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# ENVIRONMENTAL LIFE-CYCLE BASED METHODS TO SUPPORT THE TRANSITION TOWARDS CIRCULAR ECONOMY IN THE AGRI-FOOD SECTOR


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# Abstract

The holistic approach of Life Cycle Thinking (LCT) can support the transition towards sustainable production and consumption patterns, in a circular economy approach. The objective of this dissertation thesis is to critically analyse some peculiar aspects of the application of environmental life-cycle based methods to the agri-food sector and to identify opportunities and obstacles of the LCT approach through the testing of some methods and tools.

The main critical methodological problems of the application of Life Cycle Assessment (LCA) to agri-food sector were described. In particular, since on-field emissions cannot be directly measured at a reasonable effort, the analysis was focused on the necessity of using dispersion models for their estimation. Different models exist but scientific consensus lacks about the most suitable in terms of reliability and practicability. To test how this kind of models work and which data are required, PestLCI 2.0 has been applied to an experimental farm in Northern Italy, and a comprehensive set of pesticide emissions in the different compartments, which is a relevant input for the inventory phase, has been obtained. The application has required collecting detailed information about the soil characteristics, which resulted to affect the outcomes significantly, especially for groundwater emissions. Since the detailed picture of pesticides emissions is not fully captured by the current impact assessment methods, further research efforts will be needed to develop characterisation factors for groundwater emissions, in order to exploit the potential of PestLCI 2.0.

The analysis of the literature concerning the methodological problems highlighted that different scientific approaches used to solve the problems might lead to different life cycle results, thus affecting products comparability, which is important when LCA is used for calculating and communicating the environmental performance of products. In this dissertation thesis, the use of LCA for communication purposes was evaluated through the testing of the Product Environmental Footprint (PEF) method in an Italian Taleggio cheese production chain, with the aim to evaluate if it fulfils the harmonisation needs for the calculation and communication of the environmental performance of food and drink products. Although Product Environmental Footprint Category Rules for dairy products provide quite detailed guidance for some methodological issues, some other topics would require additional guidance. The application of the PEF method resulted to be resource-

intensive: this aspect could make it difficult to spread the method, especially if the goal is to involve European Small and Medium Enterprises. In general, the application of the PEF method could take advantage of the development of simplified supporting tools.

Finally, due to the significant contribution of agricultural sector to water scarcity and water pollution problems, the Water Footprint (WF) Network method was tested in an Italian tomato cultivar production with the aim to evaluate strengths and weaknesses. The study required the collection of a large number of data, some of which obtained from literature due to lack of primary data. Results highlighted that site-specific data are needed to increase the results robustness and demonstrated that the effect of the yield may penalize cultivations with low blue water use, because the model to calculate green water does not depend on the cultivation intensity, thus leading, ceteris paribus, to higher WF in extensive cultivations. Though further research will be needed to develop a common accepted WF method, agri-food companies and public decision makers can take advantage of this method to support a sustainable water management and the implementation of green marketing strategies.

## List of acronyms

AF = Allocation factor

AMD = Availability Minus the Demand

ARPAV = Veneto Regional Agency for Environmental Prevention and Protection

ARPAV = Veneto Regional Agency for Environmental Prevention and Protection

AWARE = Available WAter REmaining for area in a watershed),

B2B = Business to Business

B2C = Business to Consumers

BIB1 = Bibione soil

C = Clay

CAB1 = Caberlotto soil

CAP1 = Capitello soil

CFO1 = Ca' Fornera soil

CMT = Meteorological Centre of Teolo

CON1 = Conche soil

CRL1 = Caorle soil

CTU1 = Ca' Turcata soil

DM = Dry Matter

DNM = Data Needs Matrix

DQR = Data Quality Rating

EEA =European Environment Agency

EF = Environmental Footprint

EMEP = European Monitoring and Evaluation Programme

EPD = Environmental Product Declaration

ET = Evapotranspiration

ILCD = International Reference Life Cycle Data System

IPCC = Intergovernmental Panel on Climate Change

IPP = Integrated Product Policy

L = Loam

LCA = Life Cycle Assessment

LCI = Life Cycle Inventory analysis

LCIA = Life cycle impact assessment

LCIA = Life Cycle Impact Assessment

LCT = Life Cycle Thinking

LS = Loamy sand

MEL1 = Casa Scaramello soil

OEF = Organisation Environmental Footprint Organization

PDO = Product Designation of Origin

PEF = Product Environmental Footprint

PEFCR =Product Environmental Footprint Category Rules

QUA1 = Quarto d'Altino soil

S = Sand

SAB1 = Sabbioni soil

SCL = Sandy clay loam

SCO1 = Santa Scolastica soil

SIC = Silty clay

SICL = Silty clay loam

SIL = Silt loam

SL = Sandy loam

SOIL6 = Default soil from PestLCI 2.0 database

SMEs = Small and Medium Enterprises

STU = Soil Typological Units

TDF1 = Torre di Fine soil

VAD1 = Valcerere Dolfina soil

VAN1 = Vanzo soil

VED1 = Casa Vendramin soil

WF = Water Footprint

WFA = Water Footprint Assessment

WFN = Water Footprint Network

WULCA = Working Group on Water Use in LCA

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# 1 Introduction and objectives

Agriculture and agri-food sector satisfy one of the most important human needs, i.e. nutrition and provide significant social and economic values. As regards Italian situation, the agri-food sector is a priority sector for national economy, due to both its cultural, social and economic importance and to the peculiarities and specific characteristic of Italian food and drink products. Nevertheless, the traditional economic model, the so-called “linear economy”, applied in the last century also in the agri-food sector, as in all the other manufacturing sectors, and based on the massive exploitation of non-renewable resources and to the production of significant amount of waste, has become unsustainable (Ellen MacArthur Foundation, 2013a). In fact, food and drink supply chain has several important environmental impacts and is the main responsible for the land use change, the loss of biodiversity, the greenhouse gas emissions and the use of freshwater (European Commission, 2011). Without appropriate offsetting measures, severe consequences for both the world population and the environment could occur, such as the lack of food availability and food security and the almost complete depletion of natural resources. Because of these reasons, the agri-food sector should move from linear economy models to circular economy ones, which include sustainable production and consumption patterns and which could decouple the economic growth from the environmental impacts and the use of resources (European Commission, 2014a). These topics are included in the most important international agendas, such as the United Nations’ Global Agenda for Sustainable Development, which has been signed also by Italian government in 2015, and the related seventeen Sustainable Development Goals. At European level, the European Commission’s communication "Closing the loop – An EU Action Plan for Circular Economy" (COM (2015) 614) represents the most important policy document which defines circular economy and introduces an action plan for the implementation of a legislative framework for the development of circular economy measures in all member states.

However, the transition towards sustainable consumption and production systems in the agri-food sector, in a circular economy approach, requires the use of robust and scientific methods which can support sustainability assessment of the overall analysed system for several sustainability indicators, by appropriately evaluating the consequences of all the possible circular options, both from an economic, environmental and social point of view.

Life Cycle Thinking (LCT) approach, which takes into account the whole product's life cycle, from the extraction of raw materials, to the product manufacturing, its transport and distribution, and the final waste disposal can support this transition, because it can be used for the assessment of the impacts and benefits associated to circular solutions in the agri-food supply chain, avoiding burden shifting from a phase to another of the life cycle, and from an environmental compartment to another one.

The objective of this dissertation thesis is to critically analyse some peculiar aspects of the application of environmental life-cycle based methods and tools to the agri-food sector. The application of this holistic approach to agri-food production chains allows the evaluation of their overall ecological performance and supports product development as well as the implementation of improvement strategies. However, it presents also some technical and methodological problems that risk to limit a wide use of the approach. In this work opportunities and obstacles of the approach will be identified through the testing of some methods and tools and needs for future developments will be discussed.

Among the life-cycle based methods, Life Cycle Assessment (LCA), standardised by the ISO 14040 series (ISO 2006a, b), is recognised as a strategic and effective tool to evaluate the potential environmental impacts occurring in the whole product's life cycle as well as to identify possible areas for improvement. LCA has been used in the recent years to evaluate the environmental impacts of a wide variety of agri-food products, contributing to identify the environmental hotspots of the supply chain and the potential improvement opportunities (Notarnicola et al., 2012). Nevertheless, when applying LCA to food and drink products, the practitioner has to deal with methodological problems which stem from the peculiarities and specific characteristics of the agri-food supply chain. In fact, differently from industrial production systems, agri-food supply chains are characterised by complex relationships both between inputs (for example nutrients and soil) and outputs (crops and emissions) and more in general between biological processes and processes of technological systems, which are difficult to be modelled in LCA studies. In particular, the inventory phase of LCA studies for food and drink products involves the estimation of on-field emissions due to the use of fertilisers, pesticides, or emissions from livestock, such as ruminants' enteric fermentation, which cannot be directly measured at a reasonable cost and effort. This is one of the main critical problems in LCA studies of agri-food products, because these emissions must be calculated by dispersion models, which are based on several agricultural or livestock site-specific parameters that have to be collected or alternatively found in literature (for example the nitrogen content in manure) and which therefore require specific knowledge in this field. Several literature models are available in literature, but no scientific consensus still exists on which one should be used.

Among the main critical methodological problems of the application of LCA to agri-food sector, which will be described in this dissertation thesis, due to the relevance of the problem for the impact on ecosystems and human health, a focus will be given to the calculation of on-field emissions due to the use of pesticides at inventory level by applying the detailed PestLCI 2.0 model (Dijkman et al., 2012) to an experimental farm in Northern Italy, with the aim to verify the model's sensitiveness to soil variations.

LCA can also be used to communicate the environmental performance of products business to business (B2B) or business to consumers (B2C), representing a marketing opportunity for agri-food companies to increase competitiveness. In fact, LCA and more in general the LCT approach are the basis of many ecological labels, in particular those compliant with ISO 14020, such as the European Ecolabel or the Environmental Product Declaration (EPD), which have been increasingly applied in the last recent years to a great variety of products, including food and drink products. However, when LCA is used to communicate the environmental performance of agri-food products, the different scientific approaches used to deal with the methodological issues might lead to different LCA results, making it difficult to compare the environmental performance of products of the same category. The European Commission's Product Environmental Footprint (PEF) recommendation, developed in 2013 (European Commission, 2013a), and the integrating documents aim to fulfil the need for harmonisation to calculate and communicate environmental footprint of products. Through the development of 'category rules' for food and drink products, detailed requirements should be provided for each stage and process of the life cycle, thus supporting products comparability. In this dissertation thesis, the use of LCA for communication purposes was tested by analysing how the PEF method responds to these harmonisation needs, and applying it to an Italian Taleggio cheese supply chain, highlighting also the advantages and difficulties of this approach.

Another life-cycle based tool developed in the last years and progressively used in the agri-food sector, also for communication purposes, is Water Footprint (WF), which is an assessment of the water use by products, individuals, companies or the entire population. Agricultural sector is indeed a major contributor to water scarcity and water pollution problems, and it is therefore essential to have tools and methods to understand how water use is affected by our production and consumption choices as well as to support a better management of water resources. In this context, the Water Footprint concept has created a lively discussion within the scientific community in the last years, because two methods, both based on a LCT approach, have been developed in parallel: the WF method by the Water Footprint Network (WFN) (www.waterfootprint.org), published in 2011 (Hoekstra et al., 2011), and the ISO 14046 standard, published in 2016 (ISO, 2016). The former defines the WF as the total volume of freshwater used to produce the goods and services consumed by the individual or the community or produced by a company, whereas the latter is based on the ISO 14040 LCA method and defines the WF as a metric to quantify the potential environmental impacts of products and services related to water throughout their life cycle.

The two methods are therefore different from each other, because the method of the Water Footprint Network follows a volumetric approach focused also on the quality of water and aims to support a sustainable use of water resources, whereas the latter is focused on product's environmental impacts due to the use of water. In this work, the WF method of the WFN, which has been increasingly used in literature in the last years to evaluate the water use of several agri-food products, was applied to the production of an Italian tomato cultivar with the Product Designation of Origin label, with the aim to evaluate practicability of the method as well as its strengths and weaknesses.

The research activity performed during the PhD course and presented in this dissertation thesis was carried out in cooperation with ENEA – Italian National Agency for New Technologies, Energy and Sustainable Economic Development, Laboratory Resources Valorisation, Division Resource Efficiency, Deparment for Sustainability.

The dissertation thesis is divided in 11 chapters. Chapters 1 provides an overview of the objectives of the research activity performed during the PhD. Chapters 2 and 3 presents circular economy and its possible benefits in the agri-food sector. Chapter 4 and 5 introduce LCA and Water Footprint methods, describe the main methodological problems of the application of LCA in the agri-food sector and the available harmonised LCA guidelines for the agri-food sector. Chapter 6 presents the case study performed to test PestLCI 2.0 model and evaluates its sensitivity to soil variations. Chapters 7 and 8 describe how the Product Environmental Footprint Category Rules for dairy products fulfil the need for harmonisation to calculate and communicate environmental footprint of dairy products and present the PEF case study performed on Taleggio cheese production.

Chapter 9 outlines and compares the two available WF methods and Chapter 10 describes the WF study on Piennolo tomato production. Finally, general conclusions for the overall research activity are included in Chapter 11.

# 2 Circular economy

Chapters 2 and 3 are based on the following publication, performed during the PhD:

- Chiavetta C., Fantin V., Cascone C., 2017. **L'Economia Circolare nel settore agroalimentare e il Life Cycle Assessement come strumento a supporto: il progetto FOOD CROSSING DISTRICT**. ENEA Technical report, USER-PG64-003, June 2017 (Confidential).

The linear economic model, also known as "take-make-dispose", based on the extraction of raw materials, their transformation into manufactured products, their consumption and finally their disposal as waste, has characterized the global industrial development over the last 150 years (Ellen MacArthur Foundation, 2013a) (Figure 1). This production and consumption pattern allowed the economic growth and the improvement of the world population well-being, but it is based on the intensive exploitation of non-renewable resources and energy and has therefore become unsustainable.
According to Krausmann et al., (2009), in the last century, the use of raw materials by European industrialization has increased tenfold and the domestic energy consumption has increased by seven times. Globally, in the next decade, the economic growth will require 30% increase in the demand of oil, coal, iron and other resources, especially for the emerging countries (Ellen MacArthur Foundation, 2013a). Production efficiency can contribute to decrease the quantity of raw materials and energy necessary for the production of goods, but it cannot decouple the consumption and degradation of resources from the economic growth (Ellen MacArthur Foundation, 2013). Therefore, this linear system should refer to unlimited resources to be sustainable. The linear model presents different kind of waste and losses, both in the production phases and in the consumption and post-consumption ones, where most materials are not recovered at the end of their life, but they are disposed in landfills. For example, in Europe, around 60% of materials at the end of their lifecycle are not recycled, reused or composted. Moreover, only 5% of the resource initial value is recovered after its first use (Ellen Mac Arthur Foundation et al., 2015). Further critical issues are the energy loss, because high quantity of energy is required in the first phases of the production chain to extract and manufacture raw

materials, and the damage to terrestrial ecosystems and biodiversity (Ellen MacArthur Foundation, 2013b).

According to the forecasts, the current imbalances will tend to increase, leading to higher competition for obtaining the natural resources necessary to satisfy the linear economy model. In fact, world population will increase by 1 billion in 2030 and the global population will reach 8.5 billion individuals (United Nations, 2015).

The above-mentioned dynamics will compromise the stability of the linear economy model. In fact, without a change in production policies, in the legislation and in the people behaviour, the imbalance between supply and demand for resources will increase considerably (Ellen MacArthur Foundation, 2013a).

A production and consumption system which decouples economic growth from the intensive use of rapidly expiring resources is therefore necessary. This means that every product should enter into a cycle, which can be repeated several times, to ensure that its productivity increases. In this way, the production capacity of renewable resources will be safeguarded and the natural ecosystems will be protected, to guarantee the well-being of current and future generations (Ghisellini et al., 2016). The circular economy is "an industrial economy that is restorative by intention and design" (Ellen MacArthur Foundation, 2013a). It is based on the radical rethinking of the way in which value is created. Products' materials and components are thus designed to be part of a cycle that aims to maintain the maximum value as long as possible. This means that the design of products is aimed at facilitating their reuse, their disassembly, repair, refurbishing and recycling. Moreover, waste becomes resource for other production cycles (Jurgilevich, 2017; Ellen MacArthur Foundation, 2013a). The transition to circular economy entails a complete change over the whole value chain, including new product design, circular business models, transformation of waste into resources and new consumption patterns. These goals can be achieved by strong cooperation among industries, policy makers and consumers, and by technological, social and finance innovation (European Commission, 2014a).

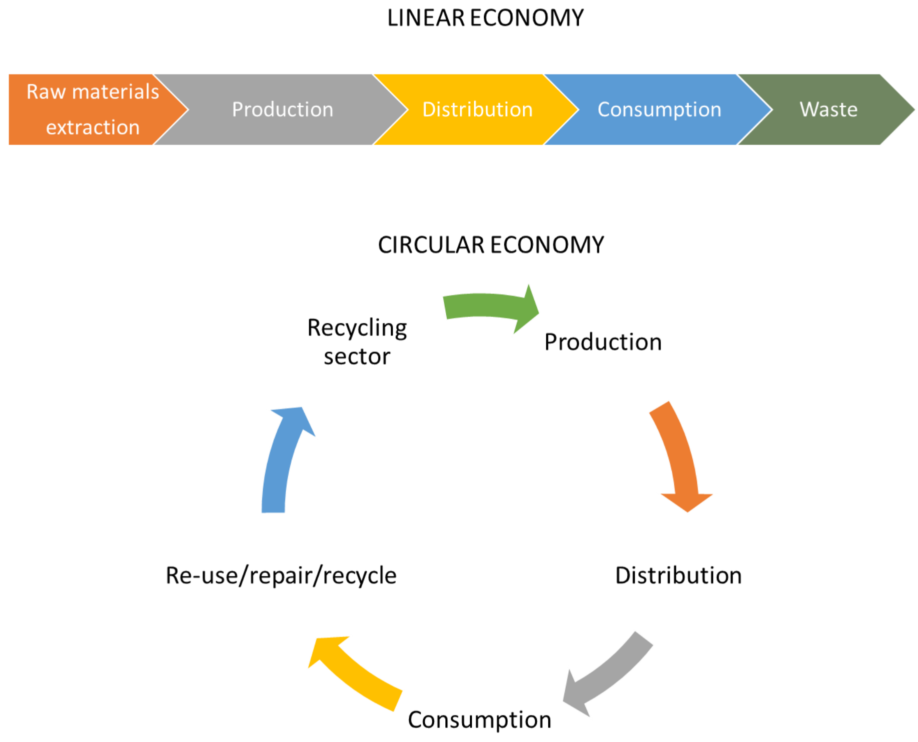



**Figure 1.** Representation of the life cycle phases in the linear and circular economy models (Source: Personal elaboration adapted from www.riciclanews.it).

This new configuration of production and consumption activities has a close correlation with non-linear systems based on frequent interactions, such as those of living beings. The fundamental principle is the systemic view. In the Cradle to Cradle philosophy, two types of cycles are distinguished: the "technical" cycles and the "biological" ones, and each of them follows a different path to close the loop (McDonough and Braungart, 2003). The "biological" components are designed to be safely released into the biosphere, they do not contain toxic substances and can be easily composted. The result is the reconstitution of natural capital, extremely important for the well-being of ecosystems and terrestrial species. On the other hand, the "technical" components are designed to recirculate at the highest possible value, without connecting to the biosphere (McDonough and Braungart, 2003). In addition to the Cradle to Cradle design, other theories have inspired the Circular Economy concept since the ‘70s, such as the, the Performance Economy (Stahel, 2006), the theories of Industrial Ecology (Lifset and Graedel, 2001), the Blue Economy (Pauli, 2010).

The circular economy model is established on three main principles (Ellen MacArthur Foundation et al., 2015):

- The protection and growth of natural capital, through the control of limited quantities of non-renewable resources and the balancing of flows of renewable resources.
- The optimization of the resource productivity by means of the permanence in the "biological" or "technical" cycles of products, components and materials, at the highest possible value.
- The centrality of the identification of environmental impacts and a design that excludes negative externalities, in order to achieve system effectiveness.

The transition to the circular economy model can lead to several opportunities and benefits from the economic, environmental and social point of views (Ellen MacArthur Foundation et al., 2015). As regards the economic advantages, they would be the revenue increase from circular economy activities and the decrease of production costs due to the increase in the utilisation rate of resources as well as the employment growth and greater innovation, by means of new technologies and higher production efficiency. Several benefits for the environment would also occur, such as the reduction of carbon dioxide emissions into the atmosphere, the lower consumption of virgin materials, the reduction of both land use and the release of toxic substances. Finally, circular economy will also benefit the society and consumers, because there would be increased spending opportunities, due to products and services lower prices, customisation of products according to people needs and reduced products obsolescence (Ellen MacArthur Foundation et al., 2015).

## 2.1 European policies for circular economy

In the framework of sustainable development policies, the European Commission has developed in 2011 a Roadmap to Resource Efficient Europe (COM (2011) 571) (European Commission, 2011), which is part of the Europe 2020 strategy (COM (2010) 2020) (European Commission, 2010) for a competitive, inclusive and sustainable European economy. It aims at both increasing the productivity of resources and reducing the environmental impacts for the decoupling of economic growth from the environmental burdens. The Roadmap foresees several actions in this context, such as the development of

new sustainable production and consumption models, with the focus on the entire products life cycles, the reduction of waste and losses by 50%, with particular attention to the agri-food sector, the valorisation of waste, by means of recovery, recycling and regeneration; the financing of eco-innovation projects; the elimination of environmentally harmful subsidies, the protection of natural ecosystems and biodiversity; the reduction of land use and improvement of soil quality; the identification of improvement solutions for those sectors with a considerable environmental impact, i.e. food, construction and mobility.
Therefore, the Roadmap includes several main topics related to circular production and consumption systems and the development of the circular economy within the European market, in order to obtain a greater global competitiveness, to support the sustainable growth and the employment growth.
A further action plan, "Closing the loop – An EU Action Plan for Circular Economy" (COM (2015) 614) (European Commission, 2015) involves all phases of the products life cycle. According to EU, “Circular economy systems keep the added value in products for as long as possible and eliminates waste. They keep resources within the economy when a product has reached the end of its life, so that they can be productively used again and again and hence create further value” (European Commission, 2014b). The solutions proposed by the Action Plan include eco-design, the choice of sustainable production techniques, the creation of industrial symbiosis projects and the adoption of the Extended Producer Responsibility policies. The consumption phase is supported by the introduction of environmental and energy labelling for products, the use of eco-design to extend the product’s life, by its reuse and repair, with the aim to avoid waste production, the reduction of household waste, the promotion of Green Public Procurement, the development and favouring of sharing economy models to share products and infrastructures and to boost the consumption of services rather than of products. The waste management actions include the compliance with the waste hierarchy, established by the EU in 2008 (prevention, preparation for re-use, recycling, energy recovery, disposal), prioritizing the waste reduction by eco-design and the recovery of the highest possible value, also increasing the recycling rate.
Furthermore, the Action Plan highlights that a market for recycled materials and secondary raw materials will be promoted, including recycled nutrients, with the aim to reduce the extraction of virgin materials. New sources of investment to finance innovative projects in the circular economy field will be introduced as well. Finally, circular actions

will be developed for some priority sectors, such as: 1) plastics, to increase their recycling and biodegradability and the recycling of plastic packaging; 2) food waste, to develop a common methodology for quantifying them as well as measures to facilitate the food donation and the use of by-products for feed production; 3) critical raw materials, to encourage their recovery; 4) construction and demolition waste; 5) biomass and biological products to promote the efficient use of bioresources and to support innovation in the bioeconomy field (European Commission, 2015).

# 3 Circular economy in the agri-food sector

## 3.1 Problems of the linear economic model in the agri-food sector

The agri-food sector involves 40% of the European land, contributes to satisfy one of the most important human needs, i.e. nutrition, provides several ecosystem services essential for our planet as well as social and cultural and economic value (European Environmental Agency, 2015; Notarnicola et al., 2017). Nevertheless, several risks threaten the stability of the linear economic model and the agri-food sector will suffer significant direct consequences, if appropriate offsetting measures will not be established. The Department of Agriculture of the U.S.A and Food and Agriculture Organisation (FAO) estimate that by 2030 there will be a greater demand for crops, equal to 40-50% higher than that of 2010 (Ellen MacArthur Foundation, 2013b). Moreover, the doubling of agricultural food production during the past 35 years was associated with a significant increase (from 1.1 to 6.87-fold) in nitrogen and phosphorus fertilization, in the amount of irrigated cropland and in land for cultivation (Fantin et al., 2017; Tilman et al., 2002). Intensive agricultural production has thus had strong impacts on the diversity, composition, and functioning of the natural world ecosystems and on their capacity to provide society with a variety of ecosystem services (Fantin et al., 2017; Tilman, 1999).

The combined use of mineral fertilizers, pesticides and massive irrigation, generated great prosperity in agriculture, satisfying the growing demand for products. Nevertheless, the trend has been reversed in the last years: the land productivity has been reduced and it was not able to satisfy the demand of the growing world population. In the next few years, hectares of fertile land will decrease by 25-35% compared to the 1.5 billion currently cultivated (Ellen MacArthur Foundation, 2013b). In addition, due to an increasingly intensive and industrialized agriculture, another important form of degradation is soil nutrient depletion. The high use of mineral fertilizers has caused negative imbalances in the soil characteristics, causing an excess of nutrients withdrawn from the soil compared to input nutrients, eutrophication phenomena, the destruction of biodiversity and the increase in the concentration of greenhouse gases in the atmosphere, due to increasingly specialized agricultural techniques based on fossil fuels consumption (Ellen MacArthur Foundation, 2013b). As regards direct environmental impacts, the food and drink production chain in the EU causes 17% of the greenhouse gases direct emissions and 28%

of the natural resource use and the consumption of meat requires involves a huge utilisation of water (European Commission, 2011).

A further critical issue in the agri-food sector is the loss of value throughout the supply chain, e.g. crops damage due to climatic and environmental conditions, losses of agricultural products which do not comply with market standards and the degradation during transport and storage (Ellen MacArthur Foundation et al., 2015). The use of water for agricultural purposes is a quarter of the total water demand, and almost 25% is lost during the transfer to the final point (European Environmental Agency, 2012). In the processing phases, 8% -12% of inputs are lost, without contributing to the final value, frequently due to non-optimised processing techniques or to strict specifications for finished products (FAO, 2011). The problem of the lost value in the distribution phase concerns mainly developing countries, where, in the post-harvest phases, the conservation and the sale of agricultural products are not efficiently managed. In developed countries, this loss occurs mainly in the use phase, where a large quantity of food is purchased but not consumed (FAO, 2011). It is estimated that around 30% of the food produced is wasted (FAO, 2011). In particular, 90 million t of food are wasted every year in Europe, equal to 180 kg per person (European Commission, 2011). Finally, a great loss of value occurs in the post-consumption phase, where large quantities of food waste are not further recovered and are treated as waste (FAO, 2011). The agricultural sector is therefore a major contributor to the waste stream stemming from the linear production and consumption model. Because of these reasons, the lack of environmental sustainability can negatively affect the functioning of the agri-food supply chain, in terms of production of safe food with a fair cost, and more in general the competitiveness of the agri-food industry (European Commission, 2014a).

Therefore, a transition towards circular economy models in the agri-food supply chain, which include sustainable production and consumption patterns, is required, which would increase system productivity while decreasing its environmental impacts (European Commission, 2014a). Without this systemic shift, the environmental impacts of the agri-food supply chain will increase significantly in the next years and probably they will exceed the planetary boundaries (Notarnicola et al., 2017).

The circular economy model aims to overcome the limits of the current system, moving from maximizing the performance of individual elements to optimizing the entire agro-food system: the increase in performance must be followed by the improvement of quality of soil, water and air (Ellen MacArthur Foundation et al., 2015).

The application of circular economy concept in the agri-food production chain involves the reduction of waste, the utilisation of by-products and food waste, the recycling of nutrients (Jurgilevich et al., 2017), the sustainable use of resources (soil, land, water, biodiversity), the use of renewable natural resources (i.e. biomass), the avoidance of food waste and surplus (Rood et al., 2017), the production and consumption of products with a better environmental performance throughout their life cycle, the application of both cleaner technologies and eco-innovation in production processes. In these ways, the resources will be used efficiently within a life cycle of a product, and waste produced will be both minimized and re-used as much as possible in other production chains, thereby providing economy with added value and causing lower environmental impacts (Rigamonti et al., 2017). All the above measures must be implemented both at the producer and consumer levels and in the waste management systems (Jurgilevich et al., 2017).

## 3.2 Benefits of the circular economy in the agri-food sector

The benefits of maintaining the components of agro-food products within "biological" cycles can be summarized in three macro-categories: the supply of new raw materials, such as the bio-chemical substances contained in waste, soil regeneration and energy production (Ellen MacArthur Foundation et al., 2015). The extraction of components with excellent chemical-physical properties takes place within bio-refineries, which process organic materials, such as agricultural residues and food waste, to obtain chemical and biofuel substances. The food industry can capture all the value contained in waste and by-products by exploiting the cascade use. The involvement of all stakeholders, such as industry associations and government bodies, as well as companies, is essential for creating favourable conditions for new business ideas (Ellen MacArthur Foundation et al., 2015). A major role to obtain these objectives is played by technological and process innovation. A main feature of the circular economy in the agri-food sector is the capacity of soil restoration in order to promote a higher fertility rate, thus increasing crop productivity. Manure and other food and animal waste can be used for this purpose, in order to avoid the massive use of chemical fertilisers. Finally, energy can be obtained from food waste through anaerobic digestion (which produces also digestate with good fertilising properties) and waste incineration (Ellen MacArthur Foundation et al., 2015).

Several advantages can stem from the redesign of the agri-food sector in a circular economy approach. The annual expenditure of food products per family would be reduced by 25% by 2030 and by 40% by 2050, thanks to the decrease in food waste (Ellen MacArthur Foundation et al., 2015). From the environmental point of view, there would be significant reductions in the use of pesticides, chemical fertilizers, energy, soil, water and in the emissions of greenhouse gases. There would be also a job growth due to the increase in the organic farming practices and to waste management systems. In economic terms, the implementation of the circular model would bring an economic benefit of € 320 billion compared to the current system, due to the reduction of costs for primary resources procurement and the decrease in externality costs (Ellen MacArthur Foundation et al., 2015). It is important to highlight that, in the transition to a new agri-food system, the introduction of innovative technologies and systems aimed at reducing waste should be coupled with policy actions to promote the resource efficiency goals, the restoration of natural capital and the production of high-quality products (Ellen MacArthur Foundation et al., 2015). Moreover, the transition to circular economy in the agri-food production chain requires joint efforts of farmers, food companies, retailers and consumers and the use of resource efficient production techniques (e.g. precision agriculture practices, organic agriculture and digitalisation of supply chains) as well as sustainable food choices and a decrease in food waste (European Commission, 2011; Ellen MacArthur Foundation et al., 2015).

# 4 Life Cycle based methods to support the transition towards circular economy in the agri-food sector

The transition towards sustainable consumption and production systems in the agri-food sector, in a circular economy approach, requires the use of robust and scientific methods and tools which can support sustainability assessment of the overall analysed system for several sustainability indicators, by appropriately evaluating the consequences of all the possible circular options, both from an economic, environmental and social point of view. For the assessment of the circular economy impacts, the application of the Life Cycle Thinking (LCT) approach, which takes into account the whole life cycle, can be an adequate solution with many benefits.

At international level, it is recognized that sustainability assessment must be based on a LCT approach, which aims to identify improvement opportunities for all phases of products life cycles, in terms of reduced environmental impacts and greater resource efficiency, thus avoiding burden shifting from a phase to another of the life cycle, and from an environmental compartment to another (Fantin, 2012). This holistic vision of the production system allows to consider the contribution of each process which fulfils the function for which it was designed. Cooperation along the value chain is essential to reach these goals, for sharing all the information and knowledge required for a complete and detailed study.

Par. 4.1 and 4.2 of this chapter are partially based on the following publications, performed during the PhD:

- Chiavetta C., Fantin V., Cascone C., 2017. **L'Economia Circolare nel settore agroalimentare e il Life Cycle Assessement come strumento a supporto: il progetto FOOD CROSSING DISTRICT**. ENEA Technical report, USER-PG64-003, June 2017 (Confidential).
- Ferrara M. Fantin V., Righi S., Chiavetta C., Buttol P., Bonoli A., 2017. **Applicazione della Water Footprint sviluppata dal WF Network: il caso del Pomodorino del Piennolo del Vesuvio DOP**. In Proceedings of XI Conference of Italian LCA Network Association, Siena, 22-23 June 2017, ISBN 978-88-8286-352-4.

## 4.1 Life Cycle Assessment

From the environmental point of view, Life Cycle Assessment (LCA) is an internationally accepted and standardised method (ISO 2006 a, b) which is recognized as a strategic and effective tool to evaluate the potential environmental impacts occurring in the whole product's life cycle as well as to identify possible areas for improvement (Fantin et al., 2014). Because of these reasons, LCA method could support the analysis of the impacts and benefits associated to circular solutions, also by a preventive approach, thus contributing to increase the sustainability of current sustainable production and consumption models (Sala et al., 2017). Since the LCA analysis considers the entire agri-food value chain, both the identification of food systems environmental impacts and the consequent improvement solutions aim to increase the resource and energy efficiency of the supply chain while decreasing their environmental burdens (Notarnicola et al., 2017).

LCA method is based on the compilation, quantification and evaluation of all inputs and outputs, in terms of materials and energy, waste and emissions, and the associated environmental impacts, throughout the entire life cycle of a product ("from cradle to grave"), thus including the extraction and processing of raw materials, the manufacturing of the product, its transport and distribution, the use, collection, storage and final disposal of the related waste (Figure 2).

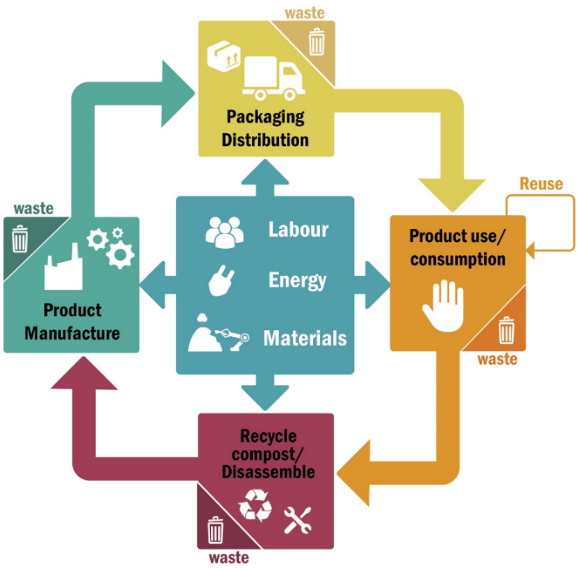



**Figure 2.** Schematic representation of product's life cycle (Source: ENEA).

LCT approach has been adopted by the European Union within the Integrated Product Policy (IPP) (European Commission, 2003) and in Sustainable production and consumption policy (European Commission, 2008) which propose the application of several actions to promote the continuous improvement of products environmental performance throughout their entire life cycle. LCA method and LCT approach are used also in environmental communication. In fact, they are the basis of both ISO 14020 compliant ecological labels, such as the European Ecolabel, and the Environmental Product Declaration (EPD) and Green Public Procurement. LCA can therefore support companies in the identification of opportunities for the improvement of the environmental performance of products, in the selection of key environmental indicators for monitoring their environmental performance, in marketing processes, e.g. to obtain ecological product labels and in the eco-design of product and processes.

The first examples of LCA method were in the 1960s and 1970s in the USA but the interest in its application grew in the 1990s. In 1993 the Society of Environmental Toxicology and Chemistry (SETAC) defined as LCA as an objective assessment of energy and environmental impacts related to a product, process or activity, carried out by means of the identification of energy and materials consumption and waste released into the environment (SETAC, 1993). The evaluation includes the entire life cycle of the product, process or activity, including the extraction and processing of raw materials, manufacturing, transportation, distribution, use, reuse, recycling and final disposal (SETAC, 1993). In 1998 SETAC created a series of guidelines that were then included in the ISO 14040 standard. According to ISO 14040 standards (ISO, 2006a, b), it consists in the "compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle" (ISO, 2006a) and involves four main phases: the goal and scope definition, the Life Cycle Inventory analysis (LCI), the Life Cycle Impact Assessment (LCIA); and the Interpretation of results (Figure 3).

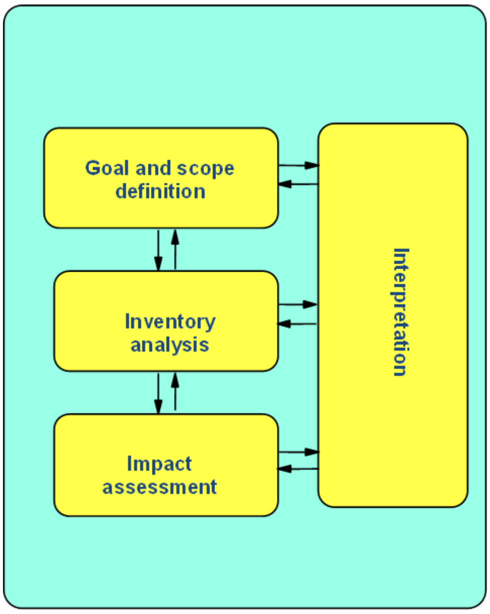



**Figure 3.** Four main phases of LCA method (Source: Personal elaboration adapted from ISO, 2006a).

LCA method shares with the circular economy the perspective connected to the consideration of the life cycle as a whole. LCA can thus support effectively the implementation of circular economy principles, because it allows to choose the solution with the lowest environmental impacts. More in detail, the application of this method can contribute to verify the hypotheses formulated in the business planning phase, highlighting for example any negative consequences due to a particular configuration. In addition, LCA can help to identify improvement opportunities by the evaluation of the possible alternatives and limits of the current scheme and then it could provide new ideas for the design phase. Finally, it can support the formulation of new objectives at a strategic level, by means of the creation and monitoring of specific performance indicators to encourage continuous environmental improvement (Chiavetta et al., 2017).

Circular economy offers a vision which can influence the way companies and governments operate. The support of a scientific method such as LCA can guarantee that this vision is translated into concrete benefits for people and for the natural capital. Ultimately, LCA provides substantial quantitative measures on which choices at product level are based, thus demonstrating its potential as a complementary tool for the circular economy (Chiavetta et al., 2017).

## 4.2 Water Footprint

Another life-cycle based tool developed in the last years and progressively used in the agri-food sector, also for communication purposes, is Water Footprint (WF), which is an assessment of the water use by products, individuals, companies or the entire population.
Water is indeed a natural resource essential to support human life and activities, although the problem of water scarcity is recognized as one of the major environmental issues at world level (Manzardo et al., 2016). According to a recent FAO report (FAO & WWC, 2015), in the next decades agriculture will continue to be the major contributor to water use and water pollution, contributing on average for more than half of water withdrawals from rivers, lakes and aquifers. In fact, the quantity of global water used for irrigation purposes is estimated to increase from 2,600 $km^3$ in 2007 to 2,900 $km^3$ in 2050, with a great increase especially in developing countries (FAO &WWC, 2015). Moreover, the increasing water demands in urban areas and businesses will decrease the volume of water available for agricultural production (FAO &WWC, 2015).
In Europe, one third of the water consumption is utilised in the agricultural sector, which influences both the quantity and quality (i.e. pollution from pesticides and fertilizers) of water available for other applications (EEA, 2012). In the southern European countries, such as Italy, Greece and Spain, about 80% of water used in agriculture is for irrigation purposes, due to the semi-arid climate conditions (EEA, 2012). Italy is one of the European countries that mostly use irrigation (ISTAT, 2014). The volume of irrigation water used in Italy in 2009-2010 was 11,618 milion of $m^3$. Most of the water use takes place in agricultural systems located in North-Western Italy (6,800 $m^3$/ha) and in North-Eastern Italy (2,500 $m^3$/ha), followed by Central and Southern regions (3,500 $m^3$/ha).
The problems of water scarcity and pollution occur locally, but the protection and efficient management of water resources must be pursued at national and transnational level, as implemented by the "Blueprint to safeguard Europe's water resources" (COM, 2012/0673). In this context, it is therefore essential to have tools and methods to understand how we are influencing the use of this resource with our production and consumption choices as well as to evaluate the results of companies and governments policies for sustainable water use (Ferrara et al., 2017).

In the last years, the WF concept has emerged, which is an assessment of the water use that has created a lively discussion within the scientific community (Pfister et al., 2017), because two methods, both based on a LCT approach, have been developed in parallel:

1) The Water Footprint Network (WFN) (www.waterfootprint.org) developed and published in 2011 the Water Footprint Assessment Manual (Hoekstra et al., 2011), which underlines the necessity to involve consumers and producers in the water management along the production chain;
2) The LCA community has developed methods to include the environmental impacts of water use throughout the product's life cycle in LCA studies and has contributed to the definition of the underlying concepts of the ISO 14046 standard (ISO, 2016).

The product WF developed by the WFN is defined as the volume of freshwater used throughout the production process. This indicator provides a measure of the amount of water available which is used by humans, dividing water into three components: blue, green and grey. The blue component refers to the consumption of water taken from a surface water body or groundwater; the green component expresses the consumption of rainwater that, once have reached the soil, does not flow or recharge the groundwater, but is used in the evapotranspiration process of the soil-plant system; the grey component is the volume of fresh water required to bring the concentration of a given load of pollutants below the maximum values established by legislation (Hoekstra et al., 2011).

The WF according to ISO 14046 is based on the LCA method (ISO 14044) and is defined as a measure that quantifies all the potential environmental impacts related to water used or influenced by a product, process or organization. According to ISO 14046, a WF evaluation means that all the potential impacts related to the use of water are taken into account, otherwise the indicator to which it refers must be specified (e.g. "water scarcity footprint" or "water eutrophication footprint"). In the framework of the Life Cycle Initiative, the UNEP/SETAC WULCA (Working Group on Water Use in LCA) has dealt with the problem of harmonizing and achieving consensus around an impact assessment method for freshwater consumption. Currently, it has developed the midpoint indicator AWARE (Available WAter REmaining for area in a watershed), which represents the water available per unit of surface which remains in a basin after having satisfied the demand from humans and ecosystems (UNEP / SETAC, 2016). Characterization factors for this method have been developed per year and country, for agricultural and non-agricultural uses. This indicator only evaluates the blue water scarcity and it does not consider rainwater (i.e. green water) (Boulay et al., 2013). Both methods have been applied to the agri-food production chain in the past 10 years (Zhang et al., 2017) and can be useful to support a sustainable management of water resources (Boulay et al., 2013) as well as the development and implementation of green marketing strategies addressed to both companies and consumers (Symeonidou & Vagiona, 2018). Moreover, some recent studies combine the use of WF Network method and ISO 14046 to compare the consistency of the obtained results and to evaluate both their strengths and weaknesses (Manzardo et al., 2016; Bai et al., 2018).

## 4.3 Application of LCA in the agri-food sector: main methodological problems

LCA method has been increasingly applied to the agri-food sector for the evaluation of the environmental impacts of a wide variety of agri-food products, contributing to both identify the environmental hotspots of the supply chain and the improvement opportunities, thus supporting political and institutional decisions (Notarnicola et al., 2012). Nevertheless, when applying LCA to food and drink products, the practitioner has to deal with methodological problems which stem from the peculiarities and specific characteristics of the agri-food supply chain. In fact, differently from industrial production systems, these systems have complex relationships, which are difficult to be modelled in LCA studies, between inputs (for example nutrients and soil) and outputs (crops and emissions), between cultivation techniques and the maintenance of long-term soil quality and in general terms between biological and technological processes (Fantin, 2012). However, when LCA is used to communicate the environmental performance of agri-food products, the different scientific approaches used to deal with the methodological issues might lead to different LCA results, making it difficult to compare the environmental performance of products of the same category. The European Commission' Product Environmental Footprint (PEF) recommendation, developed in 2013 (European Commission, 2013a), and the integrating documents aim to fulfil the need for harmonisation to calculate and communicate environmental footprint of products.

In particular, some critical issues on which different scientific approaches have been applied are the definition of the functional unit, the system boundaries and the allocation procedures, the calculation of on-field emissions from the use of fertilizers and pesticides at inventory level, the comprehensive assessment of toxicity impacts from the use of pesticides and the definition of some impact categories indicators, such as water use or land use (Notarnicola et al., 2012). Moreover, LCA method does not take into account in a comprehensive way some specific features of agri-food production chains, which are important to maintain the sustainability of food production, such as the decrease in soil quality, including the long-term effects of agricultural practices on the soil fertility, the increase in soil erosion, the reduction of ecosystem services, the loss of biodiversity and the nutritional and organoleptic properties of food products, which are difficulty to be quantified (Fantin, 2012). It is also important to point out that the analysis of some of the above aspects should require information at the landscape level, which is not accounted for in LCA method (Notarnicola et al., 2017).

Besides, other main challenges in LCAs of food systems are the the assessment of land use change, the lack of spatial and temporal characterisation in food LCA databases and life cycle impact assessment models and the exclusion of cultural values, which can affect the food production and consumption as well as the evaluation of the environmental impacts of agri-food supply chains (Hauschild et al., 2012; Notarnicola et al., 2017).

In the following paragraphs, some of the main critical methodological issues of the application of LCA method to agri-food supply chain will be analysed.

### 4.3.1 Definition of the functional unit

The functional unit is defined by ISO 14040 as the quantified performance of a product system, to be used as a reference unit. Therefore, it provides the reference to which inputs and outputs of the studied system will be connected and it is necessary to ensure the results comparability among different products with the same function. The functional unit is thus a crucial point in defining the scope of the LCA study, and is connected to the goal of the study, to the function of the product and to the analysed system.

It is important to highlight that the choice of different functional units for the same product system can affect the results of the study (Notarnicola et al., 2012).

According to Reap et al. (2008), the definition of the functional unit is one of the major methodological problems in the goal and scope definition of the LCA method. The authors explain that several sources of errors can occur when functional unit is chosen. For example, an inaccurate definition of the functions of the system can lead to a wrong representation of the reality. Moreover, products often have multiple functions, which should be considered to properly reflect reality, or their function could be not easily quantified. In this context, a further problem arises, i.e. how to choose the most proper functional unit which represents more functions (Cooper, 2003; Reap et al., 2008).

In LCA studies of agri-food systems, the main function of the system is considered to be human nutrition (Nemecek et al., 2016) and does not include multiple function that the product may have, such as the energy or nutrient content or the provision of a pleasant texture or taste (Reap et al., 2008, Shau and Fet, 2008). Therefore, the functional unit is frequently based on the mass or volume of the product (Roy et al., 2009, Schau and Fet, 2008; Kendall and Brodt, 2014; Roma et al., 2015; Petti et al., 2015). However, alternative functional unit may also be chosen, for example the nutritional or economic values of the product and land area.

The choice of mass or volume for the identification of the functional unit can be seen as a simplified approach, because the agricultural system is supposed to have one only function, i.e. the production of the main product. Moreover, this kind of functional unit can support the comparability between LCA studies of products of the same product category, but it does not consider the benefits derived from these products, especially their nutritional values (Heller et al., 2013). Moreover, the use of a physical amount of output as functional unit for food products can be misleading for consumers, because sometimes an inverse correlation exists between the yield and product's quality. In fact, several authors found that organic products, which are perceived by consumers as added value products from both environmental and health point of views, and local typical productions can have higher impacts per unit of product, due to lower production yield (Tuomisto et al., 2012). Similarly, many LCA studies highlight that intensive agriculture practices leads to lower environmental impacts per functional unit, when it is defined as mass or volume, because they have higher yields, which split the environmental impacts of the product to a larger amount of output (Notarnicola et al., 2017; Sonesson et al., 2016; Kulak et al., 2013; Kiefer et al., 2015).

Other possible functional units found in literature are the cultivated area (Shau and Fet, 2008; Petti et al., 2015), the energy or protein content (Shau and Fet, 2008; Renzulli et al., 2015), the content of glucose and fructose (Renouf et al., 2008), the serving size (Roma et al., 2015), the dry biomass production (Renzulli et al., 2015) and the economic value Notarnicola et al., 2017), although different types of external factors, which are not always connected to product's quality, can affect products' prices (Notarnicola et al., 2017).

Due to the different functions of the food product systems, it could be useful to adopt multiple functional units in order to evaluate the variability of LCA results and to increase its robustness, thus analysing the product system from different perspectives (Notarnicola et al., 2012). Therefore, several authors used multiple functional units in the same study, such as the mass of products, the energy or protein content, the cultivated area, the unit of livestock and the economic value (Sala et al., 2017**)**. For example, Salou et al. (2016) used two functional units for the assessment of environmental performance of dairy systems: t of milk and area of land occupied and demonstrated that different functional units lead to completely different results (Sala et al., 2017). The same results were obtained by Kendall and Brodt (2014) and Van Kernebeek et al. (2014) (Nemecek et al., 2016). Similar results were obtained also by Cerutti et al. (2014), which compared the same fruit production chains with different functional units. More in detail, fruit cultivars with higher yields have lower environmental impacts when using mass as a functional unit, whereas fruit cultivars with lower yields show a better environmental performance when land areas is used as functional unit.

A further aspect that should be taken into account in the definition of the functional unit of agri-food products is the quality of the product, expressed not only as nutritional content, but also as taste, texture, cultural, social and ethical values, which can be connected to the territory and landscape protection. Nevertheless, since it is not possible to include all these issues in a single functional unit, the practitioner must unambiguously declare the chosen function and analyse also the consequences of their choice (Nemecek et al., 2016). For example, 1 L of virgin olive oil cannot be compared with 1 L of olive oil, because these products are different from the qualitative point of view (Salomone et al., 2015). Consequently, it should be highlighted that the definition of the most correct functional unit is rather difficult when product quality is included in the system function (Notarnicola et al., 2012).

A further important feature for the identification of the most suitable functional unit for food products is the problem of food waste. In fact, wasted food increases the environmental burdens of the consumed food, because a higher quantity of product is required to satisfy the product’s function (Cerutti et. al., 2014; Nemecek et al., 2016). Therefore, the quantification of food losses throughout the supply chain should be included in the assessment in order to evaluate the actual environmental performance of the product consumed (Cerutti et al., 2014; Ingwersen, 2010), although the quantification of food waste in the different stages of the life cycle is often difficult and is also affected by consumers practices (Nemecek et al, 2016). In some literature studies on fruit and vegetables production, the environmental impacts of discarded products were attributed to the functional unit, which was defined as the net yield or the marketable yield (Mogensen et al., 2015; Romero-Gámez et al., 2014; Corrado et al., 2017). Manfredi M. et al. (2015) defined the functional unit as the delivery of eaten food, as suggested by Wikstrom et al., (2014).

Finally, it is important to point out that the choice of the functional unit based on mass, volume or nutrient content seems to be more adequate for the communication to consumers, in order to direct them towards consumption models with a better environmental performance (Nemecek et al., 2016; Finkbeiner, 2014). Moreover, at the moment, this approach is methodologically the most robust and investigated and it is adopted by the European Commission's Product Environmental Footprint (PEF) method (European Commission, 2013a) within the Single Market for Green Products Initiative (European Commission, 2013b) for the assessment of the environmental performance of agri-food products.

### 4.3.2 System boundaries definition

According to ISO 14040-44, the selection of the system boundaries establishes the unit processes, activities and operations to be included in the LCA study. The choice of the processes to be included or excluded depend on the goal and scope of the study and both the data quality and availability (Salomone et al., 2015). System boundaries should include all the relevant life cycle stages and processes of the studied production chain. In fact, the environmental assessment of food products should at least consider the supply chain up to the point of sale, including agricultural production, processing, by-product management, transportation, distribution, although the whole supply chain, thus including also the transport up to the consumer, the food chilling and cooking and the waste management should be taken into account for a more comprehensive evaluation of environmental performance. (Nemecek et al, 2016; Salomone et al., 2105).

Nevertheless, in literature LCA studies of agri-food products, the lack of data on some specific processes frequently lead to their exclusion from the system boundaries, and the goal and scope of the study have to be re-delineated accordingly (Salomone et al., 2015). However, it is important to highlight that ISO 14044 permits the exclusion of life cycle stages and processes only if they do not change the results of the study, and that any decision must be clearly stated and the reasons and implications for their emissions must be explained (ISO, 2006). Moreover, an inappropriate selection of system boundaries or the exclusion of some unit processes can affect final results, thereby leading to incorrect considerations and decisions, also from the decision maker point of view (Reap et al., 2008). In particular, this problem affects the comparison between different studies, which can be rather difficult (Graedel 1998; Lee et al., 1995; Reap et al., 2008; Shau and Fet, 2008).

In literature agri-food LCA studies, the most used system boundaries are the following:

1) “from cradle to farm gate”, including only the cultivation phase (for fruits and vegetables supply chains) or the cultivation and animals breeding phases (for livestock supply chains) (Roma et al., 2015; Cerutti et al., 2015; Renzulli et al., 2015; Castanheira et al., 2010; Guerci et al., 2013a, b; Zhang et al., 2013; Khoshenivesan et al; 2013;
2) “from cradle to processing factory gate”, which include both agricultural processes and transformation and processing phases (Roma et al., 2015; Cerutti et al., 2015; Renzulli et al., 2015; Fantin et al., 2017; Gonzales-Garcia et al., 2013a; Opio et al, 2013).

Most studies therefore do not consider the consumption, use and end-of life of the product (Petti et al., 2015), which however have a major importance due to the great quantity of food which is wasted at households every year, equal to 46% of the total amount of wasted food at European level in 2006 (European Commission, 2014a), considering also that one third of the edible parts of food is wasted (FAO, 2011). In this context, LCA studies of food products could have an important role because they could contribute to evaluate the potential savings of avoiding food waste, although food waste is still frequently assessed incompletely in literature (Corrado al, 2017). In addition, both the consumer transport and food preparation can have remarkable impacts (Nemecek et al., 2016) and should be included.

### 4.3.3 Allocation procedures

Allocation means “partitioning the input and output flows of a process or a product system between the product system under study and one or more other product systems” (ISO, 2006a) and must be carried out when the system under examination produces two or more products, such in the case of many agri-food systems (for example milk and meat production).

This means that allocation can divide the total environmental impacts of the multi-functional process among each function or product. As already explained for the definition of system boundaries, inaccurate allocation can affect LCA results and lead to incorrect decisions (Reap et al., 2008). ISO 14044 suggests to use the following procedure for processes which have more than one outputs:

1. Avoid allocation by (1) separating multifunctional processes into sub-processes and collecting the data for each process and/or (2) expanding the system boundaries to include the additional functions of the co-products;
2. If allocation cannot be avoided, it should be performed according the underlying physical relationships between the different products or functions of the system;
3. If allocation based on physical relationships is not viable, the environmental burdens of each product should be allocated on the basis of other relationships (e.g. the economic value of products).

However, each of the above steps can have some limitations and difficulties (Reap et al., 2009). The subdivision of multi-functional processes involves the collection of more detailed information about the processes and should be performed when this information can be collected at an affordable cost, or when the processes are independent from the economic point of view (Shau and Fet, 2008; Ekvall and Finnveden, 2001). As regards system expansion, there are mainly two approaches for performing it: 1) system boundaries enlargement; 2) avoided burdens approach (Azapagic and Clift, 1999). This means that the environmental impacts of alternative ways to produce the product or the co-products are either added to the system or subtracted from it, with the aim to assure their comparability. Nevertheless, both approaches require the identification of alternative production systems suitable for substitution and the availability of inventory data for them (Azapagic and Clift, 1999; Niederl-Schmidinger and Narodoslawsky, 2008). In addition, system expansion leads to a more complex LCA model which requires more data to be collected (with more temporal and economic efforts) and to potential uncertainty of data quality (Curran, 2007; Reap et al., 2008; Notarnicola et al., 2017). Moreover, if the practitioner applies system expansion without substituting the additional function of the co-products, the functional unit will include more products. In this way, the product could not be assigned easily to a certain product category. For example, in the application of system expansion without substitution to a milk and meat production system, the function of the system would be the production of both milk and meat, which belongs to the product category food but not to the product category milk or meat (Shau and Fet, 2008). An example of system expansion with substitution in LCA studies of agri-food supply chain is that performed by Cederberg and Stadig (2003) who compare the results obtained with those coming from the application of economic allocation. In this study, the combined milk and meat production system is expanded by including an alternative production of meat, which is beef cows producing calves and meat. The environmental impacts of the alternative meat production are then subtracted from the environmental impacts of the combined milk and meat system, in order to obtain the environmental burdens of milk production system. Results show that economic allocation between milk and meat favours meat production, whereas when system expansion is applied, milk production has a better environmental performance. This is due to milk co-products (calves and meat from culled cows), because meat production in combination with milk can be carried out with fewer animals than in meat production system. Therefore, Cederberg and Stadig (2003) conclude that system expansion should be used to obtain more robust results.

As regards the allocation based on physical relationship, it could be difficult to be evaluated, because causality could be determined in several ways, or could not be easily understandable (Finnveden et al., 2000; Reap et al, 2008). An example of physical relationship in literature LCA studies of agri-food products is the biological causality between the amount and type of feed intake and the outputs meat and milk (Cederberg and Mattsson, 2000), the demand for feed needed for lactation, body maintenance and delivering calves (Eide, 2002) or the mass of olive oil and the mass of pomace produced by olive oil production process (Rajaeifar et al., 2014).

Finally, other relationships different from physical causality could be based on energy content, mass, volume, economic value and they are frequently used in LCA studies (Ekvall and Finnveden, 2001), mainly because they could be identified more easily and data could be readily available (Reap et al, 2008). In general, economic allocation is the most used approach in food LCA, because economic data can be easily obtained and the economic value of the product is the motivation of the production process (Pelletier et al., 2015; Notarnicola et al., 2017). In addition, agri-food products often have several by-products with low economic value (e.g. fat and skin in cow slaughtering). The application of economic allocation to these by-products attributes them low environmental burdens (Corrado et al., 2017). Furthermore, economic allocation can be applied in many different contexts, because price reflects also the quality of products (e.g. cultural aspects, taste, etc.) which cannot be measured by physical criteria and which drive consumers choices. In addition, these particular characteristics connected to quality are unique and therefore potential alternative processes cannot be identified, thus excluding the possibility to apply system expansion or substitution (Ardente and Cellura, 2012).

On the other hand, economic allocation has some limitations, because prices are subject to variability and sometimes prices and physical flows are not properly correlated. Both issues lead to uncertainties and affect the reliability of LCA results (Ardente and Cellura, 2011; Marvuglia et al., 2010). In addition, ISO 14040 and International Reference Life Cycle Data System (ILCD) Handbook (European Commission – JRC-IES, 2010) suggest to use market prices of the co-products as the basis for economic allocation, although this choice could be misleading, because prices should refer to the product's value immediately after its production and not to the final price at consumers, which could reflect external factors (European Commission, 2011). According to Pelletier et el. (2015), the LCA results obtained by economic allocation reflect market relationships (via price ratios) rather than physical relationships. Problems can also occur when market prices are not available. In those case, other sources can be used, such as historical prices and expected prices (Guinée et al., 2004).

As regards food LCAs, several authors have applied economic allocation (Dalgaard et al. 2008; Blengini and Busto 2009; Gonzalez-Garcia 2013b; Beccali et al., 2010; Ayer et al., 2007). For example, Beccali et al. (2010) highlight that in the production of fruit juices and essential oils from citrus-based products, the essential oils are a small mass fraction of company's products, but they have a significant market price. In this case, substitution and system expansion are not possible because alternative production processes for essential oils cannot be identified; therefore the authors apply economic allocation because it reproduces more correctly the causality of the production process. In the same way, Gazulla et al. (2010) apply economic allocation to wine production because system expansion is not viable in this case, since grapes residues and fermentation sediments cannot be produced separately, and no detailed information is available about alternative products which could be substituted by pomace and other by-products.

In conclusion, allocation procedures and more in general the methods for solving the multi-functionality of agri-food systems are still a topic of discussion at scientific level.

### 4.3.4 Emission models for pesticides and fertilisers emissions

Pesticides are applied intensively to agricultural fields in order to increase production yields per unit of area and to control plant diseases. Nevertheless, these products can result in severe environmental contaminations (Verna et al, 2014). Agricultural pesticide use is substantial and amounts globally to about 1.2 million metric t of active ingredients (the biologically active part of a commercial pesticide formulation) per year between 2005 and 2010 (FAO 2013). As reported by Verna et al. (2014), it has been verified that pesticides applied to crops cultivation are degraded in several ways described as biotransformation, biomineralization, bioaccumulation, biodegradation, bioremediation and cometabolism (Shakoori et al., 2000; Park et al., 2003; Finley et al., 2010).

Pesticide biodegradation is a soil microbial function of critical importance for modern agriculture and its environmental impact (Dechesne et al., 2014). A few biodegradation controlling factors have tentatively been identified across pesticide classes. They include some soil characteristics as pH (Rodriguez-Cruz et al., 2006; Bending et al., 2001, 2006; El Sebai et al., 2007; Hussain et al., 2013; Rasmussen et al., 2005; Lauber et al., 2009; Franco et al., 2009; Houot et al., 2000), soil moisture (Parkin and Shelton, 1992; Stenrød et al., 2006; Rodriguez-Cruz et al., 2008; Monard et al., 2012; Hussain et al., 2013), C/N ratio and potassium (Rasmussen et al., 2005), organic matter and clay (Rodriguez-Cruz et al., 2006; Vinther et al., 2008). Other soil properties and characteristics partially related to tillage practice such as macropores (Nielsen et al., 2010), structure and fractures (Alletto et al., 2008) and soil horizons also have impact on pesticide degradation.

Pesticides and/or their degradation products are accumulated in diverse environmental compartments such as plant, soil, air, surficial and deep groundwaters and lead to severe environmental contaminations.

As explained above, LCA studies of agri-food systems have to face several main challenges, such as the modelling of emissions from pesticides and fertilisers use at inventory analysis level (Schmidt Rivera et al., 2017; Goglio et al., 2014), which can lead to important environmental impacts to freshwater and marine ecosystems as well as to terrestrial ones, due to their chemical, toxicological and eco-toxicological properties (Notarnicola et al., 2017). The criticalities in modelling are due to several factors. Firstly, since data about their production are frequently not available, the practitioner often uses estimates for the assessment, or excludes them from the study (Notarnicola et al., 2015; Fantin et al., 2017; Schmidt Rivera et al., 2017). Secondly, even when the active ingredients are considered, the pesticide and fertilizer formulation by-products (adjuvant, solvents, etc.) are omitted from the analysis (Rosenbaum et al., 2015). Finally, the calculation of on-field emissions due to the use of fertilisers and pesticides is rather difficult and frequently they must be estimated by dispersion models, because their direct measurement is not viable in LCA studies.

As regards the estimation of on-field emissions from pesticides at inventory level, the main difficulty for the LCA practitioner is to quantify the amount emitted to the different environmental media (i.e. air, soil, water), because the practitioner frequently knows only the quantity applied to the agricultural field (Rosenbaum et al., 2015). Furthermore, other problems can affect the inclusion of pesticide emissions, such as the lack of knowledge about toxicological properties of chemicals (European Commission, 2001) and the complexities involved in toxicity effect modelling (IEA Bioenergy, 2015) (Fantin et al., 2016). Moreover, the calculation of the dispersion of pesticides is particularly complex since their application can have different destinations (i.e. crop, air, soil, water, humans and the terrestrial and aquatic fauna) and different kinds of dispersion processes can occur (e.g. surface run-off, leaching, volatilization, degradation, absorption or desorption into the soil). Furthermore, other important parameters, such as on-site soil and climate conditions, the location of the water table, farming practices and pesticide's chemical and toxicological properties affect the fate of pesticides active ingredients and its interaction with the environment (Notarnicola et al., 2012).

Currently, several literature models are available for the estimation of fate and transport processes of pesticides after the field application, such as leaching, volatilisation, crop uptake as well as influencing factors such as application method, crop and soil type, temperature (Rosenbaum et al., 2015). Till the 1990s, Life Cycle Impact Assessment (LCIA) methods, such as CML 92 (Heijungs et al., 1992) or Eco-Indicator 95 (Goedkoop, 1995), did not consider pesticides' fate in the environment for their impact assessment but only their toxicity and ecotoxicity (Margni et al., 2002; Kramer, 2003). During the 1990s, the LCA scientific community started to propose methods for predicting the fate of pollutants, pesticides included, in the different environmental compartments. In 1998, Eco-Indicator 98 for assessing the average damage caused by a product in Europe was presented (Goedkoop et al., 1998). Eco-Indicator 98 included a fate model that calculated a temporal pollutant concentration increase over the European regional scale where wind speed and runoff were minimized in order to avoid transfer emissions out of the regional scale. Later, Huijbregts et al. (2000) proposed the fate, exposure and effects model USES-LCA. The model calculates the distribution of 181 substances, among 86 pesticides, through a 'distribution module' consisting of local fate models and a nested multi-media fate model with three spatial scales (regional, continental and global). USES-LCA model considers six compartments: air, freshwater, seawater, natural soil, agricultural soil, and industrial soil; the groundwater compartment is not included. Wind speed and water flow are not minimized. Two years later, Margni et al. (2002) presented a model focused on LCIA of pesticides. The model simulates pesticide fate in air, water and soil first referring to residence times, dilution volumes and transfer factors. The model assumes that 10% of the applied active ingredient remains in the air (or returns to air due to subsequent volatilization), 85% enters the soil and 5% remains on the leaves. These hypotheses concerning emissions into the different environmental comparts are based on the assumptions reported by the European Commission in its report for the harmonization of Environmental Life Cycle Assessment for Agriculture (Audsley et al., 1997). The model calculates also the active ingredient transfer to both surface waters (using the model of Leonard et al., 1987) and ground waters (using the model of Jury et al., 1987), and it determines the amount of pesticide residues in the food (from tolerance values). In the meantime, Ecoinvent Database proposed the accounting of pesticides application as full emissions into agriculture soil (Nemecek and Erzinger, 2005; Nemecek and Kägi, 2007). In 2006, a more sophisticated model called PestLCI 1.0 was developed to overcome the restrictions and data requirements of Environmental Risk Assessment models (Birkved and Hauschild, 2006). The model provides simultaneous assessment of the emission fractions of a pesticide to air, surface water and groundwater based on the application method, local climate conditions, crop type and soil data. PestLCI 1.0 provides data on one characteristic Danish soil, active ingredients in 69 pesticides and one Danish meteorological station and it is prepared for implementing and using data of other regions and other ingredients. In 2012, Dijkman et al. presented PestLCI 2.0, an updated and expanded version of PestLCI 1.0 (Dijkman et al., 2012). It was developed to overcome the limitations of the first model version and introduced new modules for macropore flow and effects of tillage. Furthermore, climate, soil and active ingredient databases were expanded widening the scope and applicability of the model to Europe (Dijkman et al., 2012). PestLCI 2.0 emission model considers that both primary and secondary processes determine the fate of the pesticide. Primary processes occur directly after pesticide application: a part of the applied pesticide drifts away from the field, the remaining part is distributed over the crops in the field and the field soil. Then, three secondary fate processes on leaves occur: volatilization, degradation and uptake (Dijkman et al., 2012).

All these approaches are often quite complex for the LCA practitioner and require the use of primary data on soil and weather conditions, the application method, the irrigation and crop characteristics, which can be difficult to obtain. Because of these reasons, the quantification of life cycle emission inventories of pesticides in LCA studies is not harmonised, and the diversity of these approaches lead to different results, thus compromising the reliability of LCA outcomes for agricultural products especially in case of comparative studies (Notarnicola et al., 2015; Rosenbaum et al., 2015). For this reason, a consensus on the delineation between pesticide emission inventory and impact assessment for LCA is necessary (Rosenbaum et al., 2015). This opinion is reiterated in the PEFCR Guidance (European Commission, 2017) where, as temporary approach, the following pesticide distributions are suggested: 90% emitted to the agricultural soil compartment, 9% emitted to air and 1% emitted to water. The PEFCR Guidance asserts that the PestLCI 2.0 model might fill in the gap between the amount of pesticide applied on the field and the amount ending up in the emission compartment in the future, but is currently still under testing (European Commission, 2017).Some recent survey conducted on LCA studies of crop production highlighted that the simplified approach by Margni et al. (2002) has been applied frequently, although several literature studies omit emissions of pesticides in air, soil and freshwater, due to the above-mentioned difficulties (Rivera et al., 2017; Fantin et al., 2017). Another simplified approach frequently adopted is that by Ecoinvent Database, one of the most used the life cycle inventory database, which assumes that 100% of the applied active ingredient is emitted to the agricultural soil (Nemecek and Kägi, 2007).

In those cases, assumptions are characterized by a high degree of generalisation (Fantin et al., 2017). For example, in the Ecoinvent approach, since no pesticide is considered to be emitted to surface water, the aquatic toxicity impact category is not affected by any applied pesticide. Evidently, these diverse approaches may lead to different results in terms of affected impact categories and absolute emission results, affecting the reliability of LCA outcomes for agricultural products of the same category, especially in case of comparative studies (Notarnicola et al., 2012).

In accordance with the statement of the PEFCRs Guidance, PestLCI 2.0 appears a very promising tool for assessing the emissions inventory of pesticides application and its use has recently increased in LCAs of food products (Niero et al., 2015a; Dijkman et al., 2017; Schmidt Rivera et al., 2017; Bacenetti et al., 2014; Bacenetti et al. 2015).

Nevertheless, while the simplified approaches defined by Margni et al. (2002) or Nemecek and Kagi (2007) do not require the detailed information in terms of on-site data, the use of more sophisticated models such as PestLCI 2.0 requires detailed data about the application method as well as soil and climate data of the studied region which can be difficult to obtain for the LCA practitioner. Therefore, further applications of PestLCI are required in order to improve the model and to increase its applicability in LCA studies. Moreover, the use of pesticides affects also the Life Cycle Impact Assessment phase, which calculates their potential toxicity anc ecotoxicity impacts. In this regard, the new version of USEtox method (USEtox 2.0, 2013) recommended by ILCD Handbook (Rosenbaum et al., 2008), provides the practitioner with characterisation factors for some pesticides, which are used to calculate their freshwater ecotoxicity and human toxicity impacts. Nevertheless, the method is affected by a high degree of uncertainty and it does not provide characterisation factors for several active ingredients (Fantin et al., 2017; Garavini et al., 2017) and for both marine and terrestrial ecotoxicity (Notarnicola et al., 2017).

As far as fertilisers are concerned, lack of scientific consensus also occurs about the calculation of nitrogen emissions from fertilisers application, with particular reference to emissions of $N_2O$, $NH_3$, $NO_x$ to air and of nitrate leaching to the groundwater. Moreover, lack of data about field emissions in food LCAs is frequent, although the production and use of fertilisers is one of the most important contributors to environmental impacts in food LCAs (Fantin et al., 2017). This deficiency is largely because soil emissions to air and water are site specific (depending on soil conditions, hydrology, climate, management practices), and very dependent on the type of fertilizer applied (Achten and Van Acker, 2016). According to some recent reviews performed on cereals production (Fantin et al., 2017; Renzulli et al., 2015), emissions from fertilisers application are in most cases quantified by the Intergovernmental Panel on Climate Change (IPCC) method (IPPC, 2006), and to a lesser extent by other methods, such as European Monitoring and Evaluation Programme (EMEP)/ European Environment Agency (EEA) guidelines

(EMEP/EEA, 2013), the model by Brentrup et al. (2000), which requires site specific soil, climate and agricultural related parameters, or a combination of two or more literature models (Fantin et al., 2017).

# 5 Harmonised LCA guidelines for agri-food production chain

On the basis of the critical methodological problems explained in par. 4.3, methodological standardization and harmonization is required for the application of LCA to agri-food production systems, especially when it is applied for communication purposes in ecological labels, both in a Business-to Business (B2B) and a Business-to Consumers (B2C) perspective, with the aim to both provide robust and consistent methods for the calculation of environmental burdens of food and drink products, and to assure comparability among different studies of the same product category. In this way, the application of LCA method could support effectively sustainable production and consumption patterns in a circular economy approach.

Because of these reasons, different harmonized methods and guidelines have been developed at both international and European level. The European Commission’s Joint Research Centre has published in 2010 the ILCD Handbook, a series of technical guidelines for the application of LCA method to products and services with the general purpose to improve the consistency, comparability and quality of the LCA studies. The ILCD Handbook is mainly based on ISO 14040-44 but includes also further provisions and requirements (European Commission JRC-IES, 2010).

In 2013, in the framework of the European Communication “Bulding a Single Market for Green Products” (European Commission, 2013b), aimed at improving the methods for assessing and communicating to consumers the environmental performance of products, the European Commission has published the Product Environmental Footprint (PEF), a harmonised method for the calculation and communication of environmental footprint of products (European Commission 2013a). The requirements of the framework are based on ILCD Handbook and the ISO 14040 standard, as well as on other methods for the environmental assessment of products (e.g. PAS 2050, WRI/WBCSD GHG protocol, Sustainability Consortium, Ecological Footprint). A pilot phase took place from 2015 to 2018, with the participation of companies and stakeholders from several agri-food product categories, to develop Product Environmental Footprint Category Rules (PEFCRs), which provide supporting methodological guidelines for the application of the PEF method to specific product categories.

In parallel with the introduction of the PEF method and the relevant pilot phase, the European Food Sustainable Consumption and Production Round Table (Food SCP Round Table), established jointly by the European Commission and the European Associations of food supply chain, has developed the Envifood Protocol (Food SCP RT, 2013), which represented the first attempt to adapt the ILCD Handbook to the specific requirements of the food production chain. The Envifood Protocol is based on specific requirements for food and drink products and was intended to be used a sectoral guidance in the context of the PEF pilot phase. Finally, the Food and Agricultural Organisation (FAO) has established in 2015 the Livestock Environmental Assessment and Performance (LEAP) Partnership (http://www.fao.org/partnerships/leap/en/), which consists of different types of stakeholders from governments, private businesses, Non-Governmental Organizations with the aim to create harmonised methods, sector specific guidelines and data to support the sustainable development of the livestock sector, on the basis of the application of ISO LCA. To date, the LEAP partnerships has published sectoral guidelines for the environmental assessment of feed, poultry, pigs, small ruminants and large ruminants production chains (see for example FAO, 2018).

Anyway, it should be highlighted that the scientific debate on some of the above-mentioned critical methodological issues is still open. Therefore, scientific community will have to work on further research and development activities with the purpose to implement robust and harmonised methods which would support the application of LCA to agri-food systems, and the communication of the environmental performance of products, in order to increase their overall sustainability.

## 5.1 ILCD Handbook

The ILCD Handbook was developed by the Joint Research Centre of the European Commission in 2010 and consists of several technical guidelines for LCA practitioners and environmental technical experts of the private and public sector for the application of a harmonized LCA method to all kinds of products and services, aiming at improving the comparability, quality and robustness of both LCA studies and data (Wolf et al., 2012). The Handbook is compliant with ISO 14040-44 but includes also more detailed provisions, technical descriptions and stricter requirements and can therefore be used to develop product-specific criteria, guidelines and tools (Lindfors et al., 2012). Since it has to cover all the different situations in which LCA studies are performed, the Handbook includes a wide set of very general provisions (Cappellaro et al, 2011).

The ILCD Handbook "General Guide for Life Cycle Assessment- Detailed Guidance" (EC- JRC-IES; 2010) identifies three main level of decisional context in which the LCA method is applied and provides guidelines for each of them (European Commission -JRC-IES, 2010; Lindfors et al., 2012):

- "Micro-level decision support": this situation is applied to most LCA studies and refers to those studies aimed at supporting decisions related to products or processes (for example improvements, comparisons, environmental product labels). The decisions taken after the LCA study have "limited or no structural consequences outside the decision context", i.e. they do not change available production capacity" (EC JRC-IES, 2010);
- "Meso/macro-level decision support": this situation includes the support of decisions at strategic level (e.g. policy development, identifications of improvement potentials) which can have "structural consequences outside the decision-context", i.e. they can "change available production capacity" (EC JRC-IES, 2010);
- "Accounting": this situation includes the description and documentation of a system's life cycle, which does not have "any potential consequences on other part of the economy" (EC JRC-IES, 2010).

In addition to the "General Guidance", further documents of the ILCD Handbook are the following:

- Specific guide for Life Cycle Inventory (LCI) data sets;
- Analysis of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment (LCA);
- Framework and requirements for Life Cycle Impact Assessment (LCIA) models and indicators;
- Recommendations for Life Cycle Impact Assessment in the European context;
- Review schemes for Life Cycle Assessment;
- Reviewer qualification.

An LCA study is fully compliant with ILCD Handbook only when all the following issues are satisfied (Cappellaro et al., 2011):

- Data quality: this aspect refers to completeness, technological, geographical and time representativeness, precision/uncertainty, appropriateness and methodological consistency;
- Method: this aspect refers to the appropriateness of the modelling phase and to the compliance with methodological provisions and their consistency during their use;
- Nomenclature: consistency and accuracy of the nomenclature used to mention the flows and processes, the use of units of measurements and the technical terminology;
- Revision: appropriateness and accuracy of the type, method and documentation of the revision phase.

## 5.2 Envifood Protocol

The goal of the Envifood Protocol is to support the assessment and communication of food and drink products environmental performance both in B2B and B2C contexts as well as the identification of potential improvements (Food SCP Round Table, 2013). The Protocol is intended to be positioned between the PEF Guide and the PEFCRs developed within the pilot phase, and should be used as a complementary guideline to the PEF Guide (Saouter et al., 2014). The Protocol was the first attempt to develop harmonised LCA guidelines for the agri-food sector and should therefore support the practitioner during the modelling phase of an LCA study as well as contribute to increase the reliability and consistency of life cycle results.

However, the Protocol requirements are quite general, because the document covers the whole agri-food supply chain, and provides broad guidelines. For example, the functional unit should be the weight or volume of the product, system boundaries should include all relevant life cycle stages, from cradle-to-gate for B2B communication applications and from cradle-to-grave for B2C purposes. Specific recommendations are given to the definition of the use phase and waste management. The ISO 14040 hierarchy must be applied for handling of multi-functionality and primary and secondary data should comply with ILCD Data Network entry level requirements. Finally, impact categories and impact assessment methods are those of the ILCD Handbook (European Commission -JRC-IES, 2010).

During the PEF pilot phase, the Food SCP Round Table provided technical support to the food pilots, with the aim to exploit the experience with the Protocol, especially for the cross-cutting issues such as the key common methodological problems (Saouter et al., 2014). Therefore, the work done during the development and implementation of the Protocol was merged with the development of the PEFCR Guidance and the PEFCRs.

## 5.3 Product Environmental Footprint method

In the last years, both companies and consumers have increased their awareness on the environmental impacts caused by consumption and production models, especially in the agri-food sector. For these reasons, several different standards and technical guidelines have been developed at international level for assessing the potential environmental impacts of products and services, such as the PAS 2050, the Greenhouse Gas Protocol and the BPX 30-323-0 (Manfredi S. et al., 2015). Furthermore, eco-labels, i.e. product labels which provide information about the overall environmental performance of products (e.g. Environmental Product Declaration according to ISO 14025 or European Ecolabel according to ISO 10424) have been used increasingly by companies to communicate their commitment towards sustainable development topics and to obtain competitive advantages in the market.

Some methods include only the impacts on climate change, such as the PAS 2050 and the Greenhouse Gas Protocol, which are focused only on the Carbon Footprint calculation, whereas other account for a limited group of environmental indicators, and therefore they do not comprehensively assess all the environmental aspects connected to products life cycle. Moreover, the results obtained by the application of those methods are not fully consistent or comparable (Manfredi S. et al, 2015; Finkbeiner et al, 2009; Laurent et al., 2012), since all these methods are not harmonised and apply different methodological choices.

One of the objectives of the “Roadmap to a Resource Efficient Europe” was to “Establish a common methodological approach to enable member states and the private sector to evaluate and communicate the environmental performance of products, services and companies based on a comprehensive assessment of environmental impacts over the life-cycle” (European Commission, 2011). Therefore in 2013 the European Commission adopted the Communication “Building a Single Market for Green Products (European Commission, 2013b) and published the “Recommendation on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations” (European Commission 2013a), both aimed at developing a common European framework for the assessment of products and organisations environmental performances throughout their entire life cycle as well as their communication to consumers, with the general purpose to facilitate and promote the development of environmentally-friendly products in the internal market and to promote competitiveness among companies (European Commission, 2013b).

The Recommendation established two harmonised methods for measuring environmental performance along the life cycle, the Product Environmental Footprint (PEF) and the Organisation Environmental Footprint Organization (OEF), both based on the standardised ISO LCA method. The PEF initiative therefore arises from the need to harmonize the methods for the environmental assessments of products and organisations as well as the related environmental labels, which have proliferated over the last years, leading to ineffective communication towards consumers and other companies, a scarce comparability among similar products, an increased difficulty in their use and a subsequent costs increase for companies.

The main purpose of the PEF method is to provide a common harmonised method which aims to increase the robustness, consistency, comparability and reproducibility of life cycle results (Manfredi S. et al., 2015). Furthermore, the PEF considers several different impact indicators related to environmental, health and resource use impacts of the product’s life cycle, thus reducing any possible burden shifting (European Commission, 2013 b).

In order to increase methodological harmonization and comparability among different studies of the same product category, the PEF method provides also general guidance on how to develop specific methodological requirements for several product categories (Product Environmental Footprint Category Rules - PEFCRs), which have been developed during a pilot phase which lasted from 2015 to 2018 and which involved several product groups, with a great participation of agri-food product categories (olive oil, dairy products, feed, beer, pasta, etc.). Moreover, a PEFCR guidance (European Commission, 2017) has been developed during the pilot phase of the PEF method, which describes the procedure to be followed fro developing the PEFCRs of a specific product category. PEF pilots involved several kinds of stakeholders who were grouped in Technical Secretariats for each pilot, consisting of technical experts such as companies and industry association (representing over 51% of the total European market for each product category), non-governmental organisations, research centres and universities. The Technical Secretariats were supported by a Steering Committee with representatives from member countries and the European Commission as well as by a Technical Advisory Board for providing technical support to specific methodological issues (Bach et al., 2018).

All PEF pilots had the following steps:

- Analysis of the existing Product Category Rules and sectoral guidance of other similar methodological standards;
- Definition of the product group and of the representative product, which is a typical average product sold on the market on which the PEFCRs will be developed;
- Screening study, a simplified PEF study for the representative product, with the aim to understand the most relevant life cycle stages, processes and environmental impacts;
- Development of a first draft of PEFCRs on the basis of the results of the screening studies, followed by a virtual consultation;
- Development of the second draft of the PEFCRs on the basis of the comments received in the consultation phase;
- Approval of the second draft of the PEFCRs by the Steering Committee;
- Supporting studies, performed for at least 3 products on the basis of the second draft of the PEFCRs;
- Development of the final version of the PEFCRs on the basis of the experience of the supporting study, followed by a virtual consultation and a review by a review panel;
- Approval of the final PEFCRs by the Steering Committee.

At the moment, the final PEFCRs are available for several product categories, with most of them representing the agri-food sector (i.e. beer, dairy, feed, packed water, pasta, wine), whereas other PEFCRs are still on-going (i.e. olive oil, wine)

In addition to the development of the PEFCRs, the pilot phase has defined, for each product category, the most relevant impact categories as well as environmental benchmarks which will be potentially compared with the results of a PEF study performed by a company on their product. Moreover, the PEF pilot has worked together with the main developers of LCA databases to develop PEF compliant LCA datasets to be used in the PEF studies, with the aim to increase the comparability of PEF results.

In the transition phase, which has started in May 2018, new PEFCRs will be developed and both the applicability of the method and the use of the PEF as an environmental label will be tested, which will include also the possible use of benchmarks for comparing different products available on the market.

The application of the harmonised PEF method in the agri-food sector, coupled with the development of the PEFCRs for several food and drink product groups, should therefore facilitate the calculation and evaluation of the environmental impacts of agri-food products, by defining common methodological rules to be applied by the practitioner, together with a detailed support in any methodological problem which can occur during the PEF study. This should result in better accuracy, reliability and reproducibility of PEF results and therefore should contribute to the transition towards circular economy models.

# 6 PestLCI 2.0 sensitivity to soil variations for the evaluation of pesticide distribution in LCA studies

As explained in par. 4.3.4, the use of PestLCI 2.0 for the calculation of pesticides emissions at life cycle inventory level has recently increased in LCAs of food products, and it has been applied to maize and wheat (Bacenetti et al., 2014; Bacenetti et al. 2015) and barley (Niero et al., 2015a; Dijkman et al., 2017; Schmidt Rivera et al., 2017) cultivation assessment. Nevertheless, in contrast with the simplified approaches defined by Margni et al. (2002) or Nemecek and Kagi (2007), the use of PestLCI 2.0 requires detailed data about the application method as well as soil and climate data of the studied region which can be difficult to obtain for the LCA practitioner. Some studies have adapted PestLCI to other climate, soil-specific and crop-specific conditions (Niero et al., 2015b; Renaud-Gentié et al., 2015; Schmidt Rivera et al., 2017), but to date no study has assessed the variations in active ingredients emissions to the different environmental compartments according to soil variations.

Therefore, PestLCI 2.0 was applied to maize cultivation in Northern Italy with the aim to verify to what extent PestLCI is sensitive to soil variations. The ultimate goal was to identify the extent to which the modelling of pesticide emissions in LCA studies can be increased in accuracy, while preserving the applicability. To this purpose, PestLCI 2.0 was applied to a maize production case-study in the experimental farm of Vallevecchia, located near Caorle (province of Venice, Northern Italy) using site-specific soil and climate data of the studied area, which were added for this purpose in PestLCI database. The application was carried out in the framework of the European project AGRICARE (Introducing innovative precision farming techniques in AGRIculture to decrease CARbon emissions), co-funded by the LIFE Programme and coordinated by Veneto Agricoltura (Italy).

The work performed on PestLCI 2.0 during the PhD has been included in the following publications, which are the basis of the whole chapter 6:

- Fantin V., Buscaroli A., Djikman T., Zamagni A., Garavini G., Bonoli A., Righi S., 2019. **PestLCI 2.0 sensitivity to soil variations for the evaluation of pesticide distribution in Life Cycle Assessment studies.** Science of the Total Environment 656, 1021–1031. https://doi.org/10.1016/j.scitotenv.2018.11.204.

- Fantin V., Righi S., Buscaroli A., Garavini G., Zamagni A., Dijkman T., Bonoli A, 2016. **Calculation of on-field pesticide emissions for maize production in Northern Italy: how much do different soil typologies affect the results of PestLCI 2.0 model?**, in Proceedings of 22nd SETAC Europe LCA Case Study Symposium. Montpellier, France, 21 September 2016.
- Fantin V., Righi S., Buscaroli A., Garavini G., Zamagni A., Dijkman T., Bonoli A., 2016. **Application of PestLCI model to site-specific soil and climate conditions: the case of maize production in Northern Italy**, in Proceedings of X Conference of Italian LCA Network Association, "Life Cycle Thinking, sostenibilità ed economia circolare". Ravenna 23 - 24 June 2016.
- Fantin V., Facibeni G., Righi S., 2017. **Emissioni da uso di pesticidi negli studi di Life Cycle Assessment: calcolo dell'inventario**. ENEA Technical report, USER-PG20-005, July 2017 (Confidential).

## 6.1 Description of PestLCI 2.0

PestLCI 2.0 model (Dijkman et al., 2012) can be used in the Life Cycle Inventory analysis (LCI) phase of LCA, to quantify pesticide emissions from an agricultural field after its application to the surrounding environmental compartments. The aim of the model development was to provide a model which requires a minimum of input data from the user, and that can be run quickly. For this purpose, modelling approaches from other models have been simplified and combined to provide one model that covers all relevant field compartments. PestLCI 2.0 is an improved version of PestLCI (Birkved & Hauschild, 2006), to which new processes to model pesticide fate have been added. In addition, the modelling of processes already included in the first version of the model have been updated. Furthermore, the geographical scope of the model was expanded to include the whole Europe. The functioning of the model is here described in brief, while a more detailed description can be found in Birkved & Hauschild (2006) and Dijkman et al. (2012).

The model boundaries of PestLCI 2.0 are formed by the 'field box', which includes the field where the pesticide is applied, the soil of that field up to 1 m depth, as well as the air above the field up to 100 m. In PestLCI 2.0, the soil, water, air and crop within the field box is considered to be part of the technosphere, i.e. part of a man-made production system, which includes human activities and the relevant material and energy flows. The ecosphere, or the environment that is the scope of LCA, then consists of the environmental compartments outside the field box. As a consequence, a pesticide only becomes an emission to the environment when it crosses the border of the field box. From that point on, Life Cycle Impact Assessment (LCIA) modelling of the fate and exposure can be performed (Birkved & Hauschild, 2006; Dijkman et al., 2012).

A pesticide is considered an emission when it crosses these boundaries. Figure 4 shows the technosphere box in which the red arrows represent all the pesticide emissions. The following emissions occur (Birkved & Hauschild, 2006; Dijkman et al., 2012):

- Emissions caused by wind drift and volatilisation of pesticides from crops and soil;
- Emissions from the soil runoff, which will pollute surface waters;
- Emissions to surface water from the field drainage system;
- Emissions to groundwater from leaching or through macropores, which are those pores with a diameter greater than 50 µm.

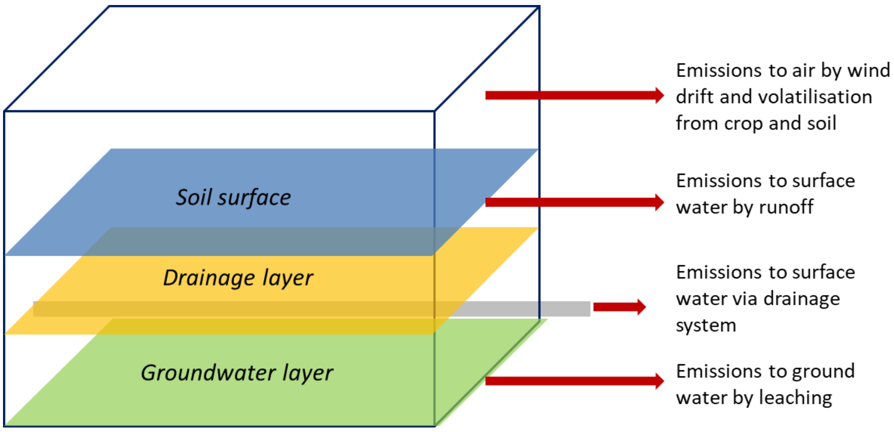



**Figure 4.** Representation of the technosphere box and of the emissions from the application of PestLCI 2.0 (Source: Personal elaboration adapted from Birkved and Hauschild, 2006).

When a pesticide is applied to an agricultural field, only a fraction of the applied dose reaches the crop; furthermore, once it has been deposited, it can subsequently be dispersed in the surrounding environment. This dispersion is defined as the "fate of a pesticide" and refers to the distribution among the different environmental compartments, such as air, surface and ground waters and soil, caused by different transport, distribution and degradation mechanisms. Emissions into the soil are not considered because soil is considered part of the technosphere (Birkved & Hauschild, 2006; Dijkman et al., 2012).

PestLCI has a modular structure in which each main process taking place in the agricultural field has its own module with its own inputs and outputs. The individual process modules calculate the fractions of the pesticide applied to the crop as well as the pesticide fractions emitted from its application. The mass conservation principle is valid in each phase of the model, i.e. the sum of the emitted pesticide fractions remains constant over the time (Birkved & Hauschild, 2006; Dijkman et al., 2012).
The emission of the pesticide in the field is modelled in two sequential phases (Birkved & Hauschild, 2006; Dijkman et al., 2012):

1) Primary distribution, which considers all the emissions during the application phase;
2) Secondary distribution, which considers all the emissions on leaves and soil after the pesticide application.

The total emission fraction of the pesticide is given by the sum of the fraction emitted to air, the fraction emitted to surface waters and that emitted to groundwater (Birkved & Hauschild, 2006; Dijkman et al., 2012).
:

$$f_{em} = \frac{m_{em}}{m_{appl}} = f_{air} + f_{sw} + f_{gw} \qquad (1)$$

Where:

$f_{em}$ = the fraction of the applied pesticide mass emitted into the surrounding environment;

$m_{em}$ = the mass of pesticide which is emitted;

$m_{appl}$ = the mass of pesticide applied to the field;

$f_{air}$ = the fraction of the applied mass emitted to the air;

$f_{sw}$ = the fraction of the applied mass emitted to surface waters;

$f_{gw}$ =the fraction of the applied mass emitted to groundwater.

PestLCI 2.0 calculates pesticide emissions from modelling primary and secondary distribution of a pesticide (Figure 5) (Birkved & Hauschild, 2006; Dijkman et al., 2012). Here, primary distribution covers the processes that occur in the first minutes after the pesticide is applied: deposition on leaves and on field soil, as well as wind drift. Wind drift is in PestLCI 2.0 the term used to describe the transport, due to wind, of pesticide droplets leaving the sprayer across the field border. In the case study presented in the next paragraphs, emissions due to wind drift are characterized as emissions to off-field surfaces. Off-field surfaces can be a surface water body, or agricultural or natural soil next to the field. This is not consistent with the boundaries of the field box: the pesticide leaves the field box through air, and therefore this emission pathways should be characterized as an emission to air. However, the drift curves used to calculate wind drift are based on measurements of off-field deposition shortly after application (see for example Holterman & Van der Zande, 2003). Characterization models such as USEtox (Rosenbaum et al., 2008) do not include this kind of agriculture-specific processes. Therefore, in the case study it was chosen to let modelling relevant processes prevail over adhering to model boundaries.

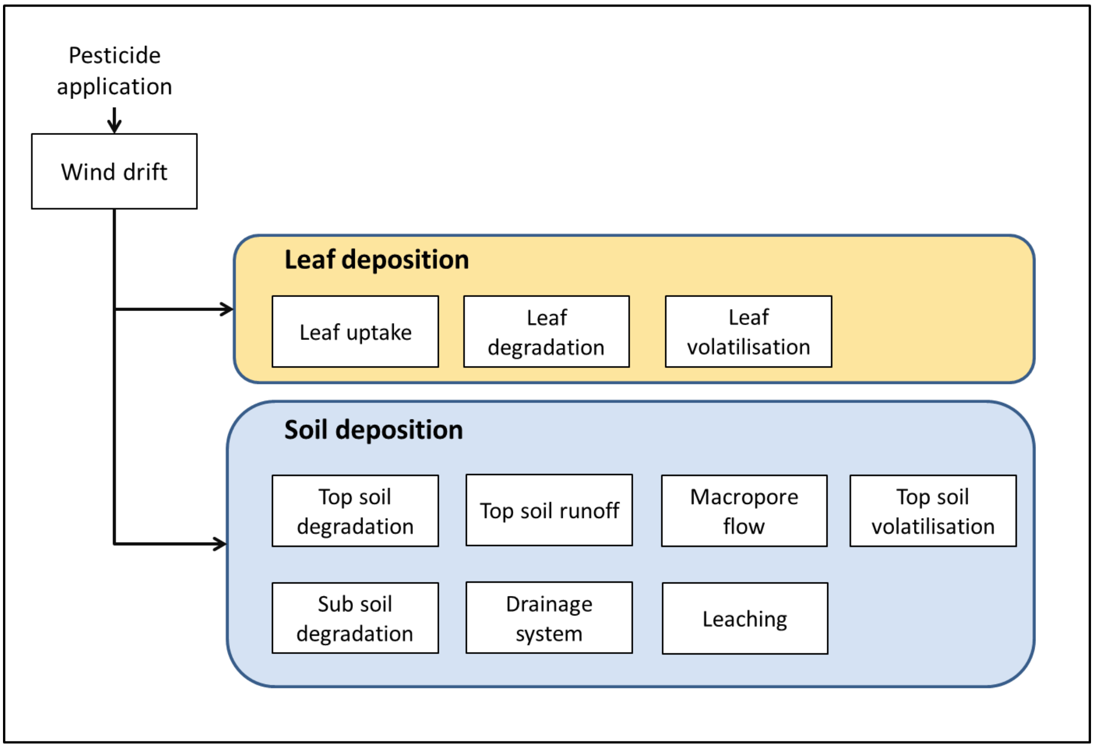



**Figure 5.** Modular structure of PestLCI 2.0 (Source: Personal elaboration adapted from Dijkman et al., 2012).

After the primary distribution has been calculated, the secondary distribution describes the fate of the pesticide that is deposited on the plants in the field and on the field soil (Birkved & Hauschild, 2006; Dijkman et al., 2012).

The secondary processes on the leaves considered in PestLCI 2.0 are pesticide degradation, uptake into the leaves, and volatilization. Among these processes, only volatilization leads to an emission, in this case to air. It is assumed that a pesticide, once volatilized, is transported immediately away from the field box by the wind. The processes on leaves are modelled until the first rainfall event after pesticide application. At that point, all remaining pesticide is assumed to wash off to the top soil (Birkved & Hauschild, 2006; Dijkman et al., 2012).

The secondary processes considered on top soil (defined as the uppermost 1 cm of soil) are pesticide degradation and volatilization, which, like the secondary processes on leaves, is modelled until the first rainfall event after application. Based on the pesticide and soil properties, only a fraction of the pesticide is available (i.e., dissolved in soil water, or present in gaseous form in pores) for these processes. The remaining fraction is sorbed to soil particles (Birkved & Hauschild, 2006; Dijkman et al., 2012).

At the first rainfall event after pesticide application, emissions to off-field surfaces due to runoff is calculated. In PestLCI 2.0, only emissions of pesticide dissolved in runoff water is calculated. Emissions of pesticide sorbed to soil particles that are washed off from the field, is not included. Thus, only dissolved pesticide is susceptible to runoff in PestLCI 2.0. The volume of runoff water depends on rainfall intensity, slope of the field, and whether the soil has a sandy texture or not (Birkved & Hauschild, 2006; Dijkman et al., 2012).

Moreover, at the first rainfall event after pesticide application, the amount of dissolved pesticide entering macropores is calculated. Macropores provide a bypass for water, allowing for a rapid downwards movement through the soil. In PestLCI 2.0, pesticides entering macropores are assumed to be emitted to groundwater. The volume of water entering the macropores is calculated from the rainfall volume and the field capacity (i.e. the soil's capacity to store water). This storage capacity is made up of two classes of pores, depending on the rate at which these pores drain. The relative presence of both pore types is calculated from the soil composition (i.e. sand, silt, and clay content). Following a tipping bucket approach, where the slow-draining pores are filled first, the model calculates how much water can be stored before macropore's flow starts to occur. Any water that cannot be stored is assumed to enter macropores (Birkved & Hauschild, 2006; Dijkman et al., 2012).

After the first rainfall event, all pesticide left in the soil is assumed to remain at the bottom of the topsoil. From here, the pesticide starts leaching through the soil until it reaches 1 m of depth. At this depth, it is considered an emission to groundwater. For modelling leaching, the soil is considered a column through which water moves downwards at a constant rate. Due to sorption to soil particles and organic carbon, the rate of pesticide movement is lower than that of the water. While moving through the soil, the pesticide is subject to degradation, which is calculated from the pesticide soil half-life, which is corrected for temperature and depth. When drainage tubes are present in the sub soil, a fraction of the pesticide is intercepted and emitted to surface water (Birkved & Hauschild, 2006; Dijkman et al., 2012).

Summarizing, emissions to off-field surfaces and groundwater depend on characteristics of both the climate and the soil of the location (studied), as well as on the pesticide properties.

It is noteworthy that PestLCI 2.0 was developed for modelling European conditions, and local circumstances may not be reflected in the modelling. This could lead to a difficulty in applying the model to different geographical areas, characterized by other soil and climate conditions, thus preventing from obtaining realistic results.

## 6.2 Soil and tillage sensitiveness evaluation method

As explained above, the aim of this case study is to evaluate to what extent PestLCI 2.0 is sensitive to soil and tillage variations. Four different tests were carried out: first of all (1) the distribution of pesticide among the environmental compartments obtained using several types of soils with similar characteristics (i.e. organic carbon content, pH, clay, silt and sand content) to the Vallevecchia soil were compared to the site-specific one (i.e. the Vallevecchia soil). Then (2), the distribution of pesticide among the environmental compartments obtained using types of soils different in their characteristics to the Vallevecchia soil were compared to the site-specific one. Moreover, (3), the distribution of pesticide among the environmental compartments obtained applying the soil data of Vallevecchia site was compared with the distribution obtained using the PestLCI 2.0 default soil contained in the model's database which was considered more similar to the site-specific one. Finally, (4) the PestLCI 2.0 results obtained with the specific soil data of the Vallevecchia area were analysed according to different types of tillage, i.e. conventional tillage, minimum tillage, no tillage.

## 6.3 Experimental farm description

The experimental farm of Vallevecchia has unique pedological and climatic characteristics for its region. It extends for 900 hectares and is bounded on the four sides by sea, lagoons and fluvial waters, which contribute to creating situations of considerable complexity and ecological interest. From the soil map of Veneto Region (ARPAV, 2018a), it is possible to notice how the Vallevecchia farm falls in a reclaimed area.

The area of the tested field is 0.35 ha, it is a small piece of land because Vallevecchia is an experimental farm. In the Vallevecchia farm, the following four crops are cultivated: maize, soy, wheat and rapeseed. Maize was chosen for the application of PestLCI 2.0 model in this case study and site-specific soil and climate parameters, described in the following paragraphs, were added in PestLCI 2.0 database for this purpose.

### 6.3.1 Climatological data

The climate of the Veneto Region is sub-continental, though the presence of the Adriatic Sea and the Alps protect it from north winds makes it temperate. Climate is characterized by significant differences between specific areas: mountains, plains, coasts, etc. The Veneto Region, due to its topographic configuration which includes Alpine reliefs, plains and a coast exposed to the sea, is conducive to heavy and long-lasting precipitation events. Climatological data have been obtained from the monitoring network of Veneto Regional Agency for Environmental Prevention and Protection (ARPAV), Meteorological Centre of Teolo (CMT) (ARPAV, 2018b). The data acquired at the station of Lugugnana (Portogruaro) are referred to the period 1994-2014. The CMT runs a dense network of automatic weather stations, counting around 200 stations over the entire territory, most of which are transmitted in almost real time. In particular, climate data were obtained from the station n° 166, 0 m above sea level, coordinates 1807248 m E, 5068864 mN, Gauss-Boaga datum (Italian Fuse Ovest).

### 6.3.2 Soil data

Soil knowledge has increased over time and nowadays several web soil databases, at different scales, are available for their consultation. In Italy, soil databases are developed and published by regional administrations or other territorial agencies. In order to perform the present study, soil data were collected from the soil map of Veneto Region (ARPAV, 2016b). Every cartographic unit in the soil map is provided with a link to the list of the included Soil Typological Units (STU). Each STU is identified by both a name and an acronym and is described in detail, with the most relevant physical and chemical soil characteristics, the landscape unit and both the Soil Taxonomy (Soil Survey Staff, 1998) and the World Reference Base (FAO, 1998) soils classifications. The studied site insists on reclaimed lagoon areas derived from Piave, Livenza and Tagliamento rivers deposits. Soils described in this area and accounted for in the study (BIB1, CAB1, CAP1, CFO1, CON1, CRL1, CTU1, MEL1, QUA1, SAB1, SCO1, TDF1, VAD1, VAN1 and VED1) are generally deep (>1 m), with a typical A/B/C or A/C horizons sequence (see explanation notes in Table 1 and Table 2), organic matter content ranging between 0.2% and 3.9% in surface horizons and pH values ranging between 6.6 and 8.7 (Table 1 and Table 2). These soils differ mainly in texture, which is silty clay loam (CAB1, CFO1, CTU1, SCO1, TDF1) or silty loam (BIB1, CON1, CRL1, QUA1) or loam (MEL1, VAN1, VED1), or sandy loam (CAP1), or sandy (SAB1, VAD1). All soils are cultivated (suffix p in horizon designation). Some soils display the presence of water in some horizons (suffix g) and also the accumulation of secondary carbonates (suffix k). Some soils (CAB1 and CTU1) display the presence of organic horizons (H) which lie, anyway, at a depth greater than 1 m, and were therefore not considered in the computation (since they are out of the field border defined by PestLCI 2.0). These STU are all suitable for maize cultivation but they have different characteristics and limitations which provide different attitudes to crop production.

The Vallevecchia studied farm is located on the Torre Di Fine (TDF1) soil, whose characteristics are summarized in Table 1.

In order to perform the case study, as explained in par. 6.2, the above-mentioned soils were divided in two classes: 1) similar soils, which have similar characteristics with regard to TDF1; 2) different soils, which have different characteristics with regard to TDF1. More in detail, the similar soils are the following (Table 1): CTU1; SCO1; CAB1; CFO1; CON1; BIB1; QUA1; CRL1. On the contrary, the following soils have different characteristics with regard to TDF1 (Table 2): VAN1; VED1; MEL1; CAP1; SAD1; VAD1.

As above-mentioned, a further comparison was performed between the results of PestLCI 2.0 with TDF1 soil and those obtained with a comparable soil already contained in the model's database. Among the seven different default soils, Soil6 of PestLCI 2.0 database has been selected on the basis of chemical and physical characteristics (Table 2). This soil in fact has the same textural class as the one present in the farm (TDF1) and also a similar organic carbon content and pH value. The other soils in the database could not be considered since they have completely different properties.

| **Soil Typological Unit (STU)** | **TDF1** | **CTU1** | **SCO1** | **CAB1** | **CFO1** | **CON1** | **BIB1** | **QUA1** | **CRL1** |
|---|---|---|---|---|---|---|---|---|---|
| Soil Name | Torre di Fine | Ca' Turcata | Santa Scolastica | Caberlotto | Ca' Fornera | Conche | Bibione | Quarto d'Altino | Caorle |
| Keys to Soil Taxonomy (SSS, 1998) | Fluvaquentic Eutrudepts fine-silty, carbonatic, mesic | Fluvaquentic Eutrudepts fine, mixed, mesic | Fluvaquentic Endoaquepts fine, mixed, calcareous, mesic | Typic Endoaquepts fine-silty, mixed, mesic | Oxyaquic Eutrudepts fine-silty, carbonatic, mesic | Oxyaquic Udifluvents coarse-loamy, mixed, calcareous, mesic | Fluvaquentic Eutrudepts coarse-silty, carbonatic, mesic | Oxyaquic Eutrochrepts fine-silty, mixed, mesic | Oxyaquic Udifluvents coarse-silty, carbonatic, mesic |
| World Reference Base (FAO, 1998) | Gleyic Fluvic Cambisols (Hypercalcaric) | Gleyic Fluvic Cambisols (Calcaric) | Calcaric Hypocalcic Gleysols | Calcaric Humic Gleysols | Gleyic Fluvic Cambisols (Hypercalcaric) | Calcaric Fluvisols | Humic Endogleyc Fluvisols (Hypercalcaric) | Calcaric Fluvic Cambisols | Hypercalcaric Endogleyic Fluvisols |
| Horizon sequence | Ap/Bg/Cg | Ap/Bg/(Ha)/Cg | Ap/B(k)g/Cg | Ap/Bg/Cg/(Ha) | Ap/Bg/Cg | Ap/C/Cg | Ap/Bg/Cg | Ap/Bg/Cg | Ap/Cg |
| Soil type | SICL | SICL | SICL | SICL | SICL | SIL | SIL | SIL | SIL |
| Horizon designation | **Ap1** | **Ap1** | **Ap** | **Ap** | **Ap1** | **Ap** | **Ap1** | **Ap1** | **Ap1** |
| Start depth (cm) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| End depth (cm) | 40 | 20 | 50 | 50 | 40 | 50 | 30 | 40 | 30 |
| Clay content (<2µm) (%) | 25.1 | 33.4 | 32 | 40.2 | 31 | 22.3 | 19 | 23.2 | 17.6 |
| Silt content (2-50 µm) (%) | 59.8 | 19.8 | 51.1 | 59.2 | 62.2 | 59 | 67.9 | 55.4 | 70.9 |
| Sand content (>50µm) (%) | 15.1 | 46.8 | 16.9 | 0.6 | 6.8 | 18.7 | 13.1 | 21.3 | 11.5 |
| Organic carbon (%) | 1.4 | 2.3 | 1.1 | 1 | 1.2 | 0.6 | 1.2 | 0.8 | 0.8 |
| pH | 8.3 | 8 | 8 | 8.1 | 8.2 | 8.2 | 8.1 | 7.9 | 8.2 |
| Soil type | SIL | SCL | SICL | SIC | SICL | SIL | SIL | SIL | SIL |
| Horizon designation | **Ap2** | **Ap2** | **Ab** | **Bg1** | **Ap2** | **C1** | **Ap2** | **Ap2** | **Ap2** |
| Start depth (cm) | 40 | 20 | 50 | 50 | 40 | 50 | 30 | 40 | 30 |
| End depth (cm) | 55 | 60 | 80 | 75 | 70 | 60 | 55 | 70 | 55 |
| Clay content (<2µm) (%) | 25.8 | 38.4 | 52.8 | 30.5 | 30.8 | 17.8 | 18.3 | 22 | 17.5 |
| Silt content (2-50 µm) (%) | 61.5 | 57.1 | 39.4 | 69.1 | 62 | 60.7 | 69.2 | 53.9 | 71.6 |
| Sand content (>50µm) (%) | 12.7 | 4.5 | 7.8 | 0.3 | 7.2 | 21.5 | 12.5 | 24.1 | 10.9 |
| Organic carbon (%) | 1.5 | 2.1 | 1.4 | 0.9 | 1.4 | 0.6 | 1.1 | 1.4 | 0.6 |
| pH | 8.2 | 7.9 | 8 | 8.2 | 8.2 | 8.3 | 8.1 | 7.9 | 8.2 |
| Soil type | SIL | SICL | C | SICL | SICL | SIL | SIL | SIL | SIL |
| Horizon designation | **Bg1** | **Bg** | **Bkg** | **Bg2** | **Bg1** | **C2** | **Bg** | **Bw** | **Cg1** |
| Start depth (cm) | 55 | 60 | 80 | 75 | 70 | 60 | 55 | 70 | 55 |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| End depth (cm) | 85 | 105 | 105 | 120 | 100 | 90 | 85 | 105 | 90 |
| Clay content (<2µm) (%) | 19.0 | 36.1 | 41.4 | 25.6 | 20.7 | 9.7 | 13.2 | 21.8 | 17.6 |
| Silt content (2-50 µm) (%) | 57.8 | 60.5 | 56.2 | 66 | 73.1 | 69.4 | 67 | 65.8 | 74.5 |
| Sand content (>50µm) (%) | 23.2 | 3.4 | 2.4 | 8.4 | 6.2 | 20.9 | 19.8 | 12.4 | 7.9 |
| Organic carbon (%) | 0.7 | 2 | 0.7 | 1 | 0.5 | 0.2 | 0.3 | 0.7 | 0.5 |
| pH | 8 | 7.8 | 8.1 | 7.9 | 8.4 | 8.3 | 8.1 | 8.2 | 8.3 |
| Soil type | SIL | SICL | SIC | SIL | SIL | SIL | SIL | SIL | SIL |
| Horizon designation | **Bg2** | | | | | | | | **Cg2** |
| Start depth (cm) | 85 | | | | | | | | 90 |
| End depth (cm) | 150 | | | | | | | | 140 |
| Clay content (<2µm) (%) | 27.1 | | | | | | | | 16.3 |
| Silt content (2-50 µm) (%) | 56.8 | | | | | | | | 69 |
| Sand content (>50µm) (%) | 16.1 | | | | | | | | 14.7 |
| Organic carbon (%)(%) | 1.4 | | | | | | | | 0.5 |
| pH | 8 | | | | | | | | 8.3 |
| Soil type | SICL | | | | | | | | SIL |

**Table 1.** Summary characteristics of considered soils (Source: Fantin et al., 2019, reproduced by permission of Elsevier).

**Horizons and suffix:** H=organic horizon saturated for prolonged periods; A=mineral horizons that have formed at the soil surface or below an O horizon; B=mineral horizons that have formed below an A, E, or O horizon; C=mineral horizons or layers that are little affected by pedogenic processes and lack the properties of O, A, E, B, or L horizons; p=tillage or other disturbance; b=buried genetic horizon; g=strong gleying; k=accumulation of secondary carbonates; w=development of color or structure.
**Soil type:** C=clay; SIC=silty clay; SICL=silty clay loam; SIL=silt loam; L=loam; SCL=sandy clay loam; SL=sandy loam; LS=loamy sand; S=sand.

| **Soil Typological Unit (STU)** | **VAN1** | **VED1** | **MEL1** | **CAP1** | **SAB1** | **VAD1** | **SOIL6** |
|---|---|---|---|---|---|---|---|
| Soil Name | Vanzo | Casa Vendramin | Casa Scaramello | Capitello | Sabbioni | Valcerere Dolfina | PestLCI 2.0 database |
| Keys to Soil Taxonomy (SSS, 1998) | Typic Calciustepts coarse-loamy, mixed, mesic | Oxyaquic Haplustepts fine-loamy,mixed, mesic | Typic Calciustepts coarse-loamy over sandy or sandy-skeletal, mixed, mesic | Typic Ustipsamments, mixed, mesic | Oxyaquic Ustipsamments sandy, mixed, mesic | Typic Ustipsamments sandy, mixed, mesic | |
| World Reference Base (FAO, 1998) | Hypocalcic Calcisols | Calcaric Cambisols | Hypercalcic Calcisols | Calcaric Regosols | Calcaric Arenic Fluvisols | Eutric Arenosols | |
| Horizon sequence | Ap/Bw/(Bk)/C | Ap/Bw/C(g) | Ap/B(k)/C | Ap/C | Ap/C | Ap/C | |
| Soil type | L | L | L | SL | S | S | |
| Horizon designation | **Ap** | **Ap** | **Ap** | **Ap** | **Ap** | **Ap** | |
| Start depth (cm) | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| End depth (cm) | 55 | 50 | 60 | 55 | 50 | 30 | 27 |
| Clay content (<2µm) (%) | 19.4 | 19 | 11.9 | 10.8 | 3.6 | 1.1 | 34 |
| Silt content (2-50 µm) (%) | 46.6 | 37 | 29.1 | 26.1 | 9.1 | 2.4 | 59 |
| Sand content (>50µm) (%) | 34 | 44 | 59 | 63.1 | 87.3 | 96.5 | 7 |
| Organic carbon (%) | 0.8 | 1.3 | 0.4 | 0.6 | 1 | 0.2 | 1.8 |
| pH | 8.2 | 8.2 | 8.2 | 8.2 | 7.8 | 8.3 | 6.6 |
| Soil type | L | L | SL | SL | S | S | SICL |
| Horizon designation | **Bk** | **Bw** | **Bk** | **C1** | **C** | **C1** | |
| Start depth (cm) | 55 | 50 | 60 | 55 | 50 | 30 | 27 |
| End depth (cm) | 75 | 90 | 85 | 100 | 95 | 60 | 47 |
| Clay content (<2µm) (%) | 19.2 | 17.8 | 7.4 | 3.6 | 4 | 0.1 | 37 |
| Silt content (2-50 µm) (%) | 50.7 | 48.5 | 55 | 3.1 | 13.5 | 1.4 | 55 |
| Sand content (>50µm) (%) | 30.1 | 33.7 | 37.7 | 93.3 | 82.5 | 98.5 | 8 |
| Organic carbon (%) | 0.5 | 1.1 | 0.3 | 0 | 0.8 | 0 | 1.8 |
| pH | 8.2 | 8.2 | 8.4 | 8.7 | 8.1 | 8.4 | 6.9 |
| Soil type | SIL | L | SIL | S | SF | S | SICL |
| Horizon designation | **C1** | **Ab** | **C** | | **Cg** | **C2** | |
| Start depth (cm) | 75 | 90 | 85 | | 95 | 60 | 47 |
| End depth (cm) | 130 | 130 | 100 | | 150 | 120 | 63 |
| Clay content (<2µm) (%) | 4.4 | 25.8 | 2.2 | | 12.9 | 0 | 39 |
| Silt content (2-50 µm) (%) | 22 | 66.2 | 8.4 | | 46.2 | 0.9 | 55 |
| Sand content (>50µm) (%) | 73.6 | 8 | 89.4 | | 40.9 | 99.1 | 6 |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Organic carbon (%) | 0.2 | 1.2 | 0 | | 0.8 | 0 | 1.3 |
| pH | 8.4 | 8.2 | 8.5 | | 8.4 | 8.4 | 7.3 |
| Soil type | SL | SIL | S | | L | S | SICL |
| Horizon designation | | | | | | | |
| Start depth (cm) | | | | | | | 63 |
| End depth (cm) | | | | | | | 90 |
| Clay content (<2µm) (%) | | | | | | | 36 |
| Silt content (2-50 µm) (%) | | | | | | | 55 |
| Sand content (>50µm) (%) | | | | | | | 9 |
| Organic carbon (%) | | | | | | | 0.9 |
| pH | | | | | | | 7.6 |
| Soil type | | | | | | | SICL |

**Table 2.** Summary characteristics of considered soils (Source: Fantin et al., 2019, reproduced by permission of Elsevier).

**Horizons and suffix:** H=organic horizon saturated for prolonged periods; A=mineral horizons that have formed at the soil surface or below an O horizon; B=mineral horizons that have formed below an A, E, or O horizon; C=mineral horizons or layers that are little affected by pedogenic processes and lack the properties of O, A, E, B, or L horizons; p=tillage or other disturbance; b=buried genetic horizon; g=strong gleying; k=accumulation of secondary carbonates; w=development of color or structure.
**Soil type:** C=clay; SIC=silty clay; SICL=silty clay loam; SIL=silt loam; L=loam; SCL=sandy clay loam; SL=sandy loam; LS=loamy sand; S=sand.

### 6.3.3 Crop and pesticide data

Maize is cultivated with three types of tillage techniques in the studied area: conventional tillage, reduced tillage, no tillage. The conventional tillage consists of ploughing the soil and applying fertilizers without optimized analysis. The reduced tillage is performed without complete inversion of the layers and at depths of less than 20 cm; the fertilizer distribution is optimised. No tillage consists of sowing without soil preparation and optimization of fertilizer application.

In the farm of Vallevecchia, three main phases for the use of pesticides can be identified: 1) the pre-emergence phase, in which the pesticides are spread before seeding or before plants emerge from the soil; 2) post-emergence, which corresponds to the leaf development of the crop; 3) treatment with insecticides which corresponds to the inflorescence development. In the case-study, three active ingredients are applied: terbuthylazine, metolachlor and cypermethrin. Terbuthylazine and metolachlor were used in both pre- and post-emergence phases, whereas cypermethrin was used only for the treatment of insects (see Table 3). All pesticides are applied with spray boom technique. Data in Table 3 refer to a maize productive cycle of one year.

| Type of treatment | Quantity | Hazard codes |
|---|---|---|
| **Pre-emergence** | | |
| Terbuthylazine (L/ha) | 0.81 | H302, H373, H400, H410 |
| Metolachlor (L/ha) | 1.13 | H317 |
| **Post-emergence** | | |
| Terbuthylazine (L/ha) | 0.77 | H302, H373, H400, H410 |
| Metolachlor (L/ha) | 1.07 | H317 |
| **Treatment for insects** | | |
| Cypermethrin (kg/ha) | 0.018 | H400, H410 |
| **Water for irrigation (mm)** | 111 | |

**Table 3.** Quantity and type of pesticide applied for maize cultivation on the Vallevecchia experimental farm. (Source: Fantin et al., 2019, reproduced by permission of Elsevier).

Pesticide hazard codes and classes: H302-Acute toxicity, oral; H317-Sensitisation, skin; H373-Specific target organ toxiticy, repeated exposure; H400-Hazardous to the aquatic environment, acute hazard; H410- Hazardous to the aquatic environment, long-term hazard (Regulation EC 1272/2008).

## 6.4 Results and Discussion

In this chapter the results of the four testss are reported and discussed, namely:

1. TDF1 (site-specific soil) versus similar soils;
2. TDF1 versus different soils;
3. TDF1 versus Soil6 (reference default soil);
4. TDF1 according to different types of tillage.

In all scenarios, the climate data were those of the Vallevecchia area described in par. 6.3.2.

The results consist in the percentage distribution of the pesticides among five fractions. The analysed fractions are: emissions to air, surface water (or off-field soils) and groundwater, degraded fraction and fraction uptaken by plant or soils. Surface water and off-field soils (i.e. natural or agricultural soils located next to the field where the pesticide is applied) are considered equivalent fate since the actual final destination depends on the presence of water or soils near the analyzed field; they include emission to surface water and wind drift. The air emissions are the sum of pesticide volatilization from leaves and top soil. The groundwater emissions are the sum of emissions through macropores and leaching through soil matrix.

### 6.4.1 Results of Test 1: comparison among TDF1 and similar soils

Table 4 shows the obtained fraction of emission to air, surface water or off-field soils, groundwater and the fractions degraded or uptaken by plants or soil of each active ingredient applied to TDF1 and the group of soils with similar characteristics (CTU1, SCO1, CAB1, CFO1, CON1, BIB1, QUA1, CRL1). Minimum, maximum and average values, standard deviation and coefficient of variation (c.v.) have been included for each row.

Firstly, it is noteworthy that the fraction degraded or uptaken by plants or soils varies greatly on the basis of the pesticide. As regards terbuthylazine in TDF1, this fraction is 0.6-0.7, therefore a significant fraction is emitted to environmental compartments. It can be observed that emissions to air increase slightly from pre-emergence to post-emergence application (0.093-0.12 respectively), while the emissions to groundwater decrease slightly from pre-emergence to post-emergence (0.25-0.17). On the contrary, the fraction of metolachlor and cypermethrin degraded or uptaken in TDF1 is always higher than 0.9 and often reaches 0.99.

| Terbuthylazine (pre-emergence) | TDF1 | CTU1 | SCO1 | CAB1 | CFO1 | CON1 | BIB1 | QUA1 | CRL1 | MIN | MAX | Average | Stand.Dev. | c.v.% |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Emission to air (fraction) | 9.3E-02 | 9.3E-02 | 9.23E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 7.2E-06 | 0.0 |
| Emission to surface water or off-field soil (fraction) | 1.1E-03 | 1.3E-03 | 1.1E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.1E-03 | 1.3E-03 | 1.2E-03 | 7.7E-05 | 6.2 |
| Emission to groundwater (fraction) | 3.0E-01 | 3.0E-01 | 2.2E-01 | 2.3E-01 | 2.7E-01 | 2.4E-01 | 3.5E-01 | 2.7E-01 | 3.4E-01 | 2.2E-01 | 3.5E-01 | 2.8E-01 | 4.6E-02 | 16.5 |
| Degradation and uptake (fraction) | 6.1E-01 | 6.1E-01 | 6.8E-01 | 6.8E-01 | 6.4E-01 | 6.7E-01 | 5.6E-01 | 6.4E-01 | 5.7E-01 | 5.6E-01 | 6.8E-01 | 6.3E-01 | 4.6E-02 | 7.4 |
| **Terbuthylazine (post-emergence)** | | | | | | | | | | | | | | |
| Emission to air (fraction) | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 3.3E-07 | 0.0 |
| Emission to surface water or off-field soil (fraction) | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 5.5E-11 | 0.0 |
| Emission to groundwater (fraction) | 1.7E-01 | 1.8E-01 | 1.6E-01 | 2.3E-01 | 2.2E-01 | 1.8E-01 | 1.6E-01 | 2.2E-01 | 1.6E-01 | 1.6E-01 | 2.3E-01 | 1.9E-01 | 2.7E-02 | 14.8 |
| Degradation and uptake (fraction) | 7.1E-01 | 7.0E-01 | 7.2E-01 | 6.5E-01 | 6.6E-01 | 7.0E-01 | 7.2E-01 | 6.6E-01 | 7.2E-01 | 6.5E-01 | 7.2E-01 | 6.9E-01 | 2.7E-02 | 4.0 |
| **Metolachlor (pre-emergence)** | | | | | | | | | | | | | | |
| Emission to air (fraction) | 7.7E-05 | 2.7E-05 | 1.0E-04 | 1.2E-04 | 8.7E-05 | 2.9E-04 | 8.7E-05 | 2.0E-04 | 1.8E-04 | 2.7E-05 | 2.9E-04 | 1.3E-04 | 7.9E-05 | 59.5 |
| Emission to surface water or off-field soil (fraction) | 9.5E-03 | 9.4E-03 | 9.5E-03 | 9.5E-03 | 9.5E-03 | 9.6E-03 | 9.5E-03 | 9.6E-03 | 9.6E-03 | 9.4E-03 | 9.6E-03 | 9.5E-03 | 4.8E-05 | 0.5 |
| Emission to groundwater (fraction) | 2.6E-03 | 2.3E-03 | 2.5E-03 | 7.1E-03 | 5.3E-03 | 4.6E-03 | 2.5E-03 | 6.2E-03 | 3.3E-03 | 2.3E-03 | 7.1E-03 | 4.1E-03 | 1.8E-03 | 44.8 |
| Degradation and uptake (fraction) | 9.9E-01 | 9.9E-01 | 9.9E-01 | 9.8E-01 | 9.9E-01 | 9.9E-01 | 9.9E-01 | 9.8E-01 | 9.9E-01 | 9.8E-01 | 9.9E-01 | 9.9E-01 | 1.9E-03 | 0.2 |
| **Metolachlor (post-emergence)** | | | | | | | | | | | | | | |
| Emission to air (fraction) | 2.5E-03 | 2.4E-03 | 2.5E-03 | 2.5E-03 | 2.5E-03 | 2.8E-03 | 2.5E-03 | 2.6E-03 | 2.6E-03 | 2.4E-03 | 2.8E-03 | 2.5E-03 | 1.1E-04 | 4.5 |
| Emission to surface water or off-field soil (fraction) | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 3.8E-05 | 0.2 |
| Emission to groundwater (fraction) | 2.0E-03 | 1.7E-03 | 1.9E-03 | 5.1E-03 | 3.8E-03 | 3.4E-03 | 1.9E-03 | 4.5E-03 | 2.5E-03 | 1.7E-03 | 5.1E-03 | 3.0E-03 | 1.3E-03 | 42.3 |
| Degradation and uptake (fraction) | 9.8E-01 | 9.8E-01 | 9.7E-01 | 9.7E-01 | 9.7E-01 | 9.7E-01 | 9.8E-01 | 9.7E-01 | 9.8E-01 | 9.7E-01 | 9.8E-01 | 9.7E-01 | 1.3E-03 | 0.1 |
| **Cypermethrin (insecticides)** | | | | | | | | | | | | | | |
| Emission to air (fraction) | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-08 | 0.0 |
| Emission to surface water or off-field soil (fraction) | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 1.3E-09 | 0.0 |
| Emission to groundwater (fraction) | 5.2E-02 | 5.5E-02 | 4.6E-02 | 7.5E-02 | 6.9E-02 | 5.6E-02 | 4.8E-02 | 6.9E-02 | 4.8E-02 | 4.6E-02 | 7.5E-02 | 5.9E-02 | 1.0E-02 | 18.5 |
| Degradation and uptake (fraction) | 9.5E-01 | 9.4E-01 | 9.5E-01 | 9.2E-01 | 9.3E-01 | 9.4E-01 | 9.5E-01 | 9.3E-01 | 9.5E-01 | 9.2E-01 | 9.5E-01 | 9.4E-01 | 1.1E-02 | 1.1 |

**Table 4.** Distributions of pesticide among the environmental compartments obtained with Pest LCI 2.0 model and using site-specific data (TDF1) and those obtained applying data of soils with similar characteristics. Figures indicate the fraction of pesticide emitted in each environmental compartment. (Source: Fantin et al., 2019, reproduced by permission of Elsevier).

As far as the comparison among TDF1 and the similar soils is concerned, results highlight that both the emissions to air and surface water (or off-field soils) are scarcely affected or completely not affected by soil variation, with the exception of metolachlor in pre-emergence.

Concerning emissions to air, it is noteworthy that pesticide volatilization from leaves and top soil depend mainly on meteorological conditions, on pesticide chemical properties and on both pH and organic carbon content of soil. Despite some differences in organic carbon content in considered soils,(from 0.2% to 2.3%, see Table 1 and Table 2), the role of soil seems negligible in these cases. Terbuthylazine and cypermethrin show virtually identical values of airborne emissions for all soils. The volatilization of terbuthylazine comes mainly from top soil (order of magnitude $10^{-1}$) and in a minor amount from leaves (order of magnitude $10^{-3}$). In spite of this, the soil properties have little influence on the emissions to air since terbuthylazine has an organic carbon partitioning coefficient ($K_{oc}$) very low at the pH of the studied soils and therefore the pesticide volatilizes easily. In contrast, for cypermethrin, volatilization from soil occurs with at a very low rate (range from 4.0E-10 to 9.0E-10), meaning that emissions to air are dominated by volatilization from leaves (range from 4.0 E-04 to 5.0E-04) and the soil variations are irrelevant. As regards metolachlor, it volatilizes mainly from leaves but its $K_{oc}$ permits a slight soil volatilization. A high differentiation among the soils is found in pre-emergence (c.v.= 59.5%) while the variation decreases in post-emergence (c.v.= 4.5%). The decrease in differentiation after emergence is explained by the fact that in metolachlor emissions to air due to volatilization from leaves (about 2.0E-03) are typically 2 orders of magnitude higher than emissions due to volatilization from soil (range from 9.0E-05 to 1.5E-04). Before emergence, there are no leaves, so that the variation in emissions resulting from differences in soil properties is visible.

Emissions to surface water (or to off-field soil) seem to be even less affected by soil characteristics if compared to emissions to air for all pesticides both in pre and post-emergence (the maximum c.v. is 6.2% observed in terbuthylazine in pre-emergence). More in detail, emissions to surface water (or to off-field soil) consist of two contributions: wind drift loss (i.e. pesticide's droplets which are transported by wind and deposited on water or soil) and runoff, where the former is always at least one order of magnitude higher than the latter. The value of wind drift loss is only correlated to the application technique and the field size that is the same for all the soils. Therefore, the results of PestLCI 2.0 show that these values are completely independent from soil type. On the contrary, runoff fraction is strictly correlated to the characteristic of soil but it has a minor contribution on the total fraction emitted to surface water (or to off-field soils). The reason for wind drift dominating the off-field surface emissions is twofold: the field size modelled is small, resulting in more drift (the larger the field, the more drifting pesticide is deposited inside the field), and the slope of the field is 0, so that water does not readily start running off.

In contrast, Table 4 shows that soil type affects remarkably the emissions to groundwater (c.v. is about 15%, 40% and 18% for terbuthylazine, cypermethrin, metolachlor, respectively), although the characteristics of the nine soils analyzed are quite similar. However, a clear relationship between soil characteristics and the fractions of pesticide reaching groundwater is very difficult to be identified. Since the emissions to this environmental matrix are leaded by many parameters (related to the type of pesticide, the type of soil and the meteorological conditions), the behaviour of the pesticide has a high variability. Emissions to groundwater consist of emissions due to: i) leaching through the soil matrix and ii) emissions through macropores. As regards leaching through the soil matrix, Pest LCI 2.0 models the soil as a column, through which water moves downwards, taking the pesticide with it. However, the pesticide moves slower than water, because it is sorbed by soil. The factor that determines how much slower the pesticide moves compared to water is calculated from the density of the soil and its organic carbon content, which differs per soil horizons. As far as the soil density is concerned, for each horizon a specific density is calculated from the sand content and the fraction of organic carbon. In addition, the fraction of pesticide sorbed (and which is thus unavailable for degradation) differs per soil horizon and also depends on the organic carbon content of the soil. Finally, the rate at which water moves downward through soil depends, amongst others, on the sand content of soil.

Regarding macropores, it is important to notice that the total pore volume is the volume of water and air in the soil and that soil pores are classified into immobile, slow mobile and fast mobile pores. The fast ones are considered macropores (with a diameter >8 mm). PestLCI 2.0 splits pores into immobile and mobile pores on the basis of the fractions of sand, silt, and clay. Sand fractions give more mobile pores than clay fractions, reflecting the fact that water moves faster through sandy soils. Next, the mobile pores are split into slow and fast pores on a fixed 70/30 basis for all soil types (which results in the macropore fraction of 0.3 as input value for the model). The assessment of the amount of pesticide which reaches the groundwater through macropores is based on a tipping bucket approach. When it rains, the pores are filled, starting with the immobile pores, then the slow mobile, and finally the fast mobile. As a consequence of this approach, sandy soils have more mobile pores and can therefore store more water (because water moves faster in mobile pores than in immobile), therefore the pesticide reaching groundwater through macropore flow is lower (Hall, 1993). However, because the split between slow and fast mobile pores is fixed, sandy soils also have more macropores (which were defined as fast mobile pores).

The weight of soil matrix contribution and the weight of macropores contribution depend on the type of pesticide. In this study, the relevance of transportation through macropores and that of leaching through the soil matrix are different for the three pesticides. The fraction of terbuthylazine which reaches groundwater through soil leaching (about from 1.5E-01 to 2.0E-01) is about one order of magnitude higher than that from macropores (about 2E-02). In the case of metolachlor, the two fractions are more or less of the same order of magnitude (about 1.5E-03-2.0E-03). Finally, the fraction of cypermethrin which reaches groundwater through macropores (from 1.0E-06 to .0E-07) is four-five orders of magnitude lower than that coming from soil matrix (about from 4.0E-02 to 7.0E-02).

Correlation analyses among leaching through the soil matrix and soil characteristics (content of clay, silt and sand, pH and organic carbon percentage) do not highlight any particular trend (Figure 6). On the contrary, it is evident for metolachlor and cypermethrin a strong inverse linear correlation between emissions through macropores and organic carbon percentage ($R^2>0.8$). There is not this evidence for terbuthylazine ($R^2<0.3$) and that could be due to its very low $K_{oc}$ at the pH of the studied soils (Figure 6).

In conclusion, Table 4 shows that soil characteristics significantly affect the fraction of pesticide which reaches groundwater. The influence of soil variation on the leaching through the soil matrix and on the emissions through macropores depends on the chemical-physical characteristic of the pesticide.

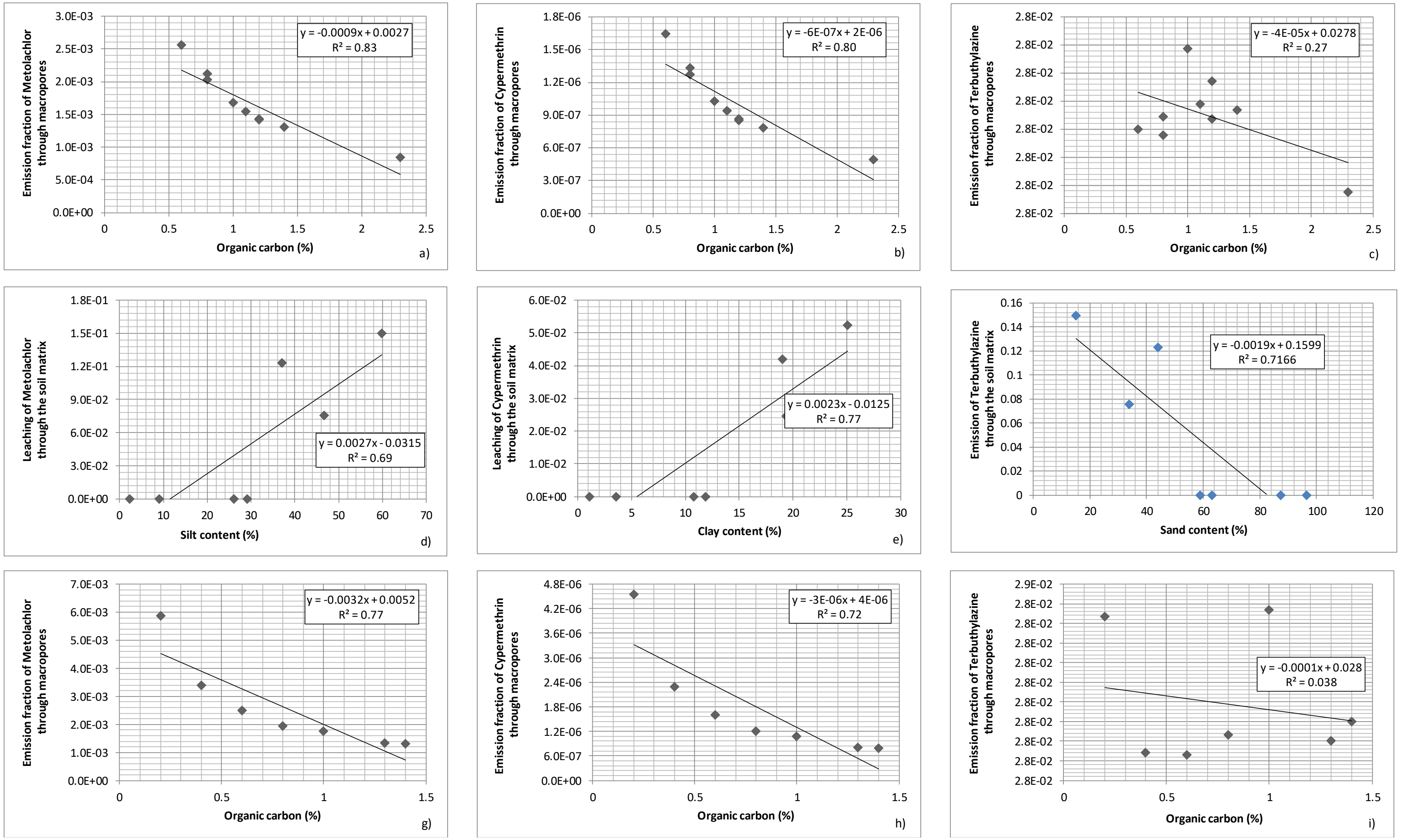



**Figure 6.** Correlation analysis among some soil characteristics and emission fractions of pesticides. Figures a), b) and c) refer to the first experiment (among similar soils); figures d), e), f), g), h) and i) refer to the second experiment (among different soils) (Source: Personal elaboration).

### 6.4.2 Results of Test 2: comparison among TDF1 and different soils

Table 5 shows the obtained fraction of emission to air, surface water or off-field soils, groundwater and the fractions degraded or uptaken by plants or soil of each active ingredient applied to TDF1 and the group of soils with different characteristics (VAN1, VED1, MEL1, CAP1, SAB1, VAD1). Minimum, maximum and average values, standard deviation and coefficient of variation (c.v.) have been included for each row.

Table 5 shows that emissions to air of terbuthylazine are not affected by the soil characteristics (c.v.= 0.1% and 0% for pre-emergence and post-emergence, respectively) and also emissions to surface water are slightly affected by soil type (2.7% and 1.8% for pre-emergence and post-emergence, respectively). On the contrary, soil typology remarkably affects the emissions to groundwater: in this case different soil characteristics lead to very different results (c.v. is over 90% in both pre and post-emergence). The comparison between Table 4 and Table 5 shows that the coefficient of variation in the emissions to groundwater of terbuthylazine increases significantly.

As regards metolachlor, a very high differentiation in emissions to air among the soils is found in pre-emergence (c.v.= 141%). As far as metolachlor in post-emergence is concerned, the results of the emissions to air regarding the different soils are slightly different (c.v.= 21.6%) from Table 4. Analogously to the results of Table 4, the decrease in differentiation of metolachlor in post-emergence is explained by the fact that emissions to air due to volatilization from leaves are much higher than emissions due to volatilization from soil. As regards emissions to surface water, metolachlor shows very similar results for all soils in pre-emergence, with the exception of VAD1. In fact, the runoff contribution in VAD1 is one order of magnitude higher than those of the other soils and this is probably due to the high sand content of this soil typology (97%). In post-emergence, wind drift contribution is about two orders of magnitude higher than runoff fraction, therefore the soil does not affect the results. As regards emissions to groundwater, the results are significantly affected by soil type. In fact, the coefficient of variation is about 55%, with a slight increase with respect to Table 4. Finally, the difference in the degraded or uptaken fraction is not significantly affected by the soil type.

| Terbuthylazine (pre-emergence) | TDF1 | VAN1 | VED1 | MEL1 | CAP1 | SAB1 | VAD1 | MIN | MAX | Average | Stand.Dev. | c.v.% |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Emission to air (fraction) | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 5.6E-05 | 0.1 |
| Emission to surface water or off-field soil (fraction) | 1.1E-03 | 1.1E-03 | 1.1E-03 | 1.1E-03 | 1.1E-03 | 1.0E-03 | 1.0E-03 | 1.0E-03 | 1.1E-03 | 1.1E-03 | 2.9E-05 | 2.7 |
| Emission to groundwater (fraction) | 3.0E-01 | 1.4E-01 | 2.1E-01 | 2.8E-02 | 2.8E-02 | 2.8E-02 | 2.8E-02 | 2.8E-02 | 3.0E-01 | 1.2E-01 | 1.1E-01 | 102 |
| Degradation and uptake (fraction) | 6.1E-01 | 7.7E-01 | 7.0E-01 | 8.8E-01 | 8.8E-01 | 8.8E-01 | 8.8E-01 | 6.1E-01 | 8.8E-01 | 7.8E-01 | 1.1E-01 | 14 |
| **Terbuthylazine (post-emergence)** | | | | | | | | | | | | |
| Emission to air (fraction) | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.3E-06 | 0.0 |
| Emission to surface water or off-field soil (fraction) | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.2E-03 | 1.2E-03 | 1.2E-03 | 1.3E-03 | 1.2E-03 | 2.2E-05 | 1.8 |
| Emission to groundwater (fraction) | 1.7E-01 | 9.7E-02 | 1.4E-01 | 2.2E-02 | 2.2E-02 | 2.2E-02 | 2.2E-02 | 2.2E-02 | 1.7E-01 | 8.2E-02 | 6.7E-02 | 91.6 |
| Degradation and uptake (fraction) | 7.1E-01 | 7.8E-01 | 7.3E-01 | 8.6E-01 | 8.6E-01 | 8.6E-01 | 8.6E-01 | 7.1E-01 | 8.6E-01 | 8.0E-01 | 6.8E-02 | 8.5 |
| **Metolachlor (pre-emergence)** | | | | | | | | | | | | |
| Emission to air (fraction) | 7.7E-05 | 1.6E-04 | 7.8E-05 | 5.0E-04 | 2.7E-04 | 1.3E-04 | 1.4E-03 | 7.7E-05 | 1.4E-03 | 3.5E-04 | 4.9E-04 | 141 |
| Emission to surface water or off-field soil (fraction) | 9.5E-03 | 9.6E-03 | 9.5E-03 | 9.7E-03 | 9.6E-03 | 9.4E-03 | 1.9E-02 | 9.4E-03 | 1.9E-02 | 1.1E-02 | 3.4E-03 | 32.3 |
| Emission to groundwater (fraction) | 2.6E-03 | 2.5E-03 | 2.1E-03 | 4.3E-03 | 3.2E-03 | 2.2E-03 | 7.5E-03 | 2.1E-03 | 7.5E-03 | 3.4E-03 | 1.9E-03 | 55 |
| Degradation and uptake (fraction) | 9.9E-01 | 9.9E-01 | 9.9E-01 | 9.9E-01 | 9.9E-01 | 9.9E-01 | 9.7E-01 | 9.7E-01 | 9.9E-01 | 9.9E-01 | 5.8E-03 | 0.6 |
| **Metolachlor (post-emergence)** | | | | | | | | | | | | |
| Emission to air (fraction) | 2.5E-03 | 2.6E-03 | 2.5E-03 | 3.0E-03 | 2.7E-03 | 2.5E-03 | 4.2E-03 | 2.5E-03 | 4.2E-03 | 2.8E-03 | 6.1E-04 | 21.6 |
| Emission to surface water or off-field soil (fraction) | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 6.4E-05 | 0.3 |
| Emission to groundwater (fraction) | 2.0E-03 | 2.0E-03 | 1.6E-03 | 3.4E-03 | 2.5E-03 | 1.8E-03 | 5.9E-03 | 1.6E-03 | 5.9E-03 | 2.6E-03 | 1.4E-03 | 55.8 |
| Degradation and uptake (fraction) | 9.8E-01 | 9.8E-01 | 9.8E-01 | 9.7E-01 | 9.8E-01 | 9.8E-01 | 9.7E-01 | 9.7E-01 | 9.8E-01 | 9.7E-01 | 2.0E-03 | 0.2 |
| **Cypermethrin (insecticides)** | | | | | | | | | | | | |
| Emission to air (fraction) | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.3E-08 | 0.0 |
| Emission to surface water or off-field soil (fraction) | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.4E-09 | 0.0 |
| Emission to groundwater (fraction) | 5.2E-02 | 2.5E-02 | 4.2E-02 | 2.3E-06 | 1.6E-06 | 1.1E-06 | 4.6E-06 | 1.1E-06 | 5.2E-02 | 2.1E-02 | 2.3E-02 | 133 |
| Degradation and uptake (fraction) | 9.5E-01 | 9.7E-01 | 9.6E-01 | 1.0E+00 | 1.0E+00 | 1.0E+00 | 1.0E+00 | 9.5E-01 | 1.0E+00 | 9.8E-01 | 2.3E-02 | 2.4 |

**Table 5.** Distributions of pesticide among the environmental compartments obtained with Pest LCI 2.0 model using site-specific data (TDF1) and those obtained applying data of soils with different characteristics. Figures indicate the fraction of pesticide emitted in each environmental compartment. (Source: Fantin et al., 2019, reproduced by permission of Elsevier).

As far as cypermethrin is concerned, the results of Table 5 show that emissions to air and to surface water are not affected by the soil characteristics, because the contribution of volatilization from leaves and wind drift are about four-five orders of magnitude higher than the contribution of top soil volatilization and runoff respectively. As regards emissions to groundwater, the soil type significantly affects these values. In fact, the coefficient of variation is 133%, with a great increase with respect to Table 4. Finally, degradation or uptake is not significantly affected by the soil type.

On the contrary of data concerning the similar soils, data of leaching through the soil matrix concerning the different soils (Table 5) shows a good direct linear correlation with clay and silt content ($R^2 \geq 0.7$ for terbuthylazine and cypermethrin; $R^2 \geq 0.5$ for metolachlor) and a good inverse linear correlation with sand content ($R^2 > 0.7$ for terbuthylazine and cypermethrin; $R^2 > 0.5$ for metolachlor) (Figure 6). In metolachlor and cypermethrin distribution, the strong inverse linear correlation between emissions through macropores and organic carbon percentage is confirmed also for the different soils group ($R^2 > 0.7$). On the contrary, there is no correlation between terbuthylazine reaching groundwater through macropores and organic carbon percentage ($R^2 < 0.03$) (Figure 6).

### 6.4.3 Results of Test 3: comparison between TDF1 and Soil6

Table 6 shows the obtained fraction of emission to air, surface water or off-field soils, groundwater and the fractions degraded or uptaken by plants or soil of each active ingredient applied to TDF1 and Soil6. Moreover, the percentage difference between the results of those two soils has been included.

Although the similar trends observed in TDF1 and Soil6 for pesticide and application period, the percentage differences between the two soils are, in some cases, significant or very high. The maximum percentage difference can be observed for metolachlor in both pre and post-emergence, where these values can be about 500%. The bigger differences concern the fraction emitted to groundwater, in particular the leaching fraction is one order of magnitude higher in Soil6 than in TDF1, while the fraction emitted to groundwater through macropores has the same order of magnitude in the two soils. Considering the discussion of Table 4 and Table 5, this large difference between the two soils in the fraction emitted to groundwater through the soil matrix should be due to the different sand content: indeed this characteristic is the one that differs the most from all the values observed in the other soils similar (Table 1) and different (Table 2).

| Terbuthylazine (pre-emergence) | TDF1 | Soil6 | % Difference |
|---|---|---|---|
| Emission to air (fraction) | 9.3E-02 | 9.3E-02 | 0% |
| Emission to surface water or off-field soil (fraction) | 1.1E-03 | 1.1E-03 | 0% |
| Emission to ground water (fraction) | 3.0E-01 | 4.1E-01 | 35% |
| Degradation and uptake (fraction) | 6.1E-01 | 5.0E-01 | -17% |
| **Terbuthylazine (post-emergence)** | | | |
| Emission to air (fraction) | 1.2E-01 | 1.2E-01 | 0% |
| Emission to surface water or off-field soil (fraction) | 1.3E-03 | 1.3E-03 | 0% |
| Emission to ground water (fraction) | 1.7E-01 | 2.8E-01 | 63% |
| Degradation and uptake (fraction) | 7.1E-01 | 6.0E-01 | -15% |
| **Metolachlor (pre-emergence)** | | | |
| Emission to air (fraction) | 7.7E-05 | 4.7E-05 | -39% |
| Emission to surface water or off-field soil (fraction) | 9.5E-03 | 9.5E-03 | 0% |
| Emission to ground water (fraction) | 2.6E-03 | 1.6E-02 | 500% |
| Degradation and uptake (fraction) | 9.9E-01 | 9.7E-01 | -1% |
| **Metolachlor (post-emergence)** | | | |
| Emission to air (fraction) | 2.5E-03 | 2.4E-03 | -2% |
| Emission to surface water or off-field soil (fraction) | 2.0E-02 | 2.0E-02 | 0% |
| Emission to ground water (fraction) | 2.0E-03 | 1.1E-02 | 455% |
| Degradation and uptake (fraction) | 9.8E-01 | 9.7E-01 | -1% |
| **Cypermethrin (insecticides)** | | | |
| Emission to air (fraction) | 4.1E-04 | 4.8E-04 | 17% |
| Emission to surface water or off-field soil (fraction) | 2.7E-04 | 2.7E-04 | 0% |
| Emission to ground water (fraction) | 5.2E-02 | 9.7E-02 | 85% |
| Degradation and uptake (fraction) | 9.5E-01 | 9.0E-01 | -5% |

**Table 6.** Results of the application of PestLCI 2.0 model for each active ingredient and for TDF1 soil and Soil6. Values indicate the fraction of pesticide emitted in each environmental compartment. (Source: Fantin et al., 2019, reproduced by permission of Elsevier).

### 6.4.4 Results of Test 4: TDF1 with different types of tillage

The effect of tillage system was tested considering three type of tillage: conventional (or traditional) tillage, minimum tillage and no-tillage (or zero-tillage). Table 7 shows, for each active ingredient and tillage system, the obtained fraction of emission to air, surface water or off-field soils, groundwater and the fraction degraded or uptaken by plants or soil. The soil considered is TDF1, which is the actual soil on which the cultivated area is located. Average values, standard deviations and coefficients of variation have been also included.

The results of Table 7 show that emissions to air and surface water and the uptake by plants are completely not affected by the tillage system. On the contrary, the emissions of terbuthylazine and metolachlor to groundwater are remarkably affected by the tillage type (see also Figure 7 and Figure 8). No effects are observed on the cypermethrin emission fractions.

It can be observed that the emission fractions to groundwater of terbuthylazine and metolachlor increase moving from conventional tillage, to minimum tillage and then to no tillage (Figure 7 and Figure 8). Conversely, the degradation fractions of these active ingredients decrease reducing the soil tillage. The highest emissions in groundwater observed in no-tillage system are caused by two main reasons: a) the organic carbon content in the soil; b) the greatest flow through the macropores.

No tillage corresponds to a high content of organic carbon in the topsoil (Alletto et al., 2007). Organic carbon binds extensively to the pesticide and keeps it on the surface, and prevents its degradation.

As far as the greatest flow through the macropores is concerned, it is caused by a minor soil treatment (Alletto et al., 2007). Tillage disturbs the soil and affects both the development and structure of macropore (Alletto et al., 2010). In fact, PestLCI 2.0 uses 3 tillage factors, based on Alletto et al. (2010): 1 for normal tillage, 3.5 for conservation tillage, and 7.5 for no tillage. Firstly, the model calculates, using a tipping-bucket approach, how much water (with dissolved pesticide) enters macropores, and thus which fraction of applied pesticide will become an emission to groundwater. Secondly, the tillage factors are applied: the emissions calculated in the first step are multiplied with (tillage factor/7.5). The result of this calculation is the amount of pesticide that is reported as an emission. Therefore, when no tillage is applied, the emissions will be 7.5 times larger than when the soil is conventionally tilled, reflecting the fact that tillage destroys macropore connectivity. This means that PestLCI 2.0 does not really consider how the soil structure changes as a consequence of tillage practice, but rather applies a fixed number on calculations for a conventionally tilled soil. Consequently, the less the soil is tilled the bigger is the fraction of pesticide that reaches groundwater. On the other hand, the less is the tillage, the more is the degradation of the pesticide.

The lack of tillage influence on cypermethrin emission fractions is due to the fact that cypermethrin reaches groundwater almost exclusively through leaching, indeed the fraction due to leaching is about 7% while that due to the macropores is of the order of

10.0E-05. Therefore, it is clear that soil compaction and the consequent reduction of the macropores have little or no influence on the pesticide fraction reaching the groundwater.

| **Terbuthylazine (pre-emergence)** | **Conventional tillage** | **Minimum tillage** | **No tillage** | **Average** | **Stand.Dev.** | **c.v.%** |
|---|---|---|---|---|---|---|
| Emission to air | 9.3E-02 | 9.3E-02 | 9.3E-02 | 9.3E-02 | 1.7E-17 | 0.0 |
| Emission to surface water or off-field soil | 1.1E-03 | 1.1E-03 | 1.1E-03 | 1.1E-03 | 0.0E+00 | 0.0 |
| Emission to ground water | 3.0E-01 | 3.5E-01 | 4.2E-01 | 3.6E-01 | 6.0E-02 | 16.9 |
| Degradation fraction | 6.1E-01 | 5.6E-01 | 4.8E-01 | 5.5E-01 | 6.6E-02 | 11.9 |
| **Terbuthylazine (post-emergence)** | | | | | | |
| Emission to air | 1.2E-01 | 1.2E-01 | 1.2E-01 | 1.2E-01 | 0.0E+00 | 0.0 |
| Emission to surface water or off-field soil | 1.3E-03 | 1.3E-03 | 1.3E-03 | 1.3E-03 | 0.0E+00 | 0.0 |
| Emission to ground water | 2.1E-01 | 2.4E-01 | 3.0E-01 | 2.5E-01 | 4.6E-02 | 18.3 |
| Degradation fraction | 6.6E-01 | 6.3E-01 | 5.7E-01 | 6.2E-01 | 4.6E-02 | 7.4 |
| Uptake fraction | 6.5E-03 | 6.5E-03 | 6.5E-03 | 6.5E-03 | 0.0E+00 | 0.0 |
| **Metolachlor (pre-emergence)** | | | | | | |
| Emission to air | 7.4E-05 | 7.4E-05 | 7.4E-05 | 7.4E-05 | 0.0E+00 | 0.0 |
| Emission to surface water or off-field soil | 9.5E-03 | 9.5E-03 | 9.5E-03 | 9.5E-03 | 0.0E+00 | 0.0 |
| Emission to ground water | 4.5E-03 | 8.7E-03 | 1.5E-02 | 9.4E-03 | 5.3E-03 | 56.2 |
| Degradation fraction | 9.9E-01 | 9.8E-01 | 9.8E-01 | 9.8E-01 | 5.8E-03 | 0.6 |
| **Metolachlor (post-emergence)** | | | | | | |
| Emission to air | 2.5E-03 | 2.5E-03 | 2.5E-03 | 2.5E-03 | 0.0E+00 | 0.0 |
| Emission to surface water or off-field soil | 2.0E-02 | 2.0E-02 | 2.0E-02 | 2.0E-02 | 0.0E+00 | 0.0 |
| Emission to ground water | 3.2E-03 | 6.5E-03 | 1.2E-02 | 7.2E-03 | 4.4E-03 | 61.5 |
| Degradation fraction | 9.7E-01 | 9.70E-01 | 9.6E-01 | 9.7E-01 | 7.2E-03 | 0.7 |
| Uptake fraction | 4.5E-04 | 4.5E-04 | 4.5E-04 | 4.5E-04 | 6.6E-20 | 0.0 |
| **Cypermethrin (insecticides)** | | | | | | |
| Emission to air | 4.1E-04 | 4.1E-04 | 4.1E-04 | 4.1E-04 | 0.0E+00 | 0.0 |
| Emission to surface water or off-field soil | 2.7E-04 | 2.7E-04 | 2.7E-04 | 2.7E-04 | 0.0E+00 | 0.0 |
| Emission to ground water | 6.6E-02 | 6.6E-02 | 6.6E-02 | 6.6E-02 | 0.0E+00 | 0.0 |
| Degradation fraction | 9.3E-01 | 9.3E-01 | 9.3E-01 | 9.3E-01 | 0.0E+00 | 0.0 |
| Uptake fraction | 1.2E-03 | 1.2E-03 | 1.2E-03 | 1.2E-03 | 0.0E+00 | 0.0 |

**Table 7.** Results of the application of PestLCI 2.0 model for each active ingredient and for TDF1 soil varying the tillage system. (Source: Fantin et al., 2019, reproduced by permission of Elsevier).

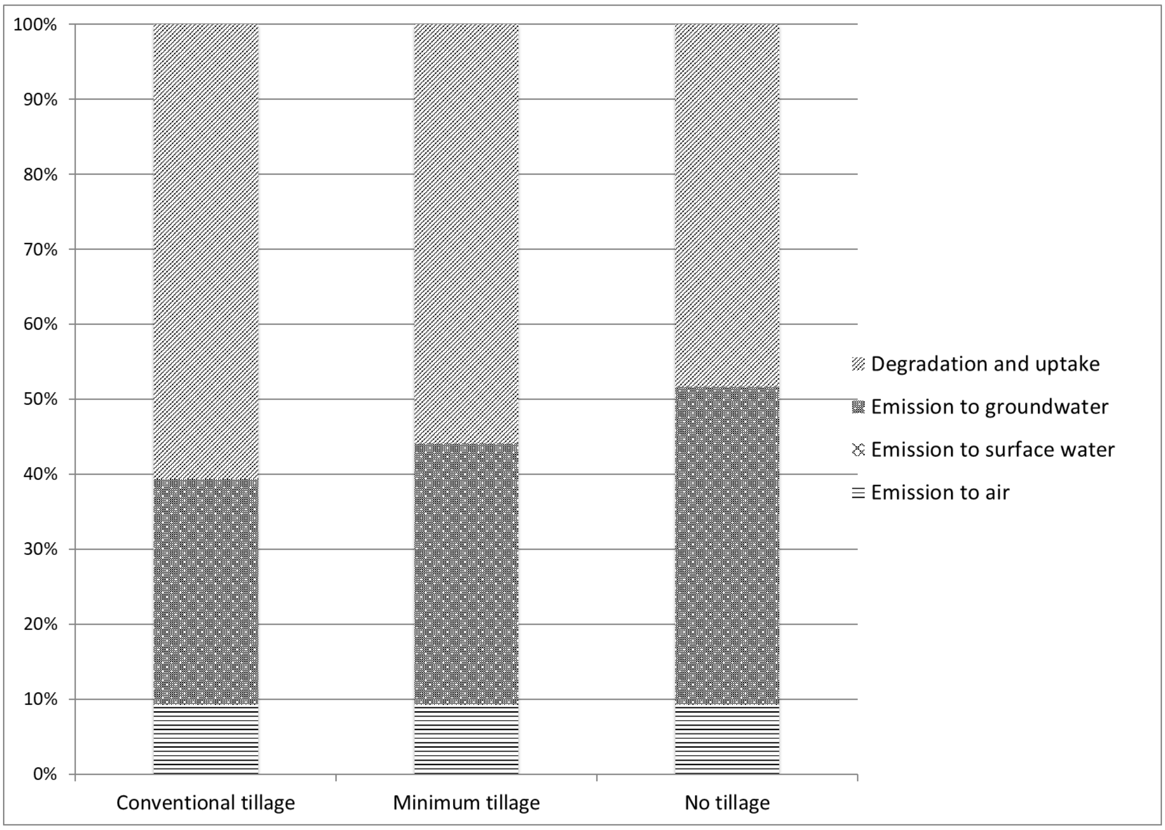



**Figure 7.** Percentage of Terbuthylazine emitted in pre-emergence according to the different types of tillage (Source: Personal elaboration).

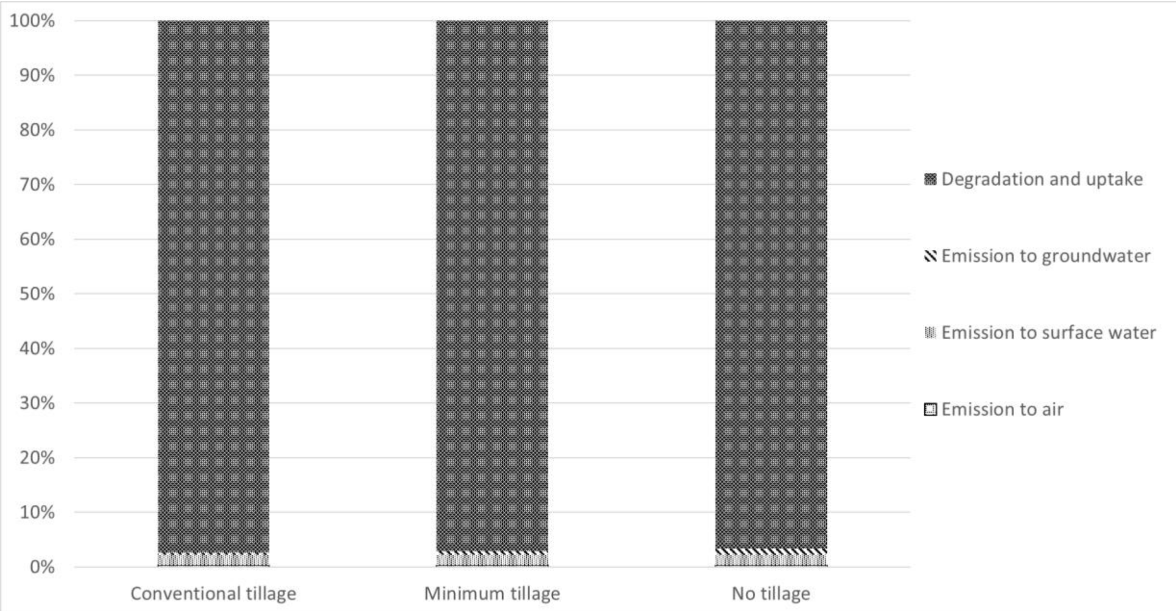



**Figure 8.** Percentage of Metolachlor emitted in post-emergence according to the different types of tillage (Source: Personal elaboration).

## 6.5 Conclusions

This study shows that little variations in soil characteristics lead to great variation of PestLCI 2.0 results concerning the distribution of the pesticides, with a significance that depends on the type of environmental compartment. The compartment most affected by soil variations is groundwater, as demonstrated by the coefficient of variation that in some cases is up to 100%. Indeed, emissions to air are dominated by meteorological conditions and pesticide physical and chemical properties, while emissions to surface water (or off-field soil) are dominated by wind drift, completely independent from soil characteristics. On the contrary, both the emissions to groundwater, i.e. leaching through the soil matrix and emissions through macropores, are strictly related to the features of the soils.

Therefore, it is evident that the use of specific soil data in PestLCI 2.0 results in the availability of a more comprehensive set of emission data in the different compartments, which represents a relevant input for the inventory phase of LCA studies

However, whether this comprehensive set of data is also accurate and reliable depends on the background assumptions of the PestLCI 2.0 modelling. In this regard, the study has allowed us to highlight some important features related to how soil characteristics are dealt with in the model and to provide further insights for improving the model itself. The assumption that the ratio between slow mobile macropores and fast mobile macropores is the same in all type of soil (0.7 and 0.3, respectively) is a strong simplification and it does not reflect the reality. The scientific literature reports that macropores occur more in structured soils (clayey and silty soils) and less in destructured soils (sandy soil) (Hall, 1993). This assumption of PestLCI 2.0 modifies the speed of water in sandy soil and the role of macropores in the emission to groundwater. The best solution would be to set the fraction of macropores to the soil type as a function of their clay, silt and sand content. Another important feature is that the model considers only the top 1 meter depth of soil. This assumption is based on two motivations. Firstly, 1 meter is sufficiently deep to draw the line between the technosphere and the ecosphere. In such way PestLCI 2.0 assumes that the field below 1 meter is not manipulated by agricultural practice. Secondly, PestLCI 2.0 assumes that pesticide degradation stops below 1 meter. This hypothesis implies that when a pesticide reaches 1 m of depth, it will at some point reach the groundwater. Therefore, the exact depth of water table is not important. It is evident that if the water table is lower than 1 m (i.e. coastal or spring areas), this assumption is misleading.

Results also showed that the conservation agricultural techniques, such as reduced tillage or no tillage, lead to higher emissions to groundwater if compared to conventional tillage. This is due to the simplified assumption that macropore leaching is reduced by a fixed factor of 7.5 when conventional tillage practices are applied. In the future development of PestLCI 2.0, the change in the soil structure as a consequence of tillage practice should therefore be considered for obtaining more reliable and accurate results.

A final observation is that PestLCI 2.0 considers organic horizons as mineral horizons. In the model, organic carbon is only used to calculate the fraction of pesticide sorbed and the density of the soil horizon. This simplification could be another limitation in the case of soils with a high level of organic carbon, because in such horizons the organic carbon has an important role especially in the downward movement of water and substances. Likewise, PestLCI 2.0 does not take into account the presence of rock fragments (soil skeleton > 2 mm) in the calculation of emissions to surface and groundwater. This might be considered for future model updates.

The application of PestLCI 2.0 with soil specific data to the four scenarios, has pointed out that it is necessary to collect detailed information on soil characteristics, and – more importantly – to be able to interpret them carefully, thus requiring a specific expertise on soil parameters and features. However, whether this high-resolution and resource-intensive data collection is worthy for the robustness of the results of the LCA studies, depends also on the capabilities of the characterization models applied in the life cycle impact assessment, such as USEtox (Rosenbaum et al., 2008), to capture them. The USEtox model is the recommended impact assessment model in the Product Environmental Footprint method (European Commission, 2017) for the toxicity-related impact assessment categories, and PestLCI 2.0 was conceived to be applied in combination with its characterization factors (CFs). However, currently a limited work has been carried out on the development of CFs for groundwater (Dijkman et al., 2012). As a consequence, the high resolution of PestLCI 2.0 in delivering pesticide distribution among the environmental compartments is not fully captured in the characterized results of USEtox.

Despite this uncomplete description of the impact of pesticides, and the lack of experimental data that can support the calculated emissions among the compartments, however the tailoring of PestLCI 2.0 to soil-specific conditions proved to enrich the information at the life cycle inventory phase, which is a fundamental step for increasing the knowledge about the behaviour of substances in a life cycle perspective, and it paves the road for future developments of the impact assessment models.

# 7 Product Environmental Footprint Category Rules for dairy products

Product Environmental Footprint Category Rules (PEFCR) for dairy products were developed by the European Dairy Association within the PEF pilot phase which took place between 2015 and 2018. This PEFCRs aims to provide harmonized guidelines for the calculation of the environmental footprint of dairy products, providing detailed requirements for each stage and process of the life cycle. In particular, it is based on the ISO 14040-44 standards and provides guidance for the use of primary and secondary data, data quality requirements, allocation rules and impact categories to be addressed (European Dairy Association, 2016). The document covers the entire life cycle (from cradle to grave) of dairy products and includes the following dairy products from cattle: liquid milk, dried whey products, cheeses, fermented milk products, butterfat products. Other products, such as creams, milk-based desserts, Greek-style yoghurts or products from goat and sheep milk, are not included in the scope of the PEFCR.

The PEFCR for dairy products should therefore represent a complete, clear and harmonized guideline for the application of the PEF method to dairy products, and should overcome the critical methodological issues of the application of LCA to this production chain, and more in general of the agri-food sector, some of which have been identified by a detailed literature review in par. 4.3. It should therefore support the practitioner in the execution of the PEF study and provide more robust and reproducible results, while assuring products comparability, which is particular important in case a company would like to use the PEF results of their own product for communication purposes, to both other companies and consumers.

The following paragraphs will critically analyse how the PEFCR for dairy products deals with the critical methodological topics identified in par. 4.3, to evaluate if the harmonization process represented by the development of PEF method has fulfilled most of the open methodological problems of the application of LCA to food and drink supply chain, or if further harmonization efforts are needed. Then, the PEF method and the PEFCR guidelines will be applied to the assessment of the environmental footprint of Taleggio cheese production in a dairy company in Northern Italy, to evaluate its environmental hotspots, in order also to support the identification of improvement potential measures, and to test the applicability of the PEF method, highlighting its strengths and weaknesses, as well as any difficulty encountered during the study.

## 7.1 Functional unit

According to the PEFCR for dairy, the default functional unit of dairy products must be based on the mass or volume of the products. However, other functional units can be used when the goal of the study is to justify alternative choices. In these cases, the serving size or the nutritional value could be selected in addition to the default functional unit. For example, the functional unit for cheeses is “10 g dry matter of cheese, consumed at home as final product without cooking or further transformation”, and that for liquid milk is “1000 ml liquid milk, consumed at home without heating or cooking” (European Dairy Association, 2016). Therefore, the PEFCR for dairy provides a detailed definition of the functional unit for this kind of products, and in case of cheese, the quantity in mass is transformed in quantity of dry matter, which is a key parameter for the execution of the PEF study and can be judged as a simplified representation of the quality of the product.

## 7.2 System boundaries

According to the PEF Guide (European Commission, 2013a), the entire product life cycle must be assessed in the PEF study. For that reason, the PEFCR for dairy requires to evaluate the whole life cycle of these products, from cradle to grave. This means that the following stages are mandatory for a PEF study on dairy products (European Dairy Association, 2016):

- Raw milk supply, which includes the on-farm feed production, the cow breeding and milking phases and the milk transport to the dairy plant;
- Dairy products processing, which accounts for the milk processing at the dairy plant, the packaging process, the on-site storage;
- Non-dairy ingredients supply, which includes the production and transport of non-dairy ingredients (e.g. salt);
- Packaging, which takes into account the production and transport of primary and secondary packaging;
- Distribution, which includes the transport to the distribution centres, to the point of sale and to consumer’s home and the related storage;

- Use, which accounts for the refrigerated storage at home;
- End of Life, which includes the packaging and food waste transport and treatment.

It is noteworthy that food losses throughout the supply chain and food waste at consumer's home must be included in the PEF study, due to their importance, using primary data when available. In case of lack of site-specific data, the PEFCR provides default food losses and waste rates to be used in the study.

## 7.3 Handling of multi-functionality

In order to deal with multi-functionality in the dairy product supply chain, the PEFCR follows the ISO 14040 decision hierarchy for multi-output processes, described at paragraph 4.3.3. This means that subdivision of unit processes must be followed as first choice, then system expansion should be applied if subdivision is not possible, and finally allocation based on physical relationship or other type of relationship should be used when allocation cannot be avoided (European Dairy Association, 2016).

The PEFCR specifies the stages in the life cycle of dairy products which have multifunctional products or multi-output processes, and for each of them provides guidance on how to deal with multi-functionality (European Dairy Association, 2016):

- Raw milk production at the dairy farm;
- Dairy products processing at the dairy unit;
- Transportation from retail to consumer home;
- Materials recycling, or incineration with energy recovery at the end-of-life.

As regards the dairy farm, different co-products can be considered in addition to the raw milk, i.e. other dairy products directly produced at farm, live animals leaving the farm for slaughter or further fattening, dead animals, manure, sold feed and arable products and energy produced on the farm (European Dairy Association, 2016). Specific requirements are provided for each of those co-products, although some of them can be quite complicated for the practitioner, for example the use of the Circular Footprint Formula when manure is considered as waste or the use of system subdivision for the production of energy, which is not always possible, for example when the farm has an anaerobic digestion plant fed with energy crops cultivated at the same farm, used also for animal feed production.

In particular, the allocation of upstream impacts between raw milk and live animals must be based on the IDF biophysical allocation method (IDF, 2015). The allocation factor (AF) between raw milk and meat must be calculated by the following formula (European Dairy Association, 2016):

$$AF = 1 - 6.04\ x\ \frac{M_{meat}}{M_{milk}} \qquad (1)$$

Equation 1: Allocation factor between milk and meat at the dairy farm (IDF, 2015)

Where:

- $M_{meat}$ is the mass of live weight of all animals sold including bull calves and culled mature animals per year;
- $M_{milk}$ is the mass of fat and protein corrected milk (FPCM) sold per year (corrected to 4% fat and 3.3% protein).

The allocation of the environmental impact of raw milk and its transport from the dairy farm to the dairy plant must be performed by mass allocation using the Dry Matter (DM) content of the studied product and its co-products, with the following formula (European Dairy Association, 2016):

$$AF_i = \frac{DM_i \text{ x } Q_i}{\sum_{i=1}^{n}(DM_i \text{ x } Q_i)} \qquad (2)$$

Equation 2: Formula for the allocation factor based on the dry matter content.

Where:

- AFi = Allocation factor for the co-product i of the dairy unit, % (dimensionless)
- DMi = Dry matter content of product i (expressed as % dry matter or as weight by mass of dry matter/weight by mass of product i).
- Qi = Quantity of product i output to the production site or from the unit operation (in kg of product i).

The allocation of energy use, all the other input materials and emissions at the dairy plant must be allocated on the basis of dry matter content of each of the dairy products produced at the dairy plant, following the above formula. As regards the dry matter content, the PEFCR requires the practitioner to use the actual dry matter content of the studied product. When these primary data are not available, default dry matter contents, provided by the PEFCR for some typical dairy products, can be used.

The transportation of dairy products from retail to consumer home by car must be performed by an allocation per item, considering that each item is allocated 5% of the journey, based on the assumption that 20 products are purchased at a time. The PEFCR then provides the average quantity (in mass) corresponding 1 item of some typical dairy products: for example, 1 item of liquid milk is equal to 1 litre, and 1 item of cheese is equal to 100 g dry matter (European Dairy Association, 2016).

Finally, for the recycling and incineration of packaging materials at the end-of-life, which are multi-output processes, the Circular Footprint Formula (CFF) from the PEFCR Guidance (European Commission, 2017) must be used, which aims to help the practitioner to deal with multi-functionality in recycling, re-use and energy recovery situations. The CFF includes the environmental impacts of production process, the impacts and benefits of both secondary material inputs and outputs and the impacts and benefits of energy recovery and disposal (Bach et al., 2018). However, the CFF has received some criticism in the scientific literature (Bach et al., 2018; Ojala et al., 2016), it is very time consuming for the practitioner and quite difficult to be applied, especially regarding the estimation of the different required parameters and the identification of proper life cycle inventory data to be used in the formula for modelling materials recycling, energy recovery and materials disposal.

## 7.4 On-farm pesticides and fertilisers emissions and livestock emissions

The PEFCR for dairy products states that the on-field emissions from the use of pesticides must be included in the PEF study, but considering the simplified approach provided by the Ecoinvent Database, according to which the total amount of pesticide is emitted to the soil (European Dairy Association, 2016). The use of more detailed models such as PestLCI 2.0 is therefore not suggested. In particular, PestLCI is not considered enough robust to evaluate the connection between the quantity of pesticide applied on the agricultural field and the quantity of active ingredient emitted to the different environmental compartments (European Commission, 2017). According to the PEFCR Guidance (European Commission, 2017), the PestLCI 2.0 model is still under testing, although it could fill in the above-mentioned gap in the future. Therefore, the PEF method and therefore the PEFCR for dairy products prefer to use a quick and ready solution for the calculation of pesticides emissions, instead of a more scientific, but also more complicated and time-consuming model, which requires the use of site-specific climate and soil data.

As regards the calculation of emissions from the use of chemical and organic fertilisers, the PEFCR for dairy provides a very long list of emitted substances to be included in the PEF study (i.e. direct $N_2O$ emissions, indirect $N_2O$ emissions due to nitrogen volatilisation and nitrogen leaching, $NH_3$ and $NO_x$, $PO_4^-$, $NO_3^-$) and the name of the model or method which has to be used for their calculation as minimum requirement (European Dairy Association, 2016). For example, IPCC Tier 1 (IPCC, 2006) must be applied as minimum requirement for the calculation of direct $N_2O$ emissions from the nitrogen fertiliser or manure application, but no detail is provided about the exact equations of the IPCC Tier 1 model to be used or the emission factors which have to applied. This means that the practitioner has to study the method required by the PEFCR and to find or calculate by themselves the emission factors needed. Moreover, the mandatory use of EMEP/EEA Tier 2 (EMEP/EEA, 2013) for the calculation of $NH_3$ and $NO_x$ emissions is even more difficult: in fact, this model requires the use of very specific data, such as the nitrogen excreted by livestock expressed as total ammoniacal nitrogen or the total nitrogen excretion rates which require specific knowledge of the livestock sector. In the same way, the use of IPCC Tier 2 model is required by the PEFCR for the calculation of methane emissions from cattle enteric fermentation, which involves a detailed computation of emission factors for each type of animal based on the daily energy intake, the animals' weight and other several parameters about the energy from feed available for the different animal activities. The use of IPCC Tier 2 model is therefore rather difficult and resource-intensive.

Further guidance should therefore be provided by the PEFCRs document, both for practitioners and especially for companies when they want to perform a PEF study for communication purposes. In particular, Small and Medium Enterprises (SMEs) seldom have the scientific knowledge necessary for the use of these guidelines. Therefore, simplified tools such as spreadsheets with formulas and emission factors that can quickly be used could be developed. Annex 1 includes an example of an electronic spreadsheet developed during the PhD for the calculation of enteric fermentation emissions from livestock for the execution of a PEF study on dairy products.

## 7.5 Water use and related impacts

In the PEFCR for dairy products, the calculation of water use at both inventory level and impact assessment level follows the approach of ISO 14046 rather than that of the WFN, being the PEF method developed by the LCA community. As regards the water use at inventory level, the PEFCR for dairy divides water consumption in the dairy supply chain into on-farm irrigation water for feed crops and drinking and cleaning water (European Dairy Association, 2016). In both cases, water must be differentiated among ground, surface and tap water and this means that only blue water use must be included in the PEF study. Moreover, the PEFCRs for dairy requires the practitioner to use a regionalised water flow in the PEF model, i.e. country-specific water flow must be used, which can be found in life cycle inventory data contained in commercial LCA softwares, although regionalised water flows cannot be always available, as stated by Bach et al. (2108). The calculation of green water used by the evapotranspiration processes of the crop-soil system is therefore not included in the requirements of the PEFCR and grey water is not mentioned in the same way, following a different approach from that of the WFN. Finally, the calculation of environmental impacts of water use must be performed by means of the AWARE impact assessment method (European Dairy Association, 2016).

# 8 PEF study on Taleggio cheese production

The PEF study was performed in the framework of PEFMED European project, coordinated by ENEA, funded by the Interreg MED Programme 2014-2020 and co-financed by European Regional Development Funds (ERDF), which aims at improving the environmental sustainability of Mediterranean agri-food supply chains by means of the application of PEF method to several agri-food regional systems. As a reference for this PEF study the PEFCR document for Dairy Products – Updated DRAFT for public consultation – July 28, 2016 was used (European Dairy Association, 2016).
The work performed on the PEF study of Taleggio cheese during the PhD was included in the following publication, which is the basis of the whole chapter 8:

- Fantin V., Cortesi S., Chiavetta C., Rinaldi C., 2019. **PEF study of Dairy product chain in Lombardy region (Italy).** ENEA Technical Report USER-PH77-001, January 2019 (Confidential).

## 8.1 Goal and scope of the study

### 8.1.1 Goal of the study

The aim of the study is to identify hotspots in the life cycle of an Italian Taleggio cheese supply chain, in order to support the identification of potential improvement scenarios. A further purpose is to test the applicability, especially in agri-food Small and Medium Enterprises in the European Mediterranean area, of the PEF Category Rules (PEFCR).

### 8.1.2 Functional unit and reference flow

The functional unit is 10 g dry matter of cheese, consumed at home as final product without cooking or further transformation, in compliance with the PEFCR for dairy products. The reference flow is the amount of packed Taleggio cheese needed to obtain 10 g dry matter of cheese, equivalent to 20.4 g Taleggio cheese. Packaging is therefore included in the functional unit, because it is an integral part of the final cheese product, as suggested by the PEFCR for dairy products.

### 8.1.3 Description of the life cycle of the analysed product

Taleggio cheese is a semi-soft Italian cheese produced with whole pasteurized cow milk, with the Product Designation of Origin (PDO) label. The minimum fat content is 48% (Dry Matter basis) and the maximum water content is 54%. Taleggio cheese is a semi-soft Italian cheese produced with whole pasteurized cow milk, with the Product Designation of Origin (PDO) label. The minimum fat content is 48% (Dry Matter basis) and the maximum water content is 54%. It is produced in few provinces within the following regions of Northern Italy: Lombardy, Piedmont and Veneto. Raw milk for Taleggio production must be produced exclusively in dairy farms located in the above-mentioned areas which have a specific quality control system. Each Taleggio cheese weighs between 1.7 and 2.2 kg, depending on the production technique and ageing duration, and it is shaped like a square slab measuring 18-20 cm in length and width and 4-7 cm in depth.

The dairy company chosen for the PEF study is located in Cremona province and processes about 16,000 t of milk per year, with a production of about 1,700 t of several types of cheese, apart from Taleggio (ex. Gorgonzola, Quartirolo Lombardo, Salva Cremasco, etc.) both from conventional and organic milk.

The dairy company produces Taleggio cheese from both conventional and organic milk, the former representing more than 97% of the Taleggio produced by the dairy and then being the type selected for the analysis.

More than 80% of the conventional raw milk for Taleggio production is produced in 11 dairy farms located in the surrounding area near the dairy company (Cremona and Lodi provinces), and is transported twice a day to the dairy farm by thermally-insulated trucks. The rest of the milk used for Taleggio cheese production is purchased from the market but must be produced within the accepted areas for PDO production.

After the reception, milk is pasteurised by heating it in order to destroy all potentially harmful pathogens and then refrigerated at 4 °C. Then milk is heated at 32-35 °C and a mix of Lactobacillus bulgaricus and Streptococcus thermophilus is added. After that, rennet is added in order to coagulate milk and separate it into curd and liquid whey. Curd is cut twice: the first time into large pieces, which are then left for 10-15 minutes, in order to release more whey and the second time into small pieces. Once this process is complete, curd is placed evenly in moulds with sides 18-20 cm long. Here whey is drained away and moulds are placed on special tray tables with raised sides which are covered with plastic mats. The cooking process is one of the most important stages. Curds and whey are shaped into cheese, followed by a process of acidification when whey is drained away. This phase, during which cheese is turned several times, can last from a minimum of 8 hours to a maximum of 16 hours, at a temperature of 22-25 °C. During this “cooking” phase Taleggio cheese is branded with a plastic brand. Every cheese producer applies its consortium number on the lower left-hand corner of each cheese, in order to identify where Taleggio was produced.

Salting process is another fundamental stage because salt helps to eliminate the remaining whey and favours the rind development, which flavours the cheese and protects its external part from harmful micro-organisms, allowing only useful ones to develop. In industrial cheese production, cheeses are submerged in a salt solution at 10 °C. Taleggio cheese remains in the solution for 8-12 hours, during which it is turned several times.

The final phase is the ageing process which, in the case of the dairy company, is carried out by ageing companies located in Bergamo and Lecco provinces which buy unripened Taleggio cheese from the dairy. Taleggio cheese is aged on wooden boards in refrigerated cells with specific temperature (2-6 °C) and humidity (85-90%) conditions. During the ageing process, cheeses are turned and sponged with a saline solution every seven days. This “washing” keeps the rind damp, eliminates any unwanted mould which may have formed and encourages the right kind of mould and yeasts which causes the characteristic pink colour. The length of the ageing process and the conditions in which the cheese is aged (temperature and humidity) depends on the type of cheese. Taleggio cheese is aged for a minimum of 35 days. Finally, Taleggio cheese is transported to distribution centres, retailers and supermarkets for the household consumption.

### 8.1.4 System boundaries and system boundaries diagram

According to the PEFCR for dairy products, the system boundaries of Taleggio cheese include seven main phases (Figure 9):

- Raw milk supply;
- Dairy processing;
- Non-dairy ingredients supply;
- Packaging;
- Distribution;
- Use;

- End-of-life.

The foreground system includes the “Dairy processing”, “Non-dairy ingredients supply” and “Packaging” stages, where primary data or semi-specific primary data were collected from the dairy and ageing companies. This choice has been performed in compliance with both the application of the Data Needs Matrix (DNM) for dairy products contained in the PEFCR Guidance 6.2 (European Commission, 2017) and the PEFCR for dairy products, considering the dairy processors perspective. In fact, in the case of Taleggio cheese produced by the studied dairy company, raw milk production is not run by the dairy company and it does not have access to company-specific information because dairy farms are not directly controlled by the dairy company. Therefore, the “Raw milk supply” phase is in Situation 3 of the DNM (i.e. the process is not run by the company applying the PEFCR and this company cannot access to company-specific information) and the “Dairy processing” phase is in Situation 1 of the DNM (i.e. the process is run by the company applying the PEFCR). It is noteworthy that, in this case study, the “Dairy processing” phase consists of the main sub-phases “Dairy company” (unripened Taleggio production) and “Ageing company” (ripened Taleggio production and packing), for both of which primary data were collected.

Therefore, the background system considers the following phases: “Raw milk supply”, “Distribution”, “Use” and “End-of-life”, all of which are in Situation 3 of the DNM (i.e. the process is not run by the company applying the PEFCR and this company has no possibility to have access to company-specific information). Consequently, secondary data from commercial life cycle databases were used to model them.

The following activities were included in the “Raw milk supply” life cycle stage:

- Crops cultivation for on-farm feed production;
- Cow breeding and milking.

The following activities were included in the “Dairy processing” stage:

- Dairy products processing at the dairy plant (including energy use and wastewater treatment, cleaning agents consumption, water consumption, refrigerant gases consumption);
- Ageing process at the ageing company (including energy use and wastewater treatment, cleaning agents consumption, water consumption, refrigerant gases consumption);

- Transportation of unripened Taleggio cheese between dairy and ageing companies
- Dairy ingredients processing (including energy use and wastewater treatment);
- Dairy ingredients transport to dairy unit;
- Product packaging phase;
- On-site storage.

The following activities were included in the “Non-dairy ingredients supply” stage:

- Production of non-dairy ingredients;
- Non-dairy ingredients transport to dairy.

The following activities were included in the “Packaging” stage:

- Raw materials production;
- Primary, secondary and tertiary packaging manufacturing;
- Packaging transport to the dairy.

The following activities were included in the “Distribution” stage:

- Transport to the distribution centres;
- Storage at distribution centres;
- Transport to point of sale;
- Storage at point of sale;
- Packaging and food waste transport and treatment;
- Transport to final users.

The following activity was included in the “Use” stage:

- Product refrigeration.

The following activity was included in the “End-of-life” stage:

- Household waste: packaging and food waste transport and treatment.

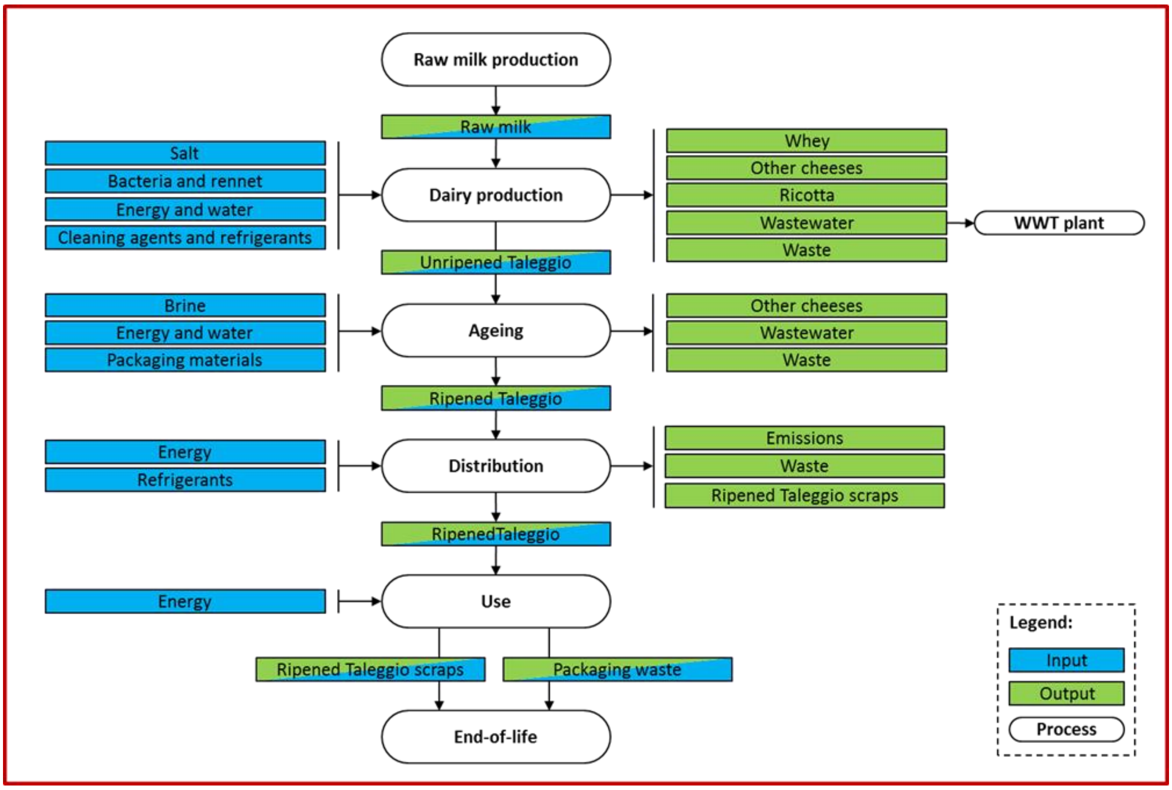



**Figure 9.** System boundaries of Taleggio cheese production (Source: Personal elaboration).

In compliance with the PEFCR for dairy products, the following processes and activities were excluded from the system boundaries:

- Transportation of input products to the dairy plant accounting for less than 1% in mass;
- Solid waste at the dairy unit;
- Capital goods at farm, at distribution centre and at retail (stables and machinery equipment and maintenance);
- Refrigerant emissions from milk cooling at farm;
- Ambient storage at consumer home;
- Cutlery for dairy products consumption at consumer home;
- Dishwashing or hand washing at the consumer home.

Lactic acid bacteria production and transport to the dairy were not included in the study because their weight is not available since they are measured by colonies and no background Life Cycle Inventory dataset is available in LCA databases. Likewise, rennet production was not included due to lack of life cycle data about its production.

### 8.1.5 Assumptions and relevant justification

The following assumptions were made in the study:

- Auxiliary materials (crate, mould, brand, mat and plastic rattan) were considered to be used only for Taleggio cheese production, although they could be used also in the production of other cheeses, because more detailed data were not available. Moreover, due to lack of data about the reuse of auxiliary materials, it was considered that they were used only once, without any reuse (conservative approach).
- The quantity of conventional whey from cow milk was obtained by subtracting the amount of cheeses and ricotta obtained from conventional cow milk from the total quantity of conventional cow milk in input to the dairy plant.
- The classes of dry matter content of goat milk cheeses, produced by the dairy together with cow milk cheeses, were assumed to be the same as those of cow milk cheeses.
- Two classes of dry matter for conventional cheese were considered: 1) soft cheeses (49% dry matter); 2) semi-hard cheeses (59.9% dry matter).
- Conventional cow milk purchased from the market can be used for Taleggio cheese production only if it is compliant with the PDO regulation. However, since the percentage of this milk compliant with the PDO regulation was not available, it was considered that 100% of it was compliant with the regulation and used also for Taleggio cheese production.
- Milk purchased from the market was considered to be produced in Treviso province (250 km of distance), which is the farthest area from the dairy company where cow milk for Taleggio cheese production can be produced, according to PDO regulation (conservative approach).

### 8.1.6 Information about the data used and data gaps

Primary data were collected from dairy and ageing companies and refer to 2016.
As regards the ageing companies, 5 ageing companies purchase Taleggio cheese from the dairy company for this production stage. However, only 1 of them was selected for the PEF study on the basis of dairy company's expert judgment, because it was considered the most representative company in terms of treated cheese quantity. Therefore, the selected

ageing company was involved in the data collection as representative for all the other ageing companies working with the dairy company. Primary data from both the dairy and the ageing company were collected on the basis of company's registers and invoices.

Ecoinvent 3.4 database and Agrifootprint database, available in SimaPro 8.5 software, were used for background data.

### 8.1.7 Impact assessment methods and indicators

The default Environmental Footprint (EF) impact category indicators were used, using the ILCD method version 1.0.9 (ILCD 2011 Midpoint+) available in SimaPro software (v. 8.5), because it includes the impact assessment methods suggested in the PEFCR for dairy products (Table 8). The ILCD 2011 Midpoint+ method was released by the European Commission – Joint Research Centre in 2012. It supports the correct use of the characterisation factors for impact assessment as recommended in the ILCD guidance document "Recommendations for Life Cycle Impact Assessment in the European context – based on existing environmental impact assessment models and factors" (European Commission-JRC-IES, 2011).

| IMPACT CATEGORY | EU CLASS | UNIT | SOURCE |
|---|---|---|---|
| Climate change | I | kg $CO_2$ eq. | IPPC 2007 |
| Ozone depletion | I | kg CFC-11 eq. | WMO 1999 |
| Human toxicity – Cancer effects | II/III | CTUh | USEtox (Rosenbaum et al., 2008) |
| Human toxicity – Non-cancer effects | II/III | CTUh | USEtox (Rosenbaum et al., 2008) |
| Particulate matter | I | kg PM2.5 eq. | Rabl and Spadaro (2004) and Greco et al (2007) |
| Ionizing radiation – Human Health | II | kBq $U^{235}$ eq. | Frischknecht et al. (2000) |
| Photochemical ozone formation | II | kg NMVOC | Van Zelm et al. (2008) |
| Acidification | II | Mol H+ eq. | Seppala et al 2006, Posch et al (2008) |
| Eutrophication terrestrial | II | Mol N eq. | Seppala et al.2006, Posch et al 2008 |
| Eutrophication freshwater | II | kg P eq. | ReCiPe2008 |
| Eutrophication marine | II | kg N eq. | ReCiPe2008 |

| IMPACT CATEGORY | EU CLASS | UNIT | SOURCE |
|---|---|---|---|
| Ecotoxicity freshwater | II/III | CTUe | USEtox (Rosenbaum et al., 2008) |
| Land use | III | kg C deficit | Mila i Canals et al (2007) |
| Resource depletion – Water | III | $m^3$ water eq. | Swiss Ecoscarcity 2006 |
| Resource depletion – Mineral, fossils and renewables | II | Kg Sb eq. | Van Oers et al (2002) |

**Table 8.** List of impact categories and related assessment methods used.

The normalization factors for ILCD 2011 Midpoint+ V1.0.9 are based on (Benini et al., 2014) and the weighting factors are based on the EU27 2010 equal weighting method (European Commission, 2017) (all impact categories receive the same weight in the baseline approach) (Table 9).

| FACTORS | NORMALIZATION | WEIGHTING |
|---|---|---|
| Climate change | 0.000108 | 0.06667 |
| Ozone depletion | 46.2963 | 0.06667 |
| Human toxicity – Non-cancer effects | 1876.17 | 0.06667 |
| Human toxicity – Cancer effects | 27100.3 | 0.06667 |
| Particulate matter | 0.263158 | 0.06667 |
| Ionizing radiation HH | 0.000885 | 0.06667 |
| Ionizing radiation E (interim) | 0 | 0.06667 |
| Photochemical ozone formation | 0.031546 | 0.06667 |
| Acidification | 0.021142 | 0.06667 |
| Terrestrial eutrophication | 0.005682 | 0.06667 |
| Freshwater eutrophication | 0.675676 | 0.06667 |
| Marine eutrophication | 0.059172 | 0.06667 |
| Freshwater ecotoxicity | 0.000114 | 0.06667 |
| Land use | 0.000013369 | 0.06667 |
| Water resource depletion | 0.012285 | 0.06667 |
| Mineral, fossil and renewable resource depletion | 9.901 | 0.06667 |

**Table 9.** Normalization and weighting factors.

### 8.1.8 Treatment of multi-functionality

In compliance with the PEFCR for dairy products, allocation of the environmental impacts at the dairy unit level was performed by applying mass allocation using the Dry Matter

(DM) content of the Taleggio cheese and its co-products, i.e. the different types of products produced by the dairy company (soft cheeses, semi-hard cheeses, ricotta, and liquid whey).
The allocation factor corresponding to each of the products of the dairy plant could be calculated by using the Equation 2 at par. 7.3 (European Dairy Association, 2017).

## 8.2 Life cycle inventory analysis

### 8.2.1 Description and documentation of all the unit processes data collected

The key primary data collected from the dairy company, in relation to unripened Taleggio cheese production, and from the ageing company, in relation to ageing and packing, are presented in Table 10 and Table 11. respectively. Data were collected by filling in questionnaires during on-site visits and phone calls, examining company invoices, administrative documents and product labels and analysing the company offer on the market.

| DATUM | DESCRIPTION |
|---|---|
| Milk collection | Characteristics (e.g. payload, refrigeration) of the vehicle used to collect the milk from the farm and transport it to the dairy unit, along with information about the travelled route if more than one farm is visited in a single journey. |
| Description of the analysed product | Main characteristics, including production flow chart and possible different sizes. |
| Production of the analysed product | Amount of the selected product produced by the company in one year. |
| Processed milk | Annual amount of milk processed by the dairy unit. |
| Dairy unit total production | List of all company products produced in one year, amount of each one and its dry matter content. |
| Ingredients use | List of all ingredients used by the dairy unit, with annual amount of each one and information about which one is used by which product/s, in order to properly allocate them. |
| By-products generation | List and annual amount of all by-products generated by the company, along with their dry matter content and intended use or further treatment. |
| Water consumption | Annual quantity of water used by the dairy unit. |

| DATUM | DESCRIPTION |
|---|---|
| Wastewater production | Annual amount of wastewater produced by the dairy unit, its average characteristics (e.g. COD) and treatment (e.g. public sewage system, company wastewater treatment plant, external treatment including transportation). |
| Electricity consumption | Annual quantity of electricity used by the dairy unit and possible share of renewable energy generated on site. |
| Fuel consumption | Type (e.g. diesel, LPG, methane) and annual amount of fuel used by the company. |
| Packaging material | Characteristics and annual amount of the packaging material used by the company. |
| Product preservation requirements | Information about preservation of the product after its production, e.g. the need for refrigeration. |
| Product transportation | Characteristics (e.g. payload, refrigeration) of the vehicle/s used to deliver the intermediate product, i.e. unripened cheese, to the ageing companies. |
| Ageing companies | Location and annual amount of unripened cheese delivered by the dairy unit for each ageing company. |

**Table 10.** Data collected from the dairy company in relation to unripened Taleggio cheese production.

| DATUM | DESCRIPTION |
|---|---|
| Process description | Flowchart of the ageing process. |
| Produced dairy products | List of all the products produced by the ageing company in one year, amount of each one and its dry matter content. |
| Ingredients use | List of all ingredients used by the ageing company, with annual amount of each one and information about which one is used by which product/s, in order to properly allocate them. |
| Water consumption | Annual quantity of water used by the ageing company. |
| Electricity consumption | Annual quantity of electricity used by the ageing company and possible share of renewable energy generated on site. |
| Fuel consumption | Type (e.g. diesel, LPG, methane) and annual amount of fuel used by the ageing company. |
| Packaging material | Characteristics and annual amount of the packaging material used by the ageing company. |

| DATUM | DESCRIPTION |
|---|---|
| Wastewater production | Annual amount of wastewater produced by the ageing company, its average characteristics (e.g. COD) and treatment (e.g. public sewage system, company wastewater treatment plant, external treatment including transportation). |
| Product preservation requirements | Information about preservation of the product during the ageing process and after packaging, e.g. the need for refrigeration. |

**Table 11.** Data collected from the ageing company in relation to ageing and packing.

### 8.2.2 Life cycle inventory

#### *8.2.2.1 Primary data*

Input and output primary data collected at the dairy unit and at the ageing company are presented in the following tables (Table 12 and Table 13), in relation to one year of operation of the company involved in each stage, and refer to the whole yearly activities of the companies, which include different type of cheeses and co-products.

| INPUT | AMOUNT | UoM | DESCRIPTION/COMMENTS |
|---|---:|---|---|
| Raw conventional cow milk – From local farms | 8,637 | t | Amount of conventional cow milk from local dairy farms, annual basis (86% of the total amount of processed milk) |
| Milk transport – From local dairy farm to dairy unit | 30 | km | Average distance for the transport of non- organic cow milk, considering 3 trips per day with a refrigerated truck with a maximum load of 19 t |
| Raw conventional cow milk – Purchased from market | 1,449 | t | Amount of conventional cow milk purchased from the market, annual basis (14% of the total amount of processed milk) |
| Purchased milk transport | 250 | km | Average distance for the transport of purchased milk (Treviso province was considered due to lack of primary data, conservative approach) |
| Raw organic cow milk from farms | 4,916 | t | Amount of organic cow milk from dairy farms, annual basis |
| Raw conventional goat milk from farms | 1,086 | t | Amount of conventional goat milk from dairy farms, annual basis |

| INPUT | AMOUNT | UoM | DESCRIPTION/COMMENTS |
|---|---|---|---|
| Dairy unit infrastructure volume | 7,480 | $m^3$ | The dairy unit infrastructure area is 1870 m2 (primary data) and the height of the dairy unit is 4 m (primary data) |
| Salt | 102 | t | Amount of salt used for the processing of all cheeses, annual basis |
| Electricity – From grid | 574,079 | kWh | Amount of electricity consumption for the whole operations of the dairy, annual basis. All electricity is purchased from the Italian grid |
| Water | 36,000 | t | Amount of water consumption for the whole operations of the dairy, annual basis |
| Methane | 333,677 | $Sm^3$ | Amount of methane consumption for the whole operations of the dairy, annual basis. |
| Sodium carbonate | 18 | t | Amount of sodium carbonate used for cleaning all the dairy unit, annual basis |
| Peracetic acid | 13 | t | Amount of peracetic acid used for cleaning all the dairy unit, annual basis |
| Chloral 50 (Detergent) | 4 | t | Amount of detergent used for cleaning all the dairy unit, annual basis |
| Europerl 450 (Diatomaceous earth) | 0.6 | t | Amount of diatomaceous earth used for the whole operations of the dairy, annual basis |
| Steel mould | 800 | g/pc | Amount of steel used for the Taleggio mould, g per piece of Taleggio |
| Plastic (PE+PP) brand | 16 | g/pc | Amount of plastic used for the brand, g per piece of Taleggio |
| Plastic (PE+PP) mat | 100 | g/pc | Amount of plastic used for the mat on which Taleggio lays, g per piece of Taleggio |
| Plastic (PE+PP) rattan | 54 | g/pc | Amount of plastic used for the rattan, g per piece of Taleggio |
| Plastic (PE+PP) crate | 2 | kg/pc | Amount of plastic used for the crate on which Taleggio is located, g per piece of Taleggio |

| INPUT | AMOUNT | UoM | DESCRIPTION/COMMENTS |
|---|---|---|---|
| Unripened conventional Taleggio cheese transport – From dairy unit to ageing company | 71 | km | Average weighted distance for the transport of unripened Taleggio cheese to the ageing company. The average weighted distance was calculated on the basis of the yearly amount of unripened Taleggio transported to the 5 ageing companies and their distance from the dairy company |
| **OUTPUT** | **AMOUNT** | **UoM** | |
| Unripened conventional Taleggio cheese | 512 | t | Amount of unripened conventional Taleggio cheese produced by the dairy, annual basis |
| Other soft (49% DM) cheeses – From conventional cow milk | 877 | t | Amount of other soft cheeses from conventional cow milk produced by the dairy, annual basis |
| Semi-hard (59.9% DM) cheeses – From conventional cow milk | 809 | t | Amount of semi-hard cheeses from conventional cow milk produced by the dairy, annual basis |
| Ricotta – From conventional cow milk | 686 | t | Amount of ricotta from conventional cow milk produced by the dairy, annual basis |
| Soft (49% DM) cheeses – From organic cow milk | 111 | t | Amount of soft cheeses from organic cow milk produced by the dairy, annual basis |
| Ricotta – From organic cow milk | 114 | t | Amount of ricotta from organic cow milk produced by the dairy, annual basis |
| Soft (49% DM) cheeses – From goat milk | 52 | t | Amount of soft cheeses from goat milk produced by the dairy, annual basis |
| Semi-hard (59.9% DM) cheeses – From goat milk | 100 | t | Amount of semi-hard cheese from goat milk produced by the dairy, annual basis |
| Whey (organic + conventional, cow + goat) | 12,955 | t | Amount of whey from organic + conventional cow milk (12,000 t) and whey from goat milk (955 t), annual basis |
| Waste whey | 400 | t | Amount of whey which is wasted and not reused or sold, annual basis |
| Wastewater | 36,400 | t | Amount of wastewater (36,000 t) + amount of waste whey (400 t), annual basis |

**Table 12.** Inputs and outputs for the cheese production stage at the dairy company.

| INPUT | AMOUNT | UoM | DESCRIPTION/COMMENTS |
|---|---|---|---|
| Unripened conventional Taleggio cheese | 811.22 | t | Amount of unripened Taleggio cheese processed by the ageing company, annual basis |
| Other soft (49% DM) washed-rind cheeses – To be aged and packed | 309.90 | t | Amount of other soft cheeses which are aged and packed by the ageing company, annual basis |
| Other soft (49% DM) washed-rind cheeses – Only to be packed | 267.95 | t | Amount of other soft cheeses which are only packed by the ageing company, annual basis |
| Semi-hard (59.9%) cheeses – To be aged and packed | 4.47 | t | Amount of semi-hard cheeses which are aged and packed by the ageing company, annual basis |
| Fresh (23% DM) cheeses – Only to be packed | 58.00 | t | Amount of fresh cheeses which are packed by the ageing company, annual basis |
| Brine | 33.82 | t | Amount of brine used for cheese production, annual basis |
| Transport of brine | 50 | km | Average distance between the ageing company and the Consorzio Agrario of Lecco |
| Water | 2,000 | $m^3$ | Amount of water used by the ageing company, annual basis |
| Electricity – From grid | 755,479 | kWh | Amount of electricity from Italian grid used by the ageing company, annual basis |
| LPG | 73,116 | $m^3$ | Amount of Liquid Petroleum Gas used by the ageing company, annual basis |
| Cloro Foam (Foaming detergent) | 1.80 | t | Amount of foaming detergent used by the ageing company, annual basis |
| Sandik (Alcohol-based disinfectant) | 170 | kg | Amount of disinfectant used by the ageing company, annual basis |
| **OUTPUT** | **AMOUNT** | **UoM** | |
| Ripened conventional Taleggio cheese | 797 | t | Amount of ripen Taleggio cheese produced by the ageing company, annual basis |
| Other soft (49% DM) washed-rind cheeses | 573 | t | Amount of other soft cheeses produced by the ageing company, annual basis |
| Semi-hard (59.9%) cheeses | 4 | t | Amount of semi-hard cheeses produced by the ageing company, annual basis |

| INPUT | AMOUNT | UoM | DESCRIPTION/COMMENTS |
|---|---|---|---|
| Fresh (23% DM) cheeses | 52 | t | Amount of fresh cheeses produced by the ageing company, annual basis |
| By-products – Scraps from different types of cheese | 2 | t | Amount of cheese scraps which are disposed of as waste, annual basis |
| Wastewater | 2,000 | $m^3$ | Amount of wastewater produced by the ageing company, annual basis |

**Table 13.** Inputs and outputs for the ageing stage at the ageing company.

The ripened Taleggio cheese produced by the ageing company is packed in different sizes and with different types of packaging, which are shown in Table 14, together with the amount produced for each size and the total weight of primary packaging.

| SIZE OF RIPENED TALEGGIO CHEESE PRODUCED BY THE AGEING COMPANY | SHARE OF TALEGGIO CHEESE COMPANY PRODUCTION IN EACH SIZE | TYPE OF PRIMARY PACKAGING | WEIGHT OF PRIMARY PACKAGING (kg) FOR EACH SIZE |
|---|---|---|---|
| Whole cheese | 88.7% | Paper | 0.007 |
| Half cheese | 4.9% | Paper | 0.008 |
| 500 g | 1.7% | Paper | 0.014 |
| 350 g | 0.1% | Paper | 0.014 |
| 250 g | 0.3% | Paper | 0.020 |
| 200 g | 1.2% | Paper | 0.020 |
| 180 g with paper packaging | 1.5% | Paper | 0.022 |
| 180 g in PET tray | 1.6% | PET | 0.067 |

**Table 14.** Types and amount of ripened Taleggio sizes produced at the ageing company and the relevant type and amount of primary packaging.

#### *8.2.2.2 Secondary data*

Ecoinvent and Agrifootprint databases were used for secondary background data.

The “Raw milk production” phase was modelled by using the Agrifootprint dataset “Raw milk, at dairy farm, PEF compliant/NL IDF/Mass”.

Due to lack of primary data, secondary and tertiary packaging, product distribution, the use phases as well as the amount of refrigerant used in the dairy unit were modelled on the basis of the default data provided by the PEFCR for dairy products.
The list of datasets used for the PEF study is included in Table 20, par. 8.2.2.4.

#### *8.2.2.3 Calculation procedures, validation of data, including documentation and justification of allocation procedures*

Input and output data described in Table 12 and Table 13 refer to the whole operation activities of the dairy and the ageing company, which generate many products and co-products, as discussed. Therefore, these data must be referred to the production of the object of the analysis, i.e. ripened and packed Taleggio cheese produced with conventional milk. In the following paragraphs the calculation procedures to obtain the data for the Taleggio cheese production will be described, and inventory tables in relation to the production of 10 g dry matter of ripened Taleggio cheese consumed by the consumer will be presented.

##### 8.2.2.3.1 Dairy company

*Calculation of Allocation factors*

The allocation factors were calculated on the basis of Equation 2 in paragraph 7.3.
As regards the dairy company, the average dry matter contents of their products and co-products as well as the produced and sold quantities of the different product codes were provided by the dairy company.
In order to perform the allocation, all cheeses produced by the dairy company were classified in two classes of dry matter: 1) soft cheeses (49% dry matter); 2) semi-hard cheeses (59.9% dry matter). The dry matter content of ricotta and liquid sweet whey was considered to be equal to 23% and 6.8% respectively as indicated in the PEFCR Annex XII – Default DM contents. The dry matter content of whey after ricotta production (i.e. whey that is sold as animal feed) was calculated with a mass balance between the amount of dry matter of the sweet whey and that of ricotta, and is equal to 5.7%. Different allocation factors were calculated for: 1) Raw milk; 2) Electricity, water, methane, detergents, infrastructure, wastewater; 3) Diatomaceous earth; and 4) Salt; because these items are shared among different processes in the Taleggio cheese production chain.

The allocation factor for conventional raw milk necessary to produce the conventional Taleggio cheese was calculated on the basis of the amount of all the co-products obtained from conventional cow milk (see Table 15). Whey from conventional cow milk was calculated by a mass balance between the total whey and the amount of whey from organic cow milk, which was obtained by subtracting the quantity of organic cow cheeses from the quantity of organic cow milk purchased by the dairy.

The allocation factor for electricity, water, methane, detergents, infrastructure and wastewater was calculated on the basis of all the co-products produced at the dairy (i.e. including the co-products from both conventional and organic cow milk as well as those from goat milk), since these items are used for the whole production process of the dairy.

The allocation factor for diatomaceous earth was calculated on the basis of all the produced cheeses, both from cow and goat milk, but excluding ricotta and whey, because diatomaceous earth is used to regenerate brine, which, in turn, is only used for cheese production.

The allocation factor for salt was calculated on the basis of all the produced cheeses and ricotta, both from cow and goat milk, but excluding whey, because whey is discharged before salting.

| TYPE OF DAIRY PRODUCT | YEARLY QUANTITY (T/YEAR) | AVERAGE DRY MATTER CONTENT |
| --- | --- | --- |
| Unripened conventional Taleggio cheese [a, b, c, d] | 512 | 49% |
| Other soft cheeses – From conventional cow milk [a, b, c, d] | 877 | 49% |
| Semi-hard cheeses – From conventional cow milk [a, b, c, d] | 809 | 59.9% |
| Ricotta – From conventional cow milk [a, b, d] | 686 | 23% |
| Soft cheeses – From organic cow milk [b, c, d] | 111 | 49% |
| Ricotta – From organic cow milk [b, c, d] | 114 | 23% |
| Soft cheeses – From goat milk [b, c, d] | 52 | 49% |
| Semi-hard cheeses – From goat milk [b, c, d] | 100 | 59.9% |
| Whey – From conventional cow milk [a, b] | 7,309 | 5.7% |
| Whey – From organic cow milk [b] | 4,691 | 5.7% |
| Whey – From goat milk [b] | 955 | 5.7% |

**Table 15.** Inputs and outputs for the cheese production at the dairy company.

[a] Used for the calculation of the allocation factor for the conventional cow milk necessary to

produce the unripened conventional Taleggio cheese.
[b] Used for the calculation of the allocation factor for electricity, water, methane, detergents, infrastructure and wastewater necessary to produce the unripened conventional Taleggio cheese.
[c] Used for the calculation of the allocation factor for diatomaceous earth necessary to produce the unripened conventional Taleggio cheese.
[d] Used for the calculation of the allocation factor for salt necessary to produce the unripened conventional Taleggio cheese.

The final allocation factors are included in Table 16.

| INPUT | ALLOCATION FACTOR |
|---|---|
| Conventional cow raw milk | 14% |
| Electricity, water, methane, detergents, infrastructure, wastewater | 11% |
| Diatomaceous earth | 19% |
| Salt | 17% |

**Table 16.** Allocation factors at the dairy unit for conventional Taleggio cheese production.

*Calculation of the amount of steel mould, brand, rattan and crate for the Taleggio production*

In order to calculate the quantity of steel used for moulds and plastic used for brands, mats, rattans and crates, the total number of conventional Taleggio cheese pieces produced per day was calculated, which is equal to 821 pieces. This value was obtained by considering an average weight of 2 kg per piece. Due to lack of more detailed information, mould, brand, mat, rattan and crate were considered to be used only for conventional Taleggio cheese and only for one piece of Taleggio per day (conservative approach). The lifetime of these items was considered to be 1 year.

*Transports*

Due to lack of primary data, milk purchased from the market was considered to be purchased from Treviso province (250 km of distance), which is the farthest area from the dairy company where cow milk can be produced for Taleggio production, according to the PDO regulation (conservative approach).
Refrigerated transport of both unripened Taleggio cheese and the relevant crate to the ageing company was calculated on the basis of a weighted average distance (71 km) between the dairy company and the 5 ageing companies to which Taleggio cheese is sold.

This weighted distance was calculated on the basis of the annual amount of unripened Taleggio cheese transported to the 5 ageing companies and their distance from the dairy company (Table 17).

Salt was considered to be transported from Consorzio Agrario of Cremona, for a distance of 50 km. The salt packaging was not included in the study due to lack of data.

Transports of the following input products to the dairy plant were excluded because, in compliance with the PEFCR for dairy products, their contribution is lower than 1% in mass:

- Soda;
- Peracetic acid;
- Chloral 50;
- Diatomaceous earth;
- Steel mould;
- Brand;
- Rattan;
- Crate.

Solid waste at the dairy plant was not included, in compliance with the PEFCR for dairy products.

| LOCATION OF THE AGEING COMPANY | DISTANCE FROM THE DAIRY COMPANY (KM) | AMOUNT OF TALEGGIO CHEESE PURCHASED FROM THE DAIRY COMPANY (T/YEAR) |
|---|---|---|
| Company 1 (Bergamo province) | 75 | 291 |
| Company 2 (Bergamo province) | 75 | 62 |
| Company 3 (Bergamo province) | 50 | 81 |
| Company 4 (Bergamo province) | 50 | 43 |
| Company 5 (Lecco province) | 105 | 37 |

**Table 17.** Parameters for the calculation of the average weighted distance for the transport of unripened Taleggio cheese to the ageing company.

*Company wastewater treatment*

Wastewater from the dairy company is rich in organic substances, and therefore has a high Carbon Organic Demand (COD). However, no Life Cycle Inventory (LCI) data representative of such treatment is yet available in LCI databases. Therefore, according to the PEFCR for dairy products, a dilution factor was applied to evaluate the surplus energy required for the treatment of dairy wastewater due to the excess COD. The explanation is that wastewater treatment plants are characterized by an input COD reduction capacity and an output COD level; therefore, treating a higher-COD wastewater can be approximated by treating a higher volume of same-COD-level wastewater. The dilution factor was calculated as the ratio of the effluent COD at the dairy unit (2.36 g/l, primary data from the dairy company) and the COD input in a reference LCI dataset, which is the Ecoinvent dataset "Treatment, potato starch production effluent, to wastewater treatment, class 990 2/CH U " with input COD content of 2 g/l.

The dilution factor was therefore calculated in compliance with the PEFCR for dairy, which provides the following formula:

$$\text{COD\_Dilution factor} = \frac{COD_{effluent}}{COD_{ref}} = \frac{COD_{effluent}}{2}$$

with:
$COD_{effluent}$ = COD in effluent wastewater at dairy unit (g/l);
$COD_{ref}$ = COD in reference dataset = 2 g/l.

#### 8.2.2.3.2 Ageing company

*Calculation of allocation factors*

In order to perform the allocation, all cheeses produced by the ageing company were classified in three classes of dry matter: 1) soft cheeses (49% dry matter); 2) semi-hard cheeses (59.9% dry matter); 3) fresh cheeses (23% dry matter). The average dry matter contents of their products as well as the produced and sold quantities of the different product codes were provided by the ageing company.
Different allocation factors were calculated for: 1) Water, Liquid Petroleum Gas (LPG), detergents, wastewater; and 2) Brine; because these items are shared among different processes in the ageing production chain.
The ageing company produces both cheeses which need only the packing process and cheeses which need the ageing process and the packing process as well. Since electricity is used for both types of cheese and only the total amount of electricity used by the company was available, energy consumption could not be divided between the ageing process alone and the packaging process alone and the allocation factor for electricity consumption

could not be calculated. Therefore, the default data for electricity consumption in the dairy processing included in the PEFCR for dairy products were used (413 Wh/kg cheese), considering that this data refers to the sum of the electricity used at the dairy plant and at the ageing company. The electricity consumption at the ageing company was then obtained by subtracting the electricity consumption at the dairy company from the data contained in the PEFCR for dairy products (see Table 12). The allocation factor for brine was calculated on the basis of the amount of all the ripened and packed cheeses produced by the ageing company, because all these cheeses undergo the brining process.

The allocation factor for water, LPG, detergents and wastewater was calculated on the basis of the amount of all cheeses produced by the ageing company (Table 18), including both the ripened and packed cheeses and the only packed cheeses, because these items are used for the whole production process of the ageing company.

| **PRODUCT/ CO-PRODUCT** | **RIPENED AND PACKED ANNUAL QUANTITY (T/YEAR)** | **ONLY PACKED ANNUAL QUANTITY (T/YEAR)** | **DRY MATTER CONTENT** |
|---|---|---|---|
| Ripened Taleggio cheese | 797 [a, b] | - | 49% |
| Other soft washed-rind cheeses | 3045 [a, b] | 268[a,] | 49% |
| Semi-hard cheeses | 4 [a, b] | - | 59.9% |
| Fresh cheeses | - | 52 [a,] | 23% |
| By-products – Scraps from different types of cheese | 2 [a, b] | - | 48.1% |

**Table 18.** Inputs and outputs for the ageing stage at the ageing company.

[a] Used for the calculation of the allocation factor for water, LPG and detergent necessary to produce the ripened Taleggio cheese.
[b] Used for the calculation of the allocation factor for the brine necessary to produce the ripened Taleggio cheese.

The obtained allocation factors are included in Table 19.

| **INPUT** | **ALLOCATION FACTOR** |
|---|---|
| Water, LPG, detergents, wastewater | 57% |
| Brine | 72% |

**Table 19.** Allocation factors at the ageing company for ripened Taleggio cheese production.

The cheese scraps attributed to Taleggio cheese were calculated to be equal to 0.2% in mass with respect to the total ripened Taleggio cheese production. However, they were considered to be included in the food loss of 5% which takes place at retail, according to the PEFCR for dairy products.

Solid waste at the ageing company was not included, in compliance with the PEFCR for dairy products.

*Transports*

Transport of packaging paper (primary packaging) and of secondary packaging (carton boxes, separators, LDPE plastic wrap, pallets) was not considered, in compliance with the PEFCR for dairy products, because their contribution is lower than 1% in mass.

Since the ageing company produces different sizes of ripened Taleggio cheese, with different types of packaging (paper and PET tray), one main type of size and packaging were chosen for the study, namely the whole Taleggio cheese (2 kg) wrapped with packaging paper. This means that the total Taleggio cheese production of the ageing company was considered packed in 2 kg size with packaging paper. The quantity of packaging paper needed in this e was calculated accordingly, on the basis of the total Taleggio cheese production and on the weight of the packaging paper, which were provided by the ageing company.

The quantity of secondary and tertiary packaging was calculated from the following default data from the PEFCR for dairy products, because primary data were not available:

- Carton boxes: 24 g/kg cheese;
- Separators: 1.6 g/kg cheese;
- LDPE Plastic wrap: 1.5 g/kg cheese;
- Pallets: 6 g/kg cheese.

The infrastructure of the ageing company was not included due to lack of data.

#### 8.2.2.3.3 Distribution phase

Due to lack of primary data, the following default data from the PEFCR for dairy products for the distribution from the ageing company up to the consumer were used for the distribution of ripened packed Taleggio cheese:

- From ageing company to distribution centre: 150 km, by refrigerated truck, allocation per mass;

- From distribution centre to point of sale: 50 km by refrigerated truck, allocation per mass;
- From point of sale to consumer home: 65% by car, 4.8 km, 35% by other means of transport, which are neglected. Allocation per item, considering 20 items purchased (i.e. 5% of the journey is allocated to each item), considering that 1 item of cheese is equal to 100 g dry matter;
- The energy and refrigerants consumption at the distribution centre and at retail were calculated in compliance with the following default data from the PEFCR for dairy products:
  - General electricity consumption at distribution centre: 6 kWh/$m^3$*y, for 1 week
  - Refrigerated storage at distribution centre (additional electricity): 40 kWh/$m^3$*y, for 1 week
  - General energy at distribution centre (natural gas burned in boiler): 72 MJ/ $m^3$*y for 1 week
  - General electricity consumption at retail: 200 kWh /$m^3$ *y, for 5 days
  - Refrigerated storage at retail (additional electricity): 950 kWh /$m^3$*y, for 5 days
  - Refrigerant gases (leaks): 0.0145 kg R404A /$m^3$*y
  - Electricity use for chilled storage at consumer home: 1350 kWh /$m^3$*y for 10 days.

The storage volume at distribution centre and at retail was considered to be equal to 3 times the product's volume, in compliance with the PEFCR for dairy products. The product volume for the whole cheese (2 kg) was 0.0072 $m^3$ (0.02 m* 0.02 m*0.06 m, data from PDO regulation) (9% of the product volume of the whole cheese and considering a uniform density in the cheese size). The volume of the packaging was considered negligible. The total storage volume of the entire Taleggio cheese production of the ageing company (797 t) was thus equal to 2,871 $m^3$.

#### 8.2.2.3.4 Use phase

Due to lack of primary data, the electricity consumption at the consumer home was calculated in compliance with the following default data from the PEFCR for dairy products:

- Electricity use for chilled storage: 1,350 kWh/$m^3$*y, for 10 days. The storage volume was considered to be 3 times the product's volume.

Food losses from farm to retail and at consumer home were considered 5% and 7% respectively, in compliance with the PEFCR for dairy products.

##### 8.2.2.3.5 End-of-life phase

The transportation to waste treatment was modelled in compliance with PEFCR, considering a distance of 30 km to reach the incineration, landfill or composting facility, and a distance of 100 km to reach the recycling facility.

#### *8.2.2.4 Inventory tables*

After the elaboration of the collected data, the inventory tables were built and then used to model the product life cycle stages (Table 20), in relation to the production of 10 g Dry Matter of ripened Taleggio cheese.

| **RAW MILK SUPPLY** | | | |
|---|---|---|---|
| Input | Amount | UoM | Dataset |
| Raw conventional cow milk – From local farms | 5.72E-02 | kg | Raw milk, at dairy farm, PEF compliant/NL IDF/Mass |
| Milk transport – From local dairy farm to dairy unit | 1.72E+00 | kgkm | Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}\| market for \| Cut-off, U |
| Raw conventional cow milk – Purchased from market | 9.59E-03 | kg | Raw milk, at dairy farm, PEF compliant/NL IDF/Mass |
| Milk transport – From market to dairy unit | 2.40E+00 | kgkm | Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}\| market for \| Cut-off, U |
| **DAIRY PROCESSING** | | | |
| **Dairy company** | | | |
| Input | Amount | UoM | Dataset |
| Infrastructure | 1.93E-06 | $m^3$ | Dairy {GLO}\| market for \| Cut-off, U |
| Water | 1.86E-01 | kg | Tap water {Europe without Switzerland}\| market for \| Cut-off, U |
| Sodium carbonate | 9.18E-05 | kg | Sodium hydroxide, without water, in 50% solution state {GLO}\| market for \| Cut-off, U |

| | | | |
|---|---|---|---|
| Peracetic acid | 4.59E-05 | kg | Acetic acid, without water, in 98% solution state {GLO}\| market for \| Cut-off, U |
| Detergent | 1.84E-05 | kg | Soap {GLO}\| market for \| Cut-off, U |
| Diatomaceous earth | 4.59E-09 | kg | Activated silica {GLO}\| market for \| Cut-off, U |
| Refrigerant R404 | 1.17E-10 | kg | Refrigerant R134a {GLO}\| market for \| Cut-off, U[1] |
| Electricity | 2.97E-03 | kWh | Electricity, medium voltage {IT}\| market for \| Cut-off, U |
| Methane | 6.40E-02 | MJ | Heat, district or industrial, natural gas {RER}\| market group for \| Cut-off, U |
| Steel mould | 3.02E-05 | kg | Steel, chromium steel 18/8, hot rolled {GLO}\| market for \| Cut-off, U |
| Plastic brand | 2.98E-07 | kg | Polyethylene, high density, granulate {GLO}\| market for \| Cut-off, U + Polypropylene, granulate {GLO}\| market for \| Cut-off, U |
| | 2.98E-07 | kg | Polypropylene, granulate {GLO}\| market for \| Cut-off, U |
| Plastic mat | 1.88E-06 | kg | Polyethylene, high density, granulate {GLO}\| market for \| Cut-off, U + Polypropylene, granulate {GLO}\| market for \| Cut-off, U |
| | 1.88E-06 | kg | Polypropylene, granulate {GLO}\| market for \| Cut-off, U |
| Plastic rattan | 1.01E-06 | kg | Polyethylene, high density, granulate {GLO}\| market for \| Cut-off, U + Polypropylene, granulate {GLO}\| market for \| Cut-off, U |
| | 1.01E-06 | kg | Polypropylene, granulate {GLO}\| market for \| Cut-off, U |
| Plastic crate | 5.09E-06 | kg | Polyethylene, high density, granulate {GLO}\| market for \| Cut-off, U + Polypropylene, granulate {GLO}\| market for \| Cut-off, U |
| | 5.09E-06 | kg | Polypropylene, granulate {GLO}\| market for \| Cut-off, U |
| Output | Amount | UoM | Dataset |
| Wastewater | 2.22E-04 | $m^3$ | Wastewater from potato starch production {GLO}\| market for \| Cut-off, U |

[1] This dataset was used due to lack of inventory dataset for R404 production.

| | | | |
|---|---|---|---|
| R404 refrigerant emissions | 5.19E-11 | kg | Ethane, pentafluoro-, HFC-125 |
| | 6.06E-12 | kg | Ethane, 1,1,1,2-tetrafluoro-, HFC-134a |
| | 6.10E-11 | kg | Ethane, 1,1,2-trifluoro-, HFC-143 |
| **Ageing** | | | |
| Input | Amount | UoM | Dataset |
| LPG | 2.40E-06 | kg | Liquefied petroleum gas {RoW}\| market for \| Cut-off, U |
| Sanitizing detergent (with 12.5% sodium hypochlorite) | 3.71E-06 | kg | Sodium hypochlorite, without water, in 15% solution state {GLO}\| market for \| Cut-off, U |
| | 2.60E-05 | kg | Tap water {Europe without Switzerland}\| market for \| Cut-off, U |
| Disinfectant detergent (with 23% ethanol) | 6.46E-07 | kg | Ethanol, without water, in 99.7% solution state, from ethylene {GLO}\| market for \| Cut-off, U |
| | 2.16E-06 | kg | Tap water {Europe without Switzerland}\| market for \| Cut-off, U |
| Water | 3.30E-05 | kg | Tap water {Europe without Switzerland}\| market for \| Cut-off, U |
| Electricity | 4.14E-03 | kWh | Electricity, medium voltage {IT}\| market for \| Cut-off, U |
| Output | Amount | UoM | Dataset |
| Wastewater | 6.31E-01 | $m^3$ | Wastewater from potato starch production {GLO}\| market for \| Cut-off, U |
| **NON DAIRY INGREDIENTS** | | | |
| **Dairy production** | | | |
| Input | Amount | UoM | Dataset |
| Salt | 7.80E-04 | kg | Sodium chloride, powder {GLO}\| market for \| Cut-off, U |
| Transport of salt to the dairy company | 3.95E-02 | kgkm | Transport, freight, light commercial vehicle {GLO}\| market for \| Cut-off, U |
| **Ageing company** | | | |
| Input | Amount | UoM | Dataset |
| Brine (26% salt) | 1.83E-05 | kg | Sodium chloride, powder {GLO}\| market for \| Cut-off, U |
| | 5.21E-04 | kg | Tap water {Europe without Switzerland}\| market for \| Cut-off, U |
| Transport of brine to the ageing company | 3.52E-02 | kgkm | Transport, freight, lorry 3.5-7.5 metric ton, EURO4 {GLO}\| market for \| Cut-off, U |
| **PACKAGING** | | | |
| Input | Amount | UoM | Dataset |

| | | | |
|---|---|---|---|
| Paper primary packaging | 1.68E-04 | kg | Kraft paper, bleached {GLO}\| market for \| Cut-off, U |
| Cardboard boxes | 5.54E-04 | kg | Corrugated board box {GLO}\| market for corrugated board box \| Cut-off, U |
| Cardboard separators | 3.70E-05 | kg | Corrugated board box {GLO}\| market for corrugated board box \| Cut-off, U |
| Pallets | 6.27E-06 | p | EUR-flat pallet {GLO}\| market for \| Cut-off, U |
| LDPE plastic film | 3.46E-05 | kg | Packaging film, low density polyethylene {GLO}\| market for \| Cut-off, U |
| **DISTRIBUTION** | | | |
| **Storage at Distribution Centre** | | | |
| Input | Amount | UoM | Dataset |
| General electricity | 9.59E-06 | kWh | Electricity, medium voltage {IT}\| market for \| Cut-off, U |
| Additional electricity for refrigeration | 6.40E-05 | kWh | Electricity, medium voltage {IT}\| market for \| Cut-off, U |
| Natural gas burned in a boiler | 1.15E-04 | MJ | Heat, district or industrial, natural gas {RER}\| market group for \| Cut-off, U |
| R404a refrigerant | 2.32E-08 | kg | Refrigerant R134a {GLO}\| market for \| Cut-off, U |
| Output | Amount | UoM | Dataset |
| Refrigerant emissions | 1.01E-08 | kg | Ethane, pentafluoro-, HFC-125 |
| | 8.69E-10 | kg | Ethane, 1,1,1,2-tetrafluoro-, HFC-134° |
| | 1.22E-08 | kg | Ethane, 1,1,2-trifluoro-, HFC-143 |
| **Storage at Point of Sale** | | | |
| Input | Amount | UoM | Dataset |
| General electricity | 2.28E-04 | kWh | Electricity, medium voltage {IT}\| market for \| Cut-off, U |
| Additional electricity for refrigeration | 1.08E-03 | kWh | Electricity, medium voltage {IT}\| market for \| Cut-off, U |
| R404a refrigerant | 1.74E-08 | kg | Refrigerant R134a {GLO}\| market for \| Cut-off, U |
| Output | Amount | UoM | Dataset |
| Refrigerant emissions | 7.24E-09 | kg | Ethane, pentafluoro-, HFC-125 |
| | 5.79E-10 | kg | Ethane, 1,1,1,2-tetrafluoro-, HFC-134a |
| | 8.69E-09 | kg | Ethane, 1,1,2-trifluoro-, HFC-143 |
| Cardboard packaging waste | 5.91E-04 | g | Treatment of Secondary packaging waste – Cardboard |
| Plastic film waste | 3.46E-05 | g | Treatment of tertiary packaging waste – Plastic (LDPE) film |
| Wood pallet waste | 1.39E-04 | g | Treatment of tertiary packaging waste – Wood pallet |

| | | | |
|---|---|---|---|
| Cheese waste | 1.15E-03 | kg | Treatment of organic waste |
| Paper primary packaging waste | 8.08E-06 | g | Treatment of primary packaging waste – Paper |
| **Transports from Ageing company to Consumer** | | | |
| Input | Amount | UoM | Dataset |
| Transport from Ageing company to Distribution Centre | 3.61E+00 | kgkm | Transport, freight, lorry with refrigeration machine, 7.5-16 ton, EURO4, R134a refrigerant, cooling {GLO}\| market for \| Cut-off, U |
| Transport from Distribution Centre to Point of Sale | 1.20E+00 | kgkm | Transport, freight, lorry with refrigeration machine, 7.5-16 ton, EURO4, R134a refrigerant, cooling {GLO}\| market for \| Cut-off, U |
| Transport from Point of Sale to Consumer | 1.68E+01 | kgkm | Transport, passenger car, EURO 4 {RER}\| market for \| Cut-off, U |
| **USE** | | | |
| Input | Amount | UoM | Dataset |
| Electricity for Taleggio cheese refrigeration at consumer's house | 2.04E-04 | kWh | Electricity, low voltage {IT}\| market for \| Cut-off, U |
| **END-OF-LIFE** | | | |
| Output | Amount | UoM | Dataset |
| Cheese waste | 1.54E-03 | kg | Treatment of organic waste |
| Paper primary packaging waste | 1.54E-04 | kg | Treatment of primary packaging waste – Paper |

**Table 20.** Inventory table in relation to 10 g Dry Matter of ripened Taleggio cheese.

#### *8.2.2.5 Description of the application of the Circular Footprint Formula*

In accordance with the PEFCR for dairy products and the PEFCR Guidance, the waste of products used during manufacturing, distribution, retail, the use stage or after use was included in the overall modelling of the life cycle of the product, and reported at the life cycle stage where the waste occurs.

To model product waste the "Circular Footprint Formula" (CFF) from the PEFCR Guidance 6.2 (European Commission, 2017) was used. The CFF is a combination of "material + energy + disposal", i.e.:

Material $$(1-R_1)E_V + R_1 \times \left(AE_{recycled} + (1-A)E_V \times \frac{Q_{Sin}}{Q_p}\right) + (1-A)R_2 \times \left(E_{recyclingEoL} - E_V^{*} \times \frac{Q_{Sout}}{Q_P}\right)$$

Energy $$(1-B)R_3 \times (E_{ER} - LHV \times X_{ER,heat} \times E_{SE,heat} - LHV \times X_{ER,elec} \times E_{SE,elec})$$

Disposal $$(1-R_2-R_3) \times E_D$$

The parameters of the CFF are described as follows:

**A**: allocation factor of burdens and credits between supplier and user of recycled materials.

**B**: allocation factor of energy recovery processes: it applies both to burdens and credits.

$\mathbf{Q_{sin}}$: quality of the ingoing secondary material, i.e. the quality of the recycled material at the point of substitution.

$\mathbf{Q_{sout}}$: quality of the outgoing secondary material, i.e. the quality of the recyclable material at the point of substitution.

$\mathbf{Q_p}$: quality of the primary material, i.e. quality of the virgin material.

$\mathbf{R_1}$: it is the proportion of material in the input to the production that has been recycled from a previous system.

$\mathbf{R_2}$: it is the proportion of the material in the product that will be recycled (or reused) in a subsequent system. R2 shall therefore take into account the inefficiencies in the collection and recycling (or reuse) processes. R2 shall be measured at the output of the recycling plant.

$\mathbf{R_3}$**:** it is the proportion of the material in the product that is used for energy recovery at EoL.

$\mathbf{E_{recycled}}$ **(**$\mathbf{E_{rec}}$**)**: specific emissions and resources consumed (per functional unit) arising from the recycling process of the recycled (reused) material, including collection, sorting and transportation process.

$\mathbf{Er_{ecyclingEoL}}$ **(**$\mathbf{E_{recEoL}}$**)**: specific emissions and resources consumed (per functional unit) arising from the recycling process at EoL, including collection, sorting and transportation process.

$\mathbf{E_v}$: specific emissions and resources consumed (per functional unit) arising from the acquisition and pre-processing of virgin material.

$\mathbf{E^*_v}$: specific emissions and resources consumed (per functional unit) arising from the acquisition and pre-processing of virgin material assumed to be substituted by recyclable materials.

$\mathbf{E_{ER}}$: specific emissions and resources consumed (per functional unit) arising from the energy recovery process (e.g. incineration with energy recovery, landfill with energy recovery, …).

**$E_{SE,heat}$** and **$E_{SE,elec}$**: specific emissions and resources consumed (per functional unit) that would have arisen from the specific substituted energy source, heat and electricity respectively.

**$E_D$**: specific emissions and resources consumed (per functional unit) arising from disposal of waste material at the EoL of the analysed product, without energy recovery.

**$X_{ER,heat}$** and **$X_{ER,elec}$**: the efficiency of the energy recovery process for both heat and electricity.

**LHV**: Lower Heating Value of the material in the product that is used for energy recovery.

In the following paragraphs, the values assigned to the relevant parameters above are reported for each waste stream within the model.

The transportation to waste treatment was modelled in compliance with PEFCR for dairy products, i.e. considering:

- A distance of 30 km to reach the incineration, landfill or composting facility, travelled by a waste collection truck (Municipal waste collection service by 21 metric ton lorry {RoW}| market for municipal waste collection service by 21 metric ton lorry | Cut-off, U);
- A distance of 100 km to reach the recycling facility, travelled by a truck (Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}| market for | Cut-off, U).

#### 8.2.2.5.1 Organic waste

The parameters presented in Table 21 were used to model the treatment of organic waste generated by Taleggio cheese discarded at point of sale and by the consumer. Following the PEFCR Guidance 6.2 (European Commission, 2017), the waste treatment of the latter one was modelled in the End-of-life stage.

| PARAMETER | VALUE |
|---|---|
| A | 0.5 |
| B | 0 |
| $Qs_{in}/ Q_p$ | 0 |
| $Qs_{out}/ Q_p$ | 0.016 |
| $R_1$ | 0 |

| PARAMETER | VALUE |
|---|---|
| $R_2$ | 0.5 |
| $R_3$ | 0.18 |
| $E_{recycled}$ ($E_{rec}$) | Not applicable. |
| $E_{recyclingEoL}$ ($E_{recEoL}$) | Biowaste {RoW}\| treatment of biowaste, industrial composting |
| $E_v$ | Taleggio cheese (as modelled in stages 1, 2 and 3) |
| $E^*_v$ | Urea, as N {RER}\| production |
| $E_{ER}$ | Municipal solid waste {IT}\| treatment of, incineration |
| $E_{SE,heat}$ | Heat, central or small-scale, natural gas {Europe without Switzerland}\| market for heat, central or small-scale, natural gas |
| $E_{SE,elec}$ | Electricity, medium voltage {IT}\| market for |
| ED | Municipal solid waste {RoW}\| treatment of, sanitary landfill |
| $X_{ER,heat}$ | 24% |
| $X_{ER,elec}$ | 12% |
| LHV | 9 |

**Table 21.** Parameters used to model the treatment of organic waste.

#### 8.2.2.5.2 Primary packaging waste – Paper

The parameters presented in Table 22 were used to model the end-of-life treatment of Taleggio primary packaging, i.e. the packaging of food losses at point of sale, of food losses at consumer's and the packaging of the consumed product. Following the PEFCR Guidance 6.2, the waste treatment of the latter two ones was modelled in the End-of-life stage.

| PARAMETER | VALUE |
|---|---|
| A | 0.2 |
| B | 0 |
| $Qs_{in}$/ $Q_p$ | 1 |
| $Qs_{out}$/ $Q_p$ | 1 |
| $R_1$ | 0 |
| $R_2$ | 0 |
| $R_3$ | 0.35 |
| $E_{recycled}$ ($E_{rec}$) | - |
| $E_{recyclingEoL}$ ($E_{recEoL}$) | - |
| $E_v$ | Kraft paper, bleached {GLO}\| market for |
| $E^*_v$ | - |
| $E_{ER}$ | Municipal solid waste {IT}\| treatment of, incineration |
| $E_{SE,heat}$ | Heat, central or small-scale, natural gas {Europe without |

| PARAMETER | VALUE |
| --- | --- |
| | Switzerland}\| market for heat, central or small-scale, natural gas |
| $E_{SE,elec}$ | Electricity, medium voltage {IT}\| market for |
| ED | Municipal solid waste {RoW}\| treatment of, sanitary landfill |
| $X_{ER,heat}$ | 24% |
| $X_{ER,elec}$ | 12% |
| LHV | 17 |

**Table 22.** Parameters used to model the treatment of paper waste.

#### 8.2.2.5.3 Secondary packaging waste – Cardboard

The parameters presented in Table 23 have been used to model the end-of-life treatment of Taleggio secondary packaging made of cardboard, i.e. boxes and separators disposed at point of sale and related to both food losses and consumed product.

| PARAMETER | VALUE |
| --- | --- |
| A | 0.2 |
| B | 0 |
| $Qs_{in}/ Q_p$ | 1 |
| $Qs_{out}/ Q_p$ | 1 |
| $R_1$ | 0 |
| $R_2$ | 0.73 |
| $R_3$ | 0.09 |
| $E_{recycled}$ ($E_{rec}$) | - |
| $E_{recyclingEoL}$ ($E_{recEoL}$) | Waste paper, sorted {GLO}\| market for |
| $E_v$ | Corrugated board box {GLO}\| market for corrugated board box |
| $E^*_v$ | Sulphate pulp {GLO}\| market for |
| $E_{ER}$ | Municipal solid waste {IT}\| treatment of, incineration |
| $E_{SE,heat}$ | Heat, central or small-scale, natural gas {Europe without Switzerland}\| market for heat, central or small-scale, natural gas |
| $E_{SE,elec}$ | Electricity, medium voltage {IT}\| market for |
| ED | Municipal solid waste {RoW}\| treatment of, sanitary landfill |
| $X_{ER,heat}$ | 24% |
| $X_{ER,elec}$ | 12% |
| LHV | 17 |

**Table 23.** Parameters used to model the treatment of cardboard waste.

#### 8.2.2.5.4 Tertiary packaging waste – Plastic (LDPE) film

The parameters presented in Table 24 were used to model the end-of-life treatment of Taleggio tertiary packaging made of plastic, i.e. LDPE film disposed at point of sale and related to both food losses and consumed product.

| PARAMETER | VALUE |
|---|---|
| A | 0.5 |
| B | 0 |
| $Qs_{in}/ Q_p$ | 0.75 |
| $Qs_{out}/ Q_p$ | 0.75 |
| $R_1$ | 0 |
| $R_2$ | 0 |
| $R_3$ | 0.35 |
| $E_{recycled}$ ($E_{rec}$) | - |
| $E_{recyclingEoL}$ ($E_{recEoL}$) | - |
| $E_v$ | Packaging film, low density polyethylene {GLO}\| market for \| Cut-off, U |
| $E^*_v$ | - |
| $E_{ER}$ | Municipal solid waste {IT}\| treatment of, incineration |
| $E_{SE,heat}$ | Heat, central or small-scale, natural gas {Europe without Switzerland}\| market for heat, central or small-scale, natural gas |
| $E_{SE,elec}$ | Electricity, medium voltage {IT}\| market for |
| ED | Municipal solid waste {RoW}\| treatment of, sanitary landfill |
| $X_{ER,heat}$ | 24% |
| $X_{ER,elec}$ | 12% |
| LHV | 46 |

**Table 24.** Parameters used to model the treatment of plastic waste.

#### 8.2.2.5.5 Tertiary packaging waste – Wood pallet

The parameters presented in Table 25 were used to model the end-of-life treatment of Taleggio tertiary packaging made of wood, i.e. pallet disposed at point of sale and related to both food losses and consumed product.

| PARAMETER | VALUE |
|---|---|
| A | 0.8 |
| B | 0 |
| $Qs_{in}/ Q_p$ | 1 |
| $Qs_{out}/ Q_p$ | 1 |

| PARAMETER | VALUE |
|---|---|
| $R_1$ | 0 |
| $R_2$ | 0.39 |
| $R_3$ | 0.21 |
| $E_{recycled}$ ($E_{rec}$) | - |
| $E_{recyclingEoL}$ ($E_{recEoL}$) | Wood chipping, industrial residual wood, stationary electric chipper {RER}\| processing |
| $E_v$ | EUR-flat pallet {RER}\| production (recalculated in order to be expressed in unit of mass instead of unit of volume) |
| $E^*_v$ | Residual wood, dry {GLO}\| market for |
| $E_{ER}$ | Municipal solid waste {IT}\| treatment of, incineration |
| $E_{SE,heat}$ | Heat, central or small-scale, natural gas {Europe without Switzerland}\| market for heat, central or small-scale, natural gas |
| $E_{SE,elec}$ | Electricity, medium voltage {IT}\| market for |
| ED | Municipal solid waste {RoW}\| treatment of, sanitary landfill |
| $X_{ER,heat}$ | 24% |
| $X_{ER,elec}$ | 12% |
| LHV | 17 |

**Table 25.** Parameters used to model the treatment of wood waste.

## 8.3 PEF impact assessment results

### 8.3.1 Characterization results

The characterised results for the entire life cycle of 10 g dry matter of Taleggio cheese are included in Table 26.

| IMPACT CATEGORY | UoM | LIFE CYCLE EXCL. USE STAGE | USE STAGE | TOTAL |
|---|---|---:|---:|---:|
| Climate change | kg $CO_2$ eq. | 1.23E-01 | 1.28E-03 | 1.24E-01 |
| Ozone depletion | kg CFC-11 eq. | 3.28E-09 | 1.38E-10 | 3.42E-09 |
| Particulate matter | kg PM2.5 eq. | 6.84E-05 | 5.32E-07 | 6.89E-05 |
| Ionizing radiation HH | kBq U235 eq. | 2.10E-03 | 2.10E-04 | 2.31E-03 |
| Ionizing radiation E (interim) | CTUe | 1.28E-08 | 5.14E-10 | 1.33E-08 |
| Photochemical ozone formation | kg NMVOC eq. | 3.36E-04 | 2.59E-06 | 3.39E-04 |
| Acidification | molc H+ eq. | 2.66E-03 | 1.05E-05 | 2.68E-03 |
| Terrestrial eutrophication | molc N eq. | 1.17E-02 | 2.70E-05 | 1.17E-02 |
| Freshwater eutrophication | kg P eq. | 1.87E-05 | 3.67E-07 | 1.91E-05 |
| Marine eutrophication | kg N eq. | 9.25E-04 | 9.86E-07 | 9.26E-04 |
| Land use | kg C deficit | 7.50E-01 | 2.94E-03 | 7.53E-01 |
| Water resource depletion | m3 water eq. | 2.04E-04 | 1.52E-05 | 2.19E-04 |

| IMPACT CATEGORY | UoM | LIFE CYCLE EXCL. USE STAGE | USE STAGE | TOTAL |
|---|---|---|---|---|
| Mineral, fossil and renewable resource depletion | kg Sb eq. | 2.02E-06 | 2.56E-08 | 2.05E-06 |

**Table 26.** Characterised values for Taleggio cheese in relation to the Functional Unit (10 g dry matter = 20.4 g of cheese).

### 8.3.2 Normalisation results

The normalised results for the entire life cycle of 10 g dry matter of Taleggio cheese are included in Table 27.

| IMPACT CATEGORY | LIFE CYCLE EXCL. USE STAGE | USE STAGE | TOTAL PER IMPACT CATEGORY |
|---|---|---|---|
| Climate change | 1.74E-05 | 1.80E-07 | 1.76E-05 |
| Ozone depletion | 2.69E-07 | 1.13E-08 | 2.80E-07 |
| Particulate matter | 1.35E-05 | 1.05E-07 | 1.36E-05 |
| Ionizing radiation HH | 8.72E-06 | 8.71E-07 | 9.59E-06 |
| Ionizing radiation E (interim) | Normalisation factors not available | | |
| Photochemical ozone formation | 7.42E-06 | 5.72E-08 | 7.48E-06 |
| Acidification | 4.75E-05 | 1.88E-07 | 4.77E-05 |
| Terrestrial eutrophication | 7.11E-05 | 1.65E-07 | 7.13E-05 |
| Freshwater eutrophication | 2.87E-06 | 5.61E-08 | 2.92E-06 |
| Marine eutrophication | 3.04E-05 | 3.24E-08 | 3.05E-05 |
| Land use | 1.44E-07 | 5.66E-10 | 1.45E-07 |
| Water resource depletion | 2.96E-06 | 2.21E-07 | 3.18E-06 |
| Mineral, fossil and renewable resource depletion | 1.05E-05 | 1.33E-07 | 1.06E-05 |
| **TOTAL LIFE CYCLE IMPACT (ALL CATEGORIES)** | 2.13E-04 | 2.02E-06 | 2.15E-04 |

**Table 27.** Normalised values for Taleggio cheese in relation to the Functional Unit (10 g dry matter = 20.4 g of cheese).

## 8.4 Interpretation of PEF results

### 8.4.1 Most relevant impact categories

As shown in Table 29, the most relevant impact categories, i.e. those representing together more than 80% of the total impact, are:

- Terrestrial eutrophication = 7.13E-05 (33.2% of the total impact);
- Acidification = 4.77E-05 (22.2% of the total impact);

- Marine eutrophication = 3.05E-05 (14.2% of the total impact);
- Climate change = 1.76E-05 (8.2% of the total impact);
- Particulate matter = 1.36E-05 (6.3% of the total impact).

### 8.4.2 Most relevant direct elementary flows

The elementary flows which contribute the most to the most relevant impact categories are presented in Table 28.

| IMPACT CATEGORY | MOST RELEVANT FLOWS | CONTRIBUTION | MAIN SOURCE |
|---|---|---|---|
| **Terrestrial eutrophication** | Ammonia | 97.9% | Emissions from the use of manure and chemical fertilisers for grass production and emissions from manure management in the stable |
| **Acidification** | Ammonia | 97.4% | |
| **Marine eutrophication** | Nitrate | 89% | Emissions from use of manure for production of maize silage and grass |
| **Climate change** | Methane, biogenic | 39.2% | Emissions from enteric fermentation and from manure management in the stable |
| | Dinitrogen monoxide | 24.8% | Direct and indirect emissions, due to use of manure for grass production and from the stable, emissions due to the production of chemical fertilisers |
| | Carbon dioxide, land transformation | 17.6% | Emissions due to land use change for the production of soybean from Brazil |
| **Particulate matter** | Ammonia | 93.8% | Emissions from the use of manure and chemical fertilisers for grass production and emissions from manure management in the stable |

**Table 28.** Most relevant elementary flows in relation to the most significant impact categories.

### 8.4.3 Most relevant processes and life cycle stages

In relation to the most significant impact categories identified in 8.4.1, the most relevant life cycle stage is always “Raw milk supply”, representing at least 82.3% of the total impact of the studied Taleggio cheese life cycle (Table 29).

The most relevant process within that stage is always the production of raw milk at dairy farm, representing at least 99.4% of the total impact of the “Raw milk supply” stage, while raw milk transportation from dairy farms to the dairy company provides a negligible contribution.

### 8.4.4 Overall assessment of data quality

Activity data used for the study and collected with the direct involvement of the dairy and the ageing companies refer to 2016. In accordance with the criteria reported in PEFCR - version 6.2 – Table 36, for those data TiR is considered equal to 2, since they refer "to maximum 2 annual administration periods with respect to the EF report publication date", and P is considered equal to 3, since the collected data have been measured or taken by literature but no revision was performed on the PEF study.

Apart from activity data directly collected from the companies, default data from PEFCR for dairy products were used, particularly for the "Packaging", "Distribution", "Use" and "End-of-life" phases. Due to a lack of information on how these default parameters were set, no evaluation on their TiR and P values was made.

The vast majority of secondary datasets used for the study comes from the Ecoinvent database. Since the database does not provide complete and clear information about the geographical, time and technological representativeness as well as precision of its datasets, no evaluation of the parameters in the DQR formula for these data was made.

Instead, the dataset to represent raw milk production (Raw milk, at dairy farm, PEF compliant/NL IDF/Mass) was taken from the Agrifootprint database, which provides sufficient information to establish that, in relation to the performed study for that secondary dataset (PEFCR - version 6.2 – Table 37):

- TiR is 2, because the report has been published not later than 2 years beyond the time validity of the dataset;
- TeR is 2, because the dataset is a proxy of the technology of the studied system;
- GR is 4, because the dataset refers to the Netherlands, a country with sufficient similarities to Italy, based on expert judgement;

In compliance with PEFCR Guidance 6.2 (European Commission, 2017), since raw milk production is the process which contributes the most to the total life cycle impact (more than 80%, see 8.4.3), the overall quality of the study is directly related to that of the secondary dataset used and the relevant activity data.

| IMPACT CATEGORY | RAW MILK SUPPLY | DAIRY PROCESSING | NON-DAIRY INGREDIENTS | PACKAGING | DISTRIBUTION | USE | END-OF-LIFE |
|---|---|---|---|---|---|---|---|
| Climate change | 82.3% | 8.4% | 0.3% | 0.2% | 7.1% | 1.0% | 0.7% |
| Particulate matter | 85.9% | 5.8% | 0.5% | 1.6% | 5.4% | 0.8% | 0.0% |
| Acidification | 96.1% | 2.0% | 0.1% | 0.2% | 1.2% | 0.4% | 0.0% |
| Terrestrial eutrophication | 97.8% | 1.1% | 0.0% | 0.1% | 0.7% | 0.2% | 0.0% |
| Marine eutrophication | 97.7% | 1.0% | 0.1% | 0.2% | 0.8% | 0.1% | 0.1% |

**Table 29.** Contribution of the life cycle stages to the overall impact for the most relevant impact categories.

Applying the following equation from PEFCR Guidance 6.2:

$$\mathrm{DQR} = \frac{\mathrm{TeR}+\mathrm{GR}+\mathrm{TiR}+\mathrm{P}}{4} \qquad (3)$$

**Equation 3**. Calculation of Data Quality Rating of the PEF study

Where:

TeR = Technological Representativeness;

GR = Geographical Representativeness;

TiR= Time-related Representativeness;

P = Precision.

and considering the following values for the raw milk production process (Table 30):

| Pad | TiRad | TiRsd | TeRsd | GRsd | Overall QUALITY |
|---|---|---|---|---|---|
| 3 | 2 | 2 | 2 | 4 | 2.625 |

**Table 30.** Considered values for the Data Quality Rating of milk production process.

Where:

Pad = Precision of Activity data;

TiRad= Time-related Representativeness of Activity data;

TiRsd= Time-related Representativeness of Secondary dataset;

TeRsd = Technological Representativeness of Secondary dataset;

GRsd = Geographical Representativeness of Secondary dataset;

the overall quality rating (DQR) of the study can be considered "good" (from 2.0 to 3.0, see PEFCR for Dairy Products – Updated DRAFT for public consultation (European Dairy Association, 2016).

## 8.5 Conclusions

This study, performed as part of the activities of the PEFMED project, has provided a valuable test for the use of PEF method and PEFCR for dairy products in the Taleggio cheese supply chain, an Italian dairy supply chain. The application of the PEF method provided a detailed picture of the environmental performance of the Taleggio cheese production, and highlighted the main hotspots in the production chain.

Nevertheless, some limitations have been identified during the study, such as the lack of relevant datasets to model the product life cycle and the uncomplete company-specific data available for some life cycle stages, e.g. ageing within “Dairy processing", resorting to default data to model some stages, e.g. “Distribution” and “Use”.

Despite these limitations, the main conclusion that can be drawn from the PEF study is that “Raw milk supply” is the most significant phase in the product life cycle, contributing to 82,3% – 97,8% of the impact in the most relevant impact categories. For climate change, the impact of the “Dairy processing” phase is not negligible as well (8.4%).

Since the environmental hotspot in the overall Taleggio cheese life cycle is raw milk production, including the crop cultivation and cattle breeding activities, improvement actions should focus on that step, such for example a better management of livestock manure, decrease in the use of fertilisers and pesticides and implementation of on-farm energy production systems. In that direction, it could be also useful to collect good quality company-specific data in relation to the “Raw milk supply” stage as well, e.g. suggesting companies part of the value chain to keep detailed records of the environmental data, even if it is not mandatory in compliance with the PEFCR for dairy products in the specific situation, in order to better direct the improvement actions in this area.

Finally, it can be highlighted that the application of both the PEF method and the PEFCR for dairy products was rather difficult and time-consuming, especially regarding the calculation of data quality requirements and data quality rating, for which the PEFCR Guidance requirements were used, and the use of Circular Footprint Formula for the End-of-Life stage, which involved the calculation of many parameters and the identification of the most proper datasets to model the recycling and disposal phases. Therefore, the application of this method was not so quick and easy as it was expected to be, also for LCA practitioners. The future availability of PEF compliant life cycle inventory datasets, currently under preparation by the European Commission, and the possible improvements of the method during the PEF transition phase could probably contribute to simplify the modelling phase of the PEF study, although the whole method will remain quite resource-intensive.

# 9 Water Footprint Network method and ISO 14046

## 9.1 Water Footprint method by the Water Footprint Network

The concept of Water Footprint (WF) was introduced by Hoekstra in 2003, with the purpose to create a consumptive indicator of freshwater use which takes into account both the direct and indirect water use stemming from the consumption and production of products and services (Hoekstra, 2003). This first approach was then implemented into the Global Standard for Water Footprint contained in the Water Footprint Assessment (WFA) Manual (Hoekstra et al., 2011). This standard has been developed in the framework of the Water Footprint Networkn (WFN), a global organisation which works in this field with the final objective to promote sustainable water use, by developing standards and tools available for free (Notarnicola et al., 2015).

According to Hoekstra et al. (2011), WF is the total volume of freshwater used to produce a certain product, measured throughout the entire production chain. WF is a multi-dimensional indicator, because it is measured in terms of water volume consumed and polluted in a specific area and time. The geographical and temporal specifications of the calculated WF enable the comparison of the WF between different sectors, between countries and across years. WF can be calculated for a process, a product, a consumer, a group of consumers or a producer.

WF can be divided in three components (Hoekstra et al., 2011) (Figure 10):

- Blue Water is the consumption of surface water and groundwater troughout the whole product's supply chain. This means that water is withdrawn from surface or groundwater in a certain basin: losses take place when withdrawn water evaporates, returns to another basin or is incorporated into a product, or returns in the same catchment area but in a different time (for example in another season).
- Green Water is the consumption of green water resources, i.e. the rainwater that does not run-off and which is used in the evapotranspiration process of soil-crop system. Therefore, it is does not become part of any surface or underground water body. In fact, rainwater is a limited resource as well, especially in some locations and months; moreover, it is fundamental for the crop growth because the blue water requirements for irrigation decreases when crops are rainfed.

- Grey Water is all water polluted by a production process and it is defined as the volume of freshwater necessary to dilute the load of pollutants, on the basis of their natural concentrations and the local water quality standards.

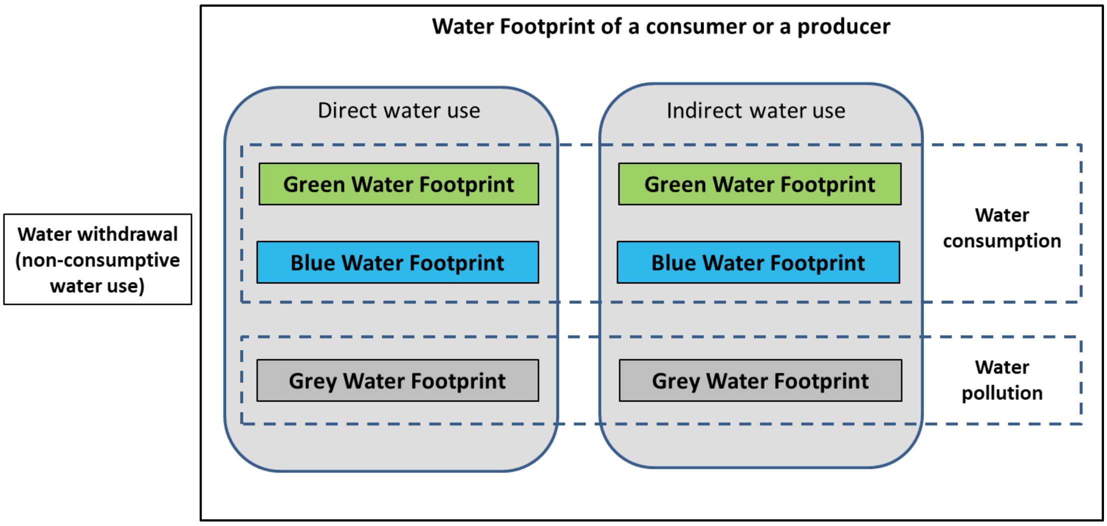



**Figure 10.** Representation of the components of a Water Footprint (Source: Personal elaboration adapted from Hoekstra et al., 2011).

Therefore, WF is a volumetric measure of water consumption and pollution which provides a comprehensive outlook on the relationship between consumers or producers and their use of freshwater resources, and can be used to evaluate and improve the water sustainability of activities and products (Hoekstra et al., 2011). In this context and with these objectives, the WFN method has been applied in the last years to several products with a main focus on agri-food production chain (Bai et al., 2018; Manzardo et al., 2016; Murphy et al., 2017; Lovarelli et al., 2016). According to Hoekstra et al. (2011), a complete WF study consists of four stages:

1. Setting goals and scope;
2. WF accounting;
3. WF sustainability assessment;
4. WF response formulation.

### 9.1.1 Setting goals and scope

This step aims to clarify the objective of the study and to evaluate the processes to be included since WF studies can have several purposes and be applied to different contexts (e.g. a product, a process, a company, a consumer, a geographical area, a country). The chosen goal will lead to a specific scope and to different choices when making assumptions for the WF calculation ) (Hoekstra et al., 2011).

The most important issues to be defined in this phase are the desired level of detail for the study (.e.g. when the objective is the identification of hotspots, a greater detail will be needed in order to assess the spatial and temporal location of this hotspot) and the system boundaries, i.e. what will be included or excluded in the study. In general, the WF of all processes within a specific production system, which significantly contribute to the overall WF, should be considered. However, in practice, only a few process phases considerably contribute to the total WF of the final product. For example, this is the case of agri-food products, where the WF of agricultural ingredients often give a major contribution to the total WF of a specific product. Industrial ingredients, which are not connected with agricultural sector, give a major contribution to water pollution and therefore to grey WF. Finally, during the goal and scope phase, the spatial and temporal coordinates to be included in the study will have to be identified (Hoekstra et al., 2011).

### 9.1.2 Water Footprint Accounting: calculation of blue, green and grey WF

In this phase, three components are quantified, in order to take into account different types of water: the blue and green WF, relative to consumptive water use, and the grey WF, related to degradative water use ) (Hoekstra et al., 2011).

Therefore, the total WF of a process ($WF_{proc}$) is given by the sum of green, blue and grey components (Hoekstra et al., 2011):

$$WF_{proc} = WF_{proc,green} + WF_{proc,blue} + WF_{proc,grey} \quad \text{[volume/mass]} \quad (4)$$

The blue WF is an indicator of blue water consumption, which is surface water or groundwater. This type of water consumption can derive from water evaporation, water incorporation into a product, water returning to the same basin or water returning in another time period (Hoekstra et al., 2011). In general, evaporation is the most significant component, although the other three components should be accounted for when applicable.

The blue WF of a process can be calculated as (Hoekstra et al., 2011):

$$WF_{proc,blue} = BlueWaterEvaporation + BlueWaterIncorporation + LostReturnFlow \quad \text{[volume/time]} \qquad (5)$$

"Lost return flows" is the amount of the return flow which cannot be reused in the same basin and in the same time period of withdrawal, because it returns to another basin, or in another period (Hoekstra et al., 2011).

Green WF is an indicator of the green water use, which is the rainwater which does not run off or restore the groundwater but is deposited in the soil or remains over the soil or plants for a certain period of time, before evaporating or transpiring through plants. In case of agriculture, green water can be used for crop growth (Hoekstra et al., 2011).

Therefore, green WF is the volumetric amount of rainwater utilised in the production process. In case of agricultural processes, it is the sum of the total rainwater evapotranspiration and the water incorporated into the crop (Hoekstra et al., 2011):

$$WF_{proc,green} = \text{GreenWaterEvaporation} + \text{GreenWaterIncorporation} \quad \text{[volume/time]} \qquad (6)$$

Differently from blue water, green water cannot be used for industrial or municipal purpose because it cannot be withdrawn or transported from a specific area. Therefore, blue water has to be utilised for industrial or municipal uses (Aldaya et al., 2010; Antonelli and Greco, 2013; Hoekstra et al., 2011). Agricultural green water consumption can be estimated with mathematical formulas or with a crop model which measures the evapotranspiration on the basis of weather, soil and plant properties (Hoekstra et al., 2011).

Grey WF of a process measures the extent to which that particular process contributes to freshwater pollution. It is the volume of freshwater needed to dilute the load of pollutants, provided their natural concentrations and current local water quality standards. This indicator can be calculated by dividing the pollutant load (L, in mass/time) by the difference between the maximum acceptable concentration of that pollutant ($c_{max}$, in mass/volume) and its natural concentration in the water body which receives this polluted water ($c_{nat}$, in mass/volume) (Hoekstra et al., 2011):

$$WF_{proc,grigia} = \frac{L}{c_{max} - c_{nat}} \quad \text{[volume/time]} \qquad (7)$$

The natural concentration is the pollutant concentration in the water body without any human contamination. As a consequence, $c_{nat}$ value is zero for human-deriving substances which are absent in water. $C_{nat}$ can also be assumed equal to zero when the actual value of natural concentrations is not available, but it is estimated to be low. Anyway, this could lead to an underestimation of grey WF when $c_{nat}$ is actually not equal to zero, because there would be an overestimation of the assimilation capacity ) (Hoekstra et al., 2011).

A specific pollutant load can lead to different grey WF because local quality standards can be site-specific. Therefore, water quality standards and natural concentrations used for the calculation of the grey WF must be specified.

A grey WF higher than zero means that part of the assimilation capacity of that water body has been exploited. When the grey WF is lower than the total water flow, water can dilute the pollutants concentration below the quality standards; on the contrary, when it is equal to water flow, the concentration will be equal to the standard (critical load $L_{crit}$) (Hoekstra et al., 2011).

In case of diffuse sources of pollution, the pollutant load is the fraction of the chemical substances total quantity which reaches the groundwater or runs off up to the surface water. Nevertheless, this percentage cannot be directly measured, because the pollutant enters the water in a diffuse way. Anyway, it can be estimated by assuming that a certain fixed percentage of the applied substance reaches the freshwater (Hoekstra et al., 2011):

$$WF_{proc,grey} = \frac{L}{c_{max} - c_{nat}} = \frac{\alpha \times Appl}{c_{max} - c_{nat}} \quad \text{[volume/time]} \qquad (8)$$

α (a dimensionless factor) represents the leaching-run-off fraction, which is the fraction of applied chemical substance which reaches the groundwater or the surface water. Appl is the amount of substance applied on soil in a certain process in a certain time period.

When several pollutants are applied on soil, only the most critical pollutant is accounted for in the calculation, which is the pollutant requiring the highest water volume to be diluted. In fact, the water volume required to take the concentration of that pollutant below water quality standard will dilute also all the other pollutants. It is noteworthy that the grey WF calculation does not consider any combined effect of two or more pollutants, which could be higher than single effects ) (Hoekstra et al., 2011).

### 9.1.3 Water Footprint sustainability assessment

The assessment of WF sustainability involves the comparison of the overall WF of human activities with what the Earth can sustain. This means that the calculated WF for a process should be compared to the amount of freshwater available in the catchment area where the process takes place (expressed in $m^3$/yr). The sustainability of the WF of a product, production process, company or consumer depends on the geographical context. Therefore, the sustainability assessment of a process, product, company or consumer WF requires the evaluation of the sustainability of the entire WF of the basin in which they are located (Hoekstra et al., 2011).

The WF sustainability in a river basin can be assessed from the environmental, social and economic perspectives, with different "sustainability criteria". The assessment of the sustainability of a WF consists of four main phases (Figure 11) (Hoekstra et al., 2011):

1. Identification and quantification of sustainability criteria;
2. The identification of "hotspots" (i.e. critical points) within the catchment area, i.e. sub-basins or time periods in which the WF is considered unsustainable (e.g. a dry period);
3. Identification and quantification of primary impacts in the hotspots, which are described in terms of change in water quality and water flow, compared to natural conditions;
4. Identification and quantification of secondary impacts in the hotspots, which are the environmental, social and economic products or services affected by primary impacts, and can be measured for example in terms of loss of biodiversity or effects on human health.

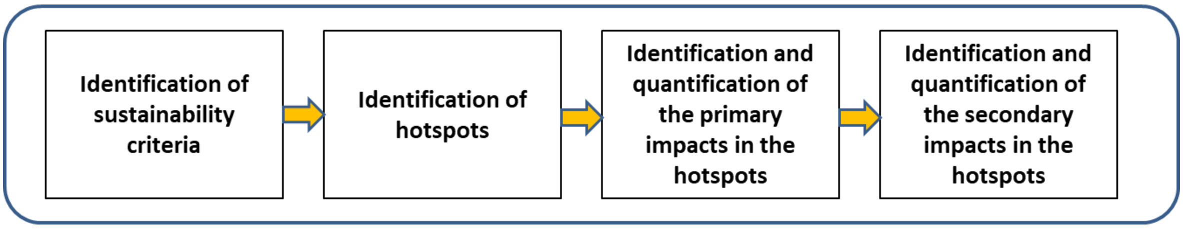



**Figure 11.** The four phases of the Water Footprint sustainability assessment in a river basin (Source: Personal elaboration adapted from Hoekstra et al., 2011).

A WF in a basin creates an environmental hotspot when the environmental water requirements are impaired or when the pollution level exceeds the assimilation capacity of the water body. The calculation of the blue and green water scarcity and the water pollution level represent a quantification of the hotspot severity: when one of these values exceeds 100%, we have an environmental hotspot (Hoekstra et al, 2011).

When a green WF in a specific basin exceeds the green water availability, it creates an environmental hotspot. The "Green Water Availability" ($WA_{green}$) in a basin x in a certain period t is defined as the total evapotranspiration of rainwater from soil ($ET_{green}$) minus the evapotranspiration from soil due to natural vegetation, which aims to preserve biodiversity and human activities ($ET_{env}$) and minus the evapotranspiration from soil in areas or time periods not suitable for crop cultivation (Hoekstra et al, 2011):

$$WA_{green}[x,t] = ET_{green}[x,t] - ET_{env}[x,t] - ET_{unprod}[x,t] \quad \text{[volume/time]} \qquad (9)$$

It is noteworthy that the availability of green water is limited, similarly to that of blue water. The level of green water scarcity ($WS_{green}$) in a basin x in a period t is the ratio between the total green water footprint of the basin ($\Sigma WF_{green}$) and the availability of green water ($WA_{green}$) (Hoekstra et al, 2011):

$$WS_{green}[x,t] = \frac{\sum WF_{green}[x,t]}{WA_{green}[x,t]} \qquad \text{[-]} \qquad (10)$$

Due to lack of literature data for the calculation of $ET_{env}$ and $ET_{unprod}$, accurate values for both the availability of green water and water scarcity cannot be calculated (Hoekstra et al, 2011).

When a blue WF in a specific basin exceeds the blue water availability in a certain time period, it creates an environmental hotspot. The "Blue Water Availability" ($WA_{blue}$) in a basin x in a certain period t is given by the basin natural runoff ($R_{nat}$) minus the "environmental flow requirement" (EFR) (Hoekstra et al, 2011):

$$WA_{blu}e[x,t] = R_{nat}[x,t] - EFR[x,t] \qquad \text{[volume/time]} \qquad (11)$$

However, the assessment of blue WF sustainability in a basin requires also to evaluate whether the levels of groundwater and lakes stay within their "sustainability boundaries" (Richter, 2010).
The “Blue Water Scarcity” ($WS_{blue}$) in a basin x is calculated by the ratio between the total blue WF of the basin ($\Sigma WF_{blue}$) and the availability of blue water ($WA_{blue}$) (Hoekstra et al, 2011):

$$WS_{blue}[x,t] = \frac{\sum WF_{blue}[x,t]}{WA_{blue}[x,t]} \qquad [-] \qquad (12)$$

A grey WF in a certain period and in a specific basin creates a hotspot when water quality standards are violated, i.e. when the pollutants assimilation capacity is completely exploited. The " Water Pollution Level” (WPL) indicator, defined as the exploited fraction of the pollutants’ assimilation capacity, measures pollution degree in a river basin. It is estimated by the ratio between the total grey WF in a basin x and at time t ($\Sigma WF_{grey}$) and the actual runoff from the basin ($R_{act}$) (Hoekstra et al, 2011):

$$WPL[x,t] = \frac{\sum WF_{grey}[x,t]}{R_{act}[x,t]} \qquad [-] \qquad (13)$$

The WF in a specific basin is considered unsustainable from the social point of view when basic water human needs, such as water requirement for drinking, washing or for food production cannot be satisfied in that area (UN, 2010) or when water is not fairly shared among people living in a certain geographical area. As regards the economic sustainability, the WF creates an economic hotspot when water is not distributed and utilised efficiently from the economic point of view. Therefore, a fair water price should be stablished, in order to include the externalities and all costs connected to the water use. Nevertheless, the abovementioned issues can be difficult to be quantified. Therefore, expert judgment should be used to evaluate the social and economic sustainability of the calculated WF for a certain process (Hoekstra et al., 2011).
According to Hoekstra et al. (2011), the sustainability of the WF of a process depends on the geographical context and on the process characteristics. This means that the WF of a process is not sustainable when it is located in a basin in a certain time period when the total WF is not environmentally, socially or economically sustainable or when it can be

reduced or avoided at a reasonable social cost. These criteria must be assessed separately for the blue, green and grey WF (Hoekstra et al, 2011).

The analysis of the first criterion (i.e. geographical context) shows that, when the WF contributes to a hotspot, the WF of that process will be unsustainable. In fact, when the overall WF in a basin is unsustainable, every single contribution will be considered unsustainable. The use of improved technologies available at fair social cost can reduce or avoid the unsustainability of the blue, green or grey WF of a process. However, an unsustainable process does not always create immediate water scarcity or pollution problems, but it is considered unsustainable because it excessively consumes water and exploits the pollutants assimilation capacity of the water body. In order to evaluate the unsustainability of a process, expert judgment is required, because no defined criteria are available in literature. Therefore, global reference parameters for the blue, green and grey components, should be developed to support the comparison between the WF of a process with that of the reference process (Hoekstra et al, 2011). According to the WFN method, the sustainability assessment of a WF requires to calculate the water availability of the catchment area and consequently the blue and green water scarcity as well as the water pollution level. Nevertheless, these parameters are rather difficult to be collected or evaluated and therefore the sustainability assessment phase is seldom applied in literature (Hoekstra et al., 2011).

### 9.1.4 Water Footprint response formulation

This last phase of a WF study comprises the evaluation of the improvement actions to reduce both the water consumption and the relevant impacts (Hoekstra et al., 2011).

Companies can reduce their WFs by decreasing the water consumption in their industrial operations and eliminating the water pollution stemming from their processes. However, the majority of the WF for many companies derives from their production chain: improvements along the supply chain are more difficult because they are not directly controlled by the company, but they can be very effective. For example, they could change suppliers or ask them to be compliant with specific standards for the reduction of the WF. A company can also aim to reduce the WF of consumers who use their product, for example by using recyclable or secondary raw materials (Hoekstra et al., 2011).

As regards the WF in agriculture, this can be reduced by adopting different irrigation techniques, such as the "deficit irrigation" which aims to achieve the maximum water productivity (t/$m^3$) instead of the maximum yield (t/ha), alternating periods where 100% of the evapotranspiration requirement is provided by irrigation and periods where water stress is imposed to the crop, decreasing the irrigation volumes by a certain rate. Finally, the grey WF can be reduced by the adoption of the organic agriculture, which eliminates or strongly decrease the use of chemical fertilizers and pesticides (Hoekstra et al., 2011).

## 9.2 The WF by ISO 14046

The ISO 14046 standard was published in 2016 and provides the principles, requirements and guidelines for the identification and quantification of potential environmental impacts related to water of products, processes and organisations throughout their life cycle and it is based on the ISO LCA method. According to ISO 14046, a WF assessment can be performed as a stand-alone study, which analyses only environmental impacts connected to water, or as part of an LCA study, which evaluates a group of different environmental impacts (ISO, 2016). It is important to highlight that the term "WF" in ISO 14046 can be used only when the calculated value is a result of an impact assessment. The ISO WF standard requires a comprehensive assessment of all the potential environmental impacts of water use, including water availability, which can consider both quality and scarcity of water resources, and water degradation, which means a decrease in water quality. Therefore, the result of an ISO WF assessment is a WF profile, i.e. a profile of impact category indicator results, where all environmental impacts related to the use of water are assessed. When a WF study is not comprehensive, the standard requires to use a qualifier, which describes the analysed impact categories (e.g. "water scarcity footprint", "water availability footprint", water eutrophication footprint", "water acidification footprint") (ISO, 2016).

A WF study carried out in compliance with ISO standard can support the identification of improvement actions for the decrease of environmental impacts related to water, water efficiency and water management as well as decision-making processes in the topics of water consumption optimisation.

Similarly to ISO LCA method, the WF study is structured into the following four steps (Figure 12) (ISO, 2016):

1. Goal and scope definition;
2. WF inventory analysis;

3. WF impact assessment;
4. Interpretation of results.

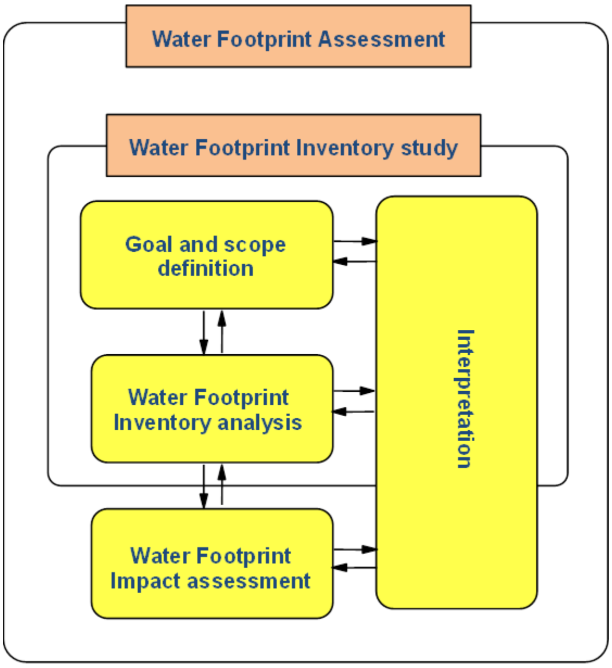



**Figure 12.** Phases of a WF assessment according to ISO 14046 (Source: Personal elaboration adpated from ISO 14046, 2016).

### 9.2.1 Goal and scope definition

In this phase, the purpose of the study, the target audience and the functional unit must be identified, together with the system boundaries as well as the geographical and temporal scope of the study. Moreover, it must be determined if the study is a stand-alone evaluation or if it is part of an LCA analysis and if the results will be used for comparative assertions among different products with the same function. The impact assessment methods and impact categories which will be included in the WF assessment must be selected in this step and it must be decided if the assessment will be comprehensive or if it will include only some impact indicator results (ISO, 2016).

System boundaries identify the unit processes to be included in the study and the criteria for their identification must be clearly explained. The exclusion of any life cycle phases, processes or elementary flows must be justified. Since water quality and water scarcity depend on site-specific conditions, these aspects should be taken into account in the selection of the unit processes for the study (ISO, 2016).

Finally, this phase requires the identification of data necessary for the data collection as well as their quality. These data include the amount and type of water used, its quality, the form of water use, any emissions to air and soil which affect the water use as well as geographical and temporal information about the water consumption. Primary data on the above issues should be used whenever possible, otherwise secondary data from literature or databases could be utilised. Data quality requirements include some specific issues of the collected data, such as the temporal, geographical and technology coverage, their precision, completeness, representativeness, consistency, reproducibility and uncertainty (ISO, 2016).

### 9.2.2 WF inventory analysis

The WF inventory must consider all the elementary flows, i.e. inputs and outputs from each unit process included in the system boundaries of the study. The calculation of the WF inventory must follow the procedures foreseen by ISO 14044. This means that all the calculation procedures must be clearly documented and any assumption must be explained, the validation of data must be performed during the data collection to comply with the data quality requirements, and appropriate reference flows must be settled for each unit process. As for ISO LCA method, this step can be iterative, therefore the system boundaries can be revised according to the goal and scope of the study. Moreover, in case of processes or systems which produce multiple outputs, allocation procedures must follow the requirements of ISO 14044 (ISO, 2016).

### 9.2.3 WF impact assessment

According to ISO 14046, this phase must be performed in compliance with ISO 14044. The potential environmental impacts related to water can be identified by the WF indicator result (e.g. “water scarcity footprint”), which is connected to a single impact category (e.g. “water scarcity”) or by the WF profile which takes into account more than one indicator results. Once having identified the impact categories, the category indicators and the characterisation models, this phase consists of the classification, which assigns the inventory results to the identified impact categories, and characterisation, which calculates the environmental impacts related to water by means of characterisation factors. Geographical and time-related parameters should be taken into account in the characterisation phase where applicable. A further optional step is the weighting, which combines the WF profile into a single parameter and that must be carried out in compliance with ISO 14044 (ISO, 2016).

#### *9.2.3.1 The WULCA impact assessment method*

According to ISO 14046, the WF impact assessment phase can include the evaluation of water availability and water degradation. The “water availability footprint” aims to evaluate the potential environmental impacts of products, processes and organisations on water availability and can include one or more impact categories. It is defined as “water scarcity footprint” when only the water quantity is accounted for and characterisation methods which consider site- specific differences in water scarcity are used. The purpose of the “water degradation footprint” is to assess the potential environmental impacts of products, processes and organisations on water quality.

In the framework of the Life Cycle Initiative, the UNEP/SETAC WULCA (Working Group on Water Use in LCA) has dealt with the problem of harmonizing and achieving consensus around an impact assessment method for evaluating freshwater consumption in LCA, because the characterisation models developed so far were based on different scarcity indicators which were not comparable (Boulay et al., 2014; Boulay et al; 2018).

The purpose of the developers of the new method was to answer the question “What is the potential to deprive another freshwater user (human or ecosystems) by consuming freshwater in this region?” (Boulay et al., 2018). Previous characterisation models were based on the “ratio of the water withdrawal-to-availability” (Frischknecht et al., 2008; Pfister et al., 2009; Boulay et al., 2018) or on the “water consumption-to-availability ratio” (Boulay et al., 2011; Hoekstra et al., 2012; Boulay et al., 2011). Therefore, a new method based on the “demand-to-availability ratio” was developed (Boulay et al., 2018), which could better answer the above question, because “both ecosystem water demand and human consumption are considered in demand” (Boulay et al, 2018).

The WULCA working group has then developed the midpoint indicator AWARE (Available WAter REmaining for area in a watershed), which represents the water available per unit of surface which remains in a basin after having satisfied the demand from humans and ecosystems (Boulay et al., 2018). This method is based on the assumption that the potential to deprive another water user is directly proportional to the amount of consumed water and inversely proportional to the available water in a basin per area and at a certain time (Boulay et al., 2018). The method is firstly calculated as the water Availability Minus the Demand of humans and aquatic ecosystems in a certain area and then the value is normalized with the world average value and inverted, thus expressing the relative value in comparison with the average volume of water consumed in the world. The indicator value ranges from 0.1 to 100, where the value 1 is the world average, and 10 for example, represents a region where the amount of available water per area is 10 times lower than the world average (Boulay et al., 2018). Characterization factors for this method have been developed per year and country, for agricultural and non-agricultural uses. This indicator only evaluates the blue water scarcity and it does not consider rainwater (i.e. green water), because it should be included in a separate indicator linked with land use (Boulay et al., 2018). In this regard, some tentative green water scarcity indicators have been developed by Núñez et al. (2012) for energy crops in Spain and by Quinteiro et al. (2015) who developed a method for the estimation of spatial and species-specific green water scarcity indicators (Quinteiro et al., 2018).

### 9.2.4 Interpretation of results

This last phase of the ISO14046 WF method includes the analysis of the obtained WF results as well as the identification of the main hotspots, i.e. the most significant life cycle phases, processes and elementary flows concerning the water-related environmental impacts. Moreover, completeness, sensitivity and consistency checks can be performed, together with qualitative or quantitative uncertainty analysis and sensitivity analysis. Finally, both the conclusions and the limitations of the WF study are determined (ISO, 2016).

## 9.3 Comparison between the WF Network method and ISO 14046

Both the WFN method and the ISO 14046, although with some specific different characteristics, follow the life cycle approach (Pfister et al., 2017) and have the common purpose to provide a framework for a better management of water resources (Boulay et al., 2013). The two frameworks were compared by Boulay et al. (2013), highlighting similarities and differences (Figure 13) and also by Pfister et al. (2017).

The comparison shows that the goal and scope phases are very similar in both methods, as well as the second phase which however has different names: “Water footprint accounting” in the former, where three types of water use (i.e. green, grey and blue WF) are calculated, and “Inventory analysis” in the latter, where inputs and outputs from the system's unit processes are accounted for. The third phase of the WFN method includes the sustainability assessment of the three WF components by means of environmental, economic and social indicators, although this involves the calculation of water availability of the catchment area which is rather difficult to be evaluated (Hoekstra et al., 2011). On the contrary, the impact assessment phase of the ISO 14046 transforms the results from the inventory analysis into environmental impacts by means of impact assessment methods. In the fourth phase, the method by WFN identifies response strategies, whereas ISO 14046 explains how the obtained results can be analysed (Boulay et al., 2013; Bai et al.; 2018).

The scientific basis of the two methods is therefore different, because WFN method follows a volumetric approach focused also on the quality of water and aims to support a sustainable and fair use of water resources, whereas ISO 14046 is focused on the environmental sustainability of products (Boulay et al., 2013). From this comparison it can be highlighted that both methods use quantitative indicators, but in different phases of the study, i.e. in the accounting phase for the WFN method, and in the impact assessment phase for ISO 14046 (Boulay et al., 2013). Furthermore, Quinteiro et al. (2018) point out how the two methods handle green, blue and grey water components in a different way.

As regards the potential synergies, both methods fulfil complementary objectives, and could take advantage of each other. For example, the blue or green WF indicator could be used in the inventory phase of ISO 14046, although impact assessment methods currently exist only for blue water use, whereas the LCA databases could be utilised also in the WFN method to increase the comprehensiveness of the WF accounting step. Moreover, joint efforts in research and development activities could develop common indicators for the water resource management (Boulay et al., 2013).

On the contrary, Pfister and Ridoutt (2014) state that both methods can be focused on products or on regional areas following a production chain perspective. Moreover, they identified inconsistencies between the two methods, such as the inclusion of green WF in the WFN method, which, according to Pfister and Ridoutt (2014), is an indicator of land use rather than of the water use. Therefore, the inclusion of green water use in LCA could lead to double counting of land use impacts or of other impacts on soil and water quality (Pfister et al., 2017; Pfister and Ridoutt, 2014). Moreover, the combination of grey and blue WF may not be meaningful from the environmental point of view for communication purposes and therefore should be translated in impact assessment indicators (Pfister and Ridoutt, 2014).

The two methods have been compared in few case studies so far, showing contrasting results. Manzardo et al. (2016) used them for comparing different food packaging alternative for tomato sauce and found that they provided consistent results for the green and blue water use and inconsistent outcomes for the grey water use. Therefore, companies should be aware that, since the two methods can result in different outcomes, they should perform a comprehensive assessment before taking decisions based on WF outcomes (Symeonidou & Vagiona, 2018). On the contrary, Bai et al. (2018) studied the WF of a swine farming company and observed that the impact assessment results from the ISO 14046 and the accounting results from the WFN had a consistent trend. According to Manzardo et al. (2016), therefore, the combined use of both methods can result in potential synergies for the definition of environmental strategies, because WFN method can support water efficiency and water management, while ISO 14046 is useful to assess water-related environmental impacts.

In conclusion, it is noteworthy that the methods for the calculation of the WF are still evolving, since they have been introduced only in the last recent years (Quinteiro et al., 2018). The development of ISO 14046 might be a step towards the establishment of a common WF method (Symeonidou & Vagiona, 2018), although both methods have created a lively discussion in the scientific community and have been subjected to mutual criticism. As discussed above, the two methods are indeed different: the WFN method accounts for a volumetric measure of the green, blue and grey water use, whereas ISO 14046 evaluates environmental impacts due to the use of blue water, because impact assessment methods have been developed so far only for blue water use. As regards environmental impacts related to green and grey water use, they are included in other environmental indicators in LCA, such as land use and eutrophication, although some tentative green water scarcity indicators have been developed in literature (Quinteiro et al., 2018)

In the next future, further research efforts should be focused on their improvement and on the identification of actual synergies (Manzardo et al., 2016; Symeonidou & Vagiona, 2018). For example, in the recent years the WFN method has tried to develop environmental indicators for blue and green water scarcity and water pollution, although affected by both lack of data at global level and some methodological problems (Quinteiro et al., 2018). As regards the WF based on ISO 14046, future improvements should be focused on the following aspects: assessment of environmental impacts related to changes in evapotranspiration; development of inventory data for blue water consumption in agriculture, which can be based on actual measurements at farm (often difficult to be recorded) or on the evapotranspiration associated to irrigation; development of spatial and time-related characterisation factors as well as their connection with inventory flows (Quinteiro et al., 2018). Despite those current methodological problems, both methods could support governments, companies and individual producers, such as farmers, in the identification of water-related environmental hotspots and improvement options and more in general in the sustainable and efficient use and management of water resources in the whole production chain. Moreover, the development of both a common accepted WF method and relevant supporting data could be useful for the implementation of green marketing strategies addressed to consumers (Symeonidou & Vagiona, 2018). In fact, it could encourage the development of transparent and reliable WF labels, which would provide consumers with information about the product's embedded water, thus helping them to adopt more sustainable and equitable consumption choices (Symeonidou & Vagiona, 2018).

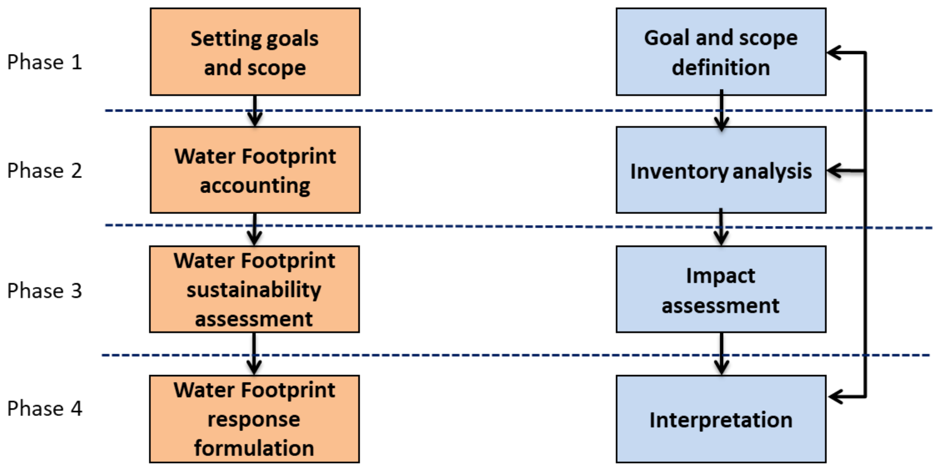



**Figure 13.** Comparison between the WFN method and ISO 14046 (Source: Personal elaboration adapted from Boulay et al., 2013).

# 10 Water Footprint of Italian tomato production

The WF method of the WFN (Hoekstra et al., 2011) was applied to the cultivation process of a Product Designation of Origin (PDO) Italian tomato, named "Pomodorino del Piennolo del Vesuvio", in a company located in the province of Naples, with the aim to test the practicability of the method and to highlight any critical problems in its application. A further aim was also to compare the results obtained with similar studies related to the Mediterranean climatic region. A survey was performed on literature WF studies on tomato production, finding that most studies apply only the first two phases of the WF method by the WF network, i.e. goal and scope definition and WF accounting (Mekonnen and Hoekstra, 2010; Aldaya et al., 2009; Evangelou et al., 2016; Chico et al., 2010¸ Chouchane et al., 2013), probably because of lack of data for determining the water availability of the river basin and also because, according to Hoekstra et al., (2011), those phases are still in development. Therefore, this case study applies those two phases of the Piennolo tomato cultivation.

This variety of tomato is produced exclusively in 18 municipalities of the Vesuvian area and has many peculiarities, such as reduced cultivation requriements and a long conservation in the winter period (up to 7-8 months), due to the high consistency of the peel, the strength of hanging on the peduncle and the high content in soluble sugars, acids and solids.

This tomato cultivar, due to the PDO regulation, requires specific methods of cultivation and conservation, typical of the area. Moreover, it can benefit from the particular environment of the Vesuvian area, (i.e. optimal exposure to sun radiation, dry climate and soils pyroclastic nature). According to the PDO regulation, the following cultivation characteristics are particularly relevant for the case study:

- The number of plants per hectare must not be higher than 45,000;
- Cultivation in greenhouses, tunnels or above ground is not admitted;
- Spray or sprinkler irrigation are not admitted; only micro-irrigation or localised irrigation are admitted.
- The maximum production is 16 t per hectare.

The Piennolo tomato is very resistant to water scarcity conditions. In fact, the contribution of rainwater is sufficient to plant growth without supplementary irrigation. Therefore, blue

WF of the cultivation phase if equal to zero, as applied also by other authors (Chico et al., 2010).

The main cultivation characteristics are included in Table 31.

| Parameter of Piennolo tomato cultivation | Value/description |
|---|---|
| **Area** | 1 ha |
| **Yield** | 16 t/ha |
| **Fertiliser** | Horse manure |
| **Amount of fertiliser** | 7,5 t/ha |
| **Irrigation** | NO |
| **Greenhouse** | NO |
| **Crop growth** | About 5 months |
| **Soil type** | Sandy |
| **Crop height** | 1 m |

**Table 31.** Primary cultivation data for the production of the PDO Italian tomato.

The work performed on the WF of Italian Piennolo tomato during the PhD was included in the following publications, which are the basis of the whole chapter 10:

- Ferrara M., Fantin V., Righi S., Chiavetta C., Buttol P., Bonoli A., 2017. **Applicazione della Water Footprint sviluppata dal WF Network: il caso del Pomodorino del Piennolo del Vesuvio DOP**. In Proceedings of XI Conference of Italian LCA Network Association, Siena, 22-23 June 2017, ISBN 978-88-8286-352-4.
- Fantin V., Ferrara M., Righi S., Chiavetta C., Buttol P., 2017. **Metodologia di Water Footprint e sua applicazione nel settore agroalimentare: il caso del Pomodorino del Piennolo del Vesuvio DOP**. ENEA Technical report, USER-PG20-004, June 2017 (Confidential).

## 10.1 Calculation of Water Footprint of crop growth

The total WF of the process of growing crops ($WF_{proc}$) is the sum of green, blue and grey WF (Hoekstra et al., 2011):

$$WF_{proc} = WF_{proc,green} + WF_{proc,blue} + WF_{proc,grey} \quad \text{[volume/mass]} \quad (14)$$

The green and blue components ($WF_{proc,green}$, $m^3/t$ and $WF_{proc,blue}$, $m^3/t$) are the green and blue components in crop water use ($CWU_{green}$, $m^3/ha$ and $CWU_{blue}$, $m^3/ha$), respectively, divided by the crop yield (Y, t/ha) (Hoekstra et al., 2011)::

$$WF_{proc,green} = \frac{CWU_{green}}{Y} \quad \text{[volume/mass]} \quad (15)$$

$$WF_{proc,blue} = \frac{CWU_{blue}}{Y} \quad \text{[volume/mass]} \quad (16)$$

The green and blue components in crop water use (CWU, $m^3/ha$) are the sum of the daily evapotranspiration (ET, mm/day) throughout the growing period, i.e. from the day of planting (day 1) to the day of harvest (lgp is the length of growing period) (Hoekstra et al., 2011):

$$CWU_{green} = 10 \times \sum_{d=1}^{lgp} ET_{green} \quad \text{[volume/area]} \quad (17)$$

$$CWU_{blue} = 10 \times \sum_{d=1}^{lgp} ET_{blue} \quad \text{[volume/area]} \quad (18)$$

The factor 10 converts water depths (mm) into water volumes per area ($m^3/ha$) (1mm/day is equal to 10 $m^3/ha/day$). The green crop water use represents the total rainwater evaporated from the field during the growing period; the blue crop water use measures the total irrigation water evaporated from the field in the same period.

The grey component ($WF_{proc,grey}$, $m^3/t$) is calculated by multiplying the chemical application rate (fertilizers and pesticides) per hectare (Appl, in mg/ha) by the leaching-runoff fraction (α) and then dividing by the difference between the water quality standard for that pollutant (the maximum acceptable concentration $c_{max}$, in mass/volume) and its natural concentration in the receiving water body ($c_{nat}$, in $kg/m^3$) and finally divided by the crop yield (Y, t/ha) (Hoekstra et al., 2011):

$$WF_{proc,grey} = \frac{(\alpha \times Appl)/(c_{max} - c_{nat})}{Y} \quad \text{[volume/mass]} \quad (19)$$

Finally, for a complete assessment, the green and blue water incorporated into the harvested crop should be taken into account, which can be estimated to be 1% of the evaporated water.

## 10.2 Calculation of Green WF

Evapotranspiration for Piennolo tomato growing was estimated by the CROPWAT model (FAO, 2010b), based on the method described in Allen et al (1998) and recommended also by the WF Manual (Hoekstra et al., 2011). The detailed "Irrigation Schedule option" was chosen for the case study as recommended by Hoekstra et al. (2011). This option includes a dynamic soil water balance, requires the irrigation typology as well as some soil-related parameters.

The evapotranspiration of a crop ($ET_{c\ adj}$) was estimated through the following formula (Allen et al., 1998):

$$ET_{c\ adj} = ET_0 \times K_c \times K_s \text{ [mm/day]} \qquad (20)$$

with:

$ET_{c\ adj}$ = evapotranspiration of the crop under stress conditions [mm/day]

$K_c$ = crop coefficient [dimensionless]

$ET_0$= reference evapotranspiration [mm/day]

$K_s$ = water stress coefficient which describes the effect of water stress on crop transpiration (for soils with water stress conditions, Ks <1, if there is no water stress Ks = 1).

$ET_0$ is the reference evapotranspiration and represents the evapotranspiration from a reference surface with specific characteristics, at a specific time and location. $ET_0$ is a purely climatic parameter, independent of the type of crop and management practices and can be calculated by the Penman-Monteith equation (Allen et al., 1998).

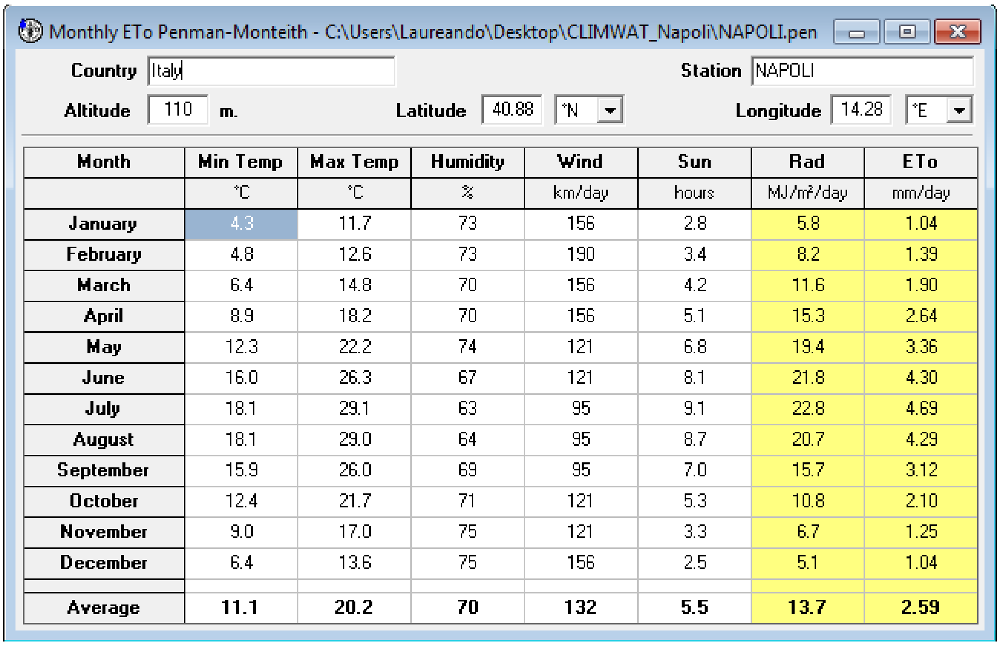
While the effects of climatic conditions are included in the estimation of $ET_0$, the characteristics of a specific crop are considered in the Kc value, which is not affected by large site and and climate variations and is thus globally accepted for the calculation of $ET_c$ (Allen et al., 1998). In particular, Kc accounts for four specific characteristics which distinguish a crop from the grass used as reference: plant height, albedo, foliage resistance and evaporation from the soil. As highlighted previously, the value of the crop coefficient varies during the crop growing period (Allen et al., 1998). In fact, there will be different Kc values in the initial phase (Kc ini), in the intermediate phase (Kc mid) and in the final phase (Kc end). In practice, the coefficient Kc represents, for each stage of the crop growth, the physical and development differences between the studied crop and the reference surface. Crops Kc values can be found in the literature (for example, in Allen et al., 1998 (Table 32)).

| Crop | $K_{c\ ini}$ | $K_{c\ med}$ | $K_{c\ end}$ | Maximum crop height (h) (m) |
|---|---|---|---|---|
| **Small vegetables** | **0.7** | **1.05** | **0.95** | |
| Broccoli | | 1.05 | 0.95 | 0.3 |
| Brussel sprouts | | 1.05 | 0.95 | 0.4 |
| Cabbage | | 1.05 | 0.95 | 0.4 |
| Carrots | | 1.05 | 0.95 | 0.3 |
| Cauliflower | | 1.05 | 0.95 | 0.4 |
| Celery | | 1.05 | 1.00 | 0.6 |
| Garlic | | 1.00 | 0.70 | 0.3 |
| Lettuce | | 1.00 | 0.95 | 0.3 |
| Onions- dry | | 1.05 | 0.75 | 0.4 |
| Spinach | | 1.00 | 0.95 | 0.3 |
| Radish | | 0.90 | 0.85 | 0.3 |
| **Vegetables – Solanum family** | **0.6** | **1.15** | **0.80** | |
| Egg plant | | 1.05 | 0.90 | 0.8 |
| Sweet peppers | | 1.05 | 0.90 | 0.8 |
| Tomato | | 1.15 | 0.70-0.90 | 0.6 |
| **Vegetables – Cucumber family** | **0.5** | **1.00** | **0.80** | |
| Cantaloupe | 0.5 | 0.85 | 0.60 | 0.3 |
| Cucumber – fresh market | 0.6 | 1.00 | 0.75 | 0.3 |
| Pumpkin | | 1.00 | 0.80 | 0.4 |
| Zucchini | | 0.95 | 0.75 | 0.3 |
| Sweet melons | | 1.05 | 0.75 | 0.4 |
| Watermelon | | 1.00 | 0.75 | 0.4 |
| **Roots and tubers** | **0.5** | **1.10** | **0.95** | |
| Beets | | 1.05 | 0.95 | 0.4 |
| Cassava – year 1 | 0.3 | 0.80 | 0.30 | 1.0 |
| Parsnip | 0.5 | 1.05 | 0.95 | 0.4 |
| Potato | | 1.15 | 0.75 | 0.6 |
| Sweet potato | | 1.15 | 0.65 | 0.4 |
| Turnip | | 1.10 | 0.95 | 0.6 |
| Sugar beet | 0.35 | 1.20 | 0.70 | 0.5 |

**Table 32.**Values for the Kc crop coefficient and maximum crop height (Source: Personal elaboration adapted from Allen et al., 1998).

When the field conditions are not optimal, a water stress coefficient (Ks) must be added, which reflects the effects of actual conditions on the evapotranspiration. In absence of water stress conditions, Ks = 1; when the soil suffers water stress, value of Ks is between 0 and 1.

As regards the climate parameters, data for the studied area were obtained by the FAO's CLIMWAT 2.0 climate database using the climatic and the rainfall values of the meteorological station closest to the studied site ("NAPOLI" station with coordinates 14.28 ° Lon, Lat 40.88 ° and altitude of 110m slmm) which were then included in CROPWAT (Figure 14 and Figure 15).


Monthly ETo Penman-Monteith - C:\Users\Laureando\Desktop\CLIMWAT_Napoli\NAPOLI.pen

Country Italy Station NAPOLI

Altitude 110 m. Latitude 40.88 °N Longitude 14.28 °E

| Month | Min Temp | Max Temp | Humidity | Wind | Sun | Rad | ETo |
|---|---|---|---|---|---|---|---|
| | °C | °C | % | km/day | hours | MJ/m²/day | mm/day |
| January | 4.3 | 11.7 | 73 | 156 | 2.8 | 5.8 | 1.04 |
| February | 4.8 | 12.6 | 73 | 190 | 3.4 | 8.2 | 1.39 |
| March | 6.4 | 14.8 | 70 | 156 | 4.2 | 11.6 | 1.90 |
| April | 8.9 | 18.2 | 70 | 156 | 5.1 | 15.3 | 2.64 |
| May | 12.3 | 22.2 | 74 | 121 | 6.8 | 19.4 | 3.36 |
| June | 16.0 | 26.3 | 67 | 121 | 8.1 | 21.8 | 4.30 |
| July | 18.1 | 29.1 | 63 | 95 | 9.1 | 22.8 | 4.69 |
| August | 18.1 | 29.0 | 64 | 95 | 8.7 | 20.7 | 4.29 |
| September | 15.9 | 26.0 | 69 | 95 | 7.0 | 15.7 | 3.12 |
| October | 12.4 | 21.7 | 71 | 121 | 5.3 | 10.8 | 2.10 |
| November | 9.0 | 17.0 | 75 | 121 | 3.3 | 6.7 | 1.25 |
| December | 6.4 | 13.6 | 75 | 156 | 2.5 | 5.1 | 1.04 |
| **Average** | **11.1** | **20.2** | **70** | **132** | **5.5** | **13.7** | **2.59** |

**Figure 14.** Climate parameters inserted in CROPWAT software (Source: CROPWAT software, personal elaboration).

Monthly rain - C:\Users\Laureando\Desktop\prova cropwat\NAPOLI.cli

Station NAPOLI    Eff. rain method USDA S.C. Method

| | Rain | Eff rain |
|---|---|---|
| | mm | mm |
| January | 116.0 | 94.5 |
| February | 85.0 | 73.4 |
| March | 73.0 | 64.5 |
| April | 62.0 | 55.8 |
| May | 44.0 | 40.9 |
| June | 31.0 | 29.5 |
| July | 19.0 | 18.4 |
| August | 32.0 | 30.4 |
| September | 64.0 | 57.4 |
| October | 107.0 | 88.7 |
| November | 147.0 | 112.4 |
| December | 135.0 | 105.8 |
| **Total** | **915.0** | **771.8** |

**Figure 15.** Rainfall values inserted in CROPWAT (Source: CROPWAT software, personal elaboration).

After having obtained the ET0 evapotranspiration value (Figure 14), crop parameters were included in the software (Figure 16), in order to calculate the crop evapotranspiration under standard conditions (ETc). The software contains the average values for each parameter and for each type of crop which can be modified if site-specific data are available. Therefore, the crop height was increased to 1 metre (primary data provided by the producer). This variation also implies a change in the Kc mid value, which is the crop coefficient in the middle stage of plant growth, which was increased from 1.15 to 1.20 (as suggested by Allen et al., 1998). The Kc values in the initial and final phases of the plant growth (Kc ini and Kc end) were fixed as 0.6 and 0.8 as suggested by Allen (1998). The number of days of each tomato growing phase with a Mediterranean climate, were divided in: 1) an initial phase of 30 days; 2) a development phase of 40 days; 3) a mid-season of 45 days; 4) a late season phase of 45 days.

The sowing and harvesting dates, provided directly by the farmer (20$^{th}$ March - 11$^{th}$ August), were included in CROPWAT. The data already available in CROPWAT were used for the rooting depth, the critical depletion level and the yield response factor. Finally, the type of soil of the studied areas were included in the software, i.e. sandy soil, according to the PDO regulation. Data suggested by the software for the soil characteristics (total available soil moisture, maximum rain infiltration date, etc.) were used, because no site-specific data were available (Figure 17).

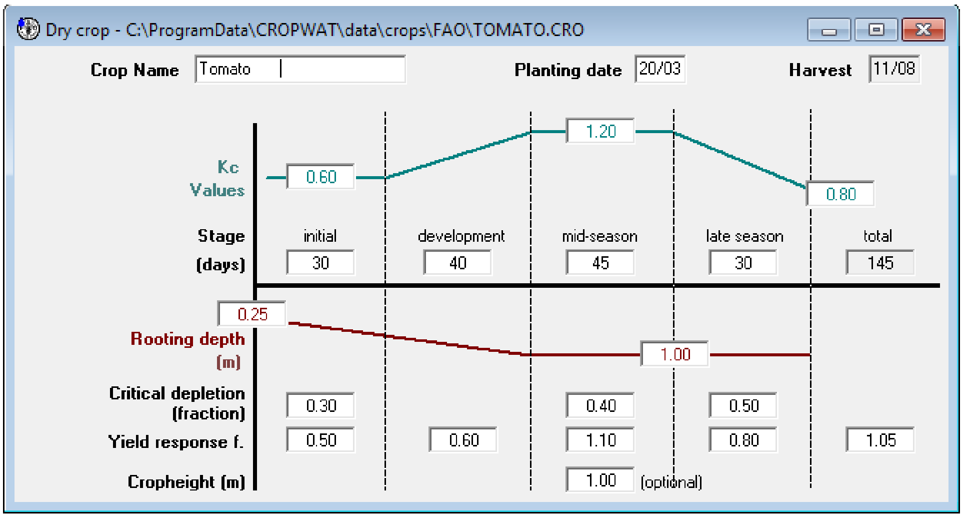



**Figure 16.** Crop parameters inserted in CROPWAT (Source: CROPWAT software, personal elaboration).

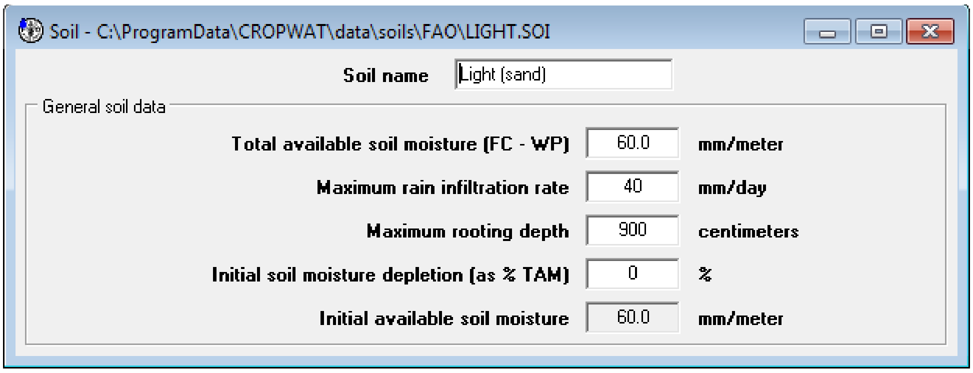



**Figure 17.** Sandy soil parameters (Source: CROPWAT software, personal elaboration).

As regards the irrigation applied, since no amount of water is used in addition to the rainfall, the option "No irrigation (rainfed)" was selected. Therefore, the amount of blue evapotranspirated water ($ET_{blue}$) is equal to zero. Consequently, the amount of evapotranspirated green water ($ET_{green}$) throughout the entire crop life cycle is equal to the total evapotranspiration (named "Actual water use by crop" in the final screen of the software, Figure 18).

It can be observed that, in the first period of crop growth (March), the water stress coefficient Ks is equal to 1 and in this case the crop evapotranspiration occurs in optimal conditions. When the precipitation decreases (first ten days of May, Figure 19), the plant suffers the effects of water stress, which means that the value of Ks decreases, up to 0.08 (early August).

The "Actual water use by crop" value (215.6 mm) is given by the sum of all evapotranspiration values during the whole crop life ($\Sigma ET_a$). This value, therefore, must be converted to $m^3$/ha by applying the conversion factor 10, and then divided by the yield of the crop (16 t/ha), following the formulas at par 10.1:

$$CWU_{green} = 10 \times \sum_{d=1}^{lgp} ET_{green} \qquad \text{[volume/area]} \qquad (21)$$

$$WF_{proc,green} = \frac{CWU_{green}}{Y} \qquad \text{[volume/mass]} \qquad (22)$$

The final obtained value is 134.75 $m^3$/t, as shown in Table 33.

In addition to the evapotranspiration, the quantity of water incorporated in the fruit should also be considered. In particular, despite a tomato water content of 94% (i.e. 0.94 liters of water per kg of tomato), the WF of the incorporated water will be 0.94 $m^3$/t, i.e. lower than 0.7% of the WF related to water lost by evapotranspiration.

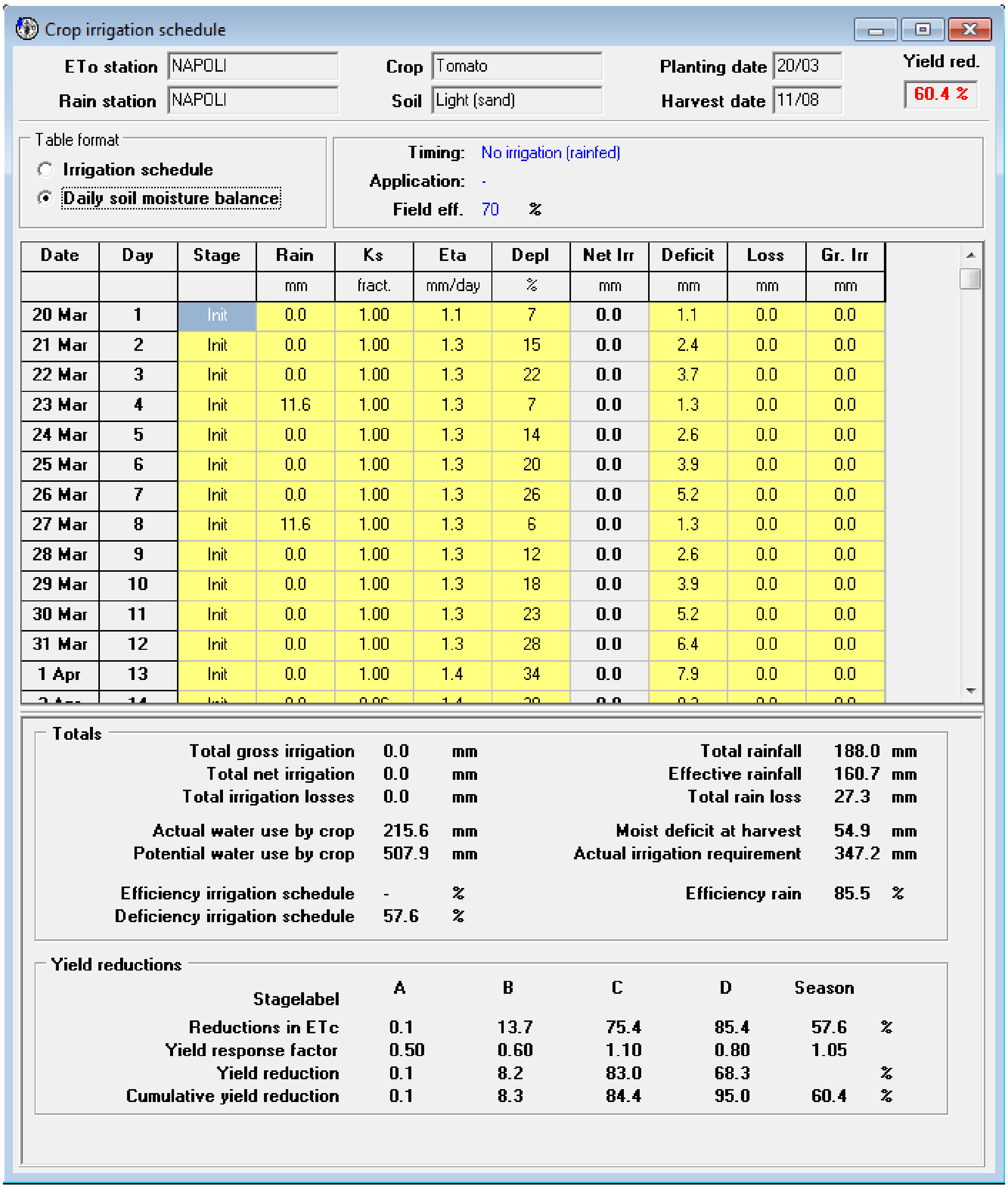



**Figure 18.** Calculation of the evapotranspiration by the “Irrigation schedule option” (Source: CROPWAT software, personal elaboration).

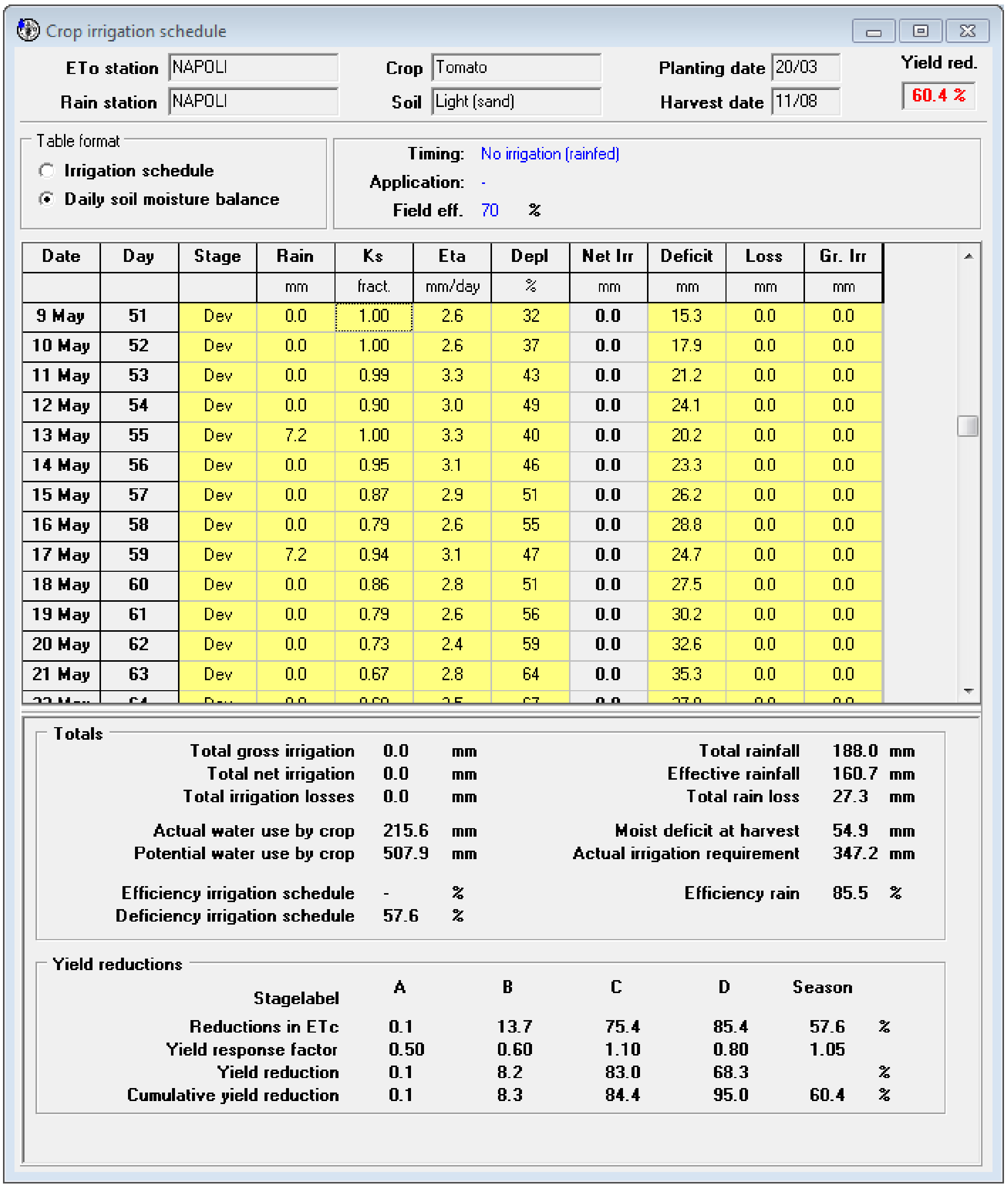



**Figure 19.** Effect of water stress conditions (Ks<1) (Source: CROPWAT software, personal elaboration).

| CROPWAT option | ETgreen | CWUgreen | Y | WFgreen |
|---|---|---|---|---|
| | mm/growing period | $m^3$/ha | t/ha | $m^3$/t |
| **Irrigation schedule option** | 215.6 | 2156 | 16 | 134.75 |

**Table 33.** Results of the evapotranspiration and green Water Footprint.

## 10.3 Calculation of Grey WF

As explained in paragraph 10.1, the grey WF can be calculated as the total amount of chemical applied to the soil (Appl) multiplied by the leaching-runoff fraction (α) divided by the maximum acceptable concentration ($c_{max}$, kg/m$^3$) minus the natural concentration for the considered contaminant ($c_{nat}$, kg/m$^3$):

$$WF_{proc,grey} = \frac{L}{c_{max} - c_{nat}} = \frac{\alpha \times Appl}{c_{max} - c_{nat}} \qquad \text{[volume/time]} \qquad (23)$$

The most difficult parameter to be quantified is the leaching-runoff fraction α, which is the percentage of a chemical which reaches to groundwater through leaching or surface water through runoff. Literature studies on tomato cultivation use a fixed fraction of 10% (Mekonnen and Hoekstra, 2010; Aldaya et al., 2009; Evangelou et al., 2016; Chico et al., 2010), according to Franke et al. (2013), which suggest to use the average α value in case of lack of detailed data (Table 34).

| | Nutrients | | Metals | Pesticides |
|---|---|---|---|---|
| | Nitrogen | Phosphorus | | |
| Minimum leaching-runoff fraction $\alpha_{min}$ | 0.01 | 0.0001 | 0.4 | 0.0001 |
| Average leaching-runoff fraction $\alpha_{avg}$ | 0.1 | 0.05 | 0.7 | 0.01 |
| Maximum leaching-runoff fraction $\alpha_{max}$ | 0.25 | 0.05 | 0.9 | 0.1 |

**Table 34.** Minimum, average and maximum values of the leaching-runoff fraction α for nutrients, metals and pesticides (Source: Personal elaboration adapted from Franke et al., 2013).

An alternative possible approach to estimate the pollutant load entering a water body, which was used in this case study, is the use of the leaching-runoff factor β which is applied to the nutrient surplus, i.e. the remaining quantity after the plant's uptake (Franke et al., 2013):

$$L = \beta \; x \; Surplus \qquad \text{[mass/time]} \qquad (24)$$

The surplus is calculated by the difference between the application rate (Appl) of the substance and the offtake rate, defined as follows (Franke et al., 2013):

$$\text{Surplus} = \text{Appl} - \text{Offtake} \qquad [\text{mass / time}] \qquad (25)$$

The offtake can be estimated by multiplying the crop yield by the chemical substance content in the crop (Franke et al., 2013):

$$\text{Offtake} = \text{Yield x Chemical substance content in the crop} \qquad [\text{mass / time}] \quad (26)$$

According to Franke et al. (2013), the value of β can be estimated on the basis of qualitative information about environmental factors (e.g. soil properties and climate conditions) and agricultural practices (e.g. the substance application rate, the type of drainage and the type of harvest) as well as the chemical-physical properties of the applied substance. Since each of the above-mentioned factor influences β differently, a weight w has to be assigned to each of them to indicate their importance.
In case of lack of site-specific information for the above factors, global average values for the leaching-runoff fraction α and β (Table 34 and Table 35) can be used.

| | **Nitrogen** | **Phosphorus** |
|---|---|---|
| Minimum leaching-runoff fraction $\beta_{min}$ | 0.08 | 0.0001 |
| Average leaching-runoff fraction $\beta_{avg}$ | 0.44 | 0.05 |
| Maximum leaching-runoff fraction $\beta_{max}$ | 0.8 | 0.1 |

**Table 35.** Minimum, average and maximum values of the β leaching fraction for nitrogen and phosphorus (Source: Personal elaboration adapted from Franke et al., 2013).

Table 36 can be used to estimate the leaching-runoff potential in a specific site (Franke et al., 2013). For each influencing factor, the potential leaching-runoff factor and the relative weight can be identified (Franke et al, 2013). The management practices can be judged on the basis of Table 37, evaluating how many practices are used in the analysed site.

| Category | Factor | | Nitrogen | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | ***Leaching-runoff fraction*** | | Very low | Low | High | Very high |
| | | | ***Score (s)*** | | 0 | 0.33 | 0.67 | 1 |
| | | | ***Weight (w)*** | | | | | |
| | | | α | β | | | | |
| ***Environmental factors*** | Atmospheric input | N deposition (g $Nm^{-2}yr^{-1}$) | 10 | 10 | <0.5 | >0.5 | <1.5 | >1.5 |
| | Soil | Texture relevant for leaching | 15 | 15 | Clay | Silt | Loam | Sand |
| | | Texture relevant for runoff | 10 | 10 | Sand | Loam | Silt | Clay |
| | | Natural drainage (relevant for leaching) | 10 | 15 | Poorly to very poorly drained | Moderately to imperfectly drained | Well drained | Excessively to extremely drained |
| | | Natural drainage (relevant for runoff) | 5 | 10 | Excessively to extremely drained | Well drained | Moderately to imperfectly drained | Poorly to very poorly drained |
| | Climate | Precipitation (mm) | 15 | 15 | 0-600 | 600-1200 | 1200-1800 | >1800 |
| ***Agricultural practice*** | N-fixation (kg/ha) | | 10 | 10 | 0 | >0 | <60 | >60 |
| | Application rate | | 10 | 0 | Very low | Low | High | Very high |
| | Plant uptake | | 5 | 0 | Very high | High | Low | Very low |
| | Management practice | | 10 | 15 | Best | Good | Average | Worst |

**Table 36.** Factors influencing leaching-runoff potential for nitrogen (Source: Personal elaboration adapted from Franke et al., 2013).

| Measure | Applied? | |
|---|---|---|
| Controlled application of chemicals | Yes | No |
| Diffuse pollution mitigation measures | Yes | No |
| Careful handling of chemicals | Yes | No |
| Application immediately before heavy rainfall or irrigation is avoided | Yes | No |
| Controlled irrigation | Yes | No |
| Field is only naturally drained | Yes | No |
| Spreading on frozen ground or foliage is avoided | Yes | No |
| Usage of winter cover crops | Yes | No |
| Soil organic matter management | Yes | No |

**Table 37.** Agricultural management practice questionnaire (Source: Personal elaboration from Franke et al., 2013).

As regards the critical contaminant to be considered, only nitrogen was taken into account, in compliance with published literature studies (Chapagain et al., 2006, Chapagain and Orr, 2009; Mekonnen and Hoekstra, 2010), because of the following reasons:

1. Nitrogen is a very dynamic element which can be easily washed away from the soil;
2. The problem of water protection from nitrates pollution from agriculture is critical.

According to Tesi and Lenzi (2005), the tomato nitrogen content was assumed 2.7 kg N/t of tomato grown in open air. The amount of nitrogen uptaken by the crop (Offtake) was obtained by multiplying the tomato nitrogen content for the crop yield, thus resulting in 43.2 kg N. The average nitrogen content of horse manure was considered to be 0.71% (Bott, 2015), which means 53.25 kg of N applied to the soil (Appl). However, the actual value of horse manure nitrogen content can be rather variable because it strongly depends on both the type of feed and of litter used. In fact, it varies from 0.24% for horses fed with low protein diet to 1.14% for horses stables which use rye straw as litter.

Then, the Surplus (i.e. the quantity of N remaining on the soil which could undergo leaching and/or runoff phenomena) was obtained by subtracting the amount of nitrogen applied to the field from the quantity of nitrogen uptaken by the crop, resulting in 10.05 kg N.

In order to quantify the actual percentage of nitrogen which can leach/runoff, the factor β was calculated according to the following formula (Franke et al., 2013):

$$\beta = \beta_{min} + \left[\frac{\sum_i s_i \times w_i}{\sum_i w_i}\right] \times (\beta_{max} - \beta_{min}) \qquad (27)$$

In compliance with Table 35, βmin and βmax were considered to be equal to 0.08 and 0.8 respectively. The scores (s) and the weights (w) were assigned according to Table 36:

- Due to lack of information about on-site nitrogen deposition, a score of 0.5 was adopted;
- The soil is sandy and therefore the score is 1 for leaching and 0 for runoff. This means that nitrogen is much more likely to reach groundwater than surface water;
- The soil is well drained, therefore the scores for this factor are 0.67 for leaching and 0.33 for runoff;
- Since the annual precipitation is 915 mm, the score is 0.33;
- Due to lack of information about on-site nitrogen fixation, a score of 0.5 was adopted;
- The weight of both the nitrogen application rate and the crop uptake for the calculation of β was assumed to be 0 because they were already taken into account in the Surplus quantification;
- The score of the agricultural management practice was assumed to be 0.33 because Italy is an industrialized country. Furthermore, the questionnaire reported in Table 37 was answered, answering "yes" to all the questions with the exception of the second and the eighth ones, which were answered "no" due to a precautionary approach (no detailed information was available), thus obtaining 7 affirmative answers which confirm the "agricultural management practice" (and therefore a score of 0.33) as "good".

Finally, the nitrogen load (L) was obtained by multiplying β (= 0.427) and the Surplus, resulting in 4.29 kg of N.

The obtained β value is very close to the average nitrogen β value nitrogen (0.44, reported by Franke et al., 2013) which would have resulted in a nitrogen load of 4.42 kg. On the contrary, if the load had been calculated using the mean value of α (0.1), a nitrogen load of 5.32 kg would have been obtained. This greater value is due to the fact that the leaching-run-off fraction α does not take into account the plant uptake, i.e. the nitrogen actually available to be transported into water is not the whole amount applied to the field, but only a fraction of this.

The maximum concentration of nitrogen in water ($c_{max}$) was assumed 10 mg/l N, as suggested by US EPA (2013) and also adopted in literature studies (Chico et al., 2010; Mekonnen and Hoekstra, 2010; Aldaya et al., 2009), and the natural nitrogen concentration in water ($c_{nat)}$ was considered zero (Chico et al., 2010; Mekonnen and Hoekstra, 2010; Aldaya et al., 2009).

The grey WF was finally calculated by dividing the load L by the difference between $c_{max}$ and $c_{nat}$ and then dividing bythe yield. Moreover, the grey WF values were also calculated with the maximum and minimum nitrogen content in horse manure. With a manure nitrogen content of 0.24%, the nitrogen applied to the field cannot fully satisfy the crop requirements (Appl is 18 kg compared with an Offtake of 43.2 kg) and therefore no nitrogen will leach/runoff. On the contrary, with a manure nitrogen content of 1.14%, the Surplus will be 42.3 kg, which will result in a grey WF of 112.99 $m^3$/t. Finally, the grey WF related to the average values of α and β were calculated, obtaining 33.28 $m^3$/t and 27.64 $m^3$/t.

The obtained grey WF values are included in Table 38.

| **Surplus** | **Area** | **N losses via leaching/runoff (42.7%)** | $c_{max}$ | **$WF_{grey}$ total** | **Yield** | **$WF_{grey}$** |
|---|---|---|---|---|---|---|
| kg/ha | ha | kg | mg/L | $m^3$ | t | $m^3$/t |
| 10.05 | 1 | 4.29 | 11.3 | 379.98 | 16 | 23.76 |
| 10.05 | 1 | 4.29 | 10 | 429 | 16 | 26.85 |

**Table 38.** Results of the grey Water Footprint.

## 10.4 Results and discussion

A specific WF is connected to temporal and site-specific conditions. The magnitude of its impact will therefore depend on several factors, such as the availability of water resources in the basin where the studied process is located, or the competition between different water users in the same area, or the amount of water required for both the ecosystems maintenance and the assimilation capacity of the water body. This means that the vulnerability of the area where the WF is located affects the WF impact (Hoekstra et al., 2008). According to Hoekstra et al. (2011), a Water Footprint is not environmentally sustainable when the environment water requirements to preserve biodiversity and support human life are violated or when the amount of pollutants exceeds the water body assimilation capacity. The sustainability assessment of a WF therefore requires to calculate the water availability of the catchment area and consequently the blue and green water scarcity as well as the water pollution level. Nevertheless, at the moment these parameters cannot be precisely calculated due to lack of relevant information (Hoekstra et al., 2011).

Because of these reasons, the calculated WF values were compared with the following literature WF of tomato cultivation in the Mediterranean region, in order to assess the results significance (Table 39):

- Mekonnen and Hoekstra (2010) for world averages;
- Aldaya et al. (2009) for Italian and Campania averages;
- Evangelou et al. (2016) for Greek averages (industrial production);
- Chouchane et al. (2013) for the Tunisian averages;
- Chico et al. (2010) for Spanish averages (rainfed, i.e. without irrigation).

The comparison between the green, blue and grey WF of Piennolo tomato and literature tomato WF is included inTable 39.

| | $WF_{green}$ | $WF_{blue}$ | $WF_{green+blue}$ | $WF_{grey}$ | $WF_{tot}$ | Y |
|---|---|---|---|---|---|---|
| | $m^3/t$ | $m^3/t$ | $m^3/ t$ | $m^3/t$ | $m^3/t$ | t/ha |
| **Piennolo tomato** | 134.75 | 0 | 134.75 | 26.85 | 161.6 | 16 |
| **World average** | 108 | 63 | 171 | 43 | 214 | - |
| **Italy** | 44 | 124 | 168 | 31 | 199 | 35 |
| **Campania** | 31 | 61 | 92 | - | - | 62 |
| **Tunisia** | 60 | 50 | 110 | 10 | 120 | 32 |
| **Greece (industrial)** | 13 | 27 | 40 | 21 | 61 | 127 |
| **Spain (rainfed)** | 158 | 0 | 158 | 808 | 966 | 13 |

**Table 39.** Water Footprint values and yield (Y) for some case studies about tomato cultivation in the mediterranean region.

Results show that the WF of the same crop can varies significantly. In fact, it ranges from 61 $m^3/t$ for the Greek industrial production to 966 $m^3$ /t of the Spanish rainfed production. This variation is due to soil conditions, to different agricultural management practices, which affect crop yields and also to the assumptions adopted during the application of the method (Evangelou et al., 2016). More in detail, the Greek industrial production (Evangelou et al., 2016) has a very low WF due to the very high crop yield.

The comparison among the grey WF highlights that the value of Piennolo tomato (134.75 $m^3/t$), is much higher than the other ones, with the exception of the Spanish case, which

amounts to 158 $m^3/t$. However, this result can be explained by the fact that crops are irrigated in all literature studies. Therefore, in order to consider the total amount of water "consumed" during the crop growth, the values of $WF_{green+blue}$ should be compared. In this case, the value for Piennolo tomato (134.75 $m^3/t$) is lower than both the Italian and world average (168 $m^3/t$ and 171 $m^3/t$, respectively), although greater than Campania average value (92 $m^3/t$). This high Piennolo tomato value is due to its very low yield and it is actually in line with the value of $WF_{green+blue}$ of the Spanish rainfed case, which has an even lower yield (158 $m^3/t$).
As regards the $WF_{grey}$, the calculated value is very low (26.85 $m^3/t$), and is very similar to the Greek industrial tomato value (21 $m^3/t$) whose yield is 8 times greater. This is due to the precise fertilization management operated by the Vesuvian farm, which applies low quantities of nitrogen which are almost totally used by the crop. It should be also pointed out that all the analysed literature studies considered only the nitrogen pollution and that, differently to the Piennolo tomato case study, all these authors used a fixed leaching-runoff value α for nitrogen, equal to 10% (as suggested by Mekonnen and Hoekstra, 2010 and Chapagain et al., 2006), which does not account for the soil properties and agricultural management practices (Evangelou et al., 2016). As a consequence, the grey WF obtained by calculating the site-specific leaching/runoff factor is more robust than that obtained by means of the average α and β mean values. Anyway, the WF values were also calculated using those values: $WF_{grey}$ with the average α value is 33.28 $m^3/t$, whereas it is 27.64 $m^3/t$ with the average value of β, which does not differ significantly from that quantified using the site-specific β factor (26.85 $m^3/t$).
The grey WF was also calculated for a very high nitrogen concentration in horse manure, obtaining a much higher value, 112.99 $m^3/t$. However, the $WF_{grey}$ of the Spanish tomato "rainfed" production is more than 8 times greater (808 $m^3/t$), which can be explained by an excessive nitrogen fertilization rate in the Spanish cultivation. In particular, Chico et al. (2010) highlight how the Water Footprint strongly depends on the production system. In fact, Figure 20 shows how, despite the same amount of nitrogen applied to the soil, the production of "rainfed" tomato has a WF extremely greater than that of both irrigated and greenhouse production, due to the yield difference.

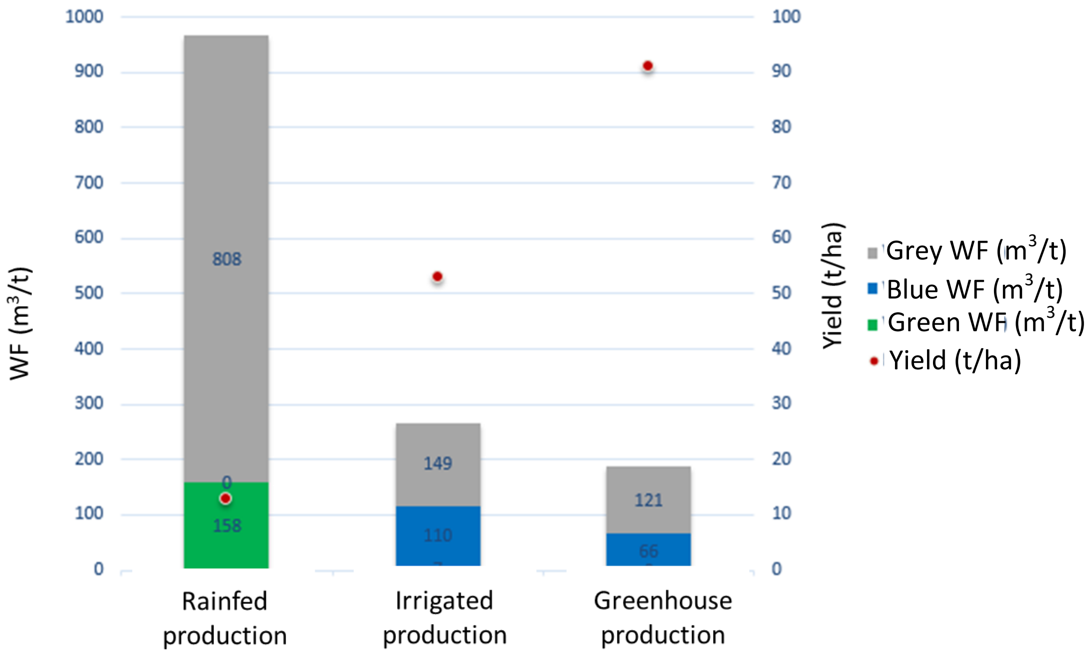



**Figure 20.** Comparison among the Green, Blue and Grey Water Footprint for open-air rainfed, open-air irrigated and greenhouse tomato production (Source: Personal elaboration adapted from Chico et al., 2010), which shows the relationship between the WF values and the type of production system.

Therefore, the variability of the grey WF is mainly due to differences in crop yield and the amount of nitrogen applied to the field. In order to reduce the nitrate load which reaches the water body, the soil nitrogen surplus should be minimized (Mekonnen and Hoekstra, 2013).

However, the reliability and robustness of the obtained WF values may be affected by the lack of appropriate site-specific data, thus increasing the results uncertainty (Evangelou et al., 2016). Therefore, the Water Footprint should be estimated on the basis of primary data, especially for the N content in horse manure, the type of soil and the actual agricultural practices.

## 10.5 Conclusions

The expected global population growth over the next few years, coupled with the growing demand for food, fibre and biofuels will cause increasing pressure on global freshwater resources. The WFN method can support a sustainable management of water resources by means of policy coordination at international, national and regional levels, because the formulation of strategies for the sustainable use of water has to be based on scientific, robust and reliable methods which evaluate the water consumption and the relevant environmental, social and economic impacts.

In this study, the WFN method was applied to the cultivation of Piennolo tomato, which is very resistant to water stress conditions, for the calculation of its WF. Despite this particular feature, the value of $WF_{green+ blue}$ is higher than the average WF of tomato production in Campania and this is due to the very low yield of Piennolo tomato. In fact, the PDO regulation sets the maximum yield ot 16 t per hectare, with a plant density not higher than 45,000 plants per hectare and forbids the use of greenhouses, which would extend the harvesting period thus increasing the overall annual harvested production. Nevertheless, the high value of $WF_{green+blue}$ does not depend on agricultural practices, since no irrigation can be applied and the $WF_{blue}$ is thus equal to zero. Therefore, no improvement measures cannot be implemented by the farm in order to decrease this value. It is noteworthy that this value, if used for environmental communication, could disadvantage this traditional cultivation if compared to industrial cultivation.

On the contrary, because of the moderate application of nitrogen fertiliser, which is slightly higher than the actual nitrogen requirement of the crop, the $WF_{grey}$ is slightly lower than the average Italian value and almost equal to the $WF_{grey}$ of Greek industrial tomato, which has 8 times greater yield. However, it is noteworthy that several assumptions were necessary for the calculation of the $WF_{grey,}$ due to lack of primary data from the farm, such as the nitrogen content in horse manure and the natural concentration of nitrogen in water bodies, but, in contrast with literature studies, it was decided to use a detailed approach for the choice of the leaching/run-off fraction. In fact, the site-specific leaching/runoff fraction β, which considers the crop's nutrient uptake was used, whereas literature studies used the fixed leaching/runoff fraction α (10%), which does not consider the soil properties and the agricultural management practices.

In this work only the first and second phases of the WFN method were applied, whereas the assessment of the environmental, social and economic sustainability and subsequent formulation of the WF reduction strategies were not carried out due to lack of information for determining the water availability of the river basin and also because, according to Hoekstra et al. (2011) and Quinteiro et al. (2018), those phases are still in development.

Another critical issue of the method is the lack of precise and internationally recognised technical and methodological rules which could result in more reliable and robust WF results. On the contrary, the study on Piennolo tomato production was based on several assumptions and, since site-specific data were not always available, literature data were used for some important issues, such as the nitrogen content in manure, the water legislation in the studied area and the natural concentration of nitrogen in the water body. Therefore, the reliability of the results obtained are affected by these assumptions and the data used.

The application of the method also highlighted how the WF, in the same way as LCA method, does not account for the quality of local extensive production, thus penalizing niche products, which nevertheless have a deeper link with the local territory, in comparison to industrial ones.

In conclusion, the calculation of the WF of Piennolo tomato production has evaluated the use of green, grey and blue water resources, providing a first step towards the identification of improvement options for more sustainable use of water in this crop cultivation. On the other side, the WFN method applied to Piennolo tomato cultivation has highlighted only the water-related aspects. Therefore, in order to obtain a comprehensive picture of the overall performance of this crop cultivation, WF should be combined with other environmental, economic and social tools, such as Carbon Footprint or Life Cycle Costing, which could support the individual producer in the transition towards sustainable production and consumption models in its supply chain.

# 11 Conclusions

Environmental life-cycle based methods and tools, such as ISO Life Cycle Assessment (LCA) method and Water Footprint (WF) can be used for the assessment of the impacts and benefits associated to circular solutions in the agri-food supply chain, avoiding burden shifting from a phase to another of the life cycle and from an environmental compartment to another one. Moreover, product environmental labels, which are based on this kind of methods, represent an important marketing opportunity for agri-food companies, both in a B2B and B2C communication perspective.

In particular, ISO LCA method has been increasingly applied in the last years to agri-food production chain. Nevertheless, an important obstacle to a wider use of this method is represented by the methodological problems deriving from the specific features of the agri-food supply chain, which the practitioner has to deal with when performing an LCA study.

The main technical and methodological issues have been identified and discussed in this dissertation thesis by means of a critical literature review, focusing on the identification of functional unit and system boundaries, the emissions from the use of fertilisers and pesticides and the allocation procedures necessary to deal with multifunctional processes. The different approaches found in literature to solve those problems were described, highlighting that they can lead different life cycle results, making it difficult to compare the environmental impacts of products of the same category, when LCA is used for communication purposes.

In this dissertation thesis, special focus was given to the calculation of on-field emissions from the use pesticides at inventory level, which is frequently one of the main methodological problems in LCA studies of food and drink products. For this purpose, the detailed PestLCI 2.0 model, which considers the climate and soil data of the studied region, was applied to the production of maize in an experimental farm in Northern Italy, with the aim to evaluate the distribution of pesticides among the environmental compartments, obtained using different types of soils. Results showed that little variations in soil characteristics can lead to great variation of PestLCI 2.0 outcomes, especially for groundwater emissions, since they are strictly related to soils features. On the contrary, emissions to air are dominated by meteorological conditions and pesticide physical and chemical properties, while emissions to surface water are dominated by wind drift, and are completely independent from soil characteristics. Moreover, the application of PestLCI 2.0 with soil specific data has pointed out that detailed information on soil characteristics and – more importantly – their interpretation are necessary, thus requiring expertise in soil science.

The use of PestLCI 2.0 in LCA studies of agri-food products, and more in detail of specific soil data within the model, therefore results in the availability of a comprehensive set of emission data in the different compartments, which is an important input for the inventory phase of LCA studies and can increase their robustness. Nevertheless, whether this high-resolution and resource-intensive data collection is worthy for the robustness of LCA results, depends also on the capability of characterization models applied in the life cycle impact assessment phase to capture them. At the moment, USEtox model, recommended by the PEF method for the toxicity-related impact assessment categories, does not address the environmental impacts due to groundwater emissions. Further research efforts are therefore needed to develop characterisation factors for groundwater emissions, in order to exploit the detailed results of PestLCI 2.0 in the impact assessment phase. The combination of detailed inventory data provided by PestLCI 2.0 and characterisation factors for groundwater emission would result in a comprehensive evaluation of pesticide emissions at inventory level and of their environmental impacts.

As regards the use of LCA for communication purposes, the PEF method and more in detail the Product Environmental Footprint Category Rules (PEFCR) for dairy products were tested (PEF study on Taleggio cheese production), with the aim to evaluate if they properly respond to the harmonisation needs for the calculation and communication of the environmental performance of food and drink products. It can be highlighted that the PEFCR for dairy products provides sufficient and quite detailed guidance for the definition of the functional unit and system boundaries as well as for the allocation procedures. On the contrary, additional guidance should be given for the calculation of both on-field fertilisers emissions and enteric fermentation from animals. In fact, the document provides only a long list of emissions to be accounted for in the study, and the name of the relevant literature model to be used as minimum requirement, but it does not provide any detail about the equations or the emission factors to be applied. The approach towards the calculation of pesticides emissions is very simplified, considering only the total amount of pesticide emitted to the soil. However, PestLCI 2.0 is quite a complicated and resource-intensive model, which is suitable for scientific purposes, but not for a quick and easy application aimed at the communication of the environmental impacts of products.

The PEF study on Taleggio cheese showed that the application of the PEF method is quite difficult and time-consuming, especially regarding the calculation of data quality requirements and data quality rating and the use of Circular Footprint Formula for the End-of-Life stage. Therefore, the application of this method seems not so quick and straightforward as it is expected to be, if the goal is to involve many companies in Europe, especially SMEs. The future availability of the PEF compliant life cycle inventory datasets, currently under preparation by the European Commission, could probably contribute to simplify the modelling phase of the PEF study, although the whole method might remain quite resource-intensive. Moreover, supporting guidance or simplified tools for the emissions calculation could be useful for both LCA practitioners without specific knowledge of the agricultural sector and for SMEs, and could in this way encourage the use of the PEF method in different types of companies.

Finally, due to the significant contribution of agricultural sector to water use and consumption, the WF method of the WFN was tested in a PDO Italian tomato cultivar in Campania, with the aim to assess the strengths and weaknesses of the method as well as its practicability.

The WFN method calculated the use of green, grey and blue water resources, thus providing a first step towards the identification of improvement options for more sustainable use of water in this crop cultivation. However, the study required great effort for both identifying and collecting the necessary data as well as for the calculation procedures. Green WF due to the evapotranspiration from the crop-soils system was calculated with the help of a detailed FAO’s software, using a combination of primary data from the company and default literature data. The calculation of grey WF was carried out through a detailed approach for the choice of the leaching/runoff fraction, considering also soil properties and agricultural management practices, in contrast with other literature studies, which used a fixed value for this fraction. Nevertheless, several assumptions were necessary for the calculation of the $WF_{grey}$, due to lack of primary data from the farm, such as the nitrogen content in horse manure and the natural concentration of nitrogen in water bodies. Therefore, the use of site-specific data could increase the overall reliability of WF results.

The application of the method also highlighted how the WF, in the same way as LCA method, does not properly consider the quality of local production, which have a deeper link with the local territory, in comparison to industrial ones. In fact, despite the advantage given by the absence of irrigation (i.e. $WF_{blue} = 0$), the low Piennolo tomato yield, fixed by the PDO regulation, leads to a $WF_{green+\ blue}$ higher than the average WF of industrial tomato production in Campania.

Moreover, it is noteworthy that the methods for the calculation of the WF are still evolving, since they have been introduced only in the last recent years. The development of ISO 14046 in 2016 might be a step towards the establishment of a common WF method, although both methods have created a lively discussion in the scientific community and have been subjected to mutual criticism. Their scientific basis is indeed different: the WFN method accounts for a volumetric measure of the green, blue and grey water use, and is focused also on the quality of water, whereas ISO 14046 evaluates environmental impacts due to the use of blue water because impact assessment methods have been developed so far only for blue water use. As regards environmental impacts related to green and grey water use, they are included in other environmental indicators in LCA, such as land use and eutrophication, although some tentative green water scarcity indicators have been developed in literature.

In the next future, further research efforts should be focused on their improvement and on the identification of actual synergies. For example, in the recent years the WFN method has tried to develop indicators for blue and green water scarcity and water pollution, although affected by both lack of data at global level and some methodological problems. As regards the WF based on ISO 14046, future improvements should be focused on the following aspects: assessment of environmental impacts related to changes in evapotranspiration; development of inventory data for blue water consumption in agriculture, which can be based on actual measurements at farm (often difficult to be recorded) or on the evapotranspiration associated to irrigation; development of spatial and time-related characterisation factors as well as their connection with inventory flows. Despite those current methodological problems, both methods could support governments, companies and individual producers, such as farmers, in the identification of water-related environmental hotspots and improvement options and more in general in the sustainable and efficient use and management of water resources in the whole production chain. Moreover, the development of both a common accepted Water Footprint method and relevant supporting data could be useful for the implementation of green marketing strategies addressed to consumers. In fact, it could encourage the development of transparent and reliable WF labels, which would provide consumers with information about the product’s embedded water, thus helping them to adopt more sustainable and equitable consumption choices.

Finally, since WF covers only water-related aspects, other tools are always needed in order to obtain a comprehensive picture of the environmental, economic and social performance of products and production processes, thus encouraging the transition towards sustainable production and consumption models in the agri-food sector.

Although affected by some methodological problems, which should be addressed in the next future, environmental life-cycle based methods are effective in supporting the transition towards circular economy models in the agri-food sector. Sharing with the circular economy the perspective of considering the product’s life cycle as a whole, they can highlight any negative consequences due to a particular configuration and allow business and decision makers to choose the solution with the lowest environmental impacts. Finally, they can be used for the development of reliable and transparent communication tools, such as environmental labels, addressed to both business and consumers.

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# ANNEX 1

This Annex includes an example of an electronic spreadsheet developed during the PhD, which can be used for the calculation of enteric fermentation emissions from livestock for the execution of a PEF study on dairy products.

| | | | | | |
|---|---|---|---|---|---|
| **CALCULATION OF ENTERIC FERMENTATION EMISSION OF THE DAIRY FARM'S HERD FOR ONE YEAR: CHAPTER 10.3 OF IPCC 2006 GUIDELINES FOR NATIONAL GREENHOUSE GAS INVENTORIES** | | | | | |
| **Dairy Farm XY** | | | | | |
| *Comment: calves were not considered because they do not have enteric emissions (see IPCC_2006_Ch10, pagina 30: "A CH4 conversion factor of zero is assumed for all juveniles consuming only milk (i.e., milk-fed lambs as well as calves)).* | | | | | |
| | | | | | |
| **Class of animals** | **Lactating cows** | **Dry cows** | **Heifers** | **Calves** | NOTES |
| **Number of animals** | 157 | 22 | 144 | 53 | Number of animals divided per category. |
| **Weight** (kg) | 650 | 650 | 400 | 230 | Weight of each animal category. |
| | | | | | |
| **Fat and Protein Corrected Milk (FPCM) production of the dairy farm** (kg/yr, with 3,6% fat and 3,5% protein) | 1,754,336.97 | 0.00 | 0.00 | 0.00 | Amount of fat and protein corrected milk produced at each farm. |
| **Milk** (kg/day) | 30.61 | 0.00 | 0.00 | 0.00 | Daily milk production for each farm. |
| **Fat** (%) | 3.60 | 0.00 | 0.00 | 0.00 | Milk fat content |
| | | | | | |
| **$Cf_i$** | 0.39 | 0.32 | 0.32 | 0.32 | It varies for each animal category (lactating/non-lactating). |
| **$C_a$** | 0.00 | 0.00 | 0.00 | 0.00 | It corresponds to animal's feeding situation (stall = 0). |
| **C** | 0.80 | 0.80 | 0.80 | 0.80 | It corresponds to 0.8 for females, 1.0 for castrates and 1.2 for bulls. |
| **WG** | 0.00 | 0.00 | 0.40 | 0.30 | It is the average daily weight gain of the animals in the population, kg day-1 (see Tab. 10A.1 |

| | | | | | |
|---|---|---|---|---|---|
| | | | | | and 10A.2 of IPCC 2006 guidelines). |
| **$C_{pregnancy}$** | 0.10 | 0.10 | 0.00 | 0.00 | It is the pregnancy coefficient (see Tab. 10.7 of IPCC 2006 guidelines). |
| **DE%** | 70.00 | 70.00 | 60.00 | 65.00 | It is the digestible energy expressed as a percentage of gross energy (see Tab. 10A.1 and 10A.2 of IPCC 2006 guidelines). |
| **$Y_m$** | 5.50 | 5.50 | 5.50 | 5.50 | It is the methane conversion factor, per cent of gross energy in feed converted to methane. Its range is 6.5% + 1.0%. When good feed is available (i.e., high digestibility and high energy value) the lower bound should be used. |
| **Energy content of methane** (MJ/kg $CH_4$) | 55.65 | 55.65 | 55.65 | 55.65 | Fixed value according to IPCC 2006 guidelines. |
| | | | | | |
| **$NE_m$=** | 49.69 | 41.45 | 28.80 | 19.02 | Net Energy for maintenance (from IPCC 2006 guidelines) |
| **$NE_a$=** | 0.00 | 0.00 | 0.00 | 0.00 | Net Energy for activity (from IPCC 2006 guidelines) |
| **$NE_g$=** | 0.00 | 0.00 | 6.62 | 3.19 | Net Energy for growth (from IPCC 2006 guidelines) |
| **$NE_l$=** | 89.09 | 0.00 | 0.00 | 0.00 | Net Energy for lactation (from IPCC 2006 guidelines) |
| **$NE_{work}$=** | 0.00 | 0.00 | 0.00 | 0.00 | Net Energy for work - NOT RELEVANT (from IPCC 2006 guidelines) |
| **$NE_{wool}$=** | 0.00 | 0.00 | 0.00 | 0.00 | Net Energy to produce wool - NOT RELEVANT (from IPCC 2006 |

| | | | | | |
|---|---|---|---|---|---|
| | | | | | guidelines) |
| **$NE_p$=** | 4.97 | 4.15 | 0.00 | 0.00 | Net Energy for pregnancy (from IPCC 2006 guidelines) |
| **REM=** | 0.53 | 0.53 | 0.49 | 0.51 | Ratio of net Energy available in a diet for Maintenance to digestible energy consumed (from IPCC 2006 guidelines) |
| **REG=** | 0.33 | 0.33 | 0.28 | 0.31 | Ratio of net Energy available in a diet for Growth to digestible energy consumed (from IPCC 2006 guidelines) |
| **GE** (MJ $head^{-1}$ $day^{-1}$)= | 388.28 | 123.16 | 136.70 | 72.84 | Gross Energy |
| **EF** (kg $CH_4$ $head^{-1}$ $yr^{-1}$)= | 140.07 | 44.43 | 49.31 | 26.28 | CH4 Emission Factors for enteric fermentation from a livestock category |
| **EF of the class** (kg $CH_4$/yr)= | 21990.44 | 977.45 | 7100.85 | 1392.63 | |
| **EF of the herd** (kg $CH_4$/yr)= | 31461.37 | | | | |

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