand, pesticide affected CC only for 1% and SI system did not exceed 5% of the N-fertiliser impact. In other words, the two fertilisers and all mechanical operations together, except for hoeing, have contributed to the CC category with 96%.

The NPK-fertiliser demonstrated the worst impact both in the terrestrial acidification (TA) and freshwater eutrophication (FE) category in contrast to the contribution of N-fertiliser (Figure 21 and 22). Although it had the second worst environmental performance for the TA category, its contribution was only 8% with respect to the NPK- fertiliser impact and even was practically zero in FE. Among mechanical operations, tilling, ploughing, and weeding were the most impacting mechanical operations in the TA category even though the highest environmental performance related to tilling did not exceed 7% of the NPK-fertiliser one. The high environmental burden caused by the NPK-fertiliser might be due to the presence of sulfur and nitrogen compound in the fertilisers used for the globe artichoke cultivation since acidification in terrestrial ecosystems is mainly caused by the atmospheric deposition of inorganic substances on Earth's surface, mainly oxides of sulfur (e.g., SO_2) and nitrogen (e.g., NO_x). The impact of the SI system on the TA category may be considered negligible compared to the environmental performance of the NPK fertiliser, which showed a contribution almost 679 times higher than the irrigation practice one.

The two fertilisers were the main responsible for FE and marine eutrophication (ME), although they demonstrated a different environmental performance within the two impact categories (Figure 21 and 22). As reported above, the NPK-fertiliser had the highest impact on FE, but its

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performance was different in the ME category in which, although showing the second worst environmental burden, it was exceeded by the N-fertiliser by 5%. The results of both impact categories highlighted the relevance of N- and P-compounds as key factors of eutrophication phenomenon. The little difference in terms of environmental incidence might have been caused by fertiliser production process and faster by N movement through runoff and leaching than the P one.

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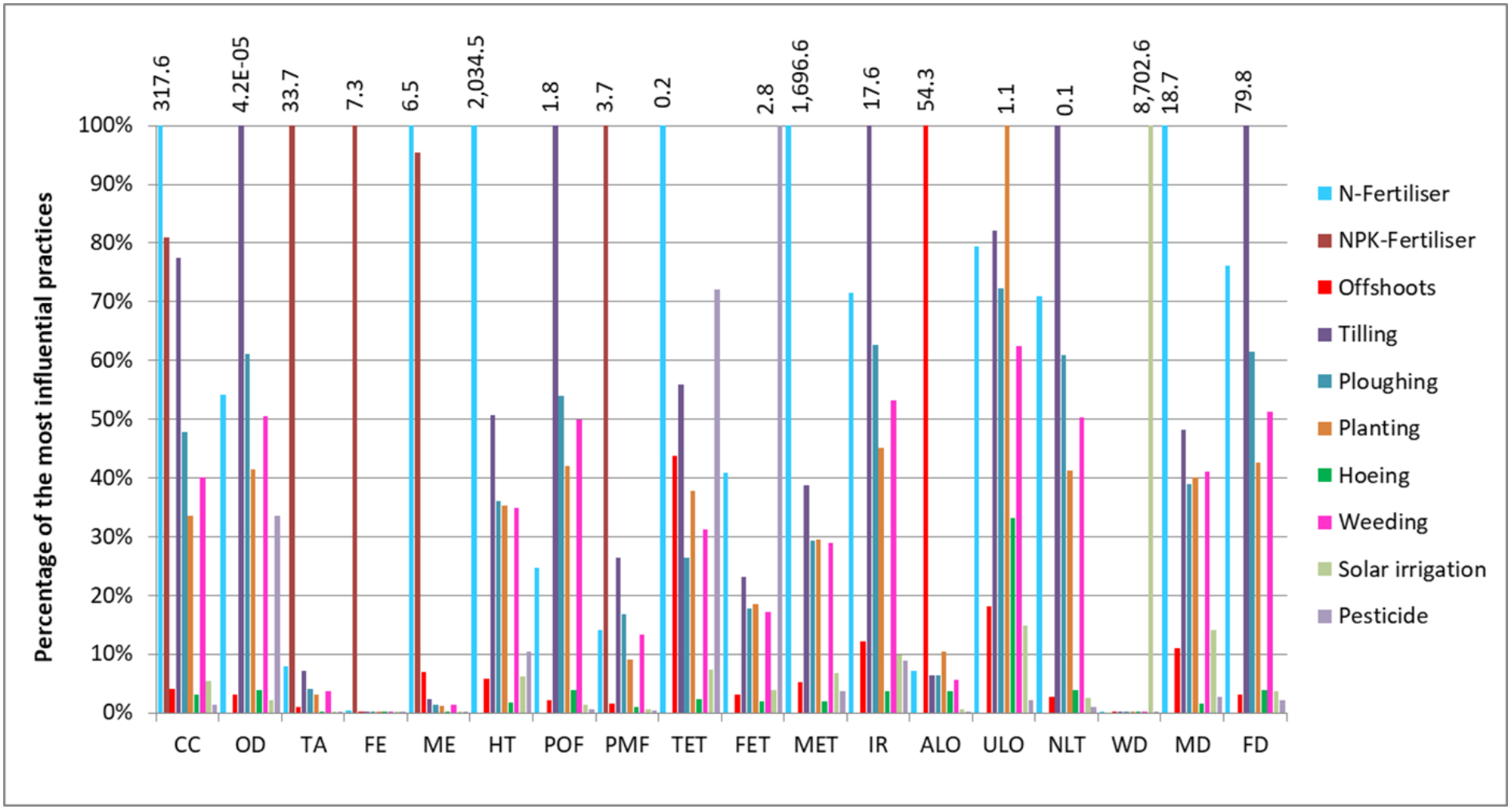

Figure 21 - Characterization of the most impacting factor on land basis (ReCiPe method). The values are expressed as percentage of the most impacting factor in each category. The standardized values in kg of substance-equivalents for all impact categories except for ALO and ULO (m2a), NLT (m2) and WD (m3) are reported on top of the histograms; the absolute values are referred to the impacts of the most impacting factor. See Table 13 for abbreviation details.

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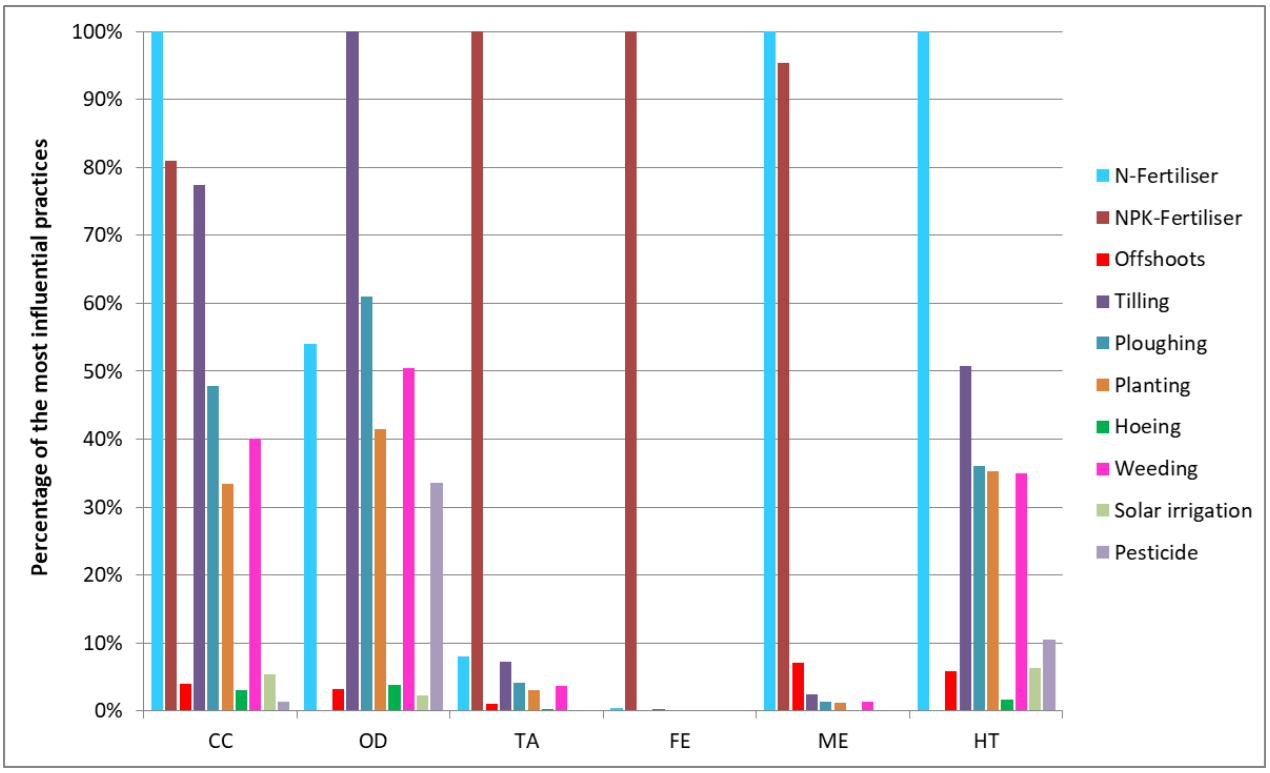

Figure 22 - Details about characterization of the most impacting factor on land basis (ReCiPe method)

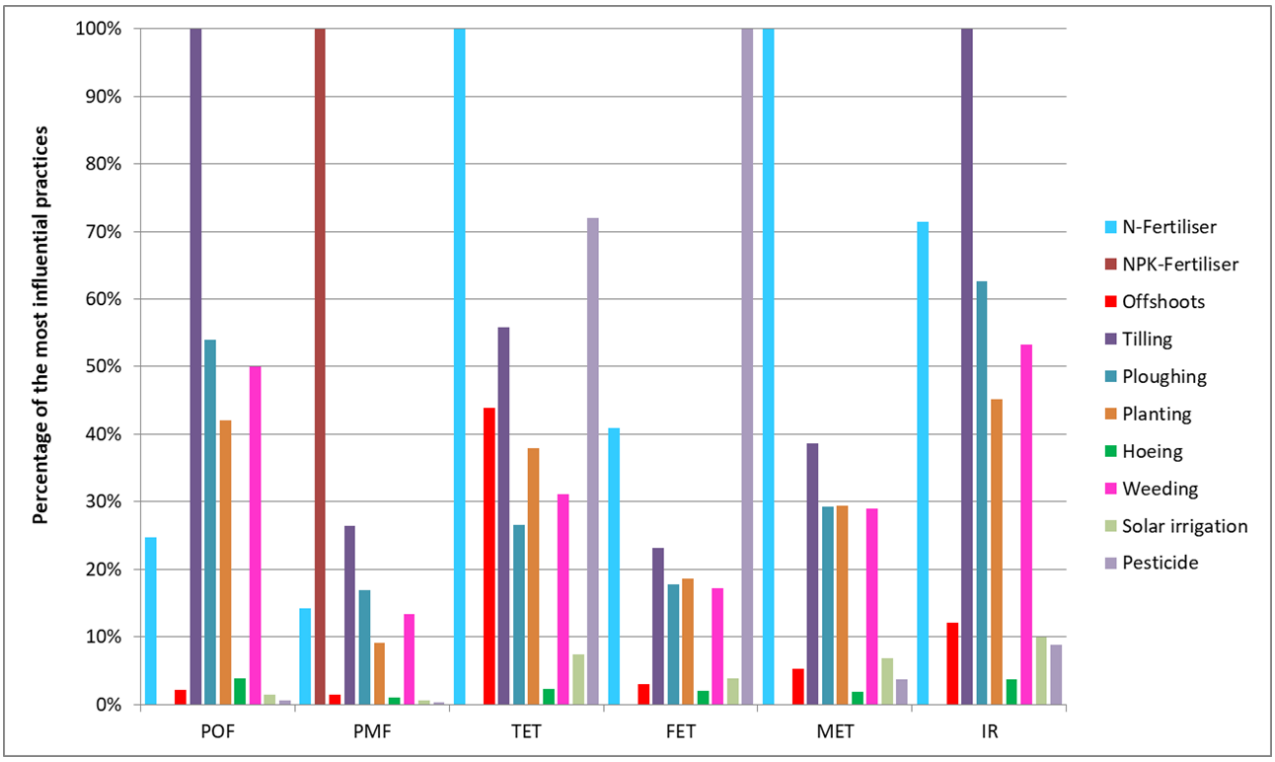

Figure 23 - Details about characterization of the most impacting factor on land basis (ReCiPe method)

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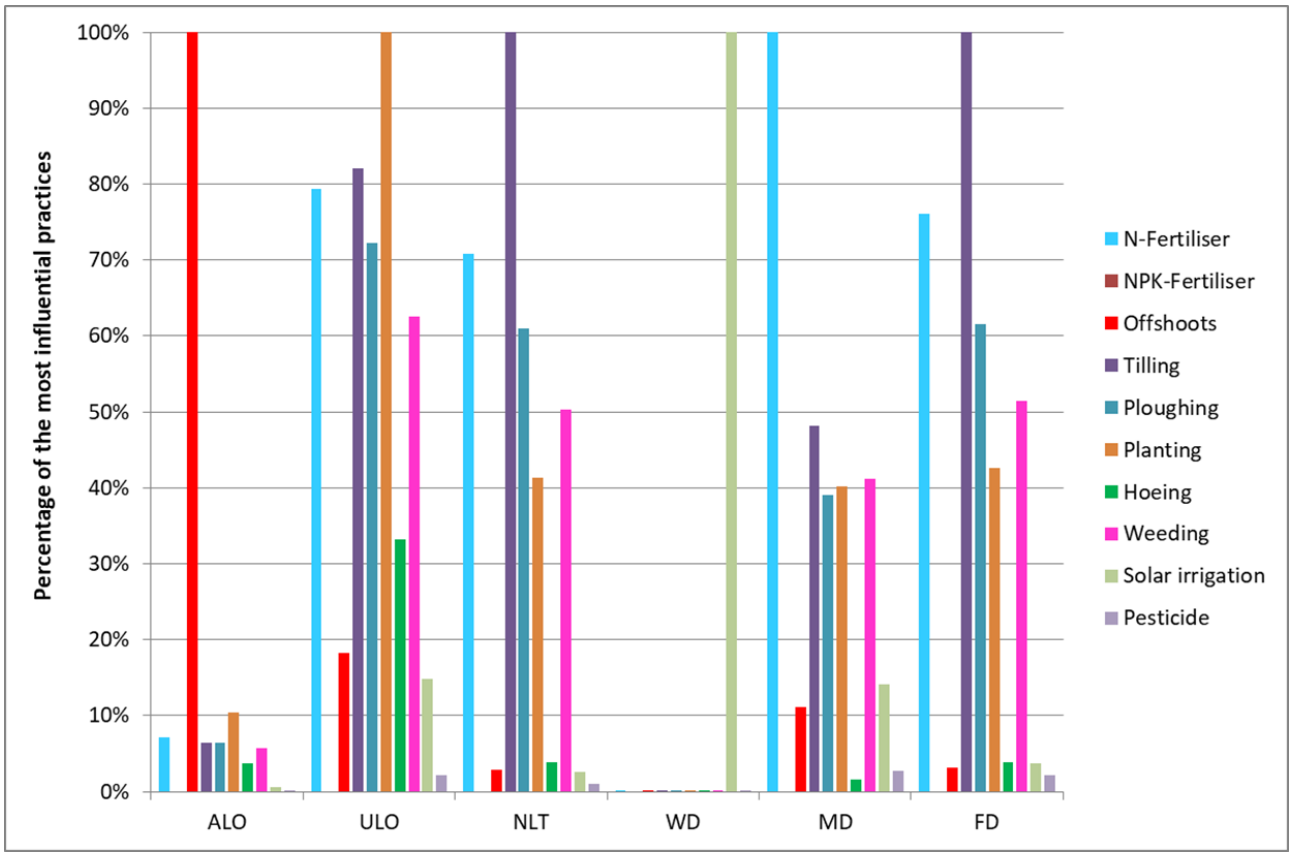

Figure 24 - Details about characterization of the most impacting factor on land basis (ReCiPe method)

Table 13 - The main impact categories based on Recipe method (Goedkoop et al., 2013b)

Impact category	Abbreviations	Unit-equivalent
Climate change	CC	kg CO_2 eq
Ozone depletion	OD	kg CFC-11 eq
Terrestrial acidification	TA	kg SO_2 eq
Freshwater eutrophication	FE	kg P eq
Marine eutrophication	ME	kg N eq
Human toxicity	HT	kg 1,4-DB eq
Photochemical oxidant formation	POF	kg NMVOC
Particulate matter formation	PMF	kg PM10 eq
Terrestrial ecotoxicity	TET	kg 1,4 DB eq
Freshwater ecotoxicity	FET	kg 1,4 DB eq
Marine ecotoxicity	MET	kg 1,4 DB eq
Ionising radiation	IR	kBq U235 eq
Agricultural land occupation	ALO	m^2a
Urban land occupation	ULO	m^2a
Natural land transformation	NLT	m^2
Water depletion	WD	m^3
Metal depletion	MD	kg Fe eq
Fossil depletion	FD	kg oil eq

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The offshoot production showed the third worst environmental impact in the ME category although its incidence was much lower than the first (N-fertiliser) and the second worst environmental performance (NPK-fertiliser) (i.e., 0.45 vs 6.45 and 6.15 kg N eq, respectively). On the other hand, the environmental contribution of the SI system on ME may be considered negligible (0.01 kg N eq).

The human toxicity (HT) category was particularly affected by N-fertiliser use, likely because of impacts of the different upstream production processes and the degree of mobility of fertiliser in the environment (Figure 21 and 22). The second worst environmental performance was provided by some mechanical operations which showed values ranging from 51% related to tilling to 35% due to planting and weeding. The pesticide incidence on HT was 10 times lower than the worst factor likely because of its half-life and movement in the environment. Also in this impact category, SI system did not have a very high effect, only 6% (the same percentage of offshoot production) with respect to N-fertiliser likely due to production processes of technical components and propagation material.

Environmental burden on the photochemical oxidant formation (POF) category was consistent with the OD trend highlighting the importance of ozone concentration for by proving the relationship between POF and OD depending on the ozone concentration especially near Earth's surface (Figures 21 and 23). In the POF category, some mechanical operations (i.e., tilling, ploughing, weeding, and planting) were the main responsible for the worst environmental performance as well as in OD. In fact, tilling showed the highest impact in both categories, followed by ploughing, weeding, and planting with an incidence on land basis equal to 54%, 50%, and 42% in the POF category with respect to the worst mechanical operation. On the other hand, hoeing showed an environmental performance quite different from the other mechanical operations with an impact equal to 4% with respect tilling likely resulting from the specific characteristics of the machine (e.g., weight). Furthermore, this operation was performed by only one machinery in contrast to the other mechanical operations were performed by tractor plus operational machinery. This condition might contribute to reduce the environmental load of hoeing compared to the other mechanical operations in all impact categories. Among technical inputs, pesticide showed the most negative environmental impact but much lower than tilling, and the same contribution of SI system that is 1%.

The particulate matter formation (PMF) category was mainly affected by NPK-fertiliser which had the most negative environmental impact (Figures 21 and 23). Tilling was the second worst performance although its impact was almost 4 times less than the NPK-fertiliser one. Other mechanical operations that have had effects on PMF were ploughing and weeding with an incidence equal to 17% and 13%, respectively with respect to the NPK-fertiliser. The N-fertiliser performance was placed in an intermediate position between the two previous mechanical operations (i.e., 14% compared to the highest impact factor). Considering that PMF is generally caused by emissions from road transport, power plants, or farming activities is easily understood the SI performance which had an impact equal to 1% that is 173 times lower than the NPK-fertiliser one.

In the ecotoxicity impact categories, namely terrestrial, freshwater, and marine ecotoxicity (TET, FET, and MET, respectively) the N-fertiliser played a relevant role in terms of environmental burden since it showed the highest value (TET and MET) and the second one (FET) on land basis (Figures 21 and 23). Other negative impacts were primarily caused by mechanical operations although with different incidence within the three categories except for TET category in which offshoot production showed the third highest negative environmental impact whereas it

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showed a low impact (3% and 5% in FET and MET, respectively) compared to the worst factor (i.e., pesticide and N-fertiliser in FET and MET, respectively). The SI system was the second best performance in all impact categories (i.e., 7%, 4%, and 7% in TET, FET, and MET, respectively) with respect to the worst factor exceeded only by hoeing (2% in all categories). Specifically, TET was affected by tilling, offshoot production, planting, and weeding with a contribution ranging from 56% (tilling) to 31% (weeding) with respect to the impact of N-fertiliser. Pesticide use was the second worst factor (72%) with respect to the N-fertiliser impact in TET whereas it showed the worst environmental performance in FET in which the roles of N-fertiliser and Pesticide were reversed. In fact, N-fertiliser contributed 41% compared to the highest impact factor. Tilling was the third worst environmental performance (57% and 23% with respect to N-fertiliser and pesticide, respectively) and the first most impacting mechanical operation in FET since the others showed an impact ranging from 19% (planting) to 17% (weeding) except for hoeing (2%) compared to N-fertiliser.

In MET category, the incidence of the previous mechanical operations was slightly different since tilling was remained the most impacting practice (39%) but planting, ploughing, and weeding had the same contribution (29%) compared to the worst environmental performance (i.e., N-fertiliser). Although FET and MET categories concern water ecosystems, the main responsible for toxicity were different for the two categories, namely pesticide and N-fertiliser, respectively. The different environmental performance might be due to a potential higher persistency of pesticide in the freshwater ecosystem compared to the N-fertiliser one which might have achieved faster the marine ecosystems via runoff and leaching. Furthermore, the environmental burden was likely affected by a different incidence of production process of the pesticide and N-fertiliser.

In the impact category of ionizing radiation (IR) the most negative performance was provided by tilling followed by N-fertiliser (71%) (Figures 21 and 23). The other mechanical operations showed values ranging from 63% for ploughing to 4% for hoeing, which had the lowest environmental burden, with respect to the tilling impact. The SI system had an impact 10 times less than the worst factor quite similar to the contribution of offshoot production and pesticide (i.e., 12% and 9%, respectively). The IR category is generally related to the release of radioactive material to the environment coming from such as the burning of coal and the extraction of phosphate rock. Therefore, the environmental burden caused by the different inputs and practices used for the globe artichoke cultivation might be likely resulted from the various extraction process of raw material used in the production of inputs and technical components of the agricultural practices applied to the cropping system.

The agricultural land occupation (ALO) category was mainly affected by the offshoot production, which may come from the cultivation of another globe artichoke system which in turn occupies additional agricultural land (Figures 21 and 24). For this reason, technical inputs and agricultural practices showed an environmental performance much lower than the worst factor in ALO. The second most negative impact was caused by planting (10%) followed by the mechanical operation one ranging from 6% (ploughing, tilling, and weeding) to 4% (hoeing) with respect to offshoot production. Pesticide and SI system showed the first and the second best environmental performance, respectively, with a contribution equal to 604 and 185 times less, respectively, than the worst factor. The environmental burden of the SI system might be mainly due to the surface of agricultural land occupied by photovoltaic plant.

The trend of urban land occupation (ULO) and natural land transformation (NLT) categories had a low incidence in absolute terms (1.1 m^2 a and 0.1 m^2, respectively) categories as well as the

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POF and TET (Figures 21 and 24). Nevertheless, planting and tilling were the most environmental impacting practices in ULO and NLT, respectively. The use of N-fertiliser and mechanical operations were the most relevant factors in both categories even though with a different environmental contribution. Specifically, the incidences of the second (tilling) and the third (N-fertiliser) worst factor were equal to 82% and 79%, respectively, compared to planting in the ULO category. In this category, hoeing reached the highest value (33% with respect to the most negative factor), with respect to its performance in the other impact categories in which it had never exceeded 4% of the corresponding worst factor. In NLT, N-fertiliser showed an incidence equal to 71% followed by the environmental impact of ploughing, weeding (61% and 50%, respectively) with respect to tilling. The performance showed by the previous inputs and practices in the ULO and NLT impact categories might be due to the upstream processes (e.g., raw material production and natural resources extraction). On the other hand, pesticide and SI system had the lowest environmental impact in both categories, namely 2% and 1% for pesticide and 15% and 3% for SI system in ULO and NLT, respectively.

As reported at the beginning of this paragraph, the SI system had the worst impact in the WD category in which its contribution achieved 8,700 m3 (Figures 21 and 24). This environmental performance was especially due to the implementation of irrigation practice which is the main factor responsible for water resource consumption. In fact, the environmental impact caused by the other inputs and agricultural practices showed an order of magnitude much lower than the SI system one (i.e., from 0.7 m^3 for N-fertiliser to 0.01 m^3 for hoeing) that expressed in percentage was equal to 0% compared to the SI system.

N-fertiliser and tilling were the worst factors in the metal depletion (MD) and fossil depletion (FD) categories even though their role was reversed in the two categories (Figures 21 and 24). In fact, N-fertiliser showed the most negative environmental performance in MD and tilling was the second worst factor (48%) whereas the opposite occurred in FD in which N-fertiliser showed an environmental impact equal to 76% compared to the tilling one. Both MD and FD were also affected by other mechanical operations by highlighting a not too low consumption of metal and fossil resources for upstream processes. In FD, the impacts range from 62% for ploughing to 43% for planting compared to tilling. In MD, the impact of N-fertiliser was 2 and 3 times higher than weeding and ploughing, respectively. The environmental performance of the SI system was not very negative in both categories (14% and 4% in MD and FD, respectively) although the most positive impact was provided by hoeing (2% in MD) and pesticide (2% in FD) with respect to N-fertiliser and tilling, respectively.

The estimated impacts normalised on the total emissions/consumption level of EU inhabitants are reported in Figure 25 and in more detail in Figure 26, 27, and 28 (expressed in inhabitant equivalent). Essentially, the ranking of the analysed factors reflects the distribution of the characterization phase. In other words, the normalisation phase confirmed the worst and the best environmental performance found in the characterization phase, although the order of magnitude of normalized values was considerably different among the categories (i.e., ranging from 0.00007 for hoeing in OD to 17,6 for NPK-fertiliser in FE). However, the environmental performance in percentage terms for each considered factor was the same as the values obtained in the characterization phase except for the WD category. As regards WD, the Recipe method do not provide normalisation values for water use (Benini et al., 2014).

This result is useful to identify the most relevant categories from the environmental point of view by making it easier to understand which technical inputs and/or agricultural practices adopted for the globe artichoke cultivation were more responsible for natural resources

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consumption and emission production. ALO and NLT were the first best and worst impact category, respectively, namely characterized by the lowest (with pesticide, 0.00002 EU inhabitant eq) and the highest (with tilling, 0.5 EU inhabitant eq) environmental performance. The SI system showed the lowest value in the OD category (0.00004 EU inhabitant eq) in which it was also the best factor in terms of environmental impact. In MET, it had the most negative performance (0.05 EU inhabitant eq), although this was not the worst impact which instead was caused by N-fertiliser (0.07 EU inhabitant eq).

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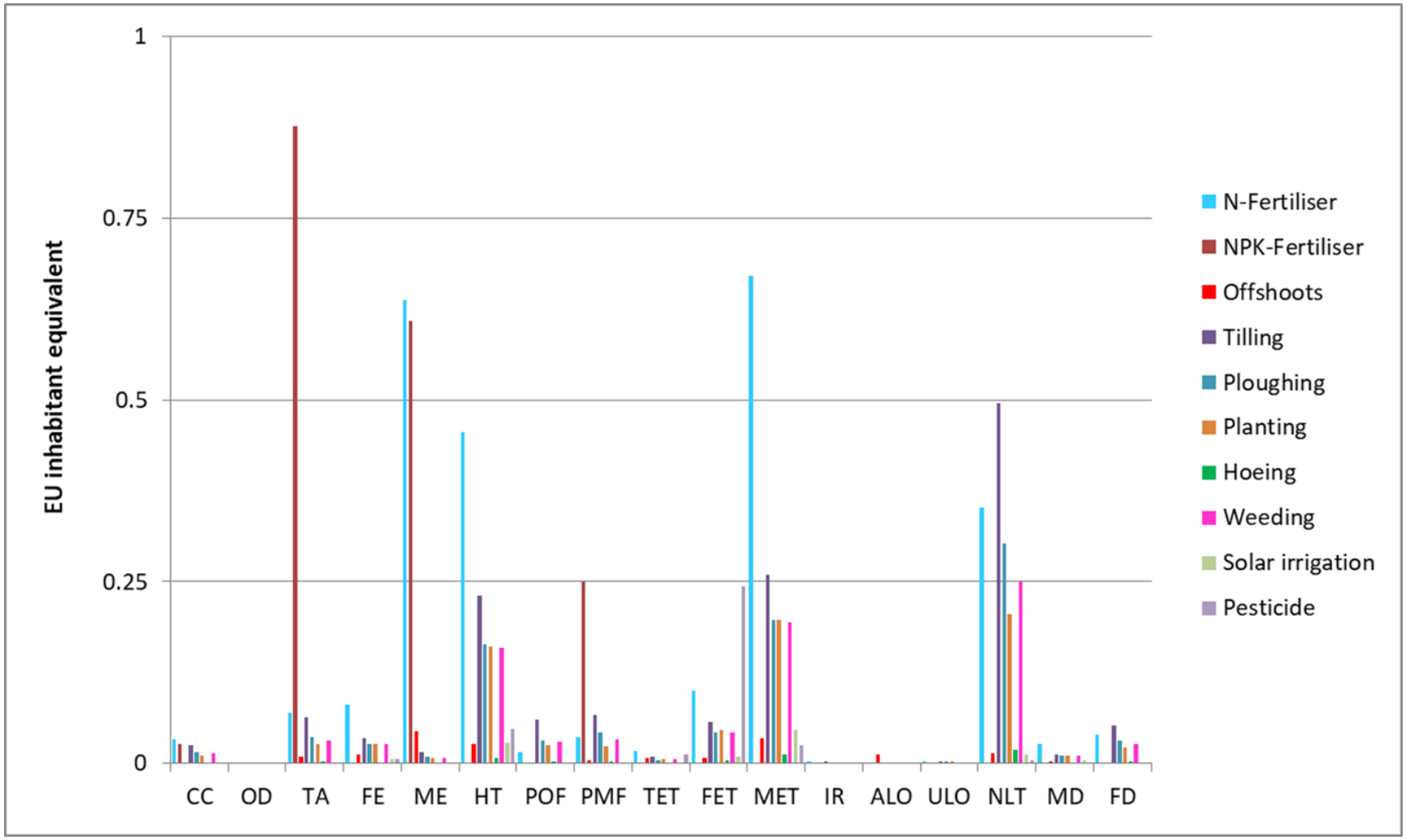

Figure 25 - Normalized impacts per unit land - EU inhabitant equivalent. See Figures 26, 27, and 28 for more details

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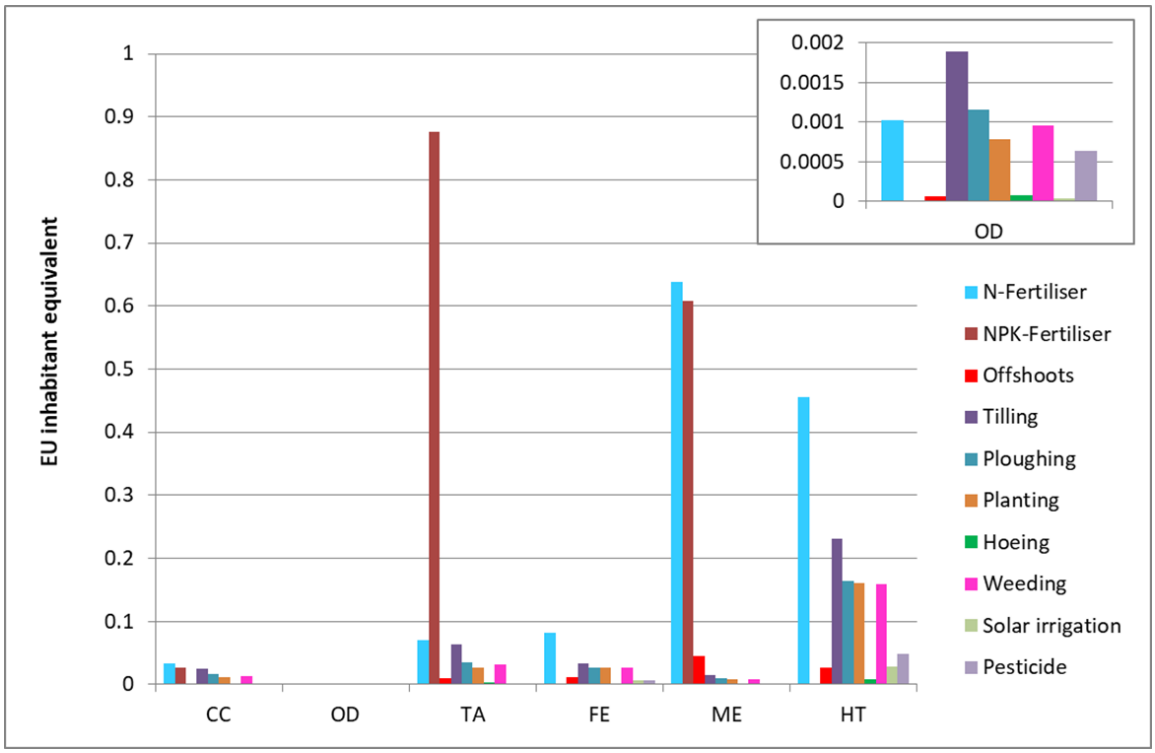

Figure 26 - The histograms report the normalized values range from 0 to 1 for the categories which are not clear in the Figure 25. The histograms on the top report the normalized values characterized by an order of magnitude lower than the one used in the main graph.

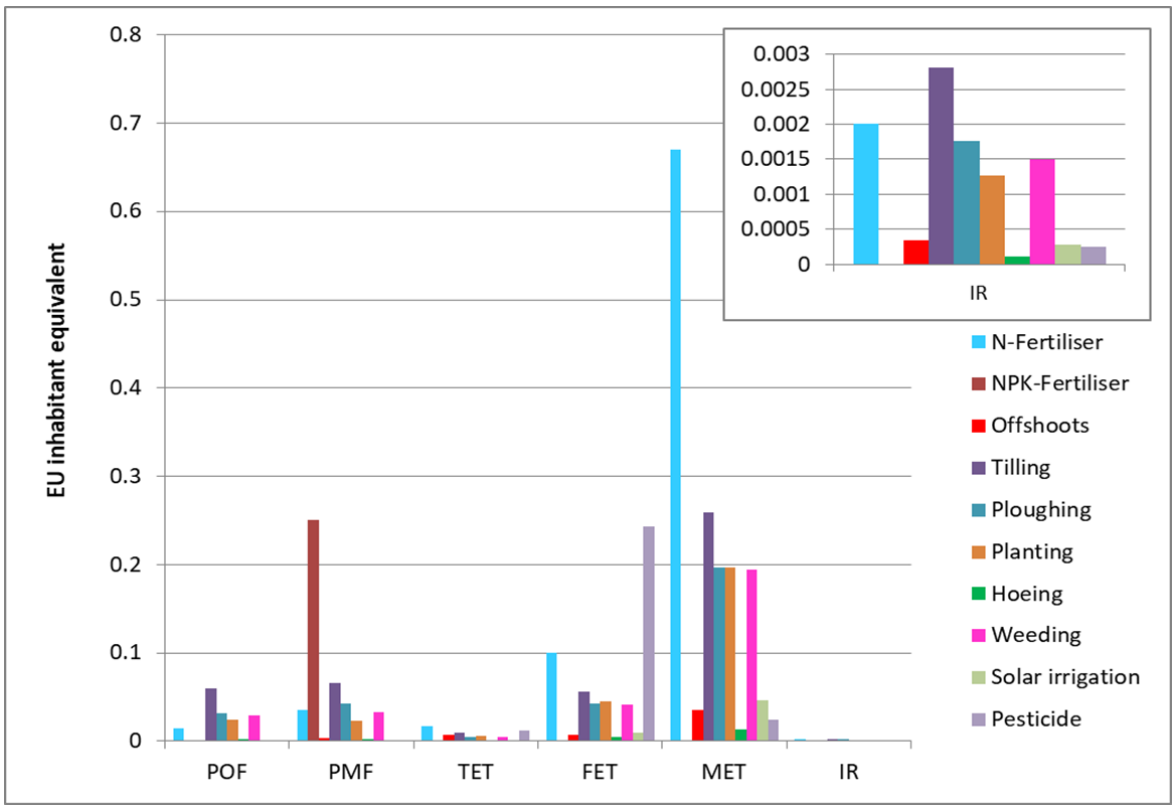

Figure 27 - The histograms report the normalized values range from 0 to 0.8 for the categories which are not clear in the Figure 25. The histograms on the top report the normalized values characterized by an order of magnitude lower than the one used in the main graph.

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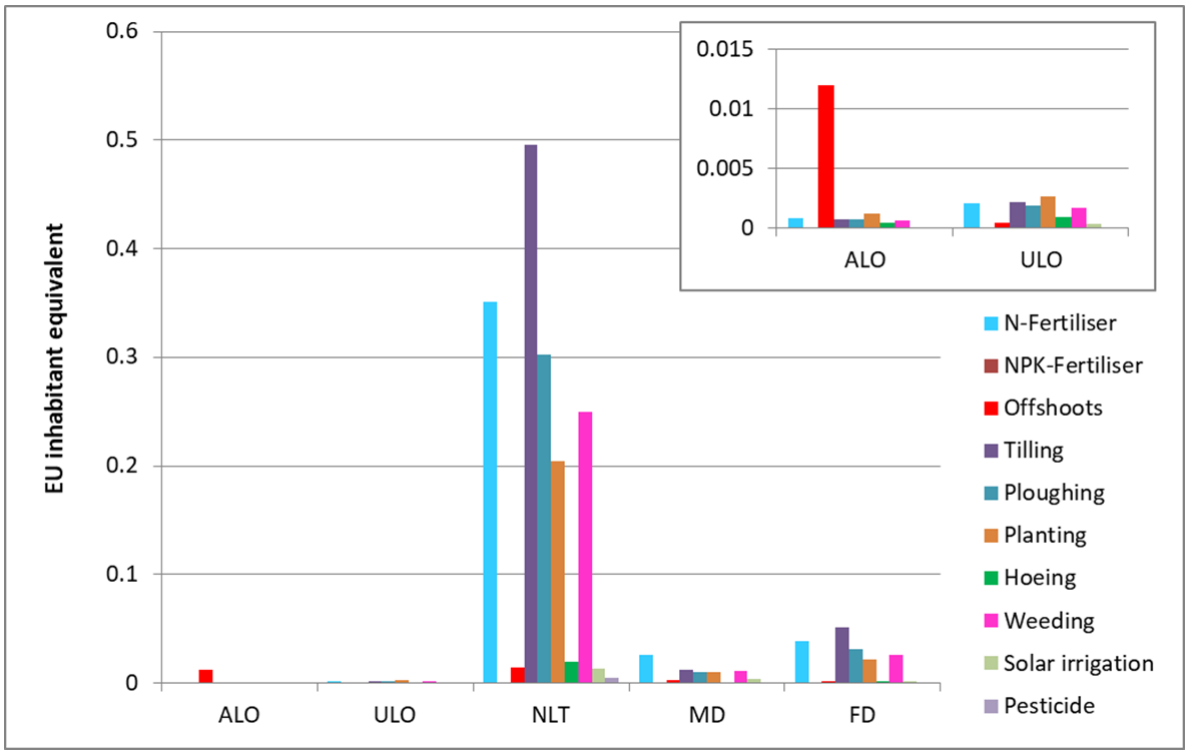

Figure 28 - The histograms report the normalized values range from 0 to 0.8 for the categories which are not clear in the Figure 25. The histograms on the top report the normalized values characterized by an order of magnitude lower than the one used in the main graph.

Damage category assessment

In order to meet the farmer's requests, the findings obtained by evaluating impact categories were expressed as a single score at endpoint level. The conversion and aggregation of eighteen impact categories previously evaluated into only three damage categories (i.e., human health (HH), ecosystem diversity (ED), and resource availability (RA)), facilitates results interpretation to the detriment of their uncertainty. Specifically, the damage categories were subjected to a normalization (i.e., the relative magnitude of each impact category) and subsequently a weighting (i.e., a conversion of normalised values to determine the relevance attributed to each damage category) phase. The aggregation of results arisen from weighting phase has enabled to express the importance of each damage category relative to each other so that they can then be summed up to get a single number for the total environmental impact (i.e., the so-called points (Pt)). This unit has enabled to compare the environmental burden caused by the different technical inputs and agricultural practices related to only three damage categories and consequently highlight the relevance related to the SI system applied to the globe artichoke cultivation.

The evaluation of damage categories was performed by using a production-based functional unit (i.e., one ton of product ha-1) in line with the goal of the present LCA analysis. Furthermore, a mass-based allocation procedure (i.e., 33% of heads and 67% of residual biomass) was used in the damage assessment in order to consider the double production purpose of the globe artichoke system.

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Considering the head production scenario, HH was the most affected damage category by each factor, with a contribution ranging from 11.3 Pt (N-fertiliser) to 0.3 Pt (hoeing) except for the offshoot production which showed the highest damage (0.9 Pt) in ED (Figure 29). On the other hand, RA was by far the best category compared to HH and ED (14.0 Pt vs 43.0 Pt and 19.0 Pt, respectively. In RA, tilling was the worst factor with a contribution equal to 3.9 Pt whereas pesticide was responsible of the lowest damage (0.09 Pt). Basically, the damage evaluation on the basis of head production highlighted that the agricultural management used for the globe artichoke was more harmful for HH than ED and RA which was damaged for 44% and 32%, respectively, compared to the worst category.

Despite the adoption of a different functional unit at the endpoint level, the damage analysis provided findings consistent with what highlighted from the impact category analysis. Indeed, human toxicity and ecosystem diversity were the most affected environmental factors, although high emission levels impacting the ecological and human toxicity categories do not necessarily mean high levels of damage.

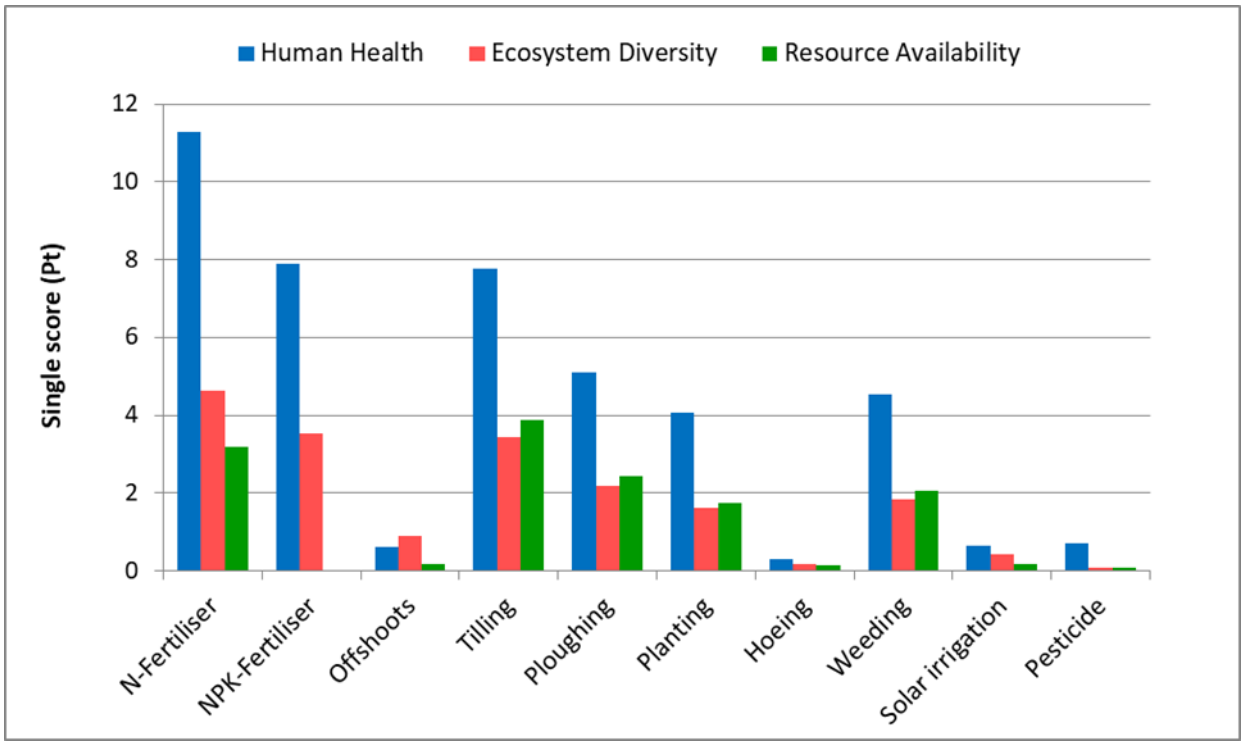

Figure 29 - Damage assessment on production basis (ton of heads)

However, the evaluation at endpoint level stressed that N-fertiliser and tilling were the most harmful factors for environment whereas the least damage was caused by hoeing and pesticide on head production basis by considering the damage categories on the whole. Specifically, tilling showed a total environmental damage equal to 79% compared to the worst factor (i.e., N-fertiliser) followed by NPK - fertiliser and ploughing (60% and 51%, respectively). On the other hand, hoeing and pesticide had contributed to the overall environmental damage with 3% and 5%, respectively. The other factors showed an overall damage ranging from 44% (weeding) to 7% (SI system) compared to the most negative factor.

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RA was the was especially affected by tilling followed by N-fertiliser which showed an incidence equal to 82% compared to the worst factor. The contribution of the other mechanical operations was between 62% (ploughing) and 4% (hoeing) compared to tilling whereas among technical inputs, the least harmful was pesticide 2% and the SI system did not exceed 5%. The potential damage of the NPK fertiliser was not evaluated in this category because of the lack of data regarding the use of raw materials in the upstream processes which were not included in the assessment as well as already occurred in the impact categories. Therefore, the NPK-fertiliser showed a damage in ED and HH on the basis of the release of emissions due to its use in the globe artichoke cultivation. Its incidence on the HH category was higher than the one of ED (69% vs 31%).

In ED, the first and the second worst factor with respect to the head production were N-fertiliser and NPK fertiliser, respectively, which showed a contribution equal to 77% compared to the most negative factor. The mechanical operations had an incidence between 75% (tilling) to 3% (hoeing). Pesticide and SI system contribute with 2% and 9% respectively, with respect to N-fertiliser. The same trend was detected in HH although with different values of incidence. The NPK-fertiliser contributed with 70% followed by tilling (69%) and the other mechanical operations that did not achieve 50% without considering hoeing which showed an incidence equal to 3% compared to N-fertiliser. Offshoot production, SI system, and pesticide were the best factors in terms of environmental damage equal to 5% (offshoots) and 6% (solar irrigation and pesticide) with respect to the most harmful factor.

The environmental performance of each factor was different within the single damage categories. For instance, N-fertiliser was the worst factor in HH and ED but not in RA in which the most harmful factor was tilling. On the other hand, hoeing showed the lowest damage in all three categories. Furthermore, all factors showed the worst performance in the HH category except for the offshoot production, which had a damage in ED 1.5 times higher than the one in HH. N-fertiliser especially contributed to HH (59%) and only 18% in RA. This result may suggest a low use of resource that does not make it too difficult to replenish the amount which was consumed. Tilling, ploughing, planting, and weeding showed the same performance within the three damage categories, namely the highest incidence was depicted in HH (ranging from 55% for planting to 51% for tilling) followed by RA (from 26% for tilling to 24% for weeding). ED showed the lowest damage, that is 23% due to tilling and 22% to planting, ploughing, and weeding. In other words, the production of emissions during the implementation of the mechanical operations and responsible for pollution and toxicity, and the consumption of raw materials for the upstream processes might have caused the higher damage in HH and RA than in ED. However, the contribution of hoeing to RA and ED was opposite with respect to the performance of the previous factors (i.e., 0.1 Pt and 0.2 Pt for RA and ED, respectively). Furthermore, hoeing was by far the least harmful factor in HH followed by pesticide in RA and ED. However, pesticide showed a considerable difference in terms of incidence on the damage categories since it contributed with 80% to HH and with 10% to ED and RA.

The potential damages caused by the offshoot production did not exceed 2.0 Pt although the highest incidence concerned ED and not HH as occurred all other factors (i.e., 54% vs 37%, respectively).

The SI system was one of the least harmful agricultural practice since its damage on the whole was equal to 1.3 Pt vs 19 Pt of the N-fertiliser with an incidence equal to 51%, 34%, and 15% in HH, ED, and RA, respectively. This performance may suggest that the SI practice might be

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characterised by technical components whose the production processes might release harmful substances and emissions for human health and ecosystem quality.

The damage evaluation based on the residual biomass production scenario showed the same trend found in the head production scenario (Figure 30).

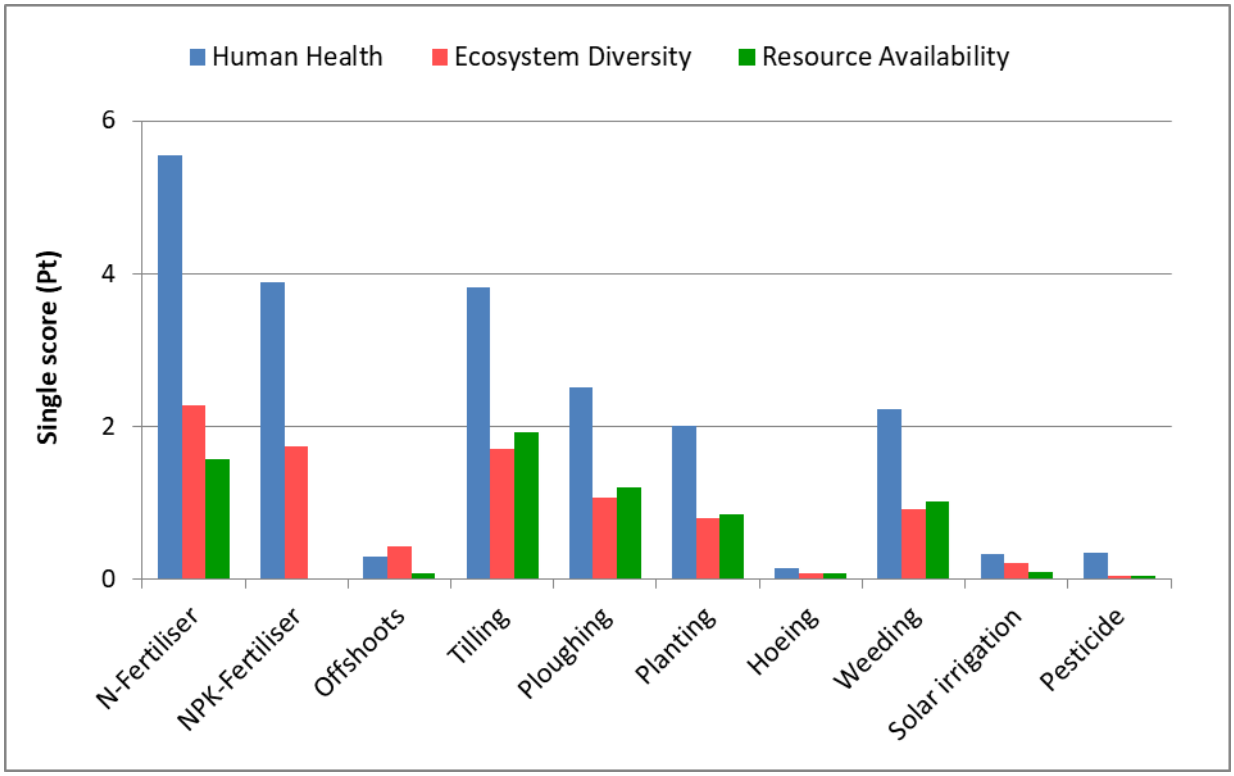

Figure 30 - Damage assessment on production basis (ton of residual biomass)

Indeed, this result could be easily predicted by considering that the residual biomass and heads are generated from the same plant. The main difference in terms of results between the two scenarios arisen from the allocation of the environmental burden between the products.

Considering that the incidence of the residual biomass production on the total environmental load was 67% vs 33% of the head production, the overall agricultural management adopted for the globe artichoke cultivation showed a twofold environmental burden in the head production scenario with respect to the residual biomass one. Therefore, HH was the most damaged category in the residual biomass production scenario as well as occurred in the other scenario. Specifically, HH was two and three times higher than ED and RA, respectively. The N-fertiliser provided the most negative contribution with 6 Pt in HH whereas hoeing showed the best environmental performance in RA with 0.07 Pt followed by the offshoot production and solar irrigation (0.08 Pt and 0.09 Pt, respectively). ED was the second most damaged category in which N-fertiliser was the worst factor (2.0 Pt) whereas pesticide was the least damaging factor with 0.05 Pt on one ton of biomass production.

The N-fertiliser was by far the worst factor in the biomass scenario as was already found in the head production scenario, followed by tilling (79%), NPK-fertiliser and some mechanical operations with an incidence ranging from 60% for NPK-fertiliser to 39% for planting. On the

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contrary, hoeing did not exceed 3% compared to N-fertiliser and together with pesticide (5%) and the SI system (7%) was by far the least damaging factor. These percentages were already found on head production basis, as previously reported, since the only difference between the two production scenarios was that the one (i.e., residual biomass production) was twice the other (i.e., head production). In other words, the evaluation of the contributions (in percentage terms) of each technical input and agricultural practice to the three damage categories regarding biomass production scenario did not differ from the results found in the head production scenario. The same consideration also applies to the incidence of the factors both in absolute terms and within each damage category. For instance, the N-fertiliser by far showed the highest damage (9.0 Pt) vs hoeing (0.3 Pt) that was the least harmful factor in the biomass scenario. The corresponding values were equal to 19 Pt and 0.6 Pt for N-fertiliser and hoeing, respectively, in the head production scenario.

The N-fertiliser was the most negative factor in HH and ED (6.0 Pt and 2.0 Pt, respectively) whereas tilling showed the worst damage (2.0 Pt) in RA. On the contrary, hoeing was the best factor in HH (0.1 Pt), pesticide showed the most positive environmental performance in ED and RA (0.05 Pt and 0.04 Pt, respectively).

As regards the SI system, its environmental performance may be considered quite positive in all damage category within biomass scenario as well as occurred for the head production. Indeed, the solar irrigation was more harmful 5 times more than pesticide in ED and 2 times more than hoeing and pesticide in HH and RA, (i.e., the least damaging factors). However, it showed a lower contribution (11, 17, and 21 times in ED, HH, and RA, respectively) than the one of the corresponding worst factor, that is N-fertiliser in ED and HH, and tilling in RA. In other words, the incidence of the SI practice compared to the agricultural management for each damage category was equal to 2% in HH and ED, and 1% in RA, that is a total damage equal to 5% in each production scenario. Considering that the same incidence evaluated for the worst factor in each damage category was equal to 26% and 25% for N-fertiliser in HH and ED, and 28% for tilling in RA, the SI system might be considered one of the less important practices in terms of environmental burden in the agricultural management of the globe artichoke (Figure 31).

Besides the SI system, the offshoot production, hoeing, and pesticide were the least harmful factors of the adopted agricultural management with an incidence ranging from 7% (offshoots) to 3% (hoeing and pesticide). ED and HH were the most damaged categories by all these factors although their contribution did not exceed 5% (in ED due to the offshoot production).

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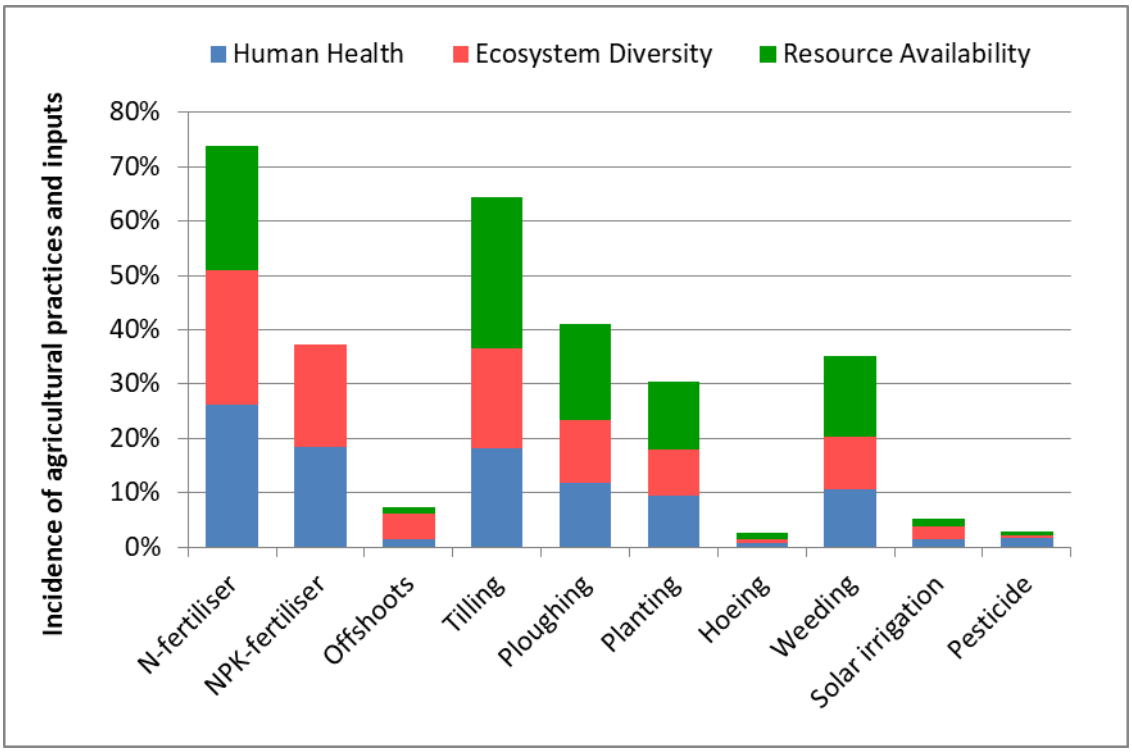

Figure 31 - Incidence of agricultural operations and inputs on the total single scores on production basis

On the contrast, the other factors showed the overall worst environmental performance with an incidence ranging from 74% (N-fertiliser) to 31% (planting). Specifically, the damage on the HH and RA categories was very similar (96% and 95%, respectively) and it was especially caused by fertilisers and mechanical operations (not previously considered) although the NPK-fertiliser was not included in the RA evaluation because of lack of data regarding the upstream processes of the fertiliser production. The contribution to RA was between 28% and 12% due to tilling and planting, respectively. In the HH category, the main responsible for damage was 26% and 11% caused by N-fertiliser and weeding, respectively. As regards ED, the damage was slightly less than the other two categories (92%) mainly owed to both fertilisers (25% and 19% for N- and NPK-fertiliser, respectively) and with a contribution of some mechanical operations ranging from 18% (tilling) and 10% (weeding).

Interpretation

The evaluation performed in the present LCA analysis has allowed to determine the relevance of the SI system in terms of environmental load compared to the other agricultural practices and technical inputs used for the globe artichoke cultivation, that is the goal of this study. The results of an LCA study are heavily dependent from availability of data as accurate as possible in qualitative and quantitative terms.

The data collected in the inventory of the present LCA analysis and used to evaluate the environmental load due to the globe artichoke system, may be considered representative of the agricultural management commonly adopted in Sardinia for the globe artichoke cultivation as widely highlighted in the goal and scope through support of scientific literature concerning the crop systems under consideration. Furthermore, the appropriateness of the definition of the

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functional unit and system boundary was ensured since both were defined directly on the basis of the information provided by the farmer and consistent with his requests which were included in the goal of the analysis.

The databases available in the SimaPro software were used to consider in the inventory and thus in the assessment of the environmental load, the upstream processes of the production of technical inputs (e.g., fertilisers and pesticides) and of structural components for the SI system and machines. As regards the NPK-fertiliser, the upstream processes related to the fertiliser production (e.g., consumption of raw materials and natural resources) were not considered by the evaluation method of impact (almost all impacts) and damage categories due to lack of data in contrast to the emissions arisen from this input which were included in the inventory. Therefore, the NPK fertiliser showed an environmental performance in impact and damage category related to emissions whereas impacts and damages due to the NPK-fertiliser and affected particularly by upstream processes were not evaluated.

The performance of the NPK-fertiliser may be considered a significant issue because the presence of the upstream processes would potentially have changed the final LCA results. Whether raw materials and natural resource use had been considered some impact categories such as terrestrial, freshwater, and marine ecotoxicity, or land transformation and occupation, or water, metal and fossil depletion they would likely be influenced. Among the damage categories, only RA did not show a damage due to the NPK-fertiliser use. In other words, the lack of data on the natural resource depletion and raw material consumption may have been responsible for the absence of damage in the RA category since RA is related to the restoration of the amount of resources that was consumed.

The LCA analysis highlighted the key role of fertiliser use from environmental point of view within the agricultural management of the globe artichoke cultivation. In fact, N- and NPK-fertiliser together showed the worst environmental performance in nine (i.e., CC, TA, FE, ME, HT, PMF, TET, MET, and MD) categories out of eighteen, that is half. The second most negative factors in terms of number of impacted category were the mechanical operations, in particular tilling affected five impact categories (i.e., OD, POF, IR, NLT, and FD) whereas the other factors showed the worst performance only in one category (i.e., offshoots, ALO; planting, ULO; pesticide, FET). The main impact due to the SI system almost exclusively concerned WD most likely because of the water volume distributed by irrigation, whereas it showed a low incidence in the remaining categories. Although WD was not evaluated in the normalisation phase, this optional step stressed a trend consistent with the one of the characterisation phase for the other impact categories.

The importance of fertilisers and mechanical operations within the adopted agricultural management was highlighted by their environmental performance with regard to the three damage categories in both production scenario (i.e., heads and residual biomass). Similarly and consistent with the impact category evaluation, the low relevance of the SI system in terms of environmental load was clear in all three damage categories. Although the SI practice showed one of the best environmental performances together with hoeing and pesticide, it was equally harmful firstly for HH, ED, and ultimately for RA. This trend demonstrates that also the SI system consumes resources and releases emissions in the environment likely due to the production of its structural components.

A Monte Carlo analysis was performed to assess the uncertainty of the LCA findings. The analysis was also performed to test for possible significant differences in the environmental load of the agricultural practices and technical inputs compared to the SI system used during the globe

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artichoke cultivation. Specifically, this statistical analysis was carried out to evaluate the reliability level of the LCA results by pairwise comparison between each of the factors characterising the agricultural management and the SI system in terms of single score for each damage category on land basis. SimaPro 8.0.4.30 software was used to run the Monte Carlo simulation (Goedkoop et al., 2013a, 2013b) at a 95% confidence interval with 1,000 reiterations.

The analysis highlighted that in all damage categories pesticide and hoeing, that is the least harmful factors, show no difference statistically significant at $\alpha = 0.05$ from SI (Table 14).

Table 14 - Results from Monte Carlo analysis (confidence interval = 95%)

Pair-to-pair comparison of MC scores										
Human Health										
	N-fert.	NPK-fert.	Offshoots	Tilling	Ploughing	Planting	Hoeing	Weeding	Solar irrig.	Pesticide
Solar irrig.	100%	100%	100%	99.2%	98.5%	93.2%	26.1%	97.1%	-	0.0%
Ecosystem Diversity										
Solar irrig.	100%	100%	100%	100%	100%	100%	0.8%	100%	-	0.0%
Resource Availability										
Solar irrig.	100%	- [a]	100%	100%	100%	100%	0.0%	100%	-	0.0%

a: missing value since RA was not evaluated because of the lack of the upstream processes of the NPK fertiliser

Furthermore, planting was not significantly different from the SI system only in the HH damage category.

The results of the Mont Carlo simulation suggest that the probability that the difference between SI and the other factors in terms of environmental damage may occur is correct 19 times out of 26, that is 73%.

The present LCA analysis has allowed to identify the most important “hotspots” within the agricultural management adopted for the globe artichoke cultivation. Specifically, the results highlighted that greater attention should be paid to the fertiliser use and mechanical operations since they were harmful in terms of toxicity for humans and ecosystems and resource depletion caused by their production processes. However, the factors responsible for a low environmental load (e.g., pesticide and solar irrigation) and not null should not be neglected as well.

On the light of the LCA findings, the potential improvements that farmer might adopted should basically cover a reduction of fertiliser/pesticide dose and the number of mechanical operations. However, these measures might not be enough to enhance the agricultural management of the globe artichoke from environmental point of view by considering that the main objective of farmer is to optimize the production. The scientific research may provide viable suggestions in order to facilitate the achievement of the maximum yield level by adopting an eco-friendly agricultural management. In fact, the adoption of principles and technologies of conservation agriculture and precision agriculture long object of study within the scientific community, might offer effective solution to foster an intensive and sustainable agriculture from environmental point of view.

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Conservation agriculture may be considered a resource-saving agricultural production system which may foster the implementation of intensive cropping system and high yields in an eco-friendly manner. Minimal soil disturbance, permanent soil cover, diversification of crop rotations, and integrated weed management may preserve soil fertility by minimizing soil erosion, water loss from runoff, and soil physical degradation (Farooq and Siddique, 2015). Nutrient management is essential to ensure crop productivity and for the adoption of conservation agriculture practice by farmers since nutrient loss may be reduced by minimizing runoff and optimizing the use of deep-rooting cover crops able to recycle nutrients leached from the topsoil (Dordas, 2015).

The use of cover crops in intercropping may be a good strategy for farmer to reduce the synthetic fertiliser application without compromising crop productivity and at the same time to prevent environmental damage such as pollution. As reported by Reddy (2016), intercropping such as legume-based cover crop may ensure a sustainable production by minimizing fossil fuel consumption and pollution risk, and there is poor investment opportunity. Specifically, the intercropping may facilitate the efficient use of resources necessary for crop growth (e.g., land, sunlight, and vertical space), improve soil fertility and fixation of large quantity of N in soil. Intercropping based on plants from Leguminosae family and non-legume plants may improve the N content in non-legume plants. Furthermore, the same author highlights that an intercropping system may ensure a greater yield stability throughout the growing seasons and economic stability to farmers, an increase in biodiversity in agro-ecosystems, reduce nitrate leaching, improve carbon sequestration, provide a permanent surface cover by protecting soil from raindrop impact, wind, and water erosion, ensure biomass addition and thus organic matter to the soil, and suppress pests.

The adoption of an intercropping system might be an opportunity for the farmer to diversify his production, namely to cultivate the globe artichoke both for food (heads) and energy (residual biomass) purpose rather than burying biomass to supply soil in terms of organic matter which would be ensured by cover crop. However, conservation agriculture is not devoid of constraints that mainly regard weed management. In fact, the spread of weed seeds on the soil surface caused by no tillage operation may exacerbate weed infestation and thus, the herbicide use is so far the only tool to face this problem by underling the need of new weed management technologies (Singh et al., 2015). On the other hand, no tillage and minimum tillage cause no or minimal soil disturbance resulting in a reduction of machine use and emission release.

Nevertheless, the benefits of the conservation agriculture, its adoption would require huge efforts and trade-offs at individual and institutional levels due to, for instance, the lack of proper information and institutional support, the potential temporary reduction in economic returns, and the need to refine nutrient and water management (Farooq and Siddique, 2015).

The application of precision agricultural practices may foster reductions in GHG emissions and increases in soil organic carbon storage since these practices may lower the intensity of tillage practices, the required N supply and other production inputs, and the consumption of fuel for mechanical operations. Specifically, these innovative practices may optimize a small amount of production inputs such as N fertilisers that, if used excessively or in a large agricultural area, may have relevant negative impacts in terms of environmental and economic sustainability (e.g., low profit margins on a land basis) (Solinas et al., 2021). As reported by Balafoutis et al. (2017), the use of precision agriculture practices (e.g., technologies that allow variable rate application of nutrients, irrigation, pesticides and planting/seeding; controlled traffic farming and machine guidance) with technological equipment may spatially and temporally optimize the use of inputs

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based on site-specific characteristics. The farmer has already adopted a precision technique, that is the irrigation system with cooling effect on the globe artichoke cultivation characterised by on soil moisture sensor system. Although this irrigation system demonstrated to foster the water saving compared to a non-precision irrigation system, the agricultural management might be further enhanced by adopting other precision technologies.

As reported by Ahmad and Mahdi (2018), precision farming technology is aimed to provide information and data to support farmers when making site-specific management decisions. The adoption of precision farming techniques are based on the collection of site-specific information and the availability and management of technologies. Detailed information on soil properties, climatic conditions, plant growth response, biotic and abiotic stressors, etc. may be used to generate different kinds of maps of the farms/villages/regions (e.g., different soil characteristics, groundwater, pest incidence, weed distribution, topography, environmental pollution, etc.) which may facilitate the farmers to regulate the supplies of inputs in different areas. Use of remote sensing and geographic information system, sensors, computers along with appropriate software (e.g., auto-steer on tractor), etc. may help in precisely identifying areas of nutrient deficiencies and other biotic and abiotic stresses, in order to develop target interventions which may lead to an increase in productivity and profitability.

Since farmers should know how to interpret available information, how to use the technology, different expertise needed for the development and dissemination of knowledge-intensive precision farming technologies. However, the adoption of these technologies may often require high investments that farmers might be more willing to make without public efforts (e.g., provision of financial aids) able to force them to introduce these innovative technologies.

Although the present LCA analysis highlighted that the SI system used for the globe artichoke cultivation showed a relatively low environmental load compared to the other factors, the results emphasized the relevant impact on water depletion due to water irrigation distribution and a not relevant damage on the three categories (HH, ED, and RA). The damages may be caused by the structural components of the SI system and they may be reduced if some raw materials used for SI are realized by recycling or into circular economy processes.

In order to better understand how water consumption is distributed over the globe artichoke system might be useful to evaluate the Water Footprint (WF) of an agricultural production on the basis of the LCA procedure. It is an environmental indicator aimed to define the total volume of freshwater that is used to produce the product by considering the volume of surface and groundwater consumed (evaporated) as a result of the production of a good (i.e., the blue water); the rainwater consumed that is the green water footprint, and the volume of freshwater that is required to assimilate the load of pollutants based on existing ambient water quality standards (i.e., grey water) (Mekonnen and Hoekstra, 2011). WF may provide useful information to the farmer in order to improve the irrigation water management.

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