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ANALYSIS OF THE WATER FOOTPRINT OF AGRICULTURE PRODUCTS AND THE RELATED MITIGATION STRATEGIES

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ANALISI DELL'IMPRONTA IDRICA DEI PRINCIPALI PRODOTTI AGRICOLI E RICERCA DELLE STRATEGIE DI MITIGAZIONE DELL'IMPATTO

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Summary

Agriculture is the major player of human appropriation of water resources. About 70% of global freshwater withdrawals are used for irrigation to sustain global crop production. In fact, irrigated areas account for 18% of global croplands which contribute for 40% of global food production. In addition, the 40% of the global irrigation practice is unsustainable because it depletes environmental flows and/or groundwater stocks. In order to preserve the water resource, the European Water Framework Directive lead to an efficient use of freshwater and it gives guidelines to preserve the qualitative natural status of water bodies. To achieve this objective, the impact assessment needs suitable indicators to be computed. The use of an environmental indicator of water shortage can enhance better the decision-making process for water saving and leading a reduction on water stress from human activities. This thesis integrates the concept of the water footprint with other environmental indicators to analyse sustainability and comparing the related performance of different agricultural practices. The main objective of this thesis is to find solutions to mitigate the impact of agricultural crop production on water resource identifying best practices quantifying the water footprint and water-related indexes. To do that, we combined different indicators of water use improving a sustainability framework for different agricultural products and field management. The first step was to classify and making a clustered analysis of agricultural products identifying their specific environmental performance and their related water footprint. Then, the definition of a framework for sustainable water use in the agriculture sector was done, considering both environmental and economic aspect of sustainability. The second step focused on the definition of best agricultural practices for a sustainable water consumption comparing different soil and field management. Results were supported by different case studies that promoted innovative solutions to mitigate water footprint and the impact on water resource by improving the agricultural production system; this can be improved by encouraging awareness and sensitivity of farmers to some ecological initiatives.

Sommario

L'agricoltura svolge un ruolo importante nello sfruttamento della risorsa idrica. Si pensi che circa il 70% del consumo di acqua a livello globale viene utilizzata per scopi agricoli, primo fra tutti l'irrigazione per la produzione di colture che potranno essere consumate sia per l'alimentazione umana, sia per quella zootecnica. A livello mondiale, le aree irrigate rappresentano circa il 18% del totale delle superfici coltivate, contribuendo per il 40% alla produzione mondiale di cibo. Se è pur vero che la pratica irrigua crea un beneficio all'agricoltura perché ne aumenta la produttività, è vero anche che a conti fatti circa il 40 per cento dell'acqua consumata a livello mondiale per scopi irrigui viene utilizzata in maniera del tutto insostenibile. Questo perché la sottrazione di volumi d'acqua all'ambiente comporta una riduzione dei deflussi ecologici e/o delle riserve d'acqua sotterranea, in particolar modo si pensi alle riserve di acqua fossile. La Direttiva Quadro sulle Acque (2000/CE/60) consente solo l'uso efficiente delle acque superficiali e di falda e fornisce delle raccomandazioni su come preservare lo stato naturale qualitativo dei corpi idrici. L'utilizzo di un indicatore ambientale, quale l'indicatore di impronta idrica, fornisce informazioni utili e volte a migliorare il processo decisionale per il risparmio idrico e a ridurre lo stress idrico causato dalle attività umane. L'obiettivo principale di questa tesi è quello di trovare soluzioni utili a mitigare l'impatto della produzione agricola sulla risorsa idrica individuando pratiche agronomiche più sostenibili. A tal fine, lo scopo generale dello studio è quello di integrare il concetto di impronta idrica con altri indicatori ambientali per confrontare le relative prestazioni di diversi processi agricoli valutando il loro grado di sostenibilità. Dapprima, si è cercato di classificare e raggruppare le famiglie dei prodotti agroalimentari in base alle loro specifiche prestazioni ambientali e la relativa impronta idrica per definire la sostenibilità della loro produzione e di conseguenza del loro consumo. Successivamente, è stato definito uno schema per lo studio della sostenibilità dell'uso dell'acqua in agricoltura, valutandone sia l'aspetto ambientale che quello economico. In secondo luogo, sono state individuate pratiche agricole che riducessero l'impronta idrica durante il processo produttivo e favorissero una maggiore salvaguardia della risorsa idrica. A supporto, diversi casi studio sono stati analizzati per valutare diverse gestioni del suolo, delle fasi agronomiche in campo e del consumo di acqua con maggiore attenzione alla pratica irrigua. Le soluzioni fornite attraverso i risultati della tesi sostengono soluzioni innovative per mitigare l'impronta idrica e l'impatto dell'attività agricola sulla risorsa idrica. Questo può ancora essere migliorato ed incentivato promuovendo la sensibilizzazione degli agricoltori alla gestione sostenibile dell'agro-ambiente e incoraggiando il mercato alimentare a premiare tali iniziative ecologiche.

1. Introduction

1.1 State of the art

Water demand will increase up to 70-90% within the next three decades (Rosegrant et al., 2002), worsened by growing food demand and the increasing population (Deurer et al., 2011), which requires improvements on life standards (D'Odorico and Rulli, 2013; Davis et al., 2016). If this trend will persist, feeding more people will require more irrigated lands (Brown and Matlock, 2011), resulting in a more intense agriculture at the expense of the natural capital and the environment (Godfray et al., 2012). At the meanwhile, the water consumption per unit of area needs to be reduced with respect to an efficient water management and environmental protection (D'Odorico et al., 2018; Playán and Mateos, 2006).

The European Union implemented strategies (Common Agriculture Policy) and proper mitigation policies (Water Framework Directive) improving the management of water resources, especially where water scarcity mostly affects water availability during irrigation season. A combination of efficient irrigation practices and innovative agronomic techniques can lead to a sustainable water management reducing water consumption, preserving water quality, and having a resilient system from climate change (Ciafani et al., 2008). Those principles are important once they are contextualized on water exploitation, especially for the impact on water resource. Such impact could be represented as the human footprint on water resource, which is translated in the concept of water footprint (WF), introduced by Allan et al. (1998) and elaborated by Chapagain and Hoekstra (2004). A more elaborated concept about water footprint is the virtual water, coined by Allan (1990), which involves the embedded water inside commodities traded around the world (Antonelli and Sartori, 2015). The virtual water connects the water consumption with the economic price of the commodity (with different cost of opportunity) (Wichelns, 2015), while the water footprint puts in relation the volume of water consumption with the crop production. The water consumption is also defined as the total water quantity used to produce goods and services, which can be evaporated and consumed along the production chain, and qualitative, as water virtually polluted per unit of time (Hoekstra et al., 2011). In terms of agricultural products, the quantitative components of the water footprint are divided in "blue" and "green" water resource. The blue water resource is the freshwater consumption (e.g. through irrigation) from surface and groundwater along the food chain which cannot return in the same catchment or it will return after a long-term period. The green water is the part of rainwater used by plants through transpiration and/or evaporated during the crop growing season. The grey water is the volume of freshwater required to assimilate the discharge of pollutants into a water body. It is carried out using the natural background concentrations and the existing ambient water quality standards of the receiving water body provided by national legislation in the regional agreements (EFSA, 2017). The grey WF is often studied as an independent indicator, thanks to its important and of high concern discipline. A guideline for the water quality indicator evaluates the hazard of pesticide dispersion and the risk for human health and environment (Franke et al., 2013). Several authors provided studies on the grey WF assessment. For example, Mekonnen and Hoekstra (2015) described the global grey WF for different sectors. They evaluated the nitrate load in the freshwater catchment and related to the water pollution level for each basin and providing a metric to understand which crop and which country have the higher contribution on water pollution. Chukalla et al. (2017) focused on grey WF estimation for different nitrogen application rate, nitrogen forms, tillage practices and irrigation strategies considering a maize crop cultivation. Differently from the aforementioned methodology, but considering the same concept of volumetric grey WF, Lamastra et al. (2014) calculated a dilution factor dividing the Predicted Environmental Concentration of contaminants by the toxicity concentration using the minimum level of No-observed effect concentration for fish, algae, and Daphnia on water contamination (Padovani et al., 2004; Trevisan et al., 2009).

Water footprint is often expressed as an indicator of water shortage and water appropriation, linking the water consumption with the local water availability to give knowledge about local water scarcity. In 2012, Hoekstra et al. (2012) mentioned the Water Footprint Sustainability describing the blue water scarcity relating the blue WF with the water availability for 201 river basins. In Vanham and Bidoglio (2013), they studied the Water Footprint for the 28 European countries, considering the blue and green WF and they explained the importance of the index in combination with other analytical means or indicators when determining integrated policy options. Furthermore, WF accounting is a measure of the sustainable water use, linking the economic costs and environmental benefits per unit of volume of water consumed for each innovative process of water saving or impact reduction (Hamdy et al., 2003). One example is the use of such innovations of water savings, like the recycling wastewater plant or an efficient irrigation system, which require investing extra money for the farm or the factory, while they give a benefit for social, economic and environmental externalities at the global scale (UNEP, 2012, 2010). Moreover, the relation between a water footprint and the water availability (water withdrawals in a basin from precipitation, streamflow and ground flow) is useful to define the water scarcity and to understand the magnitude of the local water consumption (Veettil and Mishra, 2016). Water availability is a hydrological term that refers to the water readily available to the access for human activities. It varies around the world and depends on both precipitation and evapotranspiration. Moreover, it can be the technically and economically available from the water extraction for a process, or it can be the only water technically available considering the water needed for the ecosystem function maintaining (Fekete et al., 2002). The need of determining an index of Water Footprint Sustainability is required to describe water scarcity as the relation between the human impact and the natural depletion (Davis et al., 2016; Hogeboom and Hoekstra, 2017). The water scarcity might be a physical water shortage due to the freshwater availability, or a lack on water access due to the water supply to users, or a lack of infrastructures that makes a water shortage in the area. In this case, a country could have a good water availability, but there is a weakness to maintain and manage the water supply with the infrastructure, or water is not homogeneously distributed (CGIAR, 2011).

According to the recent scientific improvements on the concept of water footprint, it is possible to distinguish two main scientific communities: the “Water Footprint Assessment” (WFA) community, which reflects the concept conceived on the “Water Footprint Assessment Manual” (Hoekstra et al., 2011), and the “Water Use Life Cycle Assessment” (WULCA) community that analyses different indicators of a “Water Footprint method” expressed in the guideline ISO 14046 (Boulay et al., 2013). The WFA method evaluates an indicator for water resource management and it consists of four steps: the goals and scope, the accounting, the sustainable assessment and the response formulation. The goal and scope for the water footprint (WF) accounting may have different purposes; therefore, it is important to specify which address the water footprint focuses. For example, the water footprint can be analysed for a product or a process, at consumers or at national level. In 2011, Mekonnen and Hoekstra proposed a national WF accounting, including the blue, green and grey WF where about 92% of the water footprint was related to the consumption of agricultural products, 5% to the consumption of industrial goods, and only the 4% to domestic water use. A first database of 127 primary and derived agricultural products at global and national scale at 5 to 5 arc minute grid was presented by Mekonnen and Hoekstra (2010) considering the period 1996-2005. They found out that a global average of WF per ton of crop is for example 200 m^3 ton^{-1} for sugar crops, 300 m^3 ton^{-1} for vegetables, roots and tubers 400 m^3 ton^{-1}, fruits 1000 m^3 ton^{-1}, cereals 1600 m^3 ton^{-1}, oil crops 2400 m^3 ton^{-1} to pulses 4000 m^3 ton^{-1}. Subsequently, Mekonnen and Hoekstra (2012) assessed the WF for livestock. This study showed that animal products from grazing systems had a smaller blue and grey water footprint than animal products from industrial systems. Different water footprint values were found for different animal products; milk had an average WF of 1000 m^3 ton^{-1}, eggs had an average of 3300 m^3 ton^{-1} and beef of 15400 m^3 ton^{-1}.

Considering the water footprint of consumers, the WF assessment calculates the summary of the single WF of products consumed by people (Hoekstra and Mekonnen, 2012). For example, in Vanham and Bidoglio, (2014) two different diet scenarios were defined using the single WF for 365 river basin with regards the consumption of agricultural products in Europe. They described a vegan and a healthy diet for European citizens considering the three components (blue, green and grey) of the WF as the import/export of virtual water. Recently, several authors started to evaluate the WF with the WFA methodology, especially for agriculture products. In this contest, for example, Zhao et al. (2014) calculated the WF of agriculture sector in China. They explored the effects of population, diet structure, urbanization and affluence on the Chinese agricultural water footprint from 1990 to 2010 using different models to examine the economic and social driving force on the water resource. The WF may also evaluate the impact at the national or at the River Basin level. For example, Chapagain and Hoekstra (2004) gave an idea of the total water footprint of nations, assessing the virtual water for the agricultural, industrial and domestic sector.

In parallel, the “WULCA” community developed the WF method for LCA users. Many research groups are working with the LCA method, as the UNEP-SETAC group that proposed guidelines in the WF LCA-based computation. LCA is a standardized and internationally recognized methodology to assess the environmental impacts of a product or service from a “cradle-to-grave” approach and standardized by the ISO 2006. Normally, LCA assesses a range of impact categories with a multicriteria method and it consists in four steps: setting goals, inventory analysis, environmental impact assessment and interpretation of results. In each stage, all the input (production factors) and output (emitted substances) resources of the process are put in an inventory and converted in an environmental impact for the midpoint evaluation step. Then, all the indicators of impacts are aggregated at endpoint level to evaluate the impact on the area of protection of Human health, Ecosystem quality and Resources.

The “WULCA” (Water Use in LCA) is a working group that provides practitioners, consistent frameworks to assess, compare and disclose the environmental performance of products and operations regarding freshwater uses and that influenced the draft of the ISO 14046. The ISO 14046 defines the WF as a multicriteria metric that quantifies the potential environmental impacts related to water (ISO, 2014), and it is represented by different impact categories of water related impact, as for example the Water Stress Index (Pfister et al., 2009). It also includes parameters of water quality and water consumption. However, the WF standard does not use the distinctions of the components as in Hoekstra et al. (2011) but refers to the hydrological nature of water flows. It defines actual water types as groundwater, surface water, brackish water, seawater, fossil water and precipitation in relation to the water cycle and hydrological mechanisms. In the case of water quality, the LCA method considers multiple indicators of water degradation as eutrophication, acidification and eco toxicity.

The computation of freshwater appropriation considers a water consumption weighted on the water availability of the area (Pfister et al., 2009) or evaluating a Water Scarcity Footprint (Boulay et al., 2017). The component of water consumption from rainwater (stored in the soil profile) received poor attention (with respect to the WFA methodology) because WULCA assumes that green water does not have direct effects on water scarcity/availability, and it is considered less environmentally relevant from a pure water consumption perspective. At the middle point level, the water consumption is generally characterized by a Characterization Factor that convert the water inventory flow into impact (Jefferies et al., 2012; Pfister et al., 2009). The mid-point indicator of water consumption represents the real pressure on water resource but considering the problem from which the impact starts and not on the damage itself (Fekete et al., 2002). The final evaluation, at the end point level (damage), addresses the impact of water depletion and water degradation to different Area of Protections, as for example to Human Health, including the effects of disease or malnutrition; to the Ecosystem addressing the damages on aquatic and terrestrial water degradation; and to Natural Resources considering the lack of renewing water in the same catchment. The result of the WF analysis with the LCA is not a volumetric measure of the water consumption, but is a weighted volume of water consumed in a specific area along with the supply chain and defined as liters (L) of water equivalent (H_2O_{eq}) (Pfister et al., 2009). The unit of measure derives from the calculation of the water stress index (WSI) using a characterization factor, where the water consumption is put in relation with water availability that gives a comparable value of water footprint between space and time (Jeswani and Azapagic, 2011). A first case study was related to the WF evaluation of two different food products considering the green, blue and grey WF (from the WFA method) and giving a stress-weighted WF in relation with the characterization factor of WSI (Ridoutt and Pfister, 2010). Ridoutt and Hodges (2017) described the WF of Australian milk production. They compare the Water Scarcity Footprint with different characterization factors, using two WSI and the new AWARE method (Boulay et al., 2017). Another implication of the WF assessment is to compute the freshwater ecosystem service. Furthermore, the European Commission worked to use those indexes to compute the freshwater ecosystem service footprint, which is a water use related to the water availability and the population of the area (European Commission, 2010). This water footprint is comparable with others indicators of LCA of social and economic sustainability (Ridoutt and Pfister, 2010), or with other environmental indicator such as the Carbon Footprint. In fact, the aim of a LCA is a multivariate analysis using a compound of indicators and determining the evaluation of sustainability for products or processes (Ridoutt and Pfister, 2013).

Therefore, there is the need to compare and combine different footprints in a “Family group” (Galli et al., 2012; Kissinger and Dickler, 2016). The aim of a Footprint Family is to cover a wide-enough spectrum of policies and to provide a metric of the sustainability for goods production and consumption (Perry, 2014). The Environmental European Agency (EEA) for policy adopted the use of a compound of indicators to facilitate the transformation of the current European food consumption and production system towards greater resilience and sustainability for: (1) consumption, (2) technological innovation and (3) organisational innovation (European Commission, 2011). There is not a standard on the definition of sustainability, while there are only some declarations (Dietz and Neumayer, 2007). Indicators must provide information about the main characteristics that affect the suitability of products and processes to be sustainable. The carbon, ecological, and water footprints (CF, EF, and WF) are the three important indicators most used for environmental issues, which have been recently grouped into a “footprint family” (Chaudhary et al., 2018; Galli et al., 2011; Herva et al., 2011). They are considered as a suite of indicators to track human pressure on the planet from different perspectives of environmental sustainability (Galli et al., 2011). An overview of the definitions and units of measurement associated with environmental, social, and economic footprints is also important since the definition of footprints varies, and it is often unclearly expressed. The LCA provides several environmental footprints, but rarely addressing deepening to an economic or a social indicator. In order to evaluate the sustainability with a compound of indexes, it is necessary to make a clear unit of measure for a sustainable system involved in a multi-criteria optimization. For example, the Barilla Company developed a water use sustainability index using the WF indicator in combination with the CF and the Ecological Footprint (Antonelli and Ruini, 2015). The WF is also used together with the CF and the nitrogen footprint and included in a Ecolabel (Leach et al., 2016). Four different ecolabel were developed in a simple and understandable label of footprint in order to guide consumers to an aware choice. Finally, the use of multicriteria indicators for sustainability analysis is applied carefully to express an overview of the environmental sustainability or for the whole aspects of sustainability. In fact, many initiatives promoted the use of ecolabel with the quantification of sustainability through a compound of environmental indicators to evaluate the impact of wine (Bonamente et al., 2016; Corbo et al., 2014). The family Footprint communicates the consumer’s responsibility as drivers of goods consumption and environmental pressure. For this reason, consumers have to be more aware of a sustainable food consumption. Steen-Olsen et al. (2012) analyzed the environmental footprint quantifying the blue WF, the CF and the land Footprint of 28 EU countries and the displacement of the environmental pressure within the food trade. The average carbon footprint at EU country scale was greater than that at worldwide. They showed also the trade network and compared the relationship among the EU countries; richest countries, usually, had a high efficiency, where goods production gained a higher CF score per unit of raw material used, also if they have a high level of consumption. Finally, the structural and natural differences affect the resource intensities and that does not depend on differences in economic efficiency of country. It is hard to evaluate a single measure from the core of indicators because the methodology must be the same and there is no threshold where each footprint indicates the level of sustainability (Čuček et al., 2012; Gu et al., 2016). For this reason, a broad scope approach can better inform the stakeholders, accounting a huge range of implications in different sectors (Tillotson et al., 2014). To monitor the humanity pressure, it is still proposed to integrate the water footprint in an appropriate set of indicators more appealing to measure the sustainability. In the context of the evaluation of the environmental impact, the analysis of life cycle is increasingly adopted, and different initiatives have been already started. In order to determine and guarantee the sustainability of a product, an Environmental Product Declaration (EPD) (Antonelli and Ruini, 2015) evaluates an independent, verified and registered document that communicates transparent and comparable information about the life-cycle environmental impact of products. The European Commission proposed a Recommendation (2013/179/UE) for the methodology to be applied during the Life Cycle Assessment of sustainability for a product or an organization. The Product Environmental Footprint (PEF) was developed by the Europe 2020 for an “efficient resource consumption”, which must follow the International Standard Organization (ISO) requirements expressed in ISO 14025 in order to certificate goods or products based on a scientific recognized methodology. PEF has the advantages to collect data from primary source without depending on secondary data from database as for example for the LCA assessment. Nevertheless, the more spread method to evaluate the environmental footprint of a product is the LCA approach that considers many different indicators for many different compartments (air, water, soil, etc). In this thesis, the concept of WF was used in relation with other environmental indicators to compare agricultural practices and agricultural products. Recent research studies lack of knowledge on water footprint for agriculture field management, especially linked to others indicator evaluating sustainability of processes. Moreover, different methodologies for WF need to be combined in order to reach meaningful and exhaustive outcomes for WF reduction. To achieve a reduction on water footprint and enhance better sustainability, we analysed different mitigation strategies (best practices).

1.2 Research questions and objectives

The Water Framework Directive aims at an efficient use of freshwater and gives guidelines to preserve the qualitative natural status of water bodies, especially from agricultural activities, which are the main factors of impact on the water resource. This thesis integrates the concept of the water footprint with other environmental indicators to analyse sustainability comparing different agricultural practices. The main objective of this thesis is to find solutions to mitigate the impact on water resource identifying best practices using WF and water-related indexes to analyse a sustainable water use. It is notable that some agricultural practices, mentioned as best practices, are suitable for water saving and for prevention from water contamination. For this reason, three main research questions were addressed:

1) How does the use of water-footprint-related indexes can change or improve water sustainability?
2) Which approach is more suitable and robust to quantify the impact on water resource?
3) Which is the most appropriate agricultural practice for a better management of water resources in a specific environment and a specific climate context?

The real issue of the water footprint is to provide an index that can be compared with other indicators addressing environmental impact. Thus, the overall purpose of the study is to define a robust and useful approach to improve water sustainability for different agricultural products also addressing concepts like water scarcity, water footprint and water efficiency.

Some specific objectives are highlighted to achieve the aims of the project:

- Definition of a trade-off between water consumption and water availability allowing a sustainable evaluation of different agricultural products;
- Definition of specific classification or clustering schemes and methods, related to water impact for different agricultural products and environments;
- Definition of the best practices for a sustainable water consumption to understand where and how much water can be saved for different agricultural processes.

All these objectives might become tools to improve the agricultural production systems, to get environmental incentives and to open the doors to a sensitive market to the quality of the food productions and a conscious consumption.

1.3 Explanation of the thesis format

The thesis is structured on six chapters, each of them is a paper taking part of the PhD project. In figure 1, the chart of the thesis structure illustrates how chapters are divided according to the main topics and related research questions.

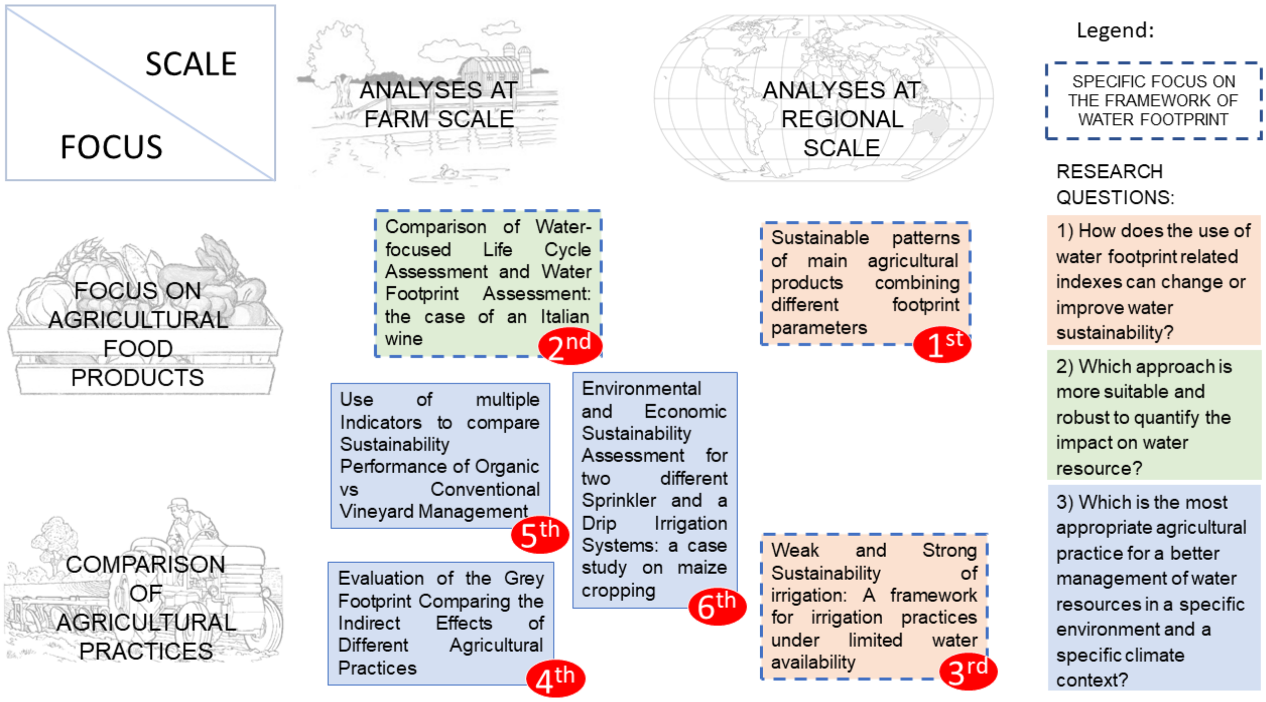

Figure 1. Chart of thesis structure including chapters divided for main topics and related research questions.

Chapter 2 is based on a first paper which has been published in the *Journal of Cleaner Production*; it aims to achieve the second objective showing a pattern of sustainability for different agricultural products and answer to the first research question. Chapter 3 relies on a second paper that has been published in the Journal of *Science of the Total Environment*; it compares two methodologies for WF from WFA and LCA communities achieving the second research question and the first objective. Chapter 4 shows a third paper submitted to the Journal of *Frontiers*, which aims to define a framework for a sustainable water use for irrigation purposes answering to the first research question and reaching the first objective. Chapter 5, which is based on a fourth paper published in *Sustainability* Journal; suggests suitable soil practices for water quality improvement, and it aims to answer to the third research question and respond to the third objective. The fifth (chapter 6) and the sixth (chapter 7) papers aim to answer to the third research question and achieve the third objective; the fifth paper has been accepted for publication to the Journal of *Science of the Total Environment*, while the sixth has been published to the Journal of *Agronomy*.

2. Sustainable patterns of main agricultural products combining different footprint parameters [1]

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2.1 Abstract:

The world population is increasing, and the human diet is becoming of considerable concern for human welfare. Furthermore, natural resources are overexploited, and governments need policies for the proper management of the environment. Sustainable agriculture can provide some solutions, as it minimizes inputs, wastes or pollution. The aim of the present study is to provide a combined analysis of different footprint approaches to allow comparison of different agricultural and livestock products regarding the efficiency of resource exploitation. Water consumption, greenhouse gas emissions and energy indexes are included in this study as footprint indicators. The study takes advantage of indexes collected from an extensive bibliography focused on different fresh agricultural products; the target is the definition of a timetable of footprints for agricultural products. Starting from a top-down perspective, an analysis of the environmental footprint for different products is an approach to understand which products can be more sustainable for the human diet. For this reason, this study distinguishes different clusters in sub-clusters of vegetable products and animal products. The comparison of the footprint indicators of water consumption regarding yield, greenhouse gas emissions equivalent, and energy provides a ranking of sustainability for a given product. Ultimately, this work seeks to propose an original pattern of food sustainability, allowing an adequate quantitative comparison of agriculture products for a more conscious human diet.

Keywords: clustering; classification; Water Footprint; Carbon Footprint; energetic content; sustainability

2.2 Introduction

The increasing urbanization and global population of two billion people for the next 50 years, the higher income expectations and changing for a better diet would make competition for food and rural demand, stressing lands, water, and resources to support the agricultural ecosystem (CGIAR, 2011). Moreover, climatic change events occur rapidly. In this context, the agricultural sector is also exposed to a profound climate change. The precipitation trend, with fast and intense rainfall events, toggles periods of droughts, especially during the crop growing season (Godfray et al., 2012). With respect to those global problems, decision makers and stakeholders have to promote policies to assure food production and encourage a sustainable management to reduce human impacts on the environment (D'Odorico and Rulli, 2013). For this reason, the Strategic Research Agenda 2006-2020 of the European Union is facing the greatest challenge to obtain healthy, secure

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[1] This chapter relies on the paper "Borsato et al., 2018. Sustainable patterns of main agricultural products combining different footprint parameters. J. Clean. Prod, 179, 357–367, doi:10.1016/j.jclepro.2018.01.044".

and environmentally sustainable food production from agro-industry (Fang et al., 2014; Pattara et al., 2017). Additionally, sustainable development and consumer awareness are supported by mutual European policies to improve and provide food preferences that follow reduced environmental impact. To promote a sustainable behaviour for environmental resource uses, the European Commission (EC) also developed environmental programmes focusing on an eco-label (European Commission, 2011; Galli et al., 2012; Leach et al., 2016). This programme of the European Commission performed a multi-criteria measure for the environmental performance of a product, called the Product Environmental Footprint (PEF). Therefore, a part of the research community is studying environmental impacts, focusing on the Greenhouse Gas (GHG) emissions and water consumption.

The GHG emission problem, recognized after the Kyoto Protocol, is getting important for the public perception. Now, ISO 14064-5-6, related to GHG emissions, and 14067, related to the Carbon Footprint (CF), have become a tool to communicate the environmental impacts of goods and services used as a label of competitiveness for the product that agrees with the increasing demand from consumers for a healthy and sustainable product (Pattara et al., 2017; UE, 2013). The methodology is standardized according to the ISO and following the life cycle assessment (LCA) tool with four steps: goals and scope definition, inventory analysis, impact assessment and interpretation (Fang et al., 2014; Lovarelli et al., 2016). In recent years, CF has become the most used indicator for the environmental performances of agro-food products evaluating the impact of consumers' diet choices. The CF index quantifies the gas emissions (carbon dioxide, methane, nitrous oxide, hydrofluorocarbons, perfluorocarbons and sulphur hexafluoride) from the life cycle of a product or service, detecting the contribution to the global warming potential (GWP) (Pandey et al., 2011).

On the other side, the main issues concerning water are its scarcity and overexploitation. These issues need mitigation strategies linked to social, ecological and economic contexts. Water scarcity is affected by the increase of higher temperatures and lower precipitation. Extreme climate events influence water availability, reducing the capacity of water supply services from ecosystems to humanity. A new concept was developed to better understand the impact of water consumption by human activities, both direct and indirect, starting from the idea of virtual water trade (Allan, 1997) in a global context (Antonelli and Sartori, 2015). This concept is the Water Footprint (WF), introduced by Hoekstra and Hung (2002), and it is a quantitative and qualitative indicator. The concept of the WF is quantitative since it calculates the volume of water consumption to produce goods or services in all their supply chains, and it is qualitative since it evaluates the amount of water required to assimilate pollutants based on the water quality standard in an ecosystem (Chapagain and Hoekstra, 2004; Mekonnen and Hoekstra, 2011). WF is an indicator used to improve policy decisions on water benchmarks and frameworks. The research community takes into account two main approaches: the volumetric methodology of Water Footprint Assessment enhanced by Hoekstra et al. (2009, 2011) and the most recent LCA, which is relevant in comparing the environmental performance of a product during its entire production chain. Mekonnen and Hoekstra (2011) and Mekonnen and Hoekstra (2012) provided the first global database of Water Footprint for agricultural products. After that, many others authors focused on the WF assessment of the agriculture sector (Castellanos et al., 2016; Nana et al., 2014). The concept evolved from the idea of a sustainability level on water resources in a country to the idea of equity and ethical water consumption to improve precautionary policy measures on the water (Tuninetti et al., 2015).

To ensure the sustainability of a process, a compound of environmental indicators is needed to provide an impact evaluation for food production. Food security needs high yield crops that affect natural resources exploitation; the use of the soil depends on the crop address that is directly related to the food choice of consumers and has different weights of impact (Cristina Rulli and D'Odorico, 2014). Different impact categories are proposed to monitor human environmental pressure through an appropriate set of indicators more appealing to measure the sustainability. Environmental, economic and social impact evaluations of particular processes, products or activities have already been developed for many different concepts and methods in single and compound indexes (European Commission, 2011). A "Family Footprint" was introduced by Galli et al. (2012) and developed by the OPEN: EU project, where they developed a group of the main footprints, taking into account the impact of multiple indicators, such as carbon, water, and ecological footprint (Corbo et al., 2014). There is no standard and precise definition of "footprint" or of the difference between the indicators of potential impacts and footprints (Čuček et al., 2012). The footprint itself does not provide helpful development strategies, but a compound of indicators gives a complementary sense of impact in different aspects of the environmental issue (Vetoné Mózner, 2014). A set of environmental indicators evaluates the sustainability addressing multiple aspects, and they can make an overview of environmental sustainability (Čuček et al., 2012). For example, these indicators evaluate the sustainability of enterprises (Herva et al., 2011); they evaluate the environmental impact of citizens using a specific model (Ewing et al., 2012; Steen-Olsen et al., 2012); or they can be compared within a food label (Leach et al., 2016). Many authors that propose a set of "Family Footprints" also promote a single measure of a composite footprint indicator (De Benedetto and Klemeš, 2009). It is hard to evaluate a single measure from the core of indicators because the methodology must be the same, and there is no threshold where each footprint indicates the level of sustainability (Čuček et al., 2012; Gu et al., 2016). For this reason, a broad scope approach can best inform the stakeholders, accounting for a huge range of implications in different sectors.

Sustainability is a broad concept that includes the main indicators for such a pillar, namely, environmental, social and economic aspects; the environmental sustainability also concerns some sub-indicators. Three environmental indicators are mostly studied when focusing on environmental sustainability. In the case of CF, the contribution of emissions is a global issue; however, for WF, the possible reduction of wasting water is due to production and consumption activities. WF and CF assessments refer to a quantitative amount of impact, but in the case of WF, the result also takes a different weight according to the local water resource availability (Galli et al., 2012; Pfister and Bayer, 2014). The approach of calculation for both environmental indicators is similar, based on a bottom-up evaluation, but the variables and components of the footprint evaluations are different (Fang et al., 2014).

There is a current and continuous interest in the water-energy-food nexus (FAO, 2018a). The nexus provides the linkage among different aspects of the sustainability of food products and their impact on the water and energy resources. In this study, we considered the CF in relation to the energy content as a ratio of energetic performance given by the product during its production, and the WF that considers the water consumption for each agricultural product from "cradle to gate". We performed data collection from a literature review for the CF indicator, mostly focused on the LCA method, and we considered the WFN database for the WF indicator as main reference, as the literature cannot supply a huge and broad dataset for the main food products. The study collects the CF, WF and energy content values data for several agricultural products. This study also

compares, for the first time, the nutritional value of energy (input) with the GHG emissions (output) of each agricultural product and then quantitatively combines with the WF. We selected these indicators because of their important role in the environmental evaluation of impacts among clusters of products. We detected the relation water-energy impact for each product, and we gave a common point of view on the sustainability of food. In the literature, other authors are studying the relation water-energy and related indexes. The literature shows many examples on the environmental evaluation impacts of food using a set of indicators. Vanham and Bidoglio (2014) studied two different diets for European citizens only considering the WF and the virtual water trade. Others gave suggestions to implement a compound of indicators with a review of the main environmental footprints, but without presenting results (Fang et al., 2014; Galli et al., 2012; Herva et al., 2011; Sala et al., 2017). Others used the carbon and Water Footprints in relation to other indicators or for other implications for computing the environmental footprint of European consumers (Steen-Olsen et al., 2012). Heller et al. (2013) provided a review of a sustainable diet considering the GHG emissions and food quality indicators. Masset et al. (2014) also studied the sustainability of food and agricultural products by analysing the CF, eutrophication and acidification as environmental footprints and nutritional quality indicators linked to the prices of foods. Davis et al. (2016) compared different footprints for different diets and provided an evaluating approach of sustainability for integrating food security and environmental impacts.
The aim of the study is to present the definition of a classification scheme that allows understanding how the sustainability of agriculture products is according to their impact. The novelty of this new case study is to provide a different point of view on the evaluation of a simple and comprehensive sustainable pattern using data collection from a literature review and considering a global average value for the indicators.

2.3 Materials and Methods

This study takes into account two footprints and a nutritional index. Carbon Footprint and Water Footprint are the two main footprint indexes included to estimate the environmental impact and to classify the sustainability of agriculture products. The other parameter is the energetic nutritional value involved in food, used for comparison with the emitted energy and to understand the energetic balance between clusters of agriculture food products. To compare and evaluate the weight of each single product, a stable unit of measure based on a kilogram of the food product is chosen. The two footprints detect the impact of water use and GHG emissions during food production and consumption, including the supply chains. The indicators are described as follows.

2.3.1 Water Footprint

The WF indicator evaluates the impact for the appropriation of natural water capital regarding volumes for human consumption (Hoekstra et al., 2011), or, in this case, to produce an amount of agriculture food products. The term of WF was coined from the concept of global virtual water flow. The WF is the environmental indicator that evaluates the volume of water applied for human activities, such as during animal and crop production. WF accounts for direct and indirect water use, and it is separated into three different components: blue, green and grey WF. The blue Water Footprint refers to the volume of surface and groundwater consumed (evaporated). It is considered as the irrigation volume supplied during the growing season. The green Water Footprint refers to the rainwater consumed; it is the evapotranspiration of crops only due to the available water in the ground after a rainfall event and absorbed by plants. The grey Water Footprint of a product relates to the volume of freshwater that is required to assimilate a load of pollutants based on existing ambient water quality standards (Mekonnen and Hoekstra, 2011). The total WF is expressed as the sum of the three components. In the case of crop production, the direct and indirect water uses concern the water consumption along the growing season, while for animal production, the indirect water uses relied on the feed crops, and the direct water is the volume of drinking water. The WF in food production is expressed by the ratio of output: input, water volume: yield or metres cubed: tonnes. The Water Footprint is an indicator of water use and resource allocation of countries within sustainability limits.

The Water Footprint is an environmental indicator of water's human competition, which was transformed into the LCA community as an indicator with water scarce weight. This transformation was due to water consumption in a rich water catchment having a different impact on the same volume in a water-scarce catchment. Actually, there are two main approaches for WF accounting: the volumetric approach developed by the Water Footprint Network (WFN) (Hoekstra et al., 2011), and the LCA approach elucidated by the LCA community. The two approaches do not follow the same methodology. The first one, computed by the WFN, focuses the evaluation of the impact on the efficiency of the system, and it usually evaluates the indicator as the volume of water consumed by the yield. The second one evaluates the impact of food products on water resources, based on the real water availability of a basin (Pfister et al., 2009; UNEP, 2010), and it is expressed as the volume of water weighted by the territorial water availability. The two water impact estimations are different because of the system boundaries they consider. In this study, the WF data collection is derived from the literature (Table 1), and mainly from one main database based on the papers by Mekonnen & Hoekstra (2011) and Mesfin M. Mekonnen & Hoekstra (2012) allowed by the Water Footprint Network. In this paper, the WFN methodology is taken into account because of the wide availability of data for many food products. The database collects several vegetal and animal products from several countries. However, the same approach here presented can be replicated taking advantage of national or regional based data. Data used from the WFN database considered a global average value of WF for many crop and animal products. They used a grid-based dynamic water balance model on a 5-by-5 arc minute grid for a time series between 1996 to 2005 (Mekonnen and Hoekstra, 2012, 2011). Since the time series is becoming obsolete, the database is the only unique exhaustive database for WF. The dataset was also considered in many other case studies (Mekonnen and Hoekstra, 2014; Owusu-Sekyere et al., 2017; Vanham and Bidoglio, 2014). Tuninetti et al. (2015) replicated the crop virtual water content using the Mekonnen and Hoekstra (2010) assessment with a time series from 2000 to 2010, and they concluded that the two time series provided a similar value for every crop at a global scale (R^2 = 0.91 for wheat, 0.76 for rice, 0.90 for maize, and 0.91 for soybean). There is a lack of case studies on WF, for which it is possible to collect data for a large number of products. In this paper, we selected data for 70 main crops and animal products at a global scale, and then we were able to compare them with the other footprint thanks to the unit of measure based on the weight of products (m^3/tonne).

It is worth noting how the calculation and clusterization method proposed in the present paper could be replicated also taking advantage of LCA methodology when a large dataset will be available. Indeed, the value of the approach here introduced is not just on the quantification of environmental performances but is rather on the definition of a sustainability pattern which allows comparison of different agricultural products or group of products.

2.3.2 *Carbon Footprint*

The CF measures the total amount of GHG emissions, both directly (on-site) and indirectly (off-site), caused by an activity or accumulated during the production and supply chain. The total amount of greenhouse gas emissions are expressed in the mass unit and not converted into an area unit for several reasons; one of the most important reasons is because the conversion needs many assumptions that give an estimation value with a high variability of error (Galli et al., 2012). GHG emissions identified by the Kyoto Protocol are evaluated in a single value expressed in kilograms of carbon dioxide equivalent per kilogram of the product according to the ISO 14067, as mentioned in the introduction. The mass unit of global warming effect referred to the CO_2 because it is the most important GHG affecting global warming. Concerning global warming, the CH_4, N_2O, HFC, PFC, and SF_6 are normalised from their global warming potentials according to the United Nations Framework of Convention on Climate Change (IPCC, 2006). The CF value supplies an additional environmental meaning to the product, adopted by corporations to improve a sustainable consumer's awareness, according to the increasingly recognized global warming issue. The Carbon Footprint is the most used environmental indicator evaluating the environmental performance and climate change. Currently, there is an increasing interest to compare the environmental impact through multiple indicators. Carbon Footprint and Water Footprint give a core of information about the atmosphere and hydrosphere. Each of the two footprints is designed to address the different environmental issues, and they are complementary to each other. The carbon equivalent, measured as the weight of the volume of gas emissions, is seen as the energy emitted and sequestrated by the Carbon Footprint, while the water consumption is reported by the amount of production. The literature is mainly focused on the LCA method, measuring the carbon dioxide equivalent emissions of each step of production and transport. In this case, we collected data considering the impact of CF "from cradle to gate", while the Carbon Footprint is calculated from the origin of the crop to the sale of the product from the farm; therefore, the system boundary is the farm gate. Data collection does not involve the market and the waste scenario of products, because the present study focuses on the agriculture products, and all the sale variables were not considered. In addition, the CF in this study is a mediated value from data collection. This value is meant to provide an average CF at a global scale. Since many considerations on the country level can be assumed, this paper avoids the variability of the assessment at the local scale. The unit of measure (functional unit) is the kg of CO_2eq per kg of product. Literature reviews include 36 articles (Table 1) for CF, since there is no open source database for this indicator.

2.3.3 *Energetic Nutritional Value*

In this study, the energetic value is taken into account to quantify the energy included inside the food product and compared with the energy emitted as GHG emissions. The energetic nutritional value is considered as an indicator of the incorporated energy per edible amount of a product. An energetic parameter of input is usually compared with other quantitative outputs. This comparison allows the understanding of how the CF, seen as emitted energy from GHG emissions, is related to the energy inside food products. It is not an energy footprint, which is normally considered as the sum of all areas used to sequestrate CO_2 emissions from the consumption of non-food and non-feed energy. For this reason, a more effective relationship between the nutritional value of the energy included in foods and the energy emitted by GHG during food production provides a meaningful indicator of energetic performance. The impact of GHG emissions depends on several productive factors, while the energy content depends on the composition of the dry matter of the product. The relationship between them considers the particular characteristic of each product to be energetically righteous with smaller energetic waste.

The energetic value data derived from website literature review are related to the website of Crea (Board of Analysis and Research on Agriculture sector), from the Italian Bank for Food Composition sites and the Composition of Foods Integrated Dataset from the Public Health England Institute (Carnovale and Marletta, 1987; Finglas et al., 2015). The energy content is measured in kilocalories per 100 grams of edible food. The value of energy content is representative of all inputs during the production chain. The food energy content depends on the composition of lipids, proteins and carbohydrates. According to the values from different sources, we selected a global average of calories included in the product.

2.3.4 Clustering and classification comparing different environmental impact indicators

Seventy products are proposed in the study, including vegetal and animal foods. No derivation of vegetal or animal foods is considered, except eggs and milk. These last food products are chosen because they are not processed goods, and the impact is directly due to the farm production. These 70 food products are selected due to their representability of the main global diet. These products represent 70% of the primary standard agricultural commodities for food consumption and transformation (FAOSTAT).

The three indicators are compared to answer the research question of what product is more suitable to reduce the consumer's impact and how commodities are placed in a sustainable scheme to understand the most sustainable and less impacting cluster of food. An index, which takes into account a compound of indicators, is more suitable because it gives more evidence about the process or production than the real environmental performance. For this reason, the energetic input-output relationship considers the CF and the energetic nutritional value evaluated as the energetic ratio (ER) with a logarithmic fraction, explained as follows:

$$ER = \log\,(\text{energetic value/Carbon Footprint}) = \log\,(\text{kcal/kgCO}_2\text{eq}) \qquad (1)$$

The ER shows, for the first time, a relationship between two different energetic metrics and compares the values of the various energetic sources in an input-output relationship.

The clustering and classification are made using a model that follows a logarithmic distribution to better detect a trend of food clusters and pool of food addresses. The logarithmic model for the WF variable is explained as follows:

$$\log \text{WF} = log(m^3/ton) \qquad (2)$$

WF was converted into $m^3\ kg^{-1}$ prior to equation 2 to better compare the impact of 1 kg of product. The logarithmic equation is used to show the trend of the ratio between three indicators. The use of the logarithm in the equation is to emphasize the data distribution at smaller values. The model better demonstrates the food cluster situation according to different footprint indicators.

A statistical analysis is done (Table 2) to compute the representativeness of clusters using the R-square index (Halkidi et al., 2001). The R-square index is used to understand the homogeneity and the Euclidean distance between groups. The index of R-square of *log WF* and *log ER* parameters (*p*) is calculated as the fraction of the sum of squares of groups and the sum of the square of the entire dataset (equation 3). The values of R-square have a range between 0 and 1. If the R-square value is zero, it indicates that no differences exist among groups. Moreover, when the value is one, there is a significant difference among groups (Halkidi et al., 2001).

$$\text{R-square} = \sum_{i=1}^{p} \sum \left(\bar{x}_g - \bar{x}_t\right)^2 \cdot n \Big/ \sum_{i=1}^{p} \sum (x_i - \bar{x}_t)^2 \quad (3)$$

Where $\bar{x}_g$ is the single group average, $\bar{x}_t$ is the total average, *n* is the number of values in the group and x_i is the single value. After this statistic group's analysis, a classification was done, putting in a ranking of top sustainable products.

The classification pattern is made using a ranking factor. The Ranking Factor (RF) is the fraction of the energetic ratio and the Water Footprint indicators as shown in equation 4:

$$\mathrm{RF} = \log(ER)/\log(\mathrm{WF}) = \mathrm{kcal} \cdot \mathrm{kg}/\mathrm{kg}_{\mathrm{CO_2eq}} \cdot \mathrm{m}^3 \quad (4)$$

In this way, a ranking of the most sustainable agriculture products is computed as a classification from higher to lower value, where higher value stands for a high sustainable cluster of agricultural products.

2.4 Results

Data collection for the WF value is combined with the assessed energetic ratio (ER) value for each agricultural product. The relationship between the WF and the ER gives a degree of sustainability of agricultural products according to their impact on the environment. All agricultural products' clusters are well represented and grouped in the graph in Figure 1. The higher the WF value, the higher the impact of water resource is, and the higher the ER value is, the better the energetic ratio for each product is.

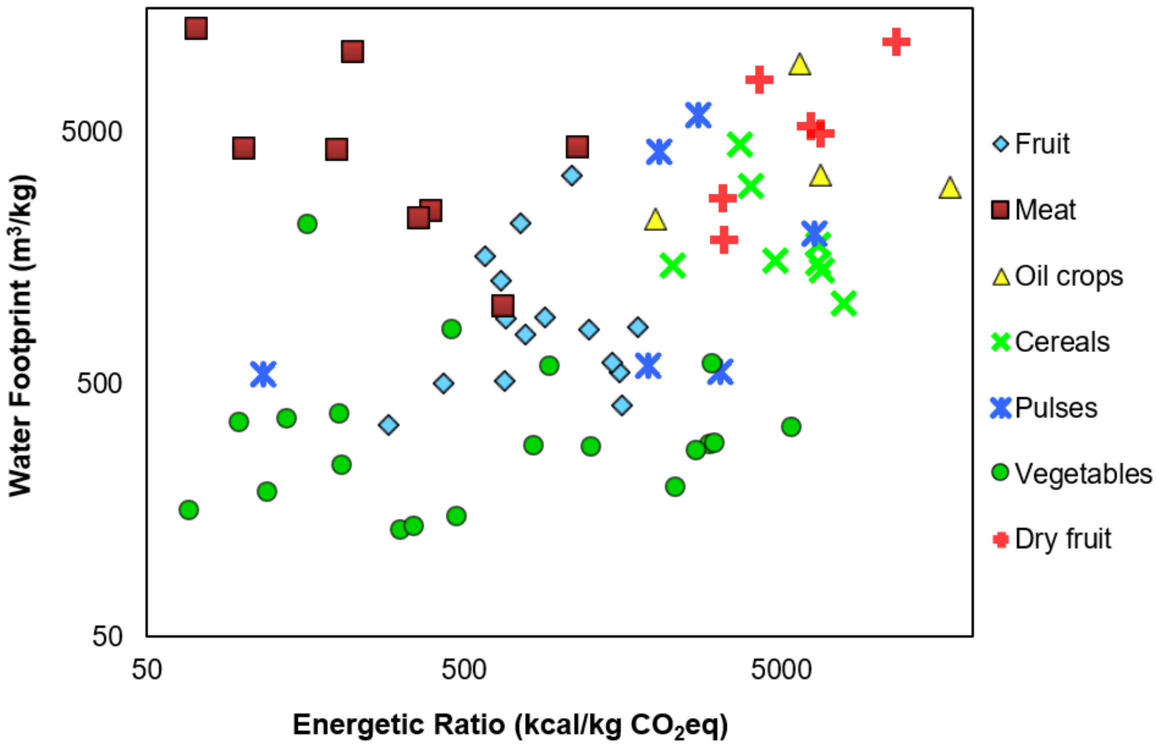

Figure 1. Graph of compared food clusters combining indicators of water and carbon footprint and energy content.

The clustering permits to compare different food products and to classify them in a ranking concerning their degree of sustainability as given in Table 1. Table 1 provides the values of the concerned variables for such products and lists each one according to its environmental performance. In addition, Table 1 discloses the awareness of the best performing food products. The clusters represented in Figure 1 focus on the main family of products. A diversification of animal and vegetal products is done. Inside these two "big families", another sub-clustering divides different varieties of vegetal foods into vegetables, pulses, oil crops, fruits, dry fruits, and cereals. The cluster of fruits occurs in the middle of the graph (Figure 1), and it could be associated with each other group of agriculture products because of its degree of sustainability and position. Indeed, fruit products also occur in the middle of the ranking in Table 1.

Differently, from the cluster of fruit, the dry fruits group has a high energetic value that corresponds to a high WF value, and the energetic ratio is also higher than the fruit cluster. For example, nuts have a WF of 1871 m^3/ton and an ER of 329.6 kcal/kgCO_2eq, and almonds have a WF of 8047 m^3/ton and an ER of 426.8 kcal/kgCO_2eq; apples have a WF of 822 m^3/ton and an ER of 123.5 kcal/kgCO_2 eq. This means a worse environmental performance in terms of water consumption, but with a better value of ER.

Pulses, dry fruits, and oil crops clusters have a high energetic distribution with a high WF. In fact, the average WF is 4506 m3/ton for oil crops, 5701 m^3/ton for dry fruits and a lower 2447 m^3/ton for pulses, but the average ER is 784 kcal/kgCO_2eq for oil crops, 585 kcal/kgCO_2eq for dry fruits and 320 kcal/kgCO_2eq for pulses. Those absolute values that seem less meaningful make sense in Figure 1, where the extent of their impact is evident. These clusters are usually considered as the main source of energy intake in a daily diet, and while their impact is not well assumed in terms of water and energy consumption, they could become important in a future sustainable diet scenario.

In the classification pattern, the animal food products have a lower chance of being sustainable, because of their water and energy consumption during their production chain. There are foods drawn up to be more sustainable for their low impact on the environment, due to their low input consumption and their resources requirements. For example, according to the ranking in Table 1, pumpkins have a WF of 336 m^3/ton and an ER of 541 kcal/kgCO_2eq, while another animal product, such as beef, has a WF of 12830 m^3/ton and an ER of 7 kcal/kgCO_2eq. The cluster of animal food products and secondary animal products occurs with a high WF and a low ratio of energy discharged. The animal cluster is quite homogeneous, except for beef meat, which is the most impacting animal product, and poultry, which is the lowest one with a WF of 2432 m^3/ton and an ER of 40 kcal/kgCO_2eq. The vegetable cluster, according to its position on the graph, has the best sustainable behaviour (Figure 1). This cluster represents the best environmentally sustainable cluster where it is possible to find some righteous products to choose for a low impact diet due to a low WF and high energy ratio involved. For example, carrots, spinach, pumpkins, and onions are positioned at the top of the ranking (Table 1). This cluster is wide and spread. Inside the vegetable cluster, there are no specific differences between vegetable products from leaves and vegetable products from fruits. It is possible to observe, in Figure 1, clusters that follow a homogeneous trend, with an exception for pulses and oilseed crops. For example, string beans have a trend similar to the vegetable cluster with a low WF and ER situation. There is no eco-friendly cluster, as shown in Figure 1. The only cluster closer to being the most sustainable is the vegetable one, as previously discussed. Table 1 shows the rankings, where 60% of the first ten sustainable products are vegetables, and only 20% cereals, 10% pulses and 10% oil crops. It is already indicated that animal products are placed in the bottom of the rank. The most suitable food product with low input and low stress on the environment are in following order: pumpkins, carrots, spinach, and potatoes.

Table 1. Animal and vegetal dataset collected from a literature review of footprint indicators (WF and CF) and a nutritional quality parameter (calories). Classification is according to combined impact indicators.

*Ranking factor is the quotient of the energetic ratio and the water footprint indicators.

Rank	Food products	WF average (m^3 ton^{-1})	CF average (kg CO_2eq kg^{-1})	Energetic value (kcal)	Energetic ratio (kcal $kgCO_2eq^{-1}$)	Ranking factor*	Reference
1	Pumpkins	336.00	0.03	18.00	541.00	1.6101	(Carnovale and Marletta, 1987; Mekonnen and Hoekstra, 2011; Schäfer and Blanke, 2012)
2	Carrots and turnips	195.00	0.15	35.00	234.00	1.2000	(Carnovale and Marletta, 1987; Jr. et al., 2013; M.M. Mekonnen A.Y. Hoekstra, 2011)
3	Spinach	292.00	0.10	31.00	310.00	1.0616	(Carnovale and Marletta, 1987; Irz and Kurppa, 2013; M.M. Mekonnen A.Y. Hoekstra, 2011)
4	Potatoes	287.00	0.29	85.00	298.00	1.0383	(Audsley et al., 2009; Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011)
5	Sweet potatoes	603.00	0.14	87.00	613.00	1.0166	(Audsley et al., 2009; Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011)
6	Onions	272.00	0.10	26.00	271.00	0.9963	(Carnovale and Marletta, 1987; Jr. et al., 2013; M.M. Mekonnen A.Y. Hoekstra, 2011)
7	Maize	1045.00	0.45	353.00	787.00	0.7531	(Audsley et al., 2009; Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011)
8	Beans	561.00	0.33	104.00	320.00	0.5704	(Carmo et al., 2016; Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011)
9	Olives	3015.00	0.08	142.00	1701.00	0.5642	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Wiedemann et al., 2017)
10	Barley	1423.00	0.48	319.00	668.00	0.4694	(Carnovale and Marletta, 1987; Korsaeth et al., 2014; M.M. Mekonnen A.Y. Hoekstra, 2011)
11	Cabbages	280.00	0.19	24.00	127.00	0.4536	(Carnovale and Marletta, 1987; Jr. et al., 2013; M.M. Mekonnen A.Y. Hoekstra, 2011)
12	Wheat	1513.00	0.48	312.00	651.00	0.4303	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Wiedemann et al., 2017)
13	Raspberries	413.00	0.22	34.00	157.00	0.3801	(Carnovale and Marletta, 1987; Girgenti et al., 2013; M.M. Mekonnen A.Y. Hoekstra, 2011)
14	Oats	1788.00	0.52	339.00	652.00	0.3647	(Carnovale and Marletta, 1987; Korsaeth et al., 2014; M.M. Mekonnen A.Y. Hoekstra, 2011)
15	Tomatoes	149.00	0.36	17.00	48.00	0.3221	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Pishgar-Komleh et al., 2017)
16	Soya beans	1981.00	0.63	398.00	634.00	0.3200	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Raucci et al., 2015)
17	Peas (green)	595.00	0.40	76.00	190.00	0.3193	(Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011; Michalský and Hooda, 2015)
18	Rye	1544.00	0.70	335.00	476.00	0.3083	(Carnovale and Marletta, 1987; Jr. et al., 2013; M.M. Mekonnen A.Y. Hoekstra, 2011)
19	Cauliflowers	285.00	0.30	25.00	83.00	0.2912	(Carnovale and Marletta, 1987; Irz and Kurppa, 2013; M.M. Mekonnen A.Y. Hoekstra, 2011)
20	Oranges	560.00	0.22	34.00	155.00	0.2768	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Yan et al., 2016)
21	Melons	137.00	0.95	33.00	35.00	0.2555	(Carnovale and Marletta, 1987; Cellura et al., 2012; M.M. Mekonnen A.Y. Hoekstra, 2011)
22	Sugar beet	132.00	0.60	19.00	32.00	0.2424	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Yousefi et al., 2014)
23	Grapes	608.00	0.42	61.00	146.00	0.2401	(Bartocci et al., 2017; Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011)
24	Blueberries	845.00	0.14	25.00	176.00	0.2083	(Carnovale and Marletta, 1987; Girgenti et al., 2013; M.M. Mekonnen A.Y. Hoekstra, 2011)
25	Sunflower seeds	3366.00	0.87	576.00	665.00	0.1976	(Finglas et al., 2015; Iriarte and Villalobos, 2013; M.M. Mekonnen A.Y. Hoekstra, 2011)
26	Nuts	1871.00	1.72	567.00	330.00	0.1764	(Finglas et al., 2015; Leach et al., 2016; M.M. Mekonnen A.Y. Hoekstra, 2011)
27	Garlic	589.00	0.44	41.00	93.00	0.1579	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Michalský and Hooda, 2015)
28	Rice	1487.00	1.49	337.00	226.00	0.1520	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Xu and Lan, 2017)
29	Apples	822.00	0.31	38.00	124.00	0.1509	(Carnovale and Marletta, 1987; Jr. et al., 2013; M.M. Mekonnen A.Y. Hoekstra, 2011)
30	Walnuts	4918.00	0.88	582.00	661.00	0.1344	(Audsley et al., 2009; Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011)

Rank	Food products	WF average (m^3 ton^{-1})	CF average (kg CO_2eq kg^{-1})	Energetic value (kcal)	Energetic ratio (kcal $kgCO_2eq^{-1}$)	Ranking factor*	Reference
31	Sorghum	3048.00	0.82	329.00	401.00	0.1316	(Carnovale and Marletta, 1987; Jain et al., 2016; M.M. Mekonnen A.Y. Hoekstra, 2011)
32	Kiwi fruit	514.00	0.66	44.00	67.00	0.1304	(Audsley et al., 2009; Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011)
33	Chestnuts	2750.00	0.43	140.00	326.00	0.1185	(Audsley et al., 2009; Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011)
34	Hazelnuts	5258.00	0.40	247.00	618.00	0.1175	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Volpe et al., 2015)
35	Pistachios	11363.00	0.53	608.00	1147.00	0.1009	(Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011; Volpe et al., 2015)
36	Bananas	790.00	0.83	65.00	78.00	0.0987	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Yan et al., 2016)
37	Pears	922.00	0.39	35.00	90.00	0.0976	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Yan et al., 2016)
38	Lettuce	237.00	0.67	14.00	21.00	0.0886	(Carnovale and Marletta, 1987; Foteinis and Chatzisymeon, 2016; M.M. Mekonnen A.Y. Hoekstra, 2011)
39	Rapeseed	2271.00	2.09	419.00	201.00	0.0885	(Audsley et al., 2009; Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011)
40	Grapefruit	506.00	0.61	26.00	43.00	0.0850	(Audsley et al., 2009; European Institute of Oncology, 2015; M.M. Mekonnen A.Y. Hoekstra, 2011)
41	Strawberries	347.00	0.94	27.00	29.00	0.0836	(Audsley et al., 2009; Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011)
42	Millet	4478.00	0.96	356.00	371.00	0.0828	(Carnovale and Marletta, 1987; Jain et al., 2016; M.M. Mekonnen A.Y. Hoekstra, 2011)
43	Peaches/nectarines	910.00	0.37	25.00	68.00	0.0747	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Yan et al., 2016)
44	Milk	1020.00	0.95	63.00	66.00	0.0647	(Finglas et al., 2015; Mekonnen and Hoekstra, 2012; Rotz et al., 2010)
45	Watermelons	186.00	1.33	16.00	12.00	0.0645	(Audsley et al., 2009; Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011)
46	Sesame seed	9371.00	1.05	598.00	570.00	0.0608	(Audsley et al., 2009; Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011)
47	Artichokes	818.00	0.48	22.00	46.00	0.0562	(Audsley et al., 2009; Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011)
48	Almonds	8047.00	1.27	542.00	427.00	0.0531	(Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011; Marvinney et al., 2015)
49	Peppers/chillies	379.00	1.08	22.00	20.00	0.0528	(Cellura et al., 2012; Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011)
50	Apricots	1287.00	0.43	28.00	65.00	0.0505	(Audsley et al., 2009; Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011)
51	Chickpeas	4177.00	1.53	316.00	207.00	0.0496	(Finglas et al., 2015; Jain et al., 2016; M.M. Mekonnen A.Y. Hoekstra, 2011)
52	Lentils	5874.00	1.06	291.00	275.00	0.0468	(Audsley et al., 2009; Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011)
53	Coffee	15897.00	0.39	287.00	731.00	0.0460	(Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011; Noponen et al., 2012)
54	Zucchini	158.00	1.60	11.00	7.00	0.0443	(Carnovale and Marletta, 1987; Cellura et al., 2012; M.M. Mekonnen A.Y. Hoekstra, 2011)
55	Eggplants	362.00	1.30	18.00	14.00	0.0387	(Audsley et al., 2009; Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011)
56	Cherries	1604.00	0.66	38.00	58.00	0.0362	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Michalský and Hooda, 2015)
57	Plums	2180.00	0.56	42.00	75.00	0.0344	(Audsley et al., 2009; Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011)
58	Figs	3350.00	0.43	47.00	109.00	0.0325	(Audsley et al., 2009; Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011)
59	Cucumbers	353.00	1.42	14.00	10.00	0.0283	(Carnovale and Marletta, 1987; M.M. Mekonnen A.Y. Hoekstra, 2011; Sami and Reyhani, 2015)
60	Chicken	4325.00	1.70	194.00	114.00	0.0264	(European Institute of Oncology, 2015; Mekonnen and Hoekstra, 2012; Wiedemann et al., 2017)
61	String beans	547.00	1.55	18.00	12.00	0.0219	(Audsley et al., 2009; European Institute of Oncology, 2015; M.M. Mekonnen A.Y. Hoekstra, 2011)
62	Poultry	2432.00	4.33	171.00	40.00	0.0164	(European Institute of Oncology, 2015; Leach et al., 2016; Mekonnen and Hoekstra, 2012)
63	Eggs	2283.00	3.54	128.00	36.00	0.0158	(Finglas et al., 2015; Leach et al., 2016; Mekonnen and Hoekstra, 2012)
64	Fish	1974.00	3.61	82.00	23.00	0.0117	(Finglas et al., 2015; Leach et al., 2016; Tillotson et al., 2014)
65	Asparagus	2150.00	1.79	29.00	16.00	0.0074	(Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011; Soode et al., 2015)
66	Pork	4245.00	5.99	119.00	20.00	0.0047	(European Institute of Oncology, 2015; Irz and Kurppa, 2013; Mekonnen and Hoekstra, 2012)
67	Cocoa beans	19928.00	4.37	355.00	81.00	0.0041	(Finglas et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011; Ortiz-Rodríguez et al., 2016)
68	Goat	4300.00	13.10	133.00	10.00	0.0023	(European Institute of Oncology, 2015; Irz and Kurppa, 2013; Mekonnen and Hoekstra, 2012)
69	Sheep	10412.00	13.10	293.00	22.00	0.0021	(European Institute of Oncology, 2015; Irz and Kurppa, 2013; Mekonnen and Hoekstra, 2012)
70	Beef	12830.00	19.83	142.00	7.00	0.0005	(European Institute of Oncology, 2015; Leach et al., 2016; Mekonnen and Hoekstra, 2012);

The statistical analysis explains the grouping choice according to a botanic family address. The statistical group analysis justifies the choice of grouping them according to their homogeneity and difference (R-square). Regarding each variable, a box plot explains the distance between groups and the distribution of products within them. In terms of ER indicator, the boxplot in Figure 2 shows a quite evident difference among groups. Intensive crops have a higher value than fruits and vegetables, while animal products show a totally different trend. The pulses group that presents few data has outliers. In the case of the energetic ratio (Figure 2), the cluster of vegetables has the largest data dispersion. The animal products' cluster has a different data population from dry fruit, cereal, oil crop and pulse clusters.

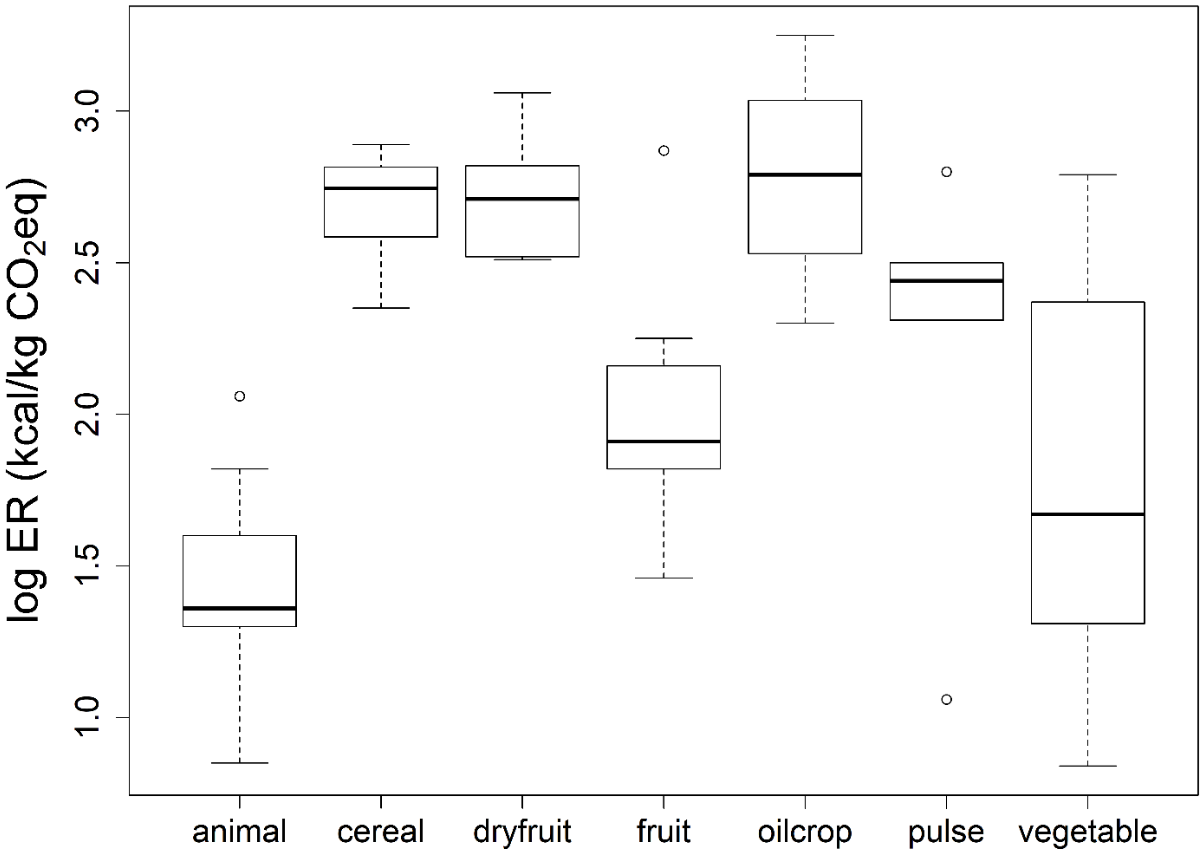

Figure 2. Boxplot of clusters for the Energetic Ratio variable.

Figure 3 considers the WF variable, in which the group of animal products has a larger inter-quartile distance than the other clusters, which means a significant dispersion of data. The cluster of vegetables has a significant different data distribution than the groups of animal products, cereal, oil crops and dry fruits. However, an overview of the boxplot demonstrates a similar average value for all the other groups.

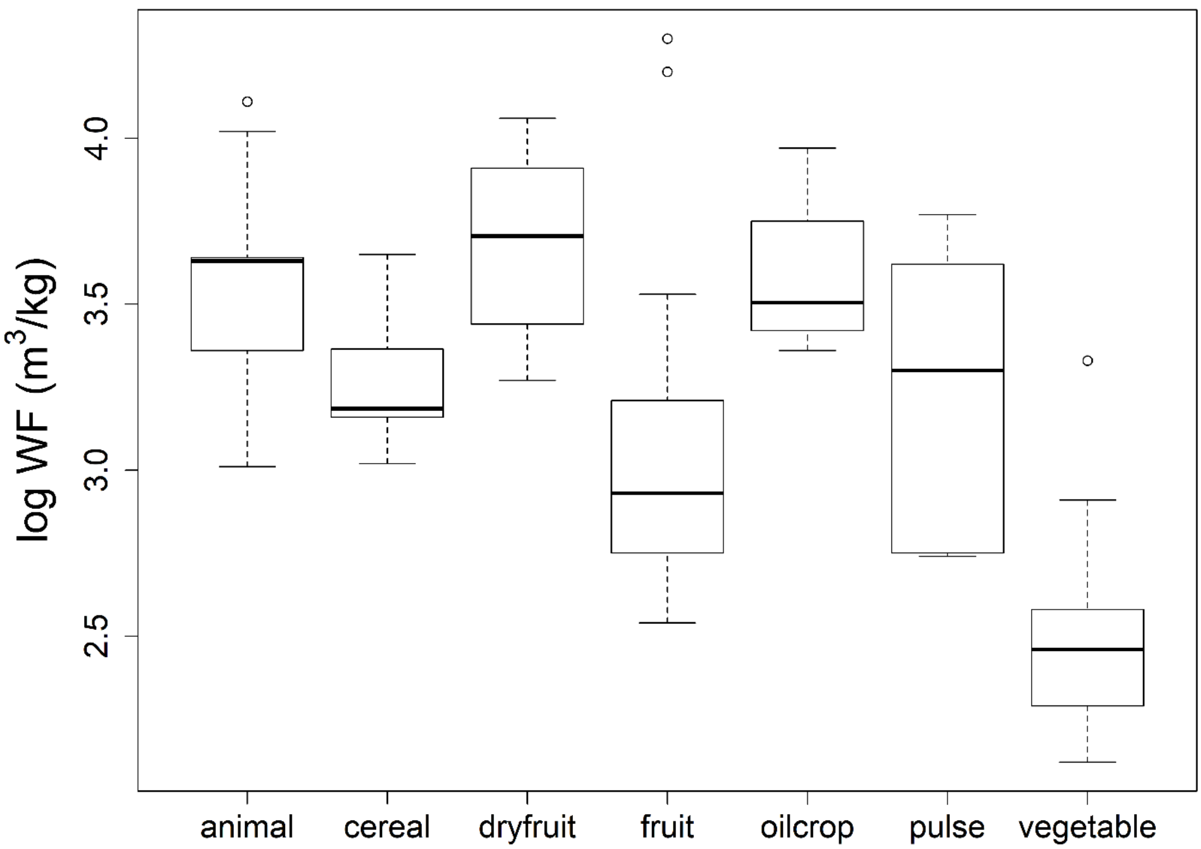

Figure 3. Boxplot of clusters for the Water Footprint variable.

Regarding the stand alone vegetal group products, a regression line is done to understand the trend of those clusters (Figure 4). In figure 4, the previous pattern presented in figure 1 is here replicated only looking to the average values for groups of vegetal products. The centroid, or the means of the cluster, was taken into account to detect the localization of clusters from the regression line. The regression line divides the graph into two sides. The groups under the line are considered to be in a more sustainable situation because of their impact indicator values. From this point of view, vegetables and cereals are more sustainable inside the big group of products of plant origin (Figure 4). The regression line should be used as benchmark for product sustainability according to the WF and ER variables. Finally, the regression line should be considered a relative threshold among considered groups, while if an extra group would be added, the regression line can be affected from the variability of the new group performance. This fact reinforces the thesis of detecting sustainable food products groups within considered clusters, because the line can move upward or downward, but it still represents a threshold between better and performant groups and less performant ones.

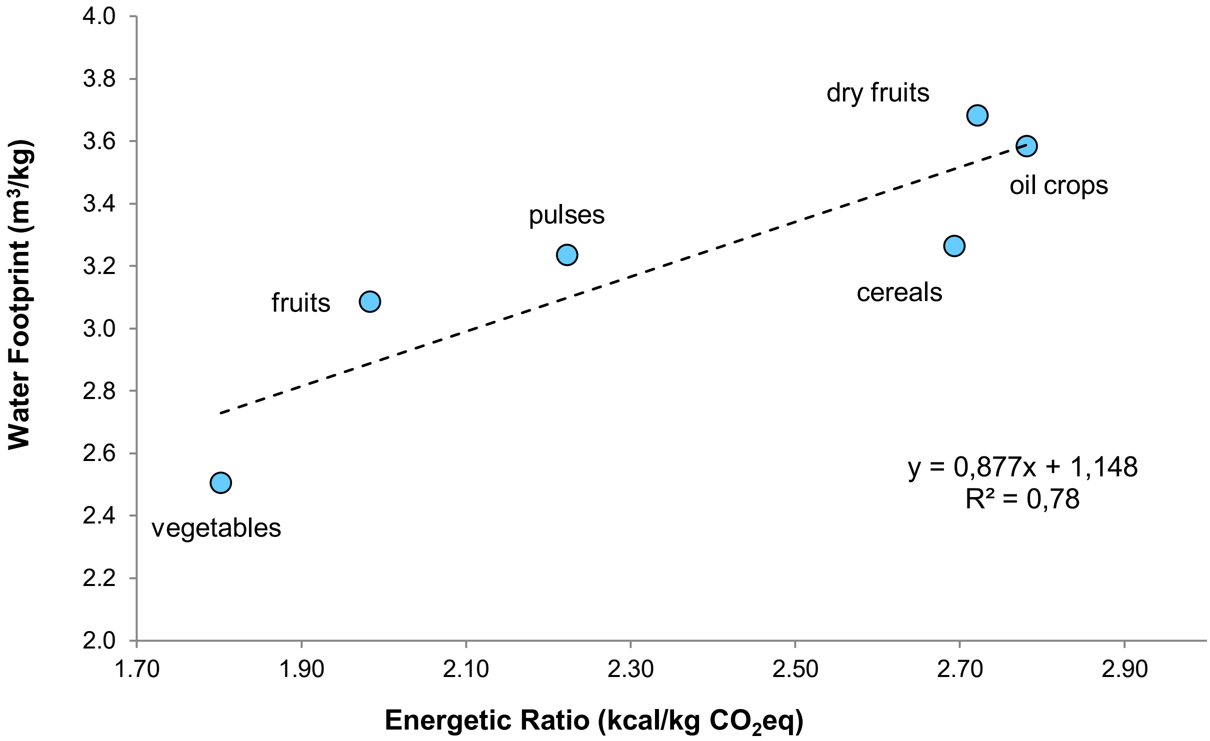

Figure 4. The trend of the vegetal clusters along the regression line.

A statistical analysis of groups is computed to highlight the significance (degree of difference among groups) of food groups' products (Table 2). The total R-square of 0.531 gives the statistical homogeneity and significance for "out of the groups", and clusters are different by a value of 53.1%. According to Table 2, the most homogeneous clusters, regarding the degree of R-square, are fruits with the other groups, and pulses with vegetables thanks to their low R-square value.

Table 2. Statistical group analysis using the R-square for the homogeneity of agriculture product groups.

R-square out of groups		0.531					
R-square inter-groups							
	fruit	*dry fruit*	*vegetable*	*pulse*	*cereal*	*oil crop*	*animal product*
fruit							
dry fruit	0.435						
vegetable	0.338	0.545					
pulse	0.040	0.596	0.391				
cereal	0.364	0.876	0.520	0.530			
oil crop	0.334	0.856	0.508	0.492	0.845		
animal product	0.428	0.792	0.531	0.551	0.777	0.759	

As described in Section 2.4, a higher the R-square coefficient means a relation among groups more homogeneous and statistically different. Thus, as proposed in Figure 1, there are few agriculture products closer to another group; for example, the string bean product is more similar to vegetables than pulses.

2.5 Discussion

The study takes into account a compound of environmental indicators to assess the impact of food consumption. The scope of the work is to describe a sustainable pattern for agricultural products according to their degree of impact. The score of the indicators is considered at a global scale, since the paper aims to normalize country specific data with more generic sources. Data collection makes the first dataset with the average value of different parameters of impact available. This study includes two different indicators of environmental impact and a nutritional quality parameter in a dataset. The clustering shown in *Figure 1* graphically highlights what Table 1 evaluates in a classification of food in order to detect which agricultural product has less impact on the environment. The graph in *Figure 1* puts in relation the water used with the energy involved according to the environmental behaviour of a single product. Water and energy are both the main resources needed for food production. Commonly, water is the most dominant limiting factor for crop production that increases the weight of food depending on the water productivity of crops. In this case, the study only considers the water consumed by crops during the growing season or consumed as animal feed and not the water included inside the products. In this context, the clusters of vegetables and cereals generally consume less water than the others do. The crop water consumption depends on several climate variables and physiological crop variables that affect the value; also, the nature of the plant origin or the genetic origin influences the variability of the value. Energy is the second most important factor limiting the yield, which is affected by sector competition. The study put the energy consumed by greenhouse gas emission in relation to the energy involved inside the agricultural products. In particular, the graph in Figure 1 demonstrates how the dry fruits, pulses, and oil crops groups occur in a high water-consumption situation, but with a better ER trend. In fact, they have high-energy content, but with a lower CF. Finally, the pulses, dry fruits, and oil crops clusters compare with a high energetic distribution with a high WF. A possible reason is that the first productive input is water, and the conversion factor into biomass is low in those foods that usually have a high value of energy content and high dry matter. The energy emitted as an amount of equivalent carbon dioxide depends on how many inputs enter during the production chain. On the other hand, the requirement for good ripening and for a good energetic food content is directly related to the amount of involved production factors. In fact, some organoleptic components, such as lipids and proteins, require more inputs, similar to dry fruits and oil crops. These agricultural products should have more attention in a future diet scenario, because they have a high ER, while in the case of the WF, we should focus on water conservation by reducing water consumption during their growing season. The recent interest to compare healthy and environmentally sustainable foods to a diet turns out to be difficult due to the multidimensional and complex nature of the nutritional quality declaration (Heller et al., 2013). To overcome to this requirement, the study gives an idea of sustainability for several food products, combining those environmental indicators that could affect the diet choice. This work provides a useful and simple set of environmental indicators to give a weight of sustainable food consumption impact to stakeholders.

The abovementioned trend in *Figure 1* demonstrates how the products with plant origins are more sustainable than the animal cluster. A diet based on vegetables, fruits and cereals is environmentally convenient and more sustainable according to its water and energetic impact. Since the graph in Figure 1 graphically demonstrates the trend of the impact of food groups, the ranking in Table 1 makes the behaviour for such products according to their degrees of impact clear. In fact, the ranking shows that the majority of vegetable products occur on the top of the classification (Table 1). Pumpkins seem to have good performance for both WF and ER, while beef meat has the lowest position due to its high-water consumption and low calories per kilogram of CO_2 emitted. Many uncertainties of the value were found during data collection because of both the methodology of each case study and the variability of each parameter taken into account. The various steps of the evaluation methodology affect the environmental impact of food product groups at the impact assessment and interpretation phases. For this reason, a dataset of the local conditions is needed (Lovarelli et al., 2016), accounting for spatial variability (Galli et al., 2012), improving the quality of available data and using a dynamic approach. The present study provides the first dataset of global average values for both WF and ER. Figure 1 is the graphical translation of values reported in Table 1. According to Table 1, the products at the top of the ranking are more sustainable, and the products at the bottom are less sustainable; this could be untrue for those in the middle of the ranking. Environmental sustainability should also take the degradation of the environment into account. Depletion of the ecosystem, pollution, toxicity, and biodiversity are other standardized indicators to develop at the food level and to better understand their impact and sustainability (Masset et al., 2014).

Considering the standalone footprint indicator, it is possible to establish foods of primary and secondary orders. This is clear when observing the boxplots in Figures 2 and 3. Regarding the single Water Footprint, the animals, pulses and oilseeds clusters have more water waste than, for example, vegetables and cereals (Figure 3). Those products require more inputs that are converted into more water consumption. Regarding the energetic ratio between the energetic nutritional value and the Carbon Footprint, it is possible to identify the first order of clusters with more energetic saving, in the case of vegetable products for example, and on the other side, the more pollutant and waster cluster of animal products (Figure 2). The difference between these clusters is possibly explained by the fact that they are usually situated in different stages of the food pyramid, and furthermore, they are placed in a different step of the food chain. The animal cluster has more pollutants in terms of GHG emissions and it requires more energy. The food supply chain is much more elaborate than the vegetarian one. Animal products, i.e., meat, are one of the most important ingredients in our daily intake, and we must not forget their role in society. In this paper, only the environmental point of view is taken into account, but in terms of the economic and social sustainability, Table 1 will have a totally different ranking. Additionally, we can appreciate animal products more than vegetable products for many organoleptic characteristics that are not considered in this analysis.

To define a sustainable pattern, we considered the distance of the centroid of clusters from the regression line (*Figure 4*). The distance from the regression line is a metric useful to understand the behaviour of a cluster from a common sustainable trend. Therefore, a cluster above the regression line is less sustainable than a cluster under the regression line. This is meaningful if we consider the vegetal clusters that have a similar trend. The regression line offers a different point of view of a “sustainable borderline” regarding the WF and ER scores that were not clearly described in Figure 1 and in the ranking of Table 1. By the way, the classification of the cluster does not provide a group of products that are eco-friendly. From an ecological point of view, it is difficult to explain when a food product is more or less sustainable, in this case regarding their water and energy consumption. In addition, the literature lacks the checking of a limit of sustainability, distinguishing between sustainable and unsustainable activities (Fang et al., 2014). For example, foods that have a higher environmental impact do not always have a better nutritional quality (Masset et al., 2014). The status of a crop field could have a different weight between energetic crops and food crops for environmental sustainability. Crop production could feed animals, such as grass, soybean or maize, and they are not directly used for human consumption. Therefore, they increase the impact on the final product. The classification of the sustainability of agricultural products should take into account their edible status due to different weights of impact. A more equitable dietary demand has beneficial solutions for food security, not only improving crop yields and ensuring a global food security but also for poor countries (Davis et al., 2014).

To improve consumers' awareness and make changes, addressing the food sector sustainably is needed, for both the production and consumption patterns. Sustainable diet awareness could be increased, giving a message of the healthiness and environmental impacts of food products and modifying the catering to this choice to change consumption habits (Vetoné Mózner, 2014). The food consumption usually reflects the income, welfare of a population and cultural trends. Social implications are very important to compute the value of sustainability. Food consumption proves to be 27% of the total environmental impacts in the European context, and a changed diet is now more suitable and required to influence the reduction of environmental impacts (Heller et al., 2013). A changeable diet towards a reduction of red meat consumption will reduce the impacts of food consumption by 8% (Heller et al., 2013). A sustainable pattern concerns produce more according to a nutritional and healthy diet.

2.6 Conclusions

In this study, 70 main agricultural products were analysed for their impact on water and energy resources. For each product, a global average score for the Water Footprint determines the water consumption. The Energetic Ratio was evaluated in order to combine the input of energy as calories of energy content and the output as the Carbon Footprint or the greenhouse gases emitted during food production. Values are considered from the cradle to the gate of the farm; the selling process was not taken into account. The aim of the paper is to show the scheme of sustainability for several agricultural products in a graph of food clusters according to the WF and ER indicators. The common trend of the impact degree for each product shows a high value of water consumption related to a high value of the energetic parameter, and on the other hand, a low water consumption is related to a low energetic ratio. In other words, oil crops, pulses and dry fruits provide a good energy trend but a wasteful water trend, while vegetables have a low water consumption linked with a low energetic ratio. However, in the cluster of animal products, a low energy ratio is related to a high water use. Since there are no eco-friendly clusters, vegetable products presented the closest sustainable cluster. A regression line was provided in order to analyse which vegetal cluster would be more sustainable. Generally, a low input product is more sustainable, as it has a low WF and a quite high ER. A ranking list makes clear the order of products into a classification of sustainability and it makes awareness about a sustainable diet composition. For the first time, this paper provides a free dataset of the WF and CF indicators available for agricultural products. The limit of the study is the source variability of data, and the robustness of the value and mark are needed to improve a local dataset of environmental indicators. However, the paper presents many strengths: it gives a primary definition of a sustainable pattern for agricultural products based on the impact on water and energy resource; the trend of the groups of products in relation with their impact is clear; and it gives awareness on the rank position in a classification of sustainability.

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3. Comparison of Water-focused Life Cycle Assessment and Water Footprint Assessment: the case of an Italian wine [2]

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3.1 Abstract

In recent decades, the debate on how to implement and measure sustainability in food production gained increasing importance and interest for agriculture. In the wine sector, producers are increasingly pursuing sustainable practices, including measures for water preservation from degradation and overuse. But methodologies for assessing and communicating the impacts on water resources need to be understood in detail to guide the selection of the most appropriate management practices, support environmental labelling and promote environmental-friendly products to consumers. This work focuses on the impacts on water resources associated with the production of Italian wine by comparing two methodologies: the Water-focused Life Cycle Assessment and the “Water” indicator included in the Italian “VIVA” certification framework, which is based on the Water Footprint Assessment. The two methodologies address the impact on freshwater consumption and degradation from a life cycle perspective. VIVA is based on a water balance method that reflects a volumetric measure of water consumption, while the LCA-based approach investigates both the freshwater consumption and depletion using different impact indicators. The study goal is to compare the two methodologies to understand how their outcomes can support and improve the management of water-related issues in wine production. One main conclusion is that the WATER indicator within VIVA framework can provide more precise recommendations for the optimal management of water use during the vineyard phase, while LCA approach highlights impact hotspots related to both direct and indirect use of water resources (e.g., it points out the relevant contribution of the bottling stage to different impact indicators). The comparative application of both methodologies can provide useful insights into the water-related impacts of different wine production processes and stages and support a comprehensive assessment of the best management practices, unless the differences in the methodological approaches and goals are well understood by assessors.

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3.2 Introduction

The sustainability of food production is called to have a central role in future national development strategies and action plans (FAO, 2018b) considering that, up to now, food production systems are among the leading drivers of impacts on the environment (Borsato et al., 2018b; Tamea et al., 2016). This is specifically relevant for food products which do not have nutritional value per se as they may address needs that are beyond the basic ones (Davis et al., 2016; Niccolucci et al., 2008; Sala et al., 2017) and involve social dimensions like drinking wine, beers or coffee (Notarnicola et al., 2017) or crops for energetic and textile proposes (Galindo et al., 2018). The food market is moving towards sustainability, having an increasing number of consumers asking for certified sustainable agricultural products (Davis et al., 2017; Reganold and Wachter, 2016). Following this trend, farmers are acting to modify and improve their production processes to respond to these new requests and, in this context, wine producers and vine growers have been increasingly engaged in sustainability initiatives driven by environmental concerns (Bastianoni et al., 2001; Ene et al., 2013; Sogari et al., 2016). This attitude is supported by recent European policy initiatives promoting the Circular Economy concept (European Commission, 2015), based on the maintenance of the value of products, materials and resources in the economy for as long as possible and on the minimisation of waste generation.

In the wine sector, many initiatives for sustainable wine production were promoted, like, for example, the Australian “Sustaining Success” strategy (Ridoutt and Hodges, 2017), the European Program of Sustainability, the New Zealand Winegrowing Program, the California Winegrowing Program, and many others. In California, the Californian Sustainable Winegrowing Alliance (CSWA) initiative promoted politics to solve the water scarcity that recently affected the area. The CSWA program identified strategies to decrease the general water use for producing 0.75 L of wine using agronomical best practices (Steenwerth et al., 2015). Another Californian sustainable winegrowing initiative, the Lodi Sustainable Winegrowing Commission, boosts politics of water saving under the Lodi Rules Sustainable Winegrowing framework. In Australia, the McLaren Vale Grape Wine & Tourism Association provides consultancy and services to farmers to improve decision-making systems for water management in order to increase the water use efficiency and reduce the impact of a wine farm. In Italy, the VIVA Sustainable Wine (Valutazione dell’Impatto in Vitivinicoltura sull’Ambiente) initiative, launched in 2011 by the Italian Ministry for Environment, Land, and Sea, developed a new framework to assess the sustainability based on four indicators, one of which, the “WATER” indicator or the “ACQUA” indicator in Italian (Corbo et al., 2014; Lamastra et al., 2016; Merli et al., 2018), will be further assessed in this paper.

The International Organization of Viticulture (OIV) defines sustainability in the wine sector as a “*global strategy on the scale of the grape production and processing systems, incorporating at the same time the economic sustainability of structures and territories, producing quality products, considering requirements of precision in sustainable viticulture, risks to the environment, products safety and consumer health and valuing of heritage, historical, cultural, ecological and landscape*” (OIV, 2011).

To assess and communicate the performance of an agricultural system under the perspective of the three “sustainability pillars” (i.e. economic, social and environmental pillar), quantitative indicators have been proposed as a suitable and effective mean. Among the indicators concerning the assessment of environmental impacts, the water footprint, describing the impacts of a system or product on water resources from both a quantitative and qualitative perspective, is considered in all the aforementioned initiatives for sustainable wines to assess the volume of freshwater consumed or impacted by polluting processes.

The water footprint concept was firstly proposed by Hoekstra et al., (2011) and it was implemented through the Water Footprint Assessment method (WFA). In the WFA method, the water footprint is the sum of three sub-indicators: green, blue and grey water footprint. The green water footprint is the rainwater stored in the root zone and used by plants through evaporation, transpiration, and incorporation in the biomass. The blue water is the irrigation water uptake by plants. The grey water is the amount of fresh water required to assimilate the critical pollutants to meet specific water quality standards. Subsequently, a new water footprint framework was developed by the research group of Water Use in Life Cycle Assessment (WULCA), a working group of the UNEP-SETAC Life Cycle Initiative that developed a water scarcity midpoint method to be used in Life Cycle Assessment (LCA) and for water scarcity footprint assessments (Ridoutt and Pfister, 2010). This LCA-based water footprint includes the quantification of water impacts related to freshwater use in terms of water availability footprint and water scarcity footprint, and to the water quality in terms of the ecotoxicity, eutrophication, and acidification (Bayart et al., 2010; Boulay et al., 2013; ISO, 2014). LCA impact categories concerning water impacts are related to the areas of protection "Human Health", "Resources", and "Ecosystems Quality" (Kounina et al., 2013). The two approaches for water footprinting were developed in parallel, and formalized in the framework ISO 14046, which defines WF as "*the metric(s) that quantifies the potential environmental impacts related to water and considers all environmentally relevant attributes or aspects of natural environment, human health and resources related to water, including water availability and water degradation*" (ISO, 2014). In this framework, WFA approach focuses on the water resource management and allocation, while the LCA community focuses on the potential impacts related to the water use by human activities that deprives others human users and the ecosystems (Boulay et al., 2017; Pacetti et al., 2017). The two methodologies have complementarities and can be used in synergy to promote and obtain a better and sustainable freshwater use (Boulay et al., 2013). For example, Herath et al., (2013) evaluated different approaches of water footprinting comparing WFA approach, LCA approach and a hydrological approach applied to two wine areas in New Zealand and highlighting their specific strengths and weaknesses. Lovarelli et al. (2016) analysed the water quality impact using the grey WF from the WFA method and an integrated water pollution indicator based on different LCA impact categories. They assessed the water pollution impact of the fertilization process on maize crop production with different crop management. In Jefferies et al. (2012), the comparison between the LCA and WFA methods at the product level showed no significant differences in the results when using the same data source. Previous works of LCA analysis of a bottle of wine discussed widely the impact contribution of different impact categories considered in the LCA method (Meneses et al., 2016), while water-related indicators were rarely discussed. Furthermore, most of the literature concerning environmental impacts of wine production focused on the Carbon Footprint assessment (Navarro et al., 2017) as the most widely used environmental indicator, while the role of the impact on water resource is less studied. This study is focused on the application of a water-focused LCA of one bottle of wine produced in a winery in North-Eastern Italy and on the comparison with the results provided by the "WATER" indicator proposed within the "VIVA Sustainable Wine" initiative (Corbo et al., 2014), which is receiving a growing interest in the Italian wine sector. The VIVA framework is based on four different indicators to assess the sustainability in viticulture: "AIR" based on the carbon footprint concept, "WATER" based on the water footprint concept, "VINEYARD" that evaluates the sustainability of the different agronomical practices, and "TERRITORY" that is a social-economic indicator (Lamastra et al., 2016). The VIVA WATER indicator reflects the water consumption in the

supply chain and processing for the grapevine and wine production, and the proposed methodology follows the Water Footprint Assessment approach (Hoekstra et al., 2011). It includes the direct water use of the production chain of grape cultivation and the direct water use in wine production up to bottling.
This study aims to assess the water footprint of a bottle of wine using different methodological approaches. The application of the VIVA WATER indicator is compared to the application of LCA-based water footprint focusing on the interpretation of results, in order to analyze how the outcomes of two different methodologies can support the identification of appropriate management strategies aimed at reducing water-related impacts and improving the sustainability of wine production. Considering the relevant number of indices and indicators currently applied in the assessment of environmental impacts of wine production, the novelty of this study lies in presenting a practical comparison of two water footprint methodologies with the objectives of highlighting differences and synergies, guiding the correct interpretation of their outcomes and supporting assessors and consumers in the understanding of the specificities of each approach.

3.3 Material and Methods

This study assesses the Water Footprint (WF) of a bottle of wine according to the ISO 14046 standard, which adopts an LCA approach, and compares the obtained results with the outcomes of the application of the VIVA "WATER" indicator. The detailed presentation of the latter application is beyond the scope of this paper, therefore only the results are reported here for comparison (they are available on the website of the VIVA Program with the certification of the selected wine as Sustainable Wine for the year 2017 (VIVA, VIVA la sostenibilità del vino, 2017)). The same functional unit and the same inventory dataset related to the year 2017 were considered for both the applications.
Both the methodologies consist of four phases. The LCA framework consists of: goal and scope definition, inventory analysis, environmental impact assessment and interpretation of the results (ISO, 2006a). The VIVA approach is based on the WFA framework, which includes: goal and scope definition, WF accounting, WF sustainability assessment and WF response (Boulay et al., 2013). In this section, the implementation of the four phases of the LCA framework will be described in detail and compared with the assessment of the VIVA "WATER" indicator.

3.3.1 Goal and scope

The scope of the study is to analyse the WF of a bottle of wine produced in a winery located in the Northeast of Italy, approximately 45.87° North, and 12.70 East. The climate is a sub-continental climate with warm temperature, characterized by a consistent humidity due to the proximity to rivers, the mountain area and the coast. The area is characterized by a sub-alkaline silty-clay-loam soil of flooding origin with the presence of gravel, with leaching and hydro retention problems. The farm holds different agricultural production, as for example the dairy production and the winery. The case study focused on a specific selection of wine production. The winery counts 114 hectares of vineyard, but only 7.04 ha were considered in this study, which are necessary to produce the wine blend. This study evaluates the water-related impact indicators of a bottle of wine until the bottling and the impact is defined as the sum of the impacts along the production chain. The functional unit is 0.75 L of wine that consists of a quantitative mass of inputs and outputs to produce a bottle of white wine and corresponds to the functional unit selected for the VIVA analysis. The system boundaries for the LCA-based Water Footprint calculation, from cradle (grape production) to gate (bottling), are described in

Fig. 1. The study includes all the processes in vineyard, cellar, and bottling during the year 2017. The WF assessment does not consider transportation, bottles delivering and end-of-life processes, according to the VIVA framework (Bonamente et al., 2016). As this study focuses on one-year production, the cultivation impact of the early age plantation is not considered. We considered the existing assessment of VIVA WATER indicator for a bottle of wine (VIVA, VIVA la sostenibilità del vino, 2017) and we performed the assessment of water-related impacts according to LCA approach using the same dataset in order to compare and interpret the implications of the outcomes of the two methodologies at the decision-making level.

3.3.2 Inventory phase

Life Cycle Inventory

The inventory phase in LCA is a very relevant and sensitive phase because it influences the quality of the entire assessment. During this phase, primary data from the winery and from the vineyard were collected. When primary data were not available for a specific process, the allocation by mass was used. In LCA, missing data can be replaced with estimation from the proportion of mass (physical) or the economic value in every process (Notarnicola et al., 2017). In this study, water and electricity consumption and waste disposal data were allocated according to the proportion of mass produced in each process. The LCA-based assessment, differently from the VIVA methodology, requires a detailed description of all processes included in the life cycle stages and the identification of inputs and output of materials, energy, and waste for each of them. Therefore, the chart of wine production describing the different phases of vineyard, cellar, and bottling including different processes with related inputs and outputs was developed (Figure 1).

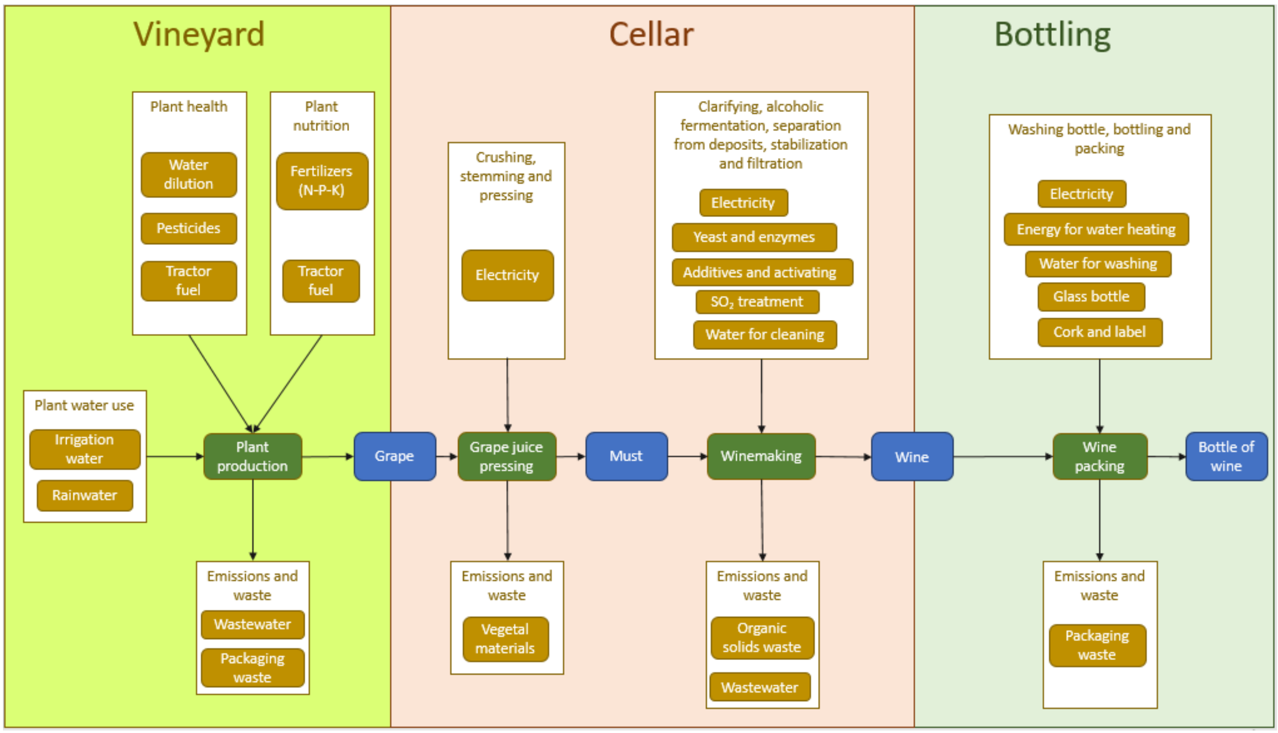

Figure 1. System boundaries of the wine bottle production. The vineyard stage includes three processes (plant health, plant nutrition, and plant water use), the cellar stage includes two processes (grape juice pressing and winemaking), and the bottling stage consists of the wine packing process.

The **vineyard stage** considers the plant production that relies on three processes, related to plant health (pest control), plant nutrition (fertilization) and plant water use (water required by the plants). The vineyard area consists of 7 ha surface, where different rates of pesticides, fertilizers, and irrigation volumes were applied. The quantification of freshwater consumption included the direct consumption associated with irrigation and the dilution volume used during the application of pesticides (plant health process). In addition, the study considers the water consumed by the soil-water system evapotranspiration in terms of rainwater and irrigation water. Two of the field plots were rainfed, while the other two were irrigated. As far as pesticides are concerned, they were inventoried following the chemical families on the basis of their chemical properties. In the case of fuel consumption, the activities considered are the use of tractors during fertilization and pesticide spraying. The methods to determine the emissions to water (L) were assessed using the method by Franke et al., (2013), including the emissions to surface and groundwater in a unique coefficient of leaching run-off fate (α), i.e. :

$$\mathrm{L}_{[\mathrm{i,p}]} = \propto * \mathrm{Appl[t]} = [\mathrm{Volume}] \tag{1}$$

where *Appl[t]* is the chemicals application at the time [t] and α is the leaching-runoff coefficient improved by Franke et al., (2013) as:

$$\alpha = \alpha_{\min} + \left[\frac{\sum_{\mathrm{i}}(\mathrm{s_i} * \mathrm{w_i})}{\sum_{\mathrm{i}}(\mathrm{w_i})}\right] * (\alpha_{\max} - \alpha_{\min}) \tag{2}$$

where α_{max} and α_{min} are the maximum and the minimum leaching-runoff factor, respectively. The s_i is the leaching-runoff potential and w_i is a weighting factor, and they vary according to the properties of the chemical substance. The α_{max} for nitrogen is fixed at 0.25, at 0.05 for phosphorus and 0.01 for pesticides, while the α_{min} is fixed at 0.01 for nitrogen, at 0.0001 for phosphorus and pesticides. The toxicity of the pesticide to the selected non-target organisms was assessed using eco-toxicological and toxicological data derived from the Pesticides Properties DataBase (University of Herthfordshire, 2013a). Climate data and soil features of the location were used for the emissions assessment. The wastewater produced during the cleaning and washing of sprayer machines is reused in the next treatment, and no emissions to environmental compartments were considered in this case.

The **cellar stage** involves two processes: grape juice pressing and winemaking. During the grape juice pressing, the stalks, pips, and skins are recycled as compost and reused as co-product in another farm production. The organic waste from cellar stage was considered as compost in the fertilization process, while the emissions from organic waste in other processes outside winery were not included in the study. The winemaking process includes the sub-processes of clarification, fermentation, stabilization, and filtration in a single process because data were unavailable for the single sub-process and allocation was not possible. In the winery, the wastewater is treated and reused outside the wine production and within the farm. The inputs used in this study are listed in Table 1 divided by different stages and processes.

The **bottling stage** is mentioned as a different step of wine production due to its importance on the impact assessment. The only process included in this stage is the wine packing that consists of three processes: washing bottles, bottling, and packing.

Table 1. Inventory of primary data used for each production stage to compute the WF LCIA. The table includes inputs of materials and energy from technophere and from nature, and outputs from each process. The (*) symbol indicates the inputs were used both in the LCA and in the assessment of VIVA WATER indicator.

Stage/Processes	input	Metric	
Grape production		*per 1 ha*	
Plant nutrition	N-fertilizer (urea)*	kg	92
	N-fertilizer (sulfate ammonium)*	kg	21
	P-fertilizer (from manure)*	kg	25
	K-fertilizer (potassium sulfate)*	kg	80
Plant health	Ametoctradin*	kg	0.3
	Boscalid*	kg	0.6
	Buprofezin*	kg	0.4
	Chlorantraniliprole*	kg	0.04
	Chlorpirifos*	kg	0.5
	Dimethomorph*	kg	0.6
	Dithianon*	kg	0.8
	Folpet*	kg	4.7
	Potassium phosphonate*	kg	2.8
	Glyphosate*	kg	0.7
	Potassium phosphite*	kg	12
	Metalaxyl-m *	kg	0.3
	Metiram*	kg	0.8
	Pyrimethanil*	kg	0.7
	Copper oxide*	kg	1
	Copper carbonate*	kg	4.6
	Copper sulfate*	kg	0.8
	Thiamethoxam*	kg	0.04
	Sulfur*	kg	33
	Fuel (diesel oil) for tractor	L	800
	Water for dilution*	m^3	0.25
Plant water use	Irrigation*	mm	90
	Evapotranspiration*	mm	387.5
	Electricity	kWh	380
Output	Grape	kg	100.8
	Packaging waste	kg	24.3
Grape juice pressing		*at cellar*	
Crushing, stemming and pressing	Grape*	kg	70950
	Electricity	kWh	630
output	Must	L	55620
	stalks, pips, and grape skins	kg	15330
Winemaking			
Clarification, fermentation and filtration	Must	L	55620
	Electricity	kWh	6360
	Electricity for cooling	kWh	5730
	Electricity from photovoltaic panels	kWh	5770
	Yeast	$g\ hL^{-1}$	20
	Activator solutions	$g\ hL^{-1}$	40
	Potassium metabisulphite	$g\ ton^{-1}$	1.5
	Water for washing*	m^3	104
output	Wine	L	53200
	Organic solid waste	kg	2420
	Solid waste	kg	690
Wine packing *washing bottle, bottling and packaging*	Wine	L	53200
	Washing water*	m^3	87
	Electricity	kWh	220
	Energy fuel for heating	MJ	212 e^4

	Number of bottle (0.75 L)*	p	70933
	Label	kg	284
	Cork	kg	426
	Capsule	kg	106
Output	Bottle of wine	p	70933
	Solid waste	kg	32

Inventory for VIVA WATER indicator

The calculation of VIVA WATER indicator is based on the same functional unit used for the LCA, that is 0.75 L wine bottle. The inventory dataset used in this study derived from the VIVA calculation during the product certification (VIVA, VIVA la sostenibilità del vino, 2017). The systems boundaries are also the same used in the LCA, i.e. a cradle to gate approach from vineyard (cradle) to winery (gate). The system boundaries are described in Figure 2, where the five sub-indicators are linked with the parameters needed to describe each specific process included in the analysis. The calculation of the sub-indicators will be further described in section 2.3.2. Specifically, the blue WF in the vineyard refers to irrigation parameters, dilution water volume for pesticides application and cleaning volume for sprayers. The green WF takes into consideration the crop features, soil and climate parameters. The grey WF in vineyard is a more complex sub-indicator, which includes chemical and soil parameters, application techniques and presence and volume of the nearby water body. The only sub-indicator concerning the winery stage is the blue WF, which includes the consumption of drinking water for washing and cleaning processes.

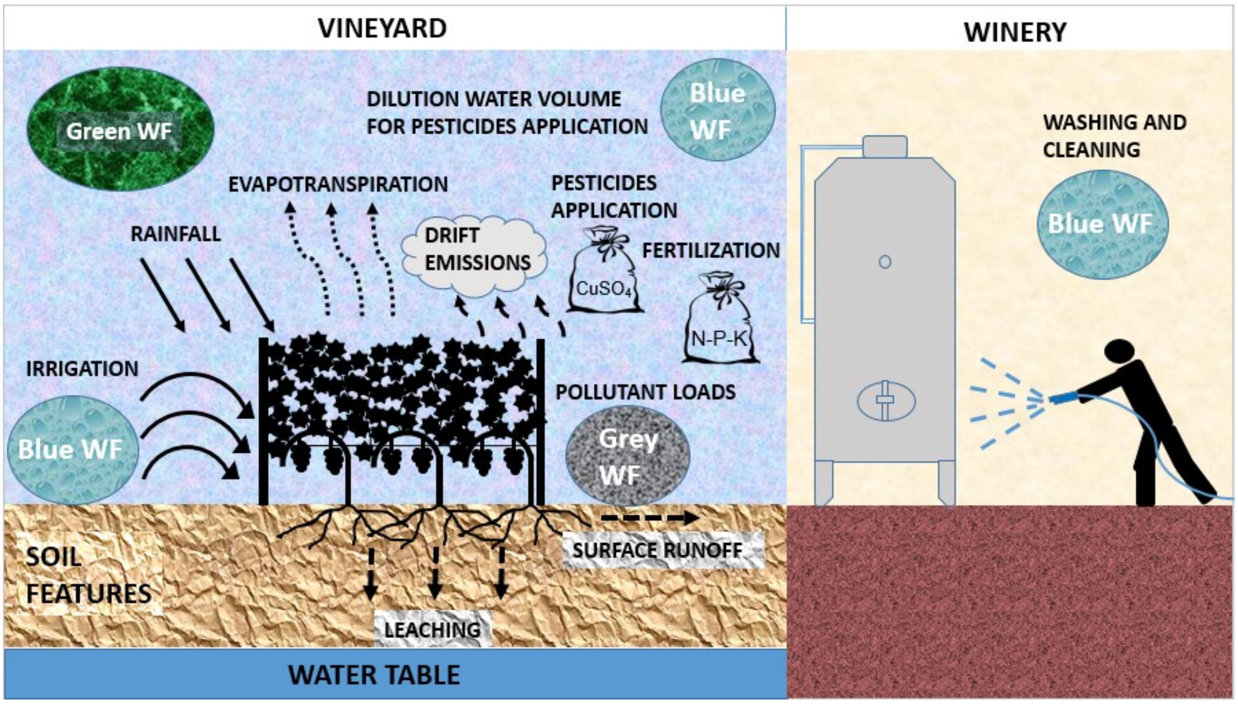

Figure 2. Chart of input and processes analysed for the VIVA WATER indicator assessment. VIVA inventory phase considers different soil, climate and crop parameters to assess water consumption and water pollution in terms of consumptive water footprint in the vineyard stage, while in the winery it considers only the blue WF due to the washing and cleaning processes.

In Table 2, the list of inputs used for the VIVA water footprint assessment is reported. The VIVA inventory dataset for the vineyard phase is more demanding than in the LCA, since more importance is given to the water pollution by pesticide application. In the vineyard phase, the distance and the volume of the nearest water body are considered to determine the concentration of the pollutant into the surface water body; the annual recharge of the water table is taken into consideration to estimate the concentration in the groundwater. The inventory phase gives more attention to agricultural activities that can produce hotspot sources of pollution. For example, the VIVA assessment requires also qualitative information such as the type of sprayer machine and relative anti-drift and runoff measures and devices; it considers the phenological state as well, in order to evaluate the interception by the canopy that can influence the drift of pesticides during spraying. The number of pesticides' applications and the interval between them is also taken into account to determine the predicted environmental concentration (Lamastra et al., 2014). The water consumed in the winery is intended as the sum of the water consumption in the cellar and in the bottling stages. The inputs analyzed in the cellar are the water volume consumed during the winemaking and wine-packing processes, while the energetic consumption and all the other indirect emissions of raw materials used in this phase are not considered in the VIVA WATER indicator. The emissions generated from the wastewater treatment are calculated only for the grape production phase, because it is assumed, accordingly to the legislative framework, that the wine company has the wastewater treatment plant, and the pollutant load is compliant with the standards of water quality before discharging in the public network or in the environment.

Table 2. Inventory list used for VIVA "WATER" for direct water use during the year of evaluation. The (*) symbol indicates that the inputs were used both in the LCA and in the assessment of VIVA WATER indicator.

	Inventory inputs	*Value*	*Unit*
Vineyard information	Grape harvesting	10000	kg ha^{-1}
	Slope	2.5	%
water body information (average of the fields)	Distance from the closest water body	359	m
	Length of the side of the water body near the vineyard	220±160	m
	Width water body	11±3	m
	Depth water body	2.55±1.4	m
	Water table depth	1.5	m
Soil parameters	Sandy	32.7	%
	Clay	22.8	%
	Organic carbon	1.6	%
	Skeleton	26	%
Irrigation	Irrigation volumes*	90	mm
Fertilization	N-fertilizer (urea)*	92	Kg ha^{-1}
	N-fertilizer (sulfate ammonium)*	21	Kg ha^{-1}
	P-fertilizer (from manure)*	25	Kg ha^{-1}
	K-fertilizer (potassium sulfate)*	80	Kg ha^{-1}
Pesticides application	Ametoctradin*	0.3	Kg ha^{-1}
	Boscalid*	0.6	Kg ha^{-1}
	Buprofezin*	0.4	Kg ha^{-1}
	Chlorantraniliprole*	0.04	Kg ha^{-1}
	Chlorpirifos*	0.5	Kg ha^{-1}
	Dimethomorph*	0.6	Kg ha^{-1}
	Dithianon*	0.8	Kg ha^{-1}
	Folpet*	4.7	Kg ha^{-1}
	Potassium phosphonate*	2.8	Kg ha^{-1}
	Glyphosate*	0.7	Kg ha^{-1}
	Potassium phosphite*	12	Kg ha^{-1}

	Metalaxyl-m *	0.3	Kg ha^{-1}
	Metiram*	0.8	Kg ha^{-1}
	Pyrimethanil*	0.7	Kg ha^{-1}
	Copper oxide*	1	Kg ha^{-1}
	Copper carbonate*	4.6	Kg ha^{-1}
	Copper sulfate*	0.8	Kg ha^{-1}
	Thiamethoxam*	0.04	Kg ha^{-1}
	Sulfur*	33	Kg ha^{-1}
	Dilution water volume for pesticide spraying applications*	0.25	m^3 ha^{-1}
	Average volume of washing sprayers machine*	0	m^3/wash
Climatic information	Calendar of weather data	-	dd/mm/yyyy
	T med	13.7	°C
	T min	8.3 ± 21	°C
	T max	19.2 ± 17	°C
	Cumulate Rainfall	1179	mm
	Average Daily Rainfall	3.2	mm
	Average Humidity	76	%
	Daily Wind speed average	2	m s^{-1}
Winery	Number of bottles produced each year *	70930	N° bottles
Washing and cleaning	Total water consumed in winery m^3*	3835	m^3

3.3.3 Impact assessment

LCIA

The Life Cycle Impact Assessment is the stage of LCA that analyses the impact of a product or a process using the inventory analysis results (Iannone et al., 2016). The LCIA phase provides information for the interpretation phase. The choice and the evaluation of the impact categories may introduce subjectivity into the LCIA (ISO, 2006a). The ISO 14040 and 14044 define the guideline to assess the environmental aspect and the potential impact for goods and services (ISO, 2006b, 2006a). The SimaPro 8.4 software was used to implement the LCIA of the selected indicators with the use of the Ecoinvent 3.3 and Agri-footprint 3.0 databases.

The LCA approach differs from the VIVA approach by the definition of impact categories for the water footprint assessment. In the LCA analysis, the water footprint assessment is based on different impact categories, related to both the consumption-to-availability perspective and the water degradation perspective (Table 3). This study followed the ISO 14046 framework and the guidelines provided by the WULCA research group of UNEP-SETAC to assess the water footprint on different areas of protection. The guideline defines the assessment of the water footprint method referring to different impact categories as the freshwater use, in terms of water scarcity and deprivation of the resource, the ecosystem quality and the toxicity on human health (Boulay et al., 2013; ISO, 2014; Kounina et al., 2013). The impact categories are chosen according to the ISO 14046 guideline for their capability to represent the impact on water related to the wine production.

Table 3. Methods involved in LCA-based Water Footprint according to ISO 14046.

Methods	Indicators
AWARE	Available Water Remaining
(Pfister et al., 2009)	Water Stress index (WSI)
Impact 2002+ (Jolliet et al., 2003)	Aquatic acidification
ILCD 2001 Midpoint (Rosenbaum et al., 2008)	Human toxicity; Freshwater ecotoxicity
ReCiPe 2016 Midpoint Hierarchic	Freshwater eutrophication

The right choice of a suitable impact category mainly depends on which question the study might want to answer. In Table 3, the set of chosen impact categories is described. The freshwater use indicator can be assessed through many approaches. A water-related impact indicator is defined as withdrawal to availability (WTA), consumption to availability (CTA), and availability minus demand (AMD) (Kounina et al., 2013). For the purpose of comparing the consumptive water use with the withdrawal of water from a region, the novel AWARE method was chosen (Vázquez-Rowe et al., 2017). The "available water remaining" assessed with the AWARE method stands for the potential impact of water deprivation by another users or compartment in a specific region (Boulay et al., 2017). The indicator is the inverse of the difference between the water availability minus water demand, therefore it defines the water remained (Jolliet et al., 2018).

In this study, we use also the Water Stress Index (Pfister et al., 2009) that calculates the water impact on the consumption-to-availability perspective of freshwater deprivation, corresponding to the blue water in the WFA methodology. The Water Stress Index assesses the freshwater consumption at the midpoint level for all three areas of protection: Resources, Ecosystems and Human Health. The cause-effect chain regarding the Human Health is connected to socio-economic data, while the effects regarding the Resource concern the impact of freshwater consumption through the freshwater renewability rate, and the cause-effect chain relevant to ecosystem services relies on the decrease of terrestrial biodiversity due to the water consumption. This Water Stress Index approach is defined as the ratio between the total annual freshwater withdrawals to hydrological availability (Pfister et al., 2009). The difference between the method AWARE and Pfister et al., (2009) lies in the different assumptions to assess and understand the Water Scarcity Footprint referring to different water use in the study area (Ridoutt and Hodges, 2017). The availability minus demand approach refers to the water remaining after water subtraction and consumption from different sub-compartments in which the environmental water requirement is met (Boulay et al., 2017); while the Water Stress Index refers to the blue water flow consumed along the production chain.

As recommended in the ISO 14046 framework, the method of Water Footprint includes also indicators dealing with the water quality. The impact categories addressing the water degradation selected in this study are aquatic acidification, freshwater eutrophication, freshwater ecotoxicity, and human toxicity (Villanueva-Rey et al., 2018, 2014). In this study, both the human toxicity and freshwater ecotoxicity are analyzed according to the ILCD framework recommended by JRC of the European Commission (JRC European commission, 2011). The freshwater ecotoxicity and human toxicity indicators are based on the USEtox model, developed by the UNEP/SETAC Working Group (Rosenbaum et al., 2008) to address the damage categories of ecosystem services and human health. The model assesses the characterization factors for freshwater ecotoxicity and human toxicity. The toxicological emissions are linked to the environmental impact through a cause-effect

chain, which considers the environmental fate, exposure, and effects for a unit of chemical emitted(Rosenbaum et al., 2008). The assessment of freshwater ecotoxicity is based on the concentration-response effects on aquatic species. In detail, the unit of measure of Comparative Toxic Unit (CTUe) estimates the potential affected fraction of aquatic species (PAF) integrated over time and volume per unit mass of a chemical emitted (PAF m^3 day kg^{-1}) in the ecosystem (Renaud-Gentié et al., 2015). The assessment of human toxicity is based on the quantity of daily human intake of a specific substance that is related to negative effects (or potential risk) to the human health through inhalation or ingestion. The human toxicity is calculated as CTUh or Comparative toxic unit and considers both chronic carcinogenic and acute non-carcinogenic toxicity (Rosenbaum et al., 2008).
The aquatic acidification is a midpoint indicator developed by the Swiss Federal Institute of Technology in Lausanne (EPFL) and refers to the impact Assessment of Chemical toxicity (Impact 2002 +) method in terms of SO_2 equivalent emitted into the air (Guinée et al., 2003). It refers specifically to the damage category of ecosystem services and the IMPACT 2002+ method calculates the potentially disappearing fraction of aquatic species affected by a unit of emission of aquatic acidification (Humbert et al., 2002; Jolliet et al., 2003).
The eutrophication indicator refers to the ReCiPe 2016 method as suggested within the framework of the ILCD recommendations. It expresses the impact on the freshwater system as kg of phosphorus (P) equivalent. The indicator is the freshwater eutrophication midpoint factor that relates the fate factor for an emitted substance into a certain compartment (surface water, groundwater, etc) with the world average fate factor for phosphorous emissions to freshwater.

VIVA framework

The "Water" indicator in VIVA is based on the methodology of the WFA, and it includes two sub-indicators of quantitative water consumption, the green water footprint and blue water footprint, and one sub-indicator for the water quality, the grey water footprint. The WF sub-indicators refer to five different outcomes: green WF in the vineyard, grey WF in the vineyard, blue WF for irrigation, blue WF for plant health in vineyard, and the blue WF in the winery. The blue water footprint is the consumption of freshwater from surface and groundwater over the supply chain of a product. It is the total annual direct freshwater consumed along the supply chain per bottle of wine (Lamastra et al., 2014). It considers the irrigation volume for the growing season, the dilution volume for active substances spraying, and the water volume for washing the equipment. The green water corresponds to precipitation and soil moisture consumed by grape. The green water volume depends on the crop water requirement and the available soil moisture. The green water is the summary of the daily crop water requirement depleted by a water stress coefficient (ks). The green water is estimated on a daily basis, by summing all the volumes of rainwater lost through crop evapotranspiration. Crop interception and rainwater percolation are subtracted daily from the precipitation value obtained from meteorological data. (Lamastra et al., 2014).

$$Etc = Et_0 * kc * ks = \frac{[Volume]}{[time]}$$

The grey water is the volume of freshwater that is required to assimilate the load of pollutants given natural background concentrations and existing ambient water quality standards. The VIVA assessment considers the

direct impact and the grey WF is given by the maximum of the dilution volumes of the different pollutants that reach the water body through leaching, runoff, and drift (Bonamente et al., 2016; Lamastra et al., 2014).
Grey WF is calculated using a stepwise procedure. The Predicted Environmental Concentration (PEC) is determined for each water compartment (surface water, groundwater) and for each contamination mechanism (leaching, run-off, drift), using models that take into account differences in environmental conditions and application factors. The PEC is compared to ecotoxicological (No Observed Effect Concentration, NOEC) or legislative threshold respectively for surface and groundwater to obtain the dilution factor (Lamastra et al, 2014).
The VIVA calculator for the WF indicator presents the novelty of a tier III method for the grey WF, where the respect of the nearest water body and different contribution for pollutants load are considered.
The parameters mostly affecting the grey WF are the distance from the water body, the fertilization rate, the amount and the eco-toxicological behavior of the active ingredients used (Bonamente et al., 2016).

3.3.4 Interpretation phase

During the interpretation phase, the results of the LCA have to be critically evaluated and discussed. In this work, both the LCA and VIVA results are first discussed individually, then, as a second step, a comparison of the outcomes of the two methodologies is provided in order to discuss their implications at the decision-making level. The two methodologies address water-related impacts using different indicators that are not directly comparable because they are based on different assumptions with various ranges and scales. This work is aimed at comparing the outcomes of the two methodologies in terms of conclusions about the possible mitigation strategies and measures to be adopted.

3.4 Results

3.4.1 Relative contributions to water-related impacts within the Life Cycle Impact Assessment

Figure 3 illustrates the relative contribution to each LCA impact category of each process within the considered life cycle stages (i.e., vineyard, cellar and bottling). The vineyard stage shows a great impact on water use. The cellar or winery stage shows the lowest contribution in all the impact categories, while the bottling stage seems the mostly impacting stage, especially for toxicity-related indicators. The seven indicators reported in Figure 3 define the total environmental impact on the water resource as estimated through LCA. The results for the AWARE indicator and the Water Stress Index are characterized by a similar contribution from individual processes. In fact, the stage of vineyard gives the greatest contribution in terms of water use with 1.316 m^3/FU for the AWARE and 8.53 E-03 m^3/FU for the WSI, with the 92% and 86% of impact on freshwater use, respectively. The water consumption from irrigation, evapotranspiration, and use of raw material for grape production has a higher impact than the water consumption during the winemaking or the packing. Regarding water pollution aspects, the vineyard stage shows a high contribution (52 %) to freshwater ecotoxicity with 2.26 E+01 CTUe/FU. This impact is mainly due to the use of chemicals during the crop season. The vineyard stage includes three different processes involved in plant production: plant water use, plant nutrition, and plant health. The plant water use from natural sources consists of the crop water consumption through irrigation and rainfall by evapotranspiration. The process of plant nutrition is responsible only for a low percentage of the overall impact. Considering only the fraction of impact associated with the vineyard stage, the plant nutrition contributes for the 40% in the case of aquatic acidification, the 27% in the case of eutrophication, and the 40.5% to human toxicity with non-carcinogen effects (Fig. 3). Nevertheless, considering the overall impact, the final contribution of plant nutrition is low, and it becomes the 7.9% for aquatic acidification, the 2.8% for the eutrophication, and the 1.5% for human toxicity (non-carcinogen effects). In addition, the plant health process gives a relevant contribution to water pollution in the vineyard. The plant health process contributes for the 93% of impacts in vineyard in the case of freshwater ecotoxicity, while it shows a smaller contribution to vineyard impacts in the case of eutrophication, aquatic acidification and human toxicity (41%, 31%, and 38%, respectively). Looking at the contribution to the overall impact, the plant health shows a quite limited contribution: 4.4% for eutrophication, 6% for aquatic acidification, and 3.6% for human toxicity (carcinogenic effects). Plant health shows a greater contribution on the ecotoxicity impact with a 48% on the total impact related to the production of a bottle of wine.

The cellar stage records only 0.124 m^3/FU for the AWARE and 0.148 E-03 m^3/FU for the WSI methods, representing the 8.6% and 15% respectively. Moreover, the cellar stage has the lowest contribution to the overall impact on WF. The grape pressing process shows even a very small impact contribution, associated mainly with the energetic consumption, due to the short time spent for the process itself. The winemaking process has also a low contribution to the overall impact. There is a low water use impact in the cellar: even if the process consumes water for washing and cleaning the working machines, the water is indeed treated and reused in other processes in the farm. The relative overall contribution of winemaking to aquatic acidification impact corresponds to the 6%, while for the wine packing process it is equal to 71%. The eutrophication gains a low impact contribution from vineyard (11%) and cellar stages (5%), while it is the 84% for the wine packing process (bottling stage).

The bottling stage has a low overall contribution equal to 7% for the AWARE indicator and 14% for the WSI. Looking at the water quality impact, the wine bottling process shows generally a high relative contribution. It is important to distinguish the high contribution to water quality impact of the bottling stage in comparison with the cellar stage. In fact, in all the impact categories related to water pollution, the bottling stage provides more than 70% of the impact, with the exception for freshwater ecotoxicity, where the bottling contribution is equal to 45%. The relative contribution of wine bottling is high for the aquatic acidification (71%), for the freshwater eutrophication (83%), and for human toxicity with carcinogenic and non- carcinogenic effects (88% and 95% respectively). This outcome demonstrates that the bottling stage is the stage that mostly contributes, with respect to previous stages, to the overall impacts on water quality associated with the production of a bottle of wine.

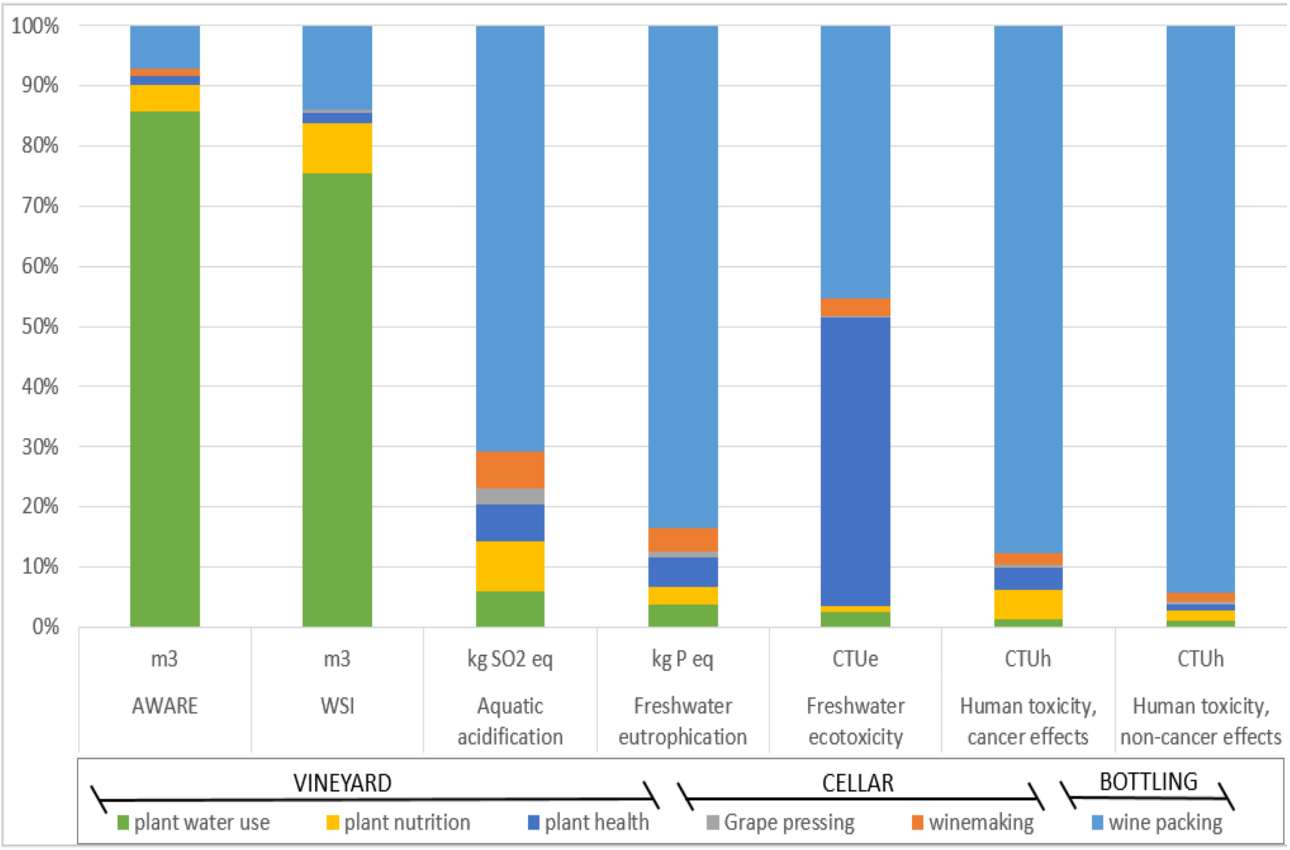

Figure 3. LCIA analysis of processes. Contribution (%) of different processes during vineyard (plant nutrition, plant health and plant water use), cellar (grape pressing, winemaking) and bottling (wine packing) stages considering different LCA impact categories.

3.4.2 Relative contributions to water-related impacts within VIVA WATER indicator

The results of the inventory analysis performed for the VIVA WATER indicator were used to identify the hotspots of water consumption and water pollution (VIVA, 2017). Fig. 4 shows the contribution of different sub-indicators to the overall WF assessment. The green WF related to the vineyard stage contributes to the 83% of the overall WF, which is associated to the plant evapotranspiration.

The blue WF is constituted by the water used for irrigation (corresponding to the 15% of the overall WF) and by water used for dilution during the phytosanitary plant treatment (corresponding to the 0.2% of the overall WF). The blue water volume consumed in the cellar represents only 0.5% of the overall WF.

The grey WF contributes only to the 2% of the total WF and represents the water volume needed to reduce the concentration of released pollutants in the water body below the selected ecotoxicological threshold.

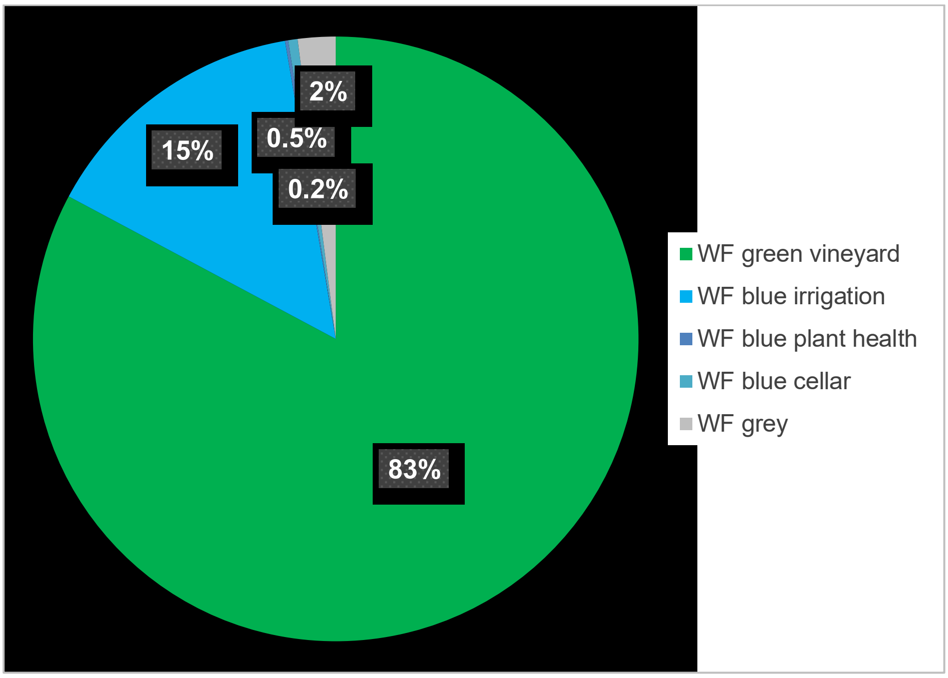

Figure 4. Relative contribution (%) of different processes to the Water Footprint (expressed in L) of 1 bottle of wine (0.75 L) according to VIVA methodology. Data available at (VIVA, 2017).

3.5 Discussion

In this section, the results of the two methodologies are compared and discussed. The LCA methodology analyses the results at the impact assessment phase, while VIVA methodology relies on water use indicators (green, blue, grey water) in the inventory phase in agreement with the WFA approach (Boulay et al., 2013). Since the two methodologies are based on different approaches and assumptions, the comparison is mainly focused on the understanding of the relative contribution of different processes and stages to the overall impacts and on the identification of the critical management aspects of the wine production as highlighted by the two methodologies.

3.5.1 *Synthesis of the LCIA and the VIVA inventory analysis and results interpretation*

Table 4 summarizes the water-related impacts associated to the production of a bottle of wine according to the VIVA and the LCA-based assessment. In this table, a different subdivision of productive stages is set up according to the VIVA inventory phase. The vineyard and the winery stages are recognized as the most important stages during wine production. The sub-total value for each stage and for each process is given; the total value of each stage is the sum of the values of processes composing that stage.

The vineyard stage includes four indicators corresponding to the processes of evapotranspiration and irrigation (plant water use), plant health, and plant nutrition. The vineyard stage causes the greatest impact on water consumption according to both VIVA and LCA methodologies. The crop evapotranspiration process gives the highest contribution to the consumptive water impact. In VIVA, the evapotranspiration process, which corresponds to the green water, is responsible for the greatest volume of freshwater use with 988 L per bottle, equals to a contribution of 83% of the total impact. In the LCIA, the AWARE indicator counts 988 L per bottle, while the WSI index calculates 6.01 L per bottle that is equal to the 69% and 60% of the total impact associated to evapotranspiration, respectively. Those values are in line with results of previous studies, where the green

WF has a relevant contribution to the overall impact on freshwater use, corresponding usually to a contribution of more than 60% (Bonamente et al., 2016; Lamastra et al., 2014). On the other hand, the water depletion in vineyard using the LCA approach counts a contribution between 40% and 80% according to the direct and indirect water resources considered in inputs (Meneses et al., 2016; Quinteiro et al., 2014; Villanueva-Rey et al., 2018).

The water volume used for irrigation is considered as an input different from rainwater and it is accounted as blue water. This process counts a water use of 173 L per bottle under the VIVA blue WF calculation and a water use of 243 L per bottle and 1.48 L per bottle under the AWARE and WSI indicators, respectively. The value differs from the methodological point of view, in VIVA, only direct consumption is considered for irrigation blue water, while in LCA even indirect water use are considered. Therefore, the WF of the irrigation process is similar according to the different methodologies in terms of percentage contribution; VIVA quantifies an overall contribution to water consumption from irrigation of the 15%, while AWARE and WSI indicators attribute, respectively, a contribution of 17% and 14% and considering direct and indirect water consumption. In the LCA, the indirect water impact of the fertilization process is taken into account within the AWARE and WSI method and it corresponds to an overall contribution of 4.4% and 8.4%, respectively. Furthermore, the plant health process is responsible for a contribution of 1.5% for the AWARE and 1.9% for the WSI, corresponding to the water use for the pesticide spreading for plant protection.

Within the VIVA approach, only the volumetric indicator of grey WF concerns impacts on water quality. The VIVA grey WF requires a great number of field data to evaluate the water pollution, especially for the processes of plant health and plant nutrition, which require the use of toxic chemical substances that might enter water bodies. The contribution of grey WF resulted to be only 2% of the total WF, but entirely associated with the vineyard stage. On the other hand, the LCA results show as vineyard stage contribution corresponds to 20% of acidification, 11% of eutrophication, and 52% of ecotoxicity. Since LCA considers also indirect impacts on water quality, several stages in fact contributed to these indicators related to water quality. Looking at previous studies, we can notice that in the work of Bonamente et al. (2016) the grey WF associated with the vineyard stage corresponds to the 15% of the overall WF. This difference in the relative contribution of the grey WF should, however, be interpreted according to the absolute value of the total WF (578 L, versus a value of 1193 L in this study) and the different system boundaries. Previous LCA studies show for the vineyard stage a contribution in the range 23-45% to aquatic acidification and in the range 29-77% to freshwater eutrophication (Meneses et al., 2016; Point et al., 2012). The significant variability might be associated with the annual variability, which depends on local climatic conditions and different territories (Ferrara and De Feo, 2018).

Looking at the winery or cellar stage, this is composed of three processes for the LCA analysis and is represented as a single stage in the VIVA assessment, as shown in Table 4. The grape pressing is the less impacting process in each of the considered LCA-based indicators, and this impact is mainly due to the electricity consumed for the process. The winery or cellar stage contributes, as blue water, for 0.5% to the overall VIVA WF with 6 L of water per bottle. In the LCA, the winery stage is responsible for 8.6% (124 L/bottle) of the total estimated impact for AWARE, and for the 14.7% (1.48 L/bottle) of the total estimated impact for WSI. According to literature, the cellar stage has the lowest contribution on water deterioration with a contribution ranging between 5% and lower than 1% depending to the indicator of water quality used in the assessment (Lamastra et al., 2014; Meneses et al., 2016; Point et al., 2012; Rinaldi et al., 2016).

The wine packing process (corresponding to the bottling stage during the LCA analysis) is typically the most impacting process in the wine production, not only in terms of water pollution, but also in terms of water use (Navarro et al., 2017; Pacetti et al., 2017; Sogari et al., 2016). The indirect impact on water quality is given by the use of glass packaging and the use of diesel for heating water, as mentioned in section 3.1. The LCA indicators related to Human toxicity are generally high for the wine packing process. For example, the overall contribution of the wine packing process is equal to 87.6% in the case of human toxicity-carcinogenic effects and 94.8% for human toxicity - non-carcinogenic effects.

Table 4. Combining the quantitative water use by processes contribution using the VIVA and LCA methodologies.

		VIVA UNIT/BOTTLE			**LCA UNIT/BOTTLE**						
	Process	green WF	blue WF	grey WF	**AWARE**	**WSI**	**Aquatic acidification**	**Freshwater eutrophication**	Freshwater ecotoxicity	Human toxicity, cancer effects	Human toxicity, non-cancer effects
	0.75 L wine	m^3	m^3	m^3	**m^3**	**m^3**	**kg SO_2 eq**	**kg P eq**	CTUe	CTUh	CTUh
VINEYARD		**0.988**	**0.175**	**0.024**	**1.351**	**8.79E-03**	**8.67E-04**	**5.56E-05**	**2.26E+01**	**8.12E-08**	**6.69E-09**
	Green water	0.988			0.988	6.01 E-03					
	Irrigation contribution		0.173		0.243	1.48 E-03	2.49E-04	1.64E-05	1.09E+00	9.63E-09	2.02E-09
	Plant health		0.002	0.024	0.021	0.19 E-03	2.60E-04	2.13E-05	2.11E+01	2.95E-08	1.84E-09
	Plant nutrition				0.064	0.85 E-03	3.36E-04	1.39E-05	4.12E-01	3.85E-08	2.63E-09
WINERY			**0.006**		**0.124**	**1.48 E-03**	**3.36E-03**	**4.26E-04**	**2.12E+01**	**7.31E-07**	**1.67E-07**
	Grape pressing				0.002	0.03 E-03	1.12E-04	5.36E-06	1.14E-01	3.85E-09	6.36E-10
	Winemaking				0.017	0.01 E-03	2.62E-04	1.91E-05	1.29E+00	1.57E-08	2.47E-09
	Wine packing				0.105	1.43 E-03	2.99E-03	4.02E-04	1.98E+01	7.11E-07	1.64E-07
TOT WF FOR 0.75 L OF WINE			1.193		1.475	10.3 E-03	4.23E-03	4.82E-04	4.38E+01	8.12E-07	1.73E-07

3.5.2 *Complementarities and synergies between VIVA and LCA approach*

The comparison of the two methods was aimed at identifying possible complementarities and synergies with respect to the user perspective. The interpretation of the results of the two methodologies shows how the different outcomes can suggest different mitigation strategies to reduce water-related impacts. Comparing the inventory phase, the two methodologies focus on different inventory elements. For example, the grey WF in VIVA takes into account different compartments (groundwater, surface water) and mechanisms (leaching, runoff, drift) for chemicals environmental concentration in water bodies. The distance to the water body and the volume of the water body are some of the features considered in VIVA to calculate the predicted environmental risk on surface water due to runoff and drift. The drift mechanism is related to the phenological crop step (crop interception) and to the use of mitigation strategies such as the use of correct nozzles, tunnel sprayers, precision instruments to reduce drift effects, or the presence of hedgerow barriers in the vineyard (Lamastra et al., 2016). The grey WF calculation considers also the interval time between the chemical applications and the half-life in the soil for the substances in order to evaluate the emissions on water. All the above-mentioned elements cannot be considered and evaluated in the LCA-oriented assessment. In addition, another VIVA sub-indicator that is not typically incorporated in an LCA analysis is the green WF. The green water calculates the rainfall water used by grapevine through the actual crop evapotranspiration assessed with a soil-water balance. In this study, the green water was considered as part of the impact on water for the vineyard stage in both the VIVA and the LCIA analysis. As mentioned in Quinteiro et al. (2014) and Villanueva-Rey et al. (2018), the green water consumption is considered as an important factor affecting the water cycle on land. It is difficult to identify innovative mitigation strategies aimed at reducing the crop evapotranspiration;

however, the use of mulching on the plant rows or the use of different grass species between rows in the vineyard can enable a reduction in green WF.
Differently, the direct water use in the WFA approach is accounted for in the blue WF, which is calculated separately for different processes. The irrigation blue water volume considers the direct water volume used by crop, while the blue water volume for plant health is the dilution volume used during spraying of chemicals and the water consumed for washing the spraying machines. In the cellar or winery stage, the blue WF is the only indicator that assesses the water impact at this production step. The LCA approach, instead, gives a more appropriate and detailed analysis of the cellar and bottling phase by including also indirect impacts on water resources (Point et al., 2012). The LCA inventory from grape harvesting to wine packing process considers, indeed, both direct and indirect water use. Furthermore, the inventory step in the LCA gives more information on post-harvesting. For example, the packaging stage includes a detailed list of materials that have a high indirect impact on water pollution. Unfortunately, in the case study presented in this work there was a limitation at the farm level to distinguish and describe additional processes within the vinification stage (winery) due to the lack of information. Therefore, the processes of fermentation, stabilization, clarification, and filtration were all included in one single process (winemaking) without further details. Moreover, the impact categories selected for the LCIA analyze different "areas of protection". The use of several impact categories at the midpoint level permits to examine different aspects of WF. In this sense, the impact assessment phase is relatively different in the two considered methodologies. The VIVA WATER indicator summarizes the WF as the volumetric water consumption for direct water use and water pollution (Bonamente et al., 2016). On the other hand, the LCIA investigates different impact categories that cannot be summarized in a single indicator. Both methodologies evaluate the impact of wine production on Water Resources, besides that, the LCA-based WF evaluates impacts also on Ecosystem Quality and Human Health (Kounina et al., 2013).
The identification of good practices might be influenced by the methodology selected for the assessment. The two methodologies can suggest different strategies to reduce impacts on water quality at the vineyard stage, where VIVA can suggest optimal device for drift regulation (nozzles), the use of hedges or best practices to reduce runoff and leaching of chemicals, while LCA approach focuses more on the choice of the most suitable pesticide according to direct and indirect impact on water resources. At this stage, water pollution impact is mainly due to plant health and plant nutrition processes. VIVA WATER indicator identifies potassium phosphonate as the critical substance, while the LCIA identifies multiple critical substances according to the impact category considered. In this study, for example, the LCA analysis evaluated different chemicals showing a significant role on the water quality impact. For example, Copper application is the greatest contributor to freshwater toxicity and ecotoxicity, while urea and potassium sulfate are the pollutants with the highest contribution to aquatic acidification, or Copper and Potassium Phosphonate to eutrophication. Even if the Copper is generally considered a heavy metal with low mobility in the soil, the analysis shows a critical situation in case of a significant application of Copper in the field.
Regarding the post-harvesting stage, the bottling stage is responsible for the highest impact contribution to water quality. In fact, the wine packing process has a great impact on aquatic acidification, eutrophication, and freshwater toxicity due to the use of the glass packaging, but also to the use of the oil burned for heating the water used to wash the bottles. In this case, a proper mitigation might consider a different material for

packaging, more environmentally sustainable. Therefore, the choice of one methodology rather than the other can affect the feedbacks the user or a technician can get and interpret.

This study focuses on a first-year analysis of the WF for both methodologies. To the best of our knowledge, only few studies in literature included a full analysis of water-related impact for a whole wine production. Further developments might consider the production over multiple years in order to analyse the variability of the two methodologies under the inter-annual variability. Future improvements might also implement a sensitivity analysis to identify the variables which influence most significantly the LCA or the VIVA assessment results.

As mentioned, the main limitations of the study refer to the inventory analysis, which does not include the transportation and the end-of-life stages. Literature shows how these steps have a higher impact on water quality rather than water use and depletion. In Bonamente et al., (2016), who performed a cradle to grave water footprint assessment, results show the distribution and the end of life stage together represent the 3.89% of total WF, with the 4.34% of grey WF and -0.45% of blue WF. Looking at the LCA literature, the average higher contribution may vary, where glass packaging shows a contribution of 22% to aquatic acidification and 4% to eutrophication, while the transportation corresponds to 8% of aquatic acidification and 5% of eutrophication, and consumer shopping corresponds to 14% and 8%, respectively (Meneses et al., 2016; Point et al., 2012).

The need to compare the two methodologies limited the analyses of extended system boundaries, with the consequence of not considering the environmental credits of recycled materials from vineyard and cellar processes or the emissions from the wastewater treatment. Moreover, missing information of processes in the cellar (e.g., organic waste disposal from grape pressing, water quality analysis in outputs) might have affected the results.

3.6 Conclusions

In conclusion, the VIVA WATER indicator and the water-focused LCA are based on different approaches but have the same scope. The main outcome of this work is that VIVA framework provides recommendations for the optimal management of direct water use during vineyard phase, while LCA approach addresses the sustainability assessment from a Life Cycle Thinking perspective. The synergic interpretation of the two methodologies can reinforce the scope of the evaluation of environmental sustainability for reducing freshwater use and water pollution. The study highlights for the first time the differences, along the assessment steps, between the Water Footprint assessment and water oriented LCA indicators for wine production. VIVA WATER indicator mainly focuses on water management aspects based on the agronomic and ecological knowledge needed at the vineyard stage, while other stages of wine production play a less significant role. The LCA approach allows the assessor to investigate in more detail the cellar and bottling stages and to understand the hotspot impacts of chemicals on water quality indicators. The assessment of different impact categories in the water-oriented LCA methodology allows to integrate VIVA WATER indicator especially in the vineyard stage, where a set of water-related impact indicators can give more information in terms of direct and indirect water depletion and water quality impact.

The present study does not provide a comparison with other wineries, and it should be considered as a first-year reference for further implementation on the same farm. A similar evaluation might be replicated in other agriculture productions. In fact, even if the VIVA Water indicator was built for the wine sector, it might be adapted to other agricultural production with only small changes (e.g., crop water requirement).

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4. Weak and Strong Sustainability of Irrigation: A framework for irrigation practices under limited water availability

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4.1 Abstract

Agriculture strongly relies on irrigation. While irrigated land accounts for roughly 20% of the global cultivated area, it contributes to about 40% of crop production. In the last few decades, the growing demand for agricultural commodities has translated into an increasing pressure on the global freshwater resources, often leading to their unsustainable use. Here we investigate the sustainability of irrigation, balancing farmers' profit generation objectives and the needs of ecological systems. We ask the question "sustainability of what?", to stress how the sustainability of irrigation is often evaluated with respect the opposing needs of humans and nature. While from the farmers' perspective irrigation is sustainable when it provides uninterrupted access to water resources at a price not exceeding the marginal revenue they generate (clearly without accounting for environmental externalities), from the standpoint of water resources, irrigation is sustainable if it does not deplete freshwater stocks or environmental flows. We invoke the notions of 'weak' and 'strong' sustainability to develop a novel framework for the evaluation of tradeoffs between human needs and the conservation of natural capital. Through the analysis of criteria of performance, we relate water deficit and irrigation overuse to the reliability and resilience of irrigation. This approach is applied to the case of Australia, a major agricultural country affected by water scarcity.

Keywords: Water Scarcity – Sustainable Irrigation – Water and Food Security – Sustainability – Australia

4.2 Introduction

The economic productivity of a number of human activities depends on access to water resources (Sullivan, 2002; D'Odorico et al., 2018). With about 24% of global land area suffering from severe water scarcity (Alcamo et al., 2003), and 35% of the global population living in areas affected by water shortages (Rockström et al., 2014), economic development often occurs at the cost of overexploitation of water resources (Savenije and van der Zaag, 2002), which ultimately leads to ecosystem degradation (Sullivan, 2002).

Agriculture is a major player in the human appropriation of water resources (Green et al., 2015). About 70% of global freshwater withdrawals are used for irrigation to sustain global crop production (Rockström et al., 2017). In fact, irrigated areas account for 18% of global croplands but contribute to about 40% of global food production (Chartzoulakis and Bertaki, 2015; Food and Agriculture Organization, 2019). At the same time 40% of global irrigation practices are unsustainable because they deplete environmental flows and/or groundwater stocks (Rosa et al., 2018; Wada and Bierkens, 2014). The strong coupling existing between economic and environmental needs for water resources raises important questions that are the core of the water sustainability debate: How can human appropriation of water resources sustain economic activities – e.g., agriculture – without depleting water stocks, aquatic habitats, or other ecosystem services? (Rosa et al., 2018; D'Odorico et al., 2018). In the specific context of agriculture, sustainable irrigation strategies need to allow for an increase in crop production to meet rising food needs, while ensuring that natural resources (e.g., groundwater stocks, freshwater ecosystems, and water quality) are not irreversibly depleted (Borsato et al., 2019; Rosa et al., 2018).

The terms “sustainability” is often used to indicate the management, use, and conservation of natural resources in a way that they remain available to future generations (Borsato et al., 2018a; FAO, 2013; Sands and Podmore, 2000). In more anthropocentric terms, “sustainability” is also used to indicate a condition that allows the needs of the present generation to be satisfied “without compromising the ability of future generations to meet their own needs” (U.N., 1987). According to this perspective, the focus is on human needs and not on the preservation of natural resources. Lack of recognition of the central role of natural resources and environmental endowments − also known as “natural capital” − in the definition of sustainability has led to a “weak” notion of sustainability (Solow, 1974). “Weak sustainability” corresponds to conditions that allow natural capital to be replaced by human-made capital, as long as their sum (i.e., natural + human-made) does not decrease over time (Hartwick, 1978; Solow, 1974). Thus, the total amount of endowments or assets future generations can enjoy is not less than that of their ancestors (Groenfeldt, 2019). In a weakly sustainable system, natural and human capital are interchangeable, in the sense that natural capital can be sacrificed in the process of producing human-made capital (i.e., human-made goods). For instance, crop production and the associated profits can occur at the cost of aquatic habitat destruction and groundwater depletion (Gerten et al., 2013; Jägermeyr et al., 2017; Rosa et al., 2018; Wada et al., 2010). The loss of natural capital, however, poses questions of intergenerational justice and often leads to socio-environmental systems that are vulnerable and prone to collapse because natural capital is the long-term foundation of humanity’s livelihoods, while human capital may vanish (Carrão et al., 2016; Gowdy and McDaniel, 1999). In contrast, “strong sustainability”, ensures that natural capital is not replaced by human capital in the sense that it is not degraded in the process of generating human capital. Thus, the ecological and economic aspects of sustainability can be analysed jointly through the notions of weak and strong sustainability to highlight the paradox of offsetting the cost of environmental deterioration with human (manufactured) capital (Rennings and Wiggering, 1997). Weak and strong sustainability can be evaluated as the depreciation cost of manufactured and natural capital, or as the impact of human activities on natural resources (Dietz and Neumayer, 2006).

With specific reference to irrigation, it is possible to aim for strong sustainability because water is overall a renewable resource. Of course, locally, water resources may be unrenewable, as in the case of desert regions with little rainfall inputs and substantial non-renewable groundwater stocks. In those regions, the use of groundwater is a classic example of unsustainable water use (often known as ‘groundwater mining’) (e.g., Konikov and Likhodedova, 2011). Sustainable irrigation needs to ensure that (1) water stocks (e.g., aquifers, rivers, or lakes) are not depleted by keeping withdrawal rates lower than those of natural replenishment; (2) withdrawals from water bodies do not lead to losses of aquatic habitat and irreversible ecosystem degradation; and (3) irrigation does not cause other forms of environmental damage (e.g., soil salinization) with associated losses of ecosystem services and functions, here collectively referred to as “natural capital” (e.g., De Perthuis and Jouvet, 2015).

Sustainability is often characterized through indicators that express the performance of an irrigation system not only in terms of its ability to deliver the water needed by agriculture with no loss of natural capital, but also from the standpoint of economic viability. Strong sustainability is achieved when irrigation does not entail the depletion of either natural or human capital (Aeschbach-Hertig and Gleeson, 2012). This means that both conditions of environmental and economic sustainability are met. The former entails that irrigation water requirement can be met while preserving environmental flows and freshwater stocks (Jägermeyr et al., 2017). Economic sustainability requires the cost of irrigation not to exceed the value of the marginal productivity of irrigation with respect to the baseline of rainfed production.

In this context suitable indicators of sustainability could be a valid tool to evaluate the (weak and strong) sustainability of irrigation and adopt adequate policy responses (Juwana et al., 2012). Moreover, a separate analysis may be necessary for surface water bodies and aquifers to ensure that both environmental flows and groundwater stocks are not depleted (Gleeson et al., 2012; Vanham et al., 2018). As defined by the 6.4 Sustainable Development Goals, indicators of water use should account for both the inter-annual and intra-annual variability of water availability (FAO, 2018, 2015; Vanham et al., 2018). Likewise, they should be spatially explicit to account for the spatial variability of water scarcity as a result of climatic, topographic, and land use conditions (Falkenmark, 1997; Vanham et al., 2018). Moreover, natural water resources should be differentiated from surface, groundwater, and non-conventional water sources (e.g. water reuse) both for withdrawal and consumption (Vanham et al., 2018). In addition to the hydrological and environmental dimensions, it is also crucial to evaluate the social aspects and the participatory capacity of water users and their ability to adapt to changes in water availability and needs (Pahl-Wostl, 2002).

Although indicators of sustainable irrigation have been developed to express the effects of water scarcity and interannual variability (Butler et al., 2017; Sandoval-Solis et al., 2011), they are not generally used to evaluate the weak and strong sustainability of irrigation. Here we use some criteria of performance based on the irrigation "water deficit" (i.e., the gap between renewable water availability and irrigation water use) to determine irrigation sustainability as the fraction of irrigation withdrawals contributing to water deficit (or "relative overuse"), the probability that in a given year no water deficit occurs (or "reliability"), and the likelihood of recovery from water deficit (or "resilience") as three criteria of performance for sustainable irrigation (Hazbavi and Sadeghi, 2017; Park and Um, 2018). Each indicator is evaluated considering both water (or "hydrologic") deficit and economic losses (Balaei et al., 2018; Holling, 1973). This work aims at providing a novel framework to analyse the sustainability of irrigation water use.

4.3 Materials and Methods

4.3.1 *Sustainability framework*

We develop a framework to evaluate water sustainability based on the analysis of indicators (Table 1) expressing the overuse, reliability, and resilience of irrigation water in terms of both hydrologic (i.e., environmental) and economic impacts.

From a **hydrologic/environmental** standpoint the deficit, dH is the difference between the volume (Ia) of irrigation water applications (withdrawn from natural sources) in a given year and the water available for irrigation in that year (AW). Overuse expresses the fraction of irrigation water applications that contributes to water resource deficit, dH (Rosa et al., 2018) or "hydrologic deficit" (Rosa et al., 2018). Reliability expresses the probability (or frequency) that in a given year irrigation will not induce a water deficit. Finally, resilience expresses the ability of the system to recover from conditions of water deficit. This may result either from year-to-year changes in precipitation (i.e., a rainy year occurring after a dry year) or from changes in crop or adoption of water saving technology (i.e., human adaptation) (Borsato et al., 2018b). Despite its simplicity, the resilience indicator used in this study aims to account for temporal autocorrelation in hydrologic conditions (i.e., that it is more likely that a drought year will follow a drought year) and the ability of the system to recover from conditions of deficit. Those indicators were calculated to understand the performance of using water for irrigation purposes and the correlated water deficit. Indicators normalize environmental and socio-economic performance with a scale from 0 to 1.

Table 1. Framework for the evaluation of water sustainability. Equation reports the deficit and the three criteria of performance both from the Hydrological and Economic aspects.

Indicator	Hydrologic	Economic
Deficit	$d_H = \begin{cases} I - AW & if\ I > AW \\ 0 & otherwise \end{cases}$	$d_E = \begin{cases} C_i - R_i & if\ C_i > R_i \\ 0 & otherwise \end{cases}$
Overuse (O_H) or Overexpenditure (O_E)	$O_H = \langle \frac{d_H}{I} \rangle$	$O_E = \langle \frac{d_E}{C_i} \rangle$
Reliability	R_l= *Prob* $[d_H = 0]$	R_l= *Prob* $[d_E = 0]$
Resilience	R_s = *Prob* $[d_H(t) = 0 \vert d_H(t-1) > 0]$	R_s = *Prob* $[d_E(t) = 0 \vert d_E(t-1) > 0]$

From the environmental perspective deficit is the positive difference between water withdrawals for irrigation and the available water (*AW*)

$$d_H = \begin{cases} I - AW & if\ I > AW \\ 0 & otherwise \end{cases} \quad (1)$$

with the available water calculated as the difference between annual runoff (*RO*) and environmental flows (*EF*) as in Rosa et al. (2018):

$$AW = RO - EF \quad (2)$$

It is important to stress that the available water (*AW*) is here defined accounting for environmental flows (eq. (2)). The hydrologic deficit as well as the indicators of overuse, reliability, and resilience defined in this relate to the natural capital services produced by hydrological systems, and are not just based on an analysis of depletion in flow conditions per se. In other words, we are not looking at deficits in water stocks and flows, but at those deficits that harm ecosystems. Likewise, reliability is here defined in relation to the risk of deficits that negatively affect environmental health.

This analysis can be performed separately for surface water and groundwater resources. Therefore, the occurrence of hydrologic deficit corresponds to water overuse by irrigation at the expenses of environmental flows and the consequent loss of aquatic habitat. In this sense hydrologic depletion is associated with losses of natural capital. The environmental flow requirement is typically expressed as a fraction, *r*, of the total runoff (i.e., EF = r × RO). For annual analysis, the value of *r* is typically taken equal to 0.6-0.8 (Gerten et al., 2013; Richter et al., 2012).

The overuse (*OH*) indicator accounts for the magnitude of the hydrologic deficit (*dH*) with respect to the irrigation water withdrawal (*I*). In other words, *OH* is the fraction of irrigation water application that cannot be met sustainably because it exceeds water availability. It is calculated by determining the *dH/I* ratio for every year and then by taking the average over the study period of *N* years (Table 1).

$$O_H = \langle \frac{d_H}{I} \rangle = \frac{\sum_i^N \frac{d_i}{I_i}}{N} \quad (3)$$

Where d_i, and I_i are the water deficit and the irrigation water withdrawals in year i, respectively.
The *reliability (R_l)* indicator is the probability (frequency) of years with no deficit (see Table 1). Thus, R_l is the fraction of the growing season in which irrigation water requirements are met and is expressed as:

$$R_l = \frac{\sum_i^N (1 - H[d_i])}{N} \quad (4)$$

where *H* is the Heaviside function (i.e., $H[d_i]=1$ if $d_i>0$; $H[d_i]=0$ otherwise).
The *resilience (R_s)* is the system's ability to recover from a failure (e.g., Holling, 1973).
Resilience is the probability that a year with water deficit is followed by a year with no deficit (Table 1):

$$R_s = Prob[d_H(t) = 0 | d_H(t-1) > 0] \qquad (t=1, \ldots, N) \qquad (5)$$

From an **economic perspective**, deficit d_E is the positive difference between irrigation costs (*C*) and the marginal revenue (*R_i*) generated by irrigation (calculated as explained in section 4.3.3)

$$d_E = \begin{cases} C_i - R_i & if\ C_i > R_i \\ 0 & otherwise \end{cases} \quad (6)$$

The marginal revenue of irrigation is the difference between the revenues from irrigated and rainfed production. *Overexpenditure* (O_E), the economic counterpart overuse, represents the fraction of irrigation costs that contributes to economic deficit averaged for all years in the study period. Therefore, O_E is calculated as:

$$O_E = \langle \frac{d_E}{C_i} \rangle \quad (7)$$

Likewise, reliability, and resilience are calculated using equations (4), and (5), with deficit expressed as in (6). In equation (1) the economic demand is expressed as the annual irrigation costs.
When such a deficit exists, farmers are using water unsustainably and depleting natural capital. We quantified the occurrence of strong and weak sustainability conditions by comparing the overuse and overexpenditure indicators, which are good metrics for the magnitude of hydrologic and economic stress in the system. If both indicators are relatively low (e.g., <0.25) the system is strongly sustainable. However, when, $O_E < 0.25$, while O_H is large ($O_H > 0.25$) the system is at most weakly sustainable because it is economically sustainable (from the producer's perspective and without considering environmental externalities), but the occurrence of water deficit entails the loss of environmental flows, water stocks, and natural capital. Unsustainable conditions occur when both overuse and overexpenditure are not negligible (Table 2). We stress that in this analysis weak sustainability does not assess whether the value of human-made capital generated by agricultural production exceeds the value of natural capital loss caused by the loss of environmental flows. We refrain from performing such a valuation effort because whether losses of ecosystem services and functions can even be valuated has been the focus of heated debates (i.e., the question of placing a "price tag" on nature). Indeed, the whole point of the notion of 'weak sustainability' is to present a critic to the substitutability between natural and human-

made capital (e.g., Jouvet and De Perthuis, 2013). We performed our analysis considering a threshold of 0.25 (first lower quartile ranking from 0 to 1), however, we recognize that decision makers might assign priority to different factors and can adjust the sustainability threshold here applied.

Table 2. The paradigm of Weak and Strong Sustainability is calculated with the following classification concerning the use of the concepts of Overuse and Overexpenditure.

Weak and Strong sustainability Paradox class	Formula
$Strongly\ sustainable$	$O_H < 0.25; O_E < 0.25;$
$at\ most\ Weak\ Environmental\ sustainable$	$O_H > 0.25; O_E < 0.25;$
$Economically\ Unsustainable$	$O_H < 0.25; O_E > 0.25;$
$Unsustainable$	$O_H > 0.25; O_E > 0.25;$

The parameters used for the deficit assessment are described further in the following sections.

4.3.2 Description of the case study

We apply this framework to the case of Australia because is a major agricultural country and global food producer that is prone severe water scarcity (ABARES, 2018). More specifically, we evaluate to what extent the water use for irrigation meets the farmers' income expectations with or without depriving water and leading to environmental consequences. Moreover, a sensitivity analysis of the framework was done. We performed the framework on a new scenario if technology efficiency using innovative irrigation systems increase by 25 % in the overall Australian continent. In addition, we also performed the framework on a scenario where using the 60% of available water instead of 20% (leaving 40% of available water to Environmental Flows).

Description of the study area

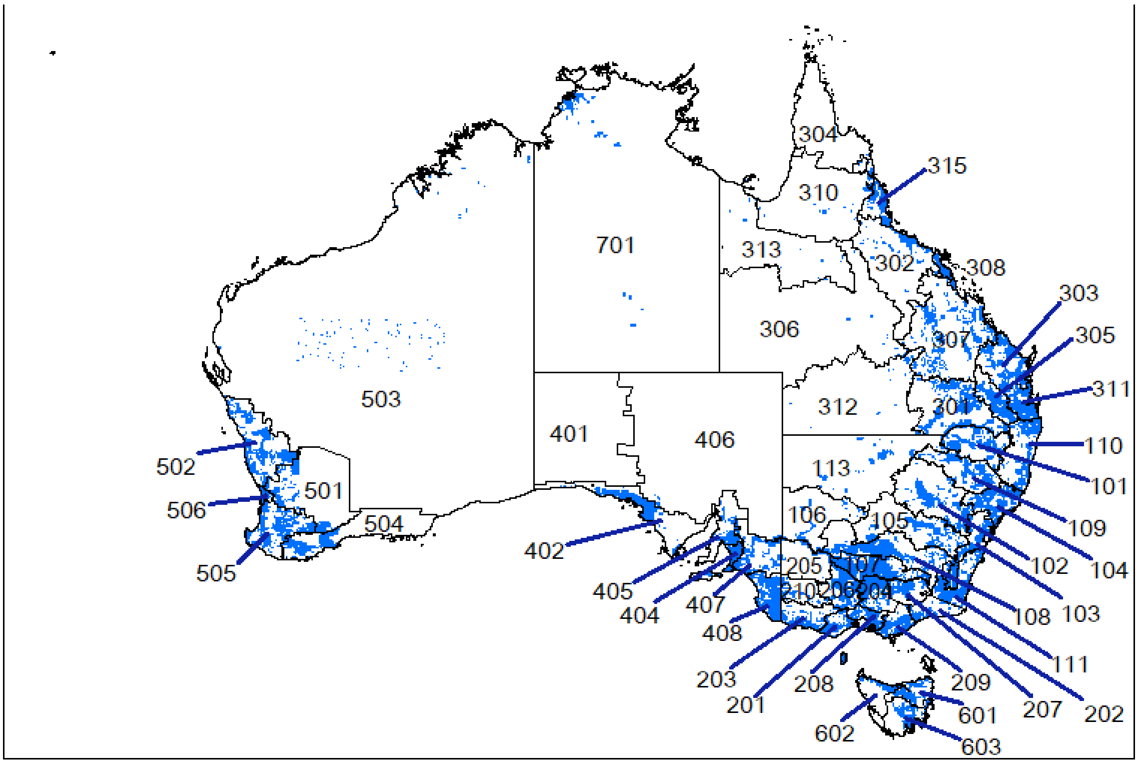
Crop production in Australia is practiced in regions with very different climate conditions ranging from subtropical in the north – where the climate is warmer with well-defined dry (winter) and wet (summer) seasons – and temperate in the south – where four distinct seasons exist, temperature contrast between winter and summer is stronger, and precipitation is more uniformly distributed throughout the year (CSIRO and Bureau of Meteorology, 2015; ABARES, 2018). South-eastern Australia has a "Mediterranean" climate with most precipitation occurring in winter and spring. Moreover, Australia exhibits relatively high interannual rainfall variability. In recent years this region was affected by a major drought (the "millennium drought", 1996-2010) followed by a period (2010-2013) with very heavy rainfall events (Van Dijk et al., 2013).

The "breadbasket" of Australia is in the region of the Murray-Darlin Basin, where irrigation is used to mitigate the effects of a very variable climate (Williams, 2017). In this basin the irrigated area planted with seasonal crops exceeds the irrigated area cultivated with permanent crops. After the 2009 drought some irrigated crops were replaced with drought resistant crops such as grape or citrus (ABARES, 2018). Irrigation is for most part based on drip systems, especially in permanent crops, while most seasonal crops use surface irrigation (Sue Argus and Research, 2015). Irrigation allows for the attainment of yields that are up to three to five times greater than with rainfed production (Meyer, 2005). Hence, many irrigated enterprises differ in their gross return per ha or per m^3, depending on land management and investments in technology.

Data sources

Irrigation and crop production data were taken from the database of the Australian Bureau of Statistic (ABS, 2018), which provides information on annual water withdrawals for irrigation both from surface water bodies and groundwater. Moreover, ABS provides province-level and region-level data on irrigated area, the number of farm holders practicing irrigation, the irrigation water source (surface water and/or ground water), the type of irrigation system used per unit of area (i.e., surface, sprinkler or drip), and indicators of the cost and benefits of irrigation, including the cost of water, operation, administration, infrastructure and equipment, and the irrigation revenue. Estimates of water availability and total evapotranspiration in irrigated areas were based on outputs from the Australian Water Availability Project model (AWAP), available from the CSIRO website (Rapauch et al., 2009). Surface water availability is determined as surface runoff minus environmental flow requirements, while the (sustainable) groundwater availability was estimated as the rate of groundwater recharge or "deep drainage". The AWAP model is a model-data fusion based on remote sensing data assimilation (vegetation greenness and land surface temperature) and assembling weather data and discharge measurements from 200 unimpaired catchments (Rapauch et al., 2009). The model predicts soil moisture in two layers and terrestrial water fluxes due to rainfall, transpiration, soil evaporation, surface runoff and deep drainage using a 5km spatial resolution and daily time resolution. The model calculates the soil water balance considering inflows as precipitation (P) and possible irrigation (I), and outflows as plant transpiration (T), soil evaporation (E), runoff (RO) and drainage to groundwater (DG). Model prediction is therefore combined with the remote sensing observations providing maps of land processed on the Australian continent at 5 km spatial and monthly resolution from 1900 to 2018 (Raupach et al., 2018). In this study the AWAP values of runoff, deep drainage, and actual evapotranspiration (ET) were used.

We analyse the sustainability of irrigation through criteria of performance, based on indicators that change in space and time. We concentrate on a 15-year period from 2002 to 2016 at the spatial resolution of the Australian Natural Resources Management regions (NRM) and territories (Figure 1).



Code	NRM name region	Code	NRM name territory	Code	NRM name territory	Code	NRM name territory
100	New South Wales	101	Border Rivers-Gwydir	206	North Central	401	Alinytjara Wilurara
200	Victoria	102	Central West	207	North East	402	Eyre Peninsula
300	Queensland	103	Hawkesbury-Nepean	208	Port Phillip and Western Port	403	Kangaroo Island
400	South Australia	104	Hunter-Central Rivers	209	West Gippsland	404	Adelaide and Mount Lofty Ranges
500	Western Australia	105	Lachlan	210	Wimmera	405	Northern and Yorke
600	Tasmania	106	Lower Murray Darling	301	Border Rivers Maranoa-Balonne	406	South Australian Arid Lands
701	Northern Territory	107	Murray	302	Burdekin	407	South Australian Murray Darling Basir
801	Australian Capital Territory	108	Murrumbidgee	303	Burnett Mary	408	South East
		109	Namoi	304	Cape York	501	Avon
		110	Northern Rivers	305	Condamine	502	Northern Agricultural
		111	Southern Rivers	306	Desert Channels	503	Rangelands
		112	Sydney Metro	307	Fitzroy	504	South Coast
		113	Western	308	Mackay Whitsunday	505	South West
		201	Corangamite	310	Northern Gulf	506	Swan/Perth
		202	East Gippsland	311	South East Queensland	601	North
		203	Glenelg Hopkins	312	South West Queensland	602	North West
		204	Goulburn Broken	313	Southern Gulf	603	South
		205	Mallee	315	Wet Tropics		

Figure 1. Map of irrigated area within the Natural Resource Management regions (NRM) regions and territories. The table refers to the NRM regions codes on the map (ABS, 2018; Siebert et al., 2013).

4.3.3 *Method: Framework application*

We evaluate environmental and socio-economic performance of irrigation water use in agriculture. We consider the case of surface water and groundwater separately and calculate the hydrologic and economic deficits, dH and dE, separately for these two water sources. The monthly raster files of surface runoff (for surface water availability) and deep drainage (for groundwater availability) from the AWAP model were aggregated to the annual scale, while the monthly raster files of evapotranspiration (ET) were aggregated through the growing season. The cumulative annual values of surface runoff, deep drainage and ET were averaged within each NRM province. Equations (1) and (6) were used to calculate the hydrologic and economic deficits, dH and dE, respectively. In the case of surface water, dH considers the volume of irrigation applications I from surface water sources, while the available water (AW) is determined as the annual surface runoff, RO, minus the environmental flow requirement (EF), which is the amount of water needed to maintain ecological function in in-stream aquatic ecosystems. Here EF was taken equal to 80% of annual runoff (Gerten et al., 2013; Richter et al., 2012). In the case of irrigation from groundwater, the available water was equal to the cumulative annual deep drainage (*DG*), and I from groundwater sources.

The socio-economic deficit (*dE*) evaluates whether the cost of irrigation exceeds the net revenue generated by irrigation (see equation (6)). The ABS database provides the annual income from irrigated area and the irrigation costs (ABS, 2018). The variable irrigation costs include the cost of purchase and the cost of water access (water licence), which are increasing functions of the volume I of irrigation water applications. In this analysis the fixed costs of irrigation include the operational costs, the cost of investments in infrastructures and irrigation equipment. The cost these investments was amortized on 50 and 20 years for infrastructure and equipment, respectively. The amortization period is considered the average lifetime before replacement or update (Pietrucha-Urbanik, 2015; Sands et al., 1982). In this analysis the irrigation costs differ between irrigation with surface water and groundwater only by the costs of water pricing, while the fixed costs are assumed to be about the same.

The marginal revenue from irrigation depends on the increase in crop yield afforded by irrigation with respect to the baseline of rainfed production. If Δ is the fraction of agricultural revenues contributed by irrigation (i.e., due to the increase in irrigated crop production with respect to rainfed yields) the marginal revenue of irrigation (*Ri*) can be calculated as a function of the total agricultural income (R_T) from irrigated areas as:

$$R_i = \Delta \times R_T \qquad (8)$$

The value of Δ is here estimated as the difference between irrigated and rainfed yields divided by the irrigated yield using the Doorenbos and Pruitt (1977) equation:

$$\Delta = \frac{k_y \frac{PET-1}{ET}}{1+k_y \frac{PET-1}{ET}} \qquad (9)$$

where *PET* is potential evapotranspiration and *ET* is the actual evapotranspiration in the case of rainfed agriculture. Therefore, the difference between *PET* and *ET* is assumed to be the irrigation volume consumed by crops, I_c, (ABS, 2018), and the equation can be written as:

$$\Delta = \frac{k_y \frac{I_c}{ET}}{1+k_y \frac{I_a}{ET}} \qquad (10)$$

where k_y is a crop-specific yield response factor. Because values of I_c and R_T are provided by ABS (2018) for entire provinces or regions where multiple crops are cultivated, we are unable to estimate the marginal revenue of irrigation for individual crops. Therefore, we evaluate R_i as a regional aggregate value. To that end, we use an estimate of Δ that is not crop-specific, Thus, we adopt a value of k_y that is representative of each region. Based on the crops cultivated in Australia, k_y is expected to vary between 0.9 and 1. Hence, we considered an averaged k_y equal to 0.95 for all crops. The *ET* values were taken from the output of the AWAP model and aggregated to the annual scale and integrated over the irrigated area within all the NRM territories. Irrigation water consumption, I_c, was calculated as the annual irrigation water application I_a from the ABS (2018) database times the irrigation system efficiency (*IE*), $I_C = I_a \times IE$. *IE* was calculated as the average of the *IE* values of the main irrigation system used in each NRM region and territories, weighed on the extent of areas in which each type of irrigation systems is used. We used a value of *IE*=0.9 for drip irrigation, 0.7 for sprinkler irrigation, and 0.5 for surface irrigation (Grafton et al., 2018).

4.4 Results and Discussion

4.4.1 Water depletion trend

Secondary data from the ABS archive and AWAP simulations allow us to calculate the hydrological and socio-economic deficits and investigate trends in water use for irrigation. The data summarized in Table 3 show a reduction in the number of agricultural businesses practicing irrigation between 2002 and 2017, a trend that is not consistently paralleled by a reduction in irrigated area except during the 2006-2010 drought. Indeed, the Murray-Darling Basin of Australia is pursuing a program of acquiring existing water entitlements from farmers to reduce consumptive use by 20% and restore environmental flows in the Murray-Darling Region (Richter, 2014). Moreover, these data show that surface water use for irrigation exhibits interannual fluctuations that ultimately affect the water price (Debaere et al., 2014). Overall, the gross value of irrigated crops (AUD - Australian Dollars) rises by 66% from 2002 to 2017 and reflected the increasing gross value of agricultural production (+87%). Therefore, the irrigation revenue per hectare (AUD/ha) increased by 50% between 2002 and 2017, while irrigation costs per hectare (AUD/ha) increased by 7% as a result of changes in the variable costs of water.

Table 3. Environmental and socio-economic indicators of irrigation in Australia, based on the ABS database. Groundwater volume used is mentioned as part of the freshwater volume withdrawn from all sources. Irrigation water applied stands for the irrigation volume withdrawn and applied in the field net from supply losses.

Year	Number of agricultural businesses irrigating	Area Watered	Volume of freshwater withdrawn from all sources	Total Irrigation water applied	Groundwater volume withdrawn for irrigation	Gross Value of Irrigated Production	Gross Value of Crop Production (rainfed + irrigated land)	Irrigation revenue	Irrigation costs
	n°	10^3 ha	Km^3	Km^3	Km^3	AUD 10^6	AUD 10^6	AUD/ha	AUD/ha
2016-17	22,103	2,245	9.969	9.104	1.820	15,512	60,842	1,901	156
2015-16	22,690	2,148	9.157	8.381	2.358	15,015	55,994	1,868	163
2014-15	36,533	2,149	9.780	8.950	2.108	15,108	53,625	1,933	163
2013-14	36,155	2,361	11.562	10.731	2.088	14,599	50,866	1,906	148
2012-13	30,629	2,377	11.929	11.060	1.856	13,431	46,289	1,697	147
2011-12	34,911	2,141	9.007	8.174	1.589	13,546	46,687	1,318	164
2010-11	38,752	1,963	7.551	6.645	1.611	12,946	46,020	1,107	178
2009-10	40,817	1,840	7.359	6.596	2.325	11,485	39,707	1,484	190
2008-09	39,940	1,761	7.286	6.501	2.490	11,953	41,849	1,712	199
2007-08	39,637	1,851	7.044	6.285	2.408	12,311	43,270	1,503	189
2006-07	41,787	1,923	8.521	7.636	2.740	12,488	36,060	1,998	182
2005-06	44,826	2,546	11.689	10.737	2.392	12,257	38,527	1,214	138
2004-05	35,244	2,405	10.683	10.085	2.460	10,570	35,555	1,090	146
2003-04	40,400	2,402	11.061	10.442	2.559	10,436	36,927	1,156	146
2002-03	43,774	2,378	11.021	10.402	2.632	9,323	32,563	1,262	-

Moreover, we considered 8 NRM regions; each region comprises provinces (synonym of territories in Figure 1). The main irrigated area in Australia is situated in the Murray Darling Basin, between New South Wales, Victoria, and Queensland. In these regions, irrigation is used by a relatively large number of businesses over a large irrigated area (Table 4). As shown in table 4, surface water is the predominant source for irrigation in

Australia except for the Northern Territory which strongly relies on groundwater. The Northern Territory and the Australian Capital Territory have a small fraction of irrigated land.

The irrigation revenue differs among regions depending on the productivity of crops cultivated in each area. Irrigation costs include variable costs associated with the purchase of water, and fixed costs for operation, infrastructure, and equipment purchase (Table 4). The operation costs account for the costs of maintenance and labour during irrigation and vary between 34 AUD/ha in Queensland to 97 AUD/ha in the Northern Territory. The highest cost of water purchase is found in Victoria and New South Wales with 171 AUD/ha and 113 AUD/ha, respectively.

Table 4. Environmental and socio-economic indicators of the NRM regions. Value are reported as averages and standard deviations throughout the study period (2002-03 to 2016-17). The Standard Deviation (STD) is added for every variable.

NRM Region	Agricultural businesses irrigating		Area watered		Volume withdrawn from all sources		Total Irrigation water applied		Groundwater volume applied for Irrigation		Gross Value of Irrigated Production		Irrigation revenue		Cost of annual water purchase		Cost for equipment and infrastructure		Cost for operation in irrigation	
	n°	STD	10^3 ha	STD	Km^3	STD	Km^3	STD	Km^3	STD	AUD 10^6	STD	AUD ha^{-1}	STD	AUD ha^{-1}	STD	AUD ha^{-1}	STD	AUD ha^{-1}	STD
New South Wales	9058	1783	752	150	3.557	1.021	3.322	0.99	0.733	0.218	2789	500	867	237	113	25	14	3	76	16
Victoria	9662	1723	549	92	2.124	0.522	1.952	0.503	0.286	0.046	3657	649	2061	580	171	32	10	2	58	11
Queensland	8105	1505	523	28	2.429	0.300	2.192	0.305	0.578	0.085	3244	471	1594	444	65	4	15	1	34	2
South Australia	5163	980	186	18	0.865	0.133	0.786	0.123	0.447	0.084	1573	171	3252	702	97	9	24	2	96	9
Western Australia	2615	546	62	25	0.361	0.033	0.271	0.026	0.132	0.015	765	133	3721	1103	71	16	21	5	55	12
Tasmania	1722	310	90	7	0.26	0.033	0.237	0.034	0.019	0.007	624	123	2050	435	58	4	15	1	71	5
Northern Territory	216	72	5	1	0.045	0.021	0.02	0.005	0.033	0.014	70	20	3667	1194	12	3	21	5	97	22
Australian Capital Territory	8	5	0.23	0.14	$0.479 \cdot 10^{-3}$	0.237	$0.371 \cdot 10^{-3}$	0.267	$0.045 \cdot 10^{-3}$	0.025	3	1	11386	8956	9	7	-	-	35	29

Table 5. Average fraction of area irrigated with different type of systems.

	Surface irrigation	*Drip irrigation - above ground*	*Drip irrigation - subsurface*	*Sprinkler irrigation – (microspray and microsprinkler)*	*Sprinkler irrigation - portable irrigators*	*Sprinkler irrigation - hose irrigators*	*Sprinkler irrigation - large mobile machines*	*Sprinkler irrigation - solid set*	*Other irrigation systems*	*Total irrigation*
Name Territory	%	%	%	%	%	%	%	%	%	ha
Australia (total)	58.01	8.68	1.48	4.20	3.19	8.41	12.50	2.18	1.35	2,166,005
New South Wales	70.57	6.12	0.89	1.73	3.24	4.41	10.87	0.84	1.33	751,842
Victoria	67.78	9.61	1.16	4.32	2.32	3.87	6.88	3.39	0.67	548,445
Queensland	45.97	4.74	3.12	6.41	4.51	21.28	10.72	2.45	0.80	523,184
South Australia	17.92	36.51	0.76	6.77	1.10	3.17	22.74	5.09	5.95	186,303
Western Australia	38.70	24.82	2.01	7.75	1.15	1.04	14.87	6.33	3.31	62,440
Tasmania	2.81	3.62	0.05	2.74	12.48	31.59	41.60	2.11	2.99	90,016
Northern Territory	26.32	17.84	5.99	38.79	0.22	0.61	7.31	0.19	2.72	5,112
Australian Capital Territory	7.82	19.93	0.00	4.87	0.00	55.41	11.62	0.35	0.00	225

Fifty-eight percent of irrigated lands in Australia use surface irrigation, followed by various types of sprinkler (32%) and drip irrigation (10%) (Table 5). Our estimates of the hydrologic deficit indicate that in the Australian continent water use for irrigation tends to deplete surface water resources and environmental flows more than groundwater (Figure 2). In fact, groundwater is less frequently in a deficit than surface water (Table 6). The occurrence of hydrologic deficit conditions indicates that farmers have withdrawn more water than it would have been sustainable to do. In other words, withdrawals (Irrigation volume applied) exceed the available water, which means that they occur at the expenses of environmental flows or groundwater depletion. In the case of Australia, it seems that the main concerns should be for the loss of environmental flows. Recent news article reported that fish died from a severe water overuse in the Murray-Darling Basin (AFP, 2019).

Conditions of socio-economic deficit (on the right) do not emerge at the average country scale, but they do occur in a very low scales in New South Wales. Changes in economic deficit can result from changes in water price or in water productivity as shown in Tables 4 and 6. Economic deficit conditions correspond to situations in which the cost of water exceeds the marginal revenues generated by irrigation.

Table 6. Average hydrological deficit (d_H) and socio-economic deficit (d_E) over the period 2002-2017 for 8 NRM regions. The d_H and d_E resulted for surface and groundwater sources are shown.

		Australia	*New South Wales*	*Victoria*	*Queensland*	*South Australia*	*Western Australia*	*Tasmania*	*Northern Territory*	*Australian Capital Territory*
Surface Water (d_H)	km³	5.946	2.542	1.640	1.300	0.375	0.062	0.112	-0.064	0.0003
Surface Water (d_E)	AUD·10⁶	-2415	-410	-876	-621	-333	-140	-163	-5	-2
Groundwater(d_H)	km³	-1.734	-0.328	-0.393	-0.416	0.299	-0.373	-0.334	-0.190	0.00001
Groundwater (d_E)	AUD·10⁶	-828	-80	-165	-215	-351	-88	-11	-14	-1
Total irrigation (d_H)	km³	4.011	2.161	1.223	0.826	0.634	-3.438	-0.224	-0.271	0.000
Total irrigation (d_E)	AUD·10⁶	-403	12	-191	-125	-104	-34	-41	-1.041	-1.394
Area watered	ha·10⁶	2.166	0.741	0.547	0.523	0.186	0.062	0.090	0.005	0.000
Surface Water (d_H)	m³ ha⁻¹	2745	3431	2995	2484	2012	992	1.243	-12.505	1.422
Surface Water (d_E)	AUD ha⁻¹	-1115	-553	-1600	-1187	-1789	-2236	-1808	-1075	-8117
Groundwater (d_H)	m³ ha⁻¹	-801	-443	-717	-796	1606	-5975	-3.712	-37.075	0.054
Groundwater (d_E)	AUD ha⁻¹	-382	-108	-301	-411	-1882	-1414	-122	-2722	-3238
Total irrigation (d_H)	m³ ha⁻¹	1852	2917	2235	1579	3402	-5506	-2.488	-53.106	1.509
Total irrigation (d_E)	AUD ha⁻¹	-186	17	-348	-239	-558	-537	-459	-204	-6189

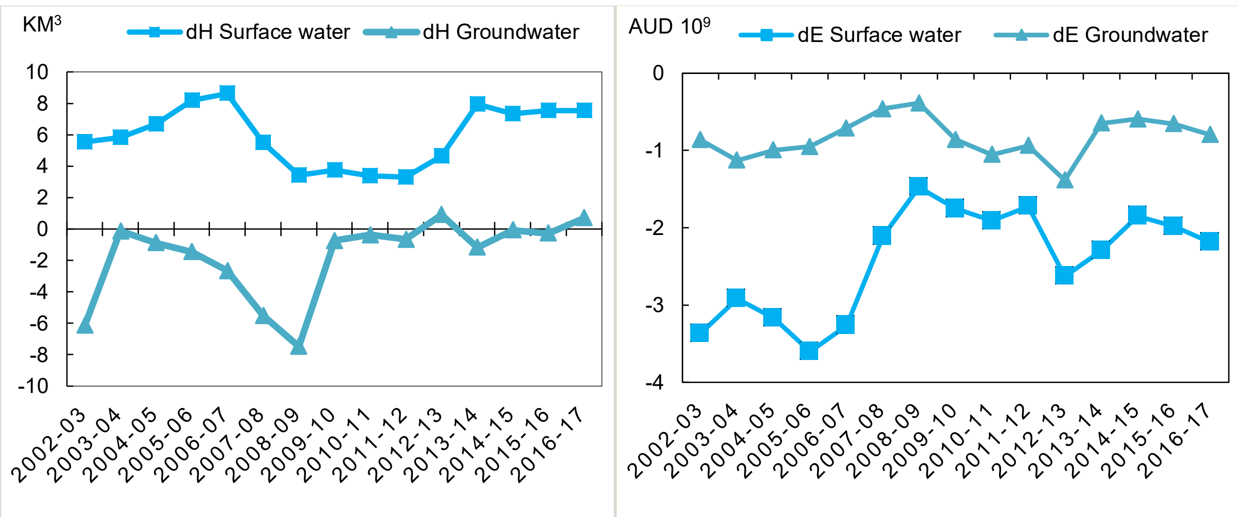


Figure 2. Trend (2002-2017 period) of environmental deficit d_H (km^3), on the left, and the socio-economic deficit d_E (AUD 10^9), on the right at the Australian continent scale.

4.4.2 Analysis of the Criteria of Performance for Surface water

We first consider the sustainability of surface water resources, which account for almost 80% of irrigation withdrawals in Australia. We look at indicators of performance of irrigation for both the hydrologic and economic aspects of sustainability (Figure 3). in other words, we look at the overuse or overexpenditure, unreliability (i.e., 1-R_l, with R_l being the Reliability; see Table 1) and the persistence of deficit conditions (i.e., 1-R_s). Considering the hydrologic/environmental aspect of sustainability, overuse is widespread across Australia (Figure 3, panel 1a) except for the Northern Territory. The unreliability of surface water use is also relatively high across the country except for the northern territories (Figure 3, panel 1b), where climate is tropical sub-humid. Likewise, water deficit conditions appear to persist across the country, as reflected by the high probability that years with hydrologic deficit are followed by deficit years (Figure 3, panel 1c), indicating inability for the system to overcome (and recover from) stress (i.e., low resilience).

The indicators of socio-economic performance show a different picture because overexpenditure is overall low across Australia (<0.25; Figure 3, panel 2a), except for the territories of Southern Gulf and the southern coast of Victoria where overexpenditure is greater than 0.5 or even 0.75. This means that in these regions the revenue of irrigated agricultural production is not sufficient to offset the cost of irrigation. However, other than in these regions, the use of surface irrigation seldom leads to an economic deficit (Table 6). Moreover, economic deficit has a low likelihood of occurrence (i.e., irrigation is economically reliable) and does not persist (i.e., irrigation is economically resilient) (Figure 3, panels 2b, 2c).

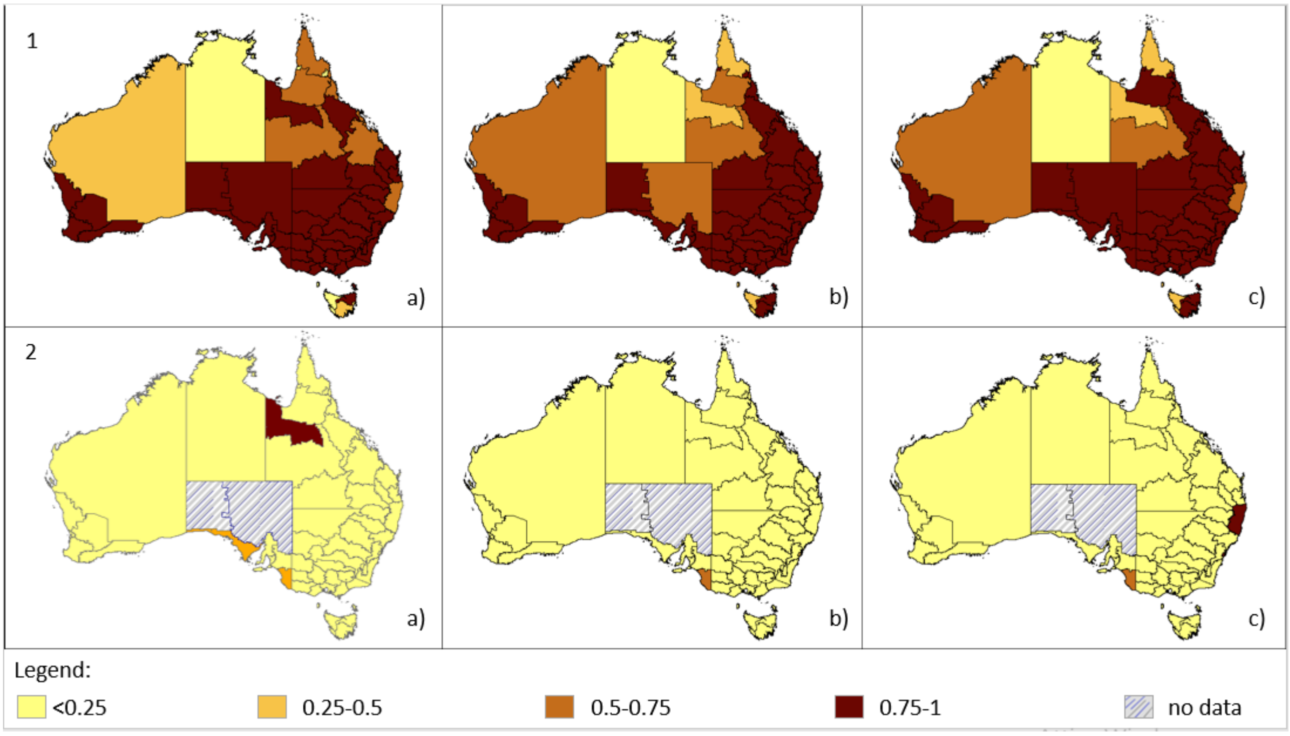


Figure 3. Maps of criteria of performance on the environmental (1) and socio-economic (2) aspects. The figure shows the Overuse/Overexpenditure (a), the non-Reliability (b), and the non-Resilience (c), of surface water use for irrigation purposes in the Australian continent.

To evaluate the strong and weak sustainability of irrigation, we map (Figure 4) areas affected by both high hydrologic and economic deficits (strongly unsustainable), high hydrologic but relatively low economic deficit (weakly sustainable) and low hydrologic and economic deficit (sustainable), as explained in the methods section (Table 2). We also consider the case in which irrigation is sustainable hydrologically but not economically. The results of this analysis (Figure 4) show that in most of Australia surface water use for irrigation is weakly sustainable, in that it can generate good profits on most years but 'usurps' environmental flows. In fact, hydrologic deficits correspond to conditions in which surface water withdrawals for irrigation exceed the available water, which is here estimated as the difference between surface runoff and environmental flows. Thus, the use of surface water for irrigation is strongly sustainable in the Northern Territory, and Northeast Tasmania. In the rest of Australia, surface water withdrawals for irrigation are weakly sustainable, except for the Southern Gulf in Queensland, Northern Rivers in New South Wales, and Eyre Peninsula and South East province in South Australia that are affected by economic unsustainability.

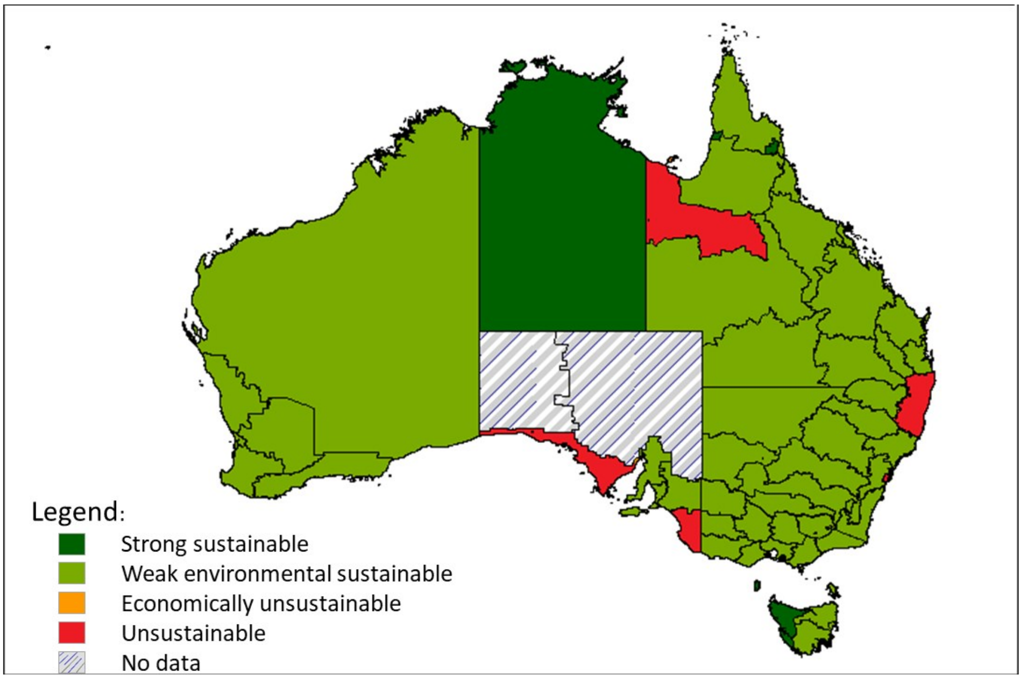


Figure 4. Map of the sustainability paradigm of the irrigation practice using surface water.

4.4.3 Analysis of the Criteria of Performance for Groundwater

Groundwater sustains about 20% of irrigation in Australia, particularly in those areas where surface water is not sufficient to meet irrigation water requirement. In Figure 5, the sustainability of groundwater use for irrigation is evaluated through the three criteria of performance. Overuse is generally higher than 0.5 in areas with lower surface water supply. In those areas, groundwater stocks are often depleted by irrigation with withdrawals (Figure 5, panel 1a). The unreliability of irrigation and persistence of groundwater deficit is relatively high in the Northern Territory, Western Australia, and in some provinces such as the Lower Murray Darlyn in South New Wales and Fitzroy in Queensland (Figure 5, panels 1b, 1c). From the standpoint of the economic performance, overexpenditure is high in the Southern Gulf and Northern Rivers, where it exceeds 0.75. In North Central Victoria and in the Eyre Peninsula overexpenditure is in the 0.50-0.75 range (Figure 5, panel 2a). The probability of occurrence of years with economic deficit (or 'unreliability') is relatively low (Figure 5, panel 2b). Likewise, probability of persistence of economic deficit is relatively small (<0.25), except for Southern Gulf in Queensland, North Central in Victoria, Eyre Peninsula in South Australia and South in Tasmania (Figure 5, panel 2c).

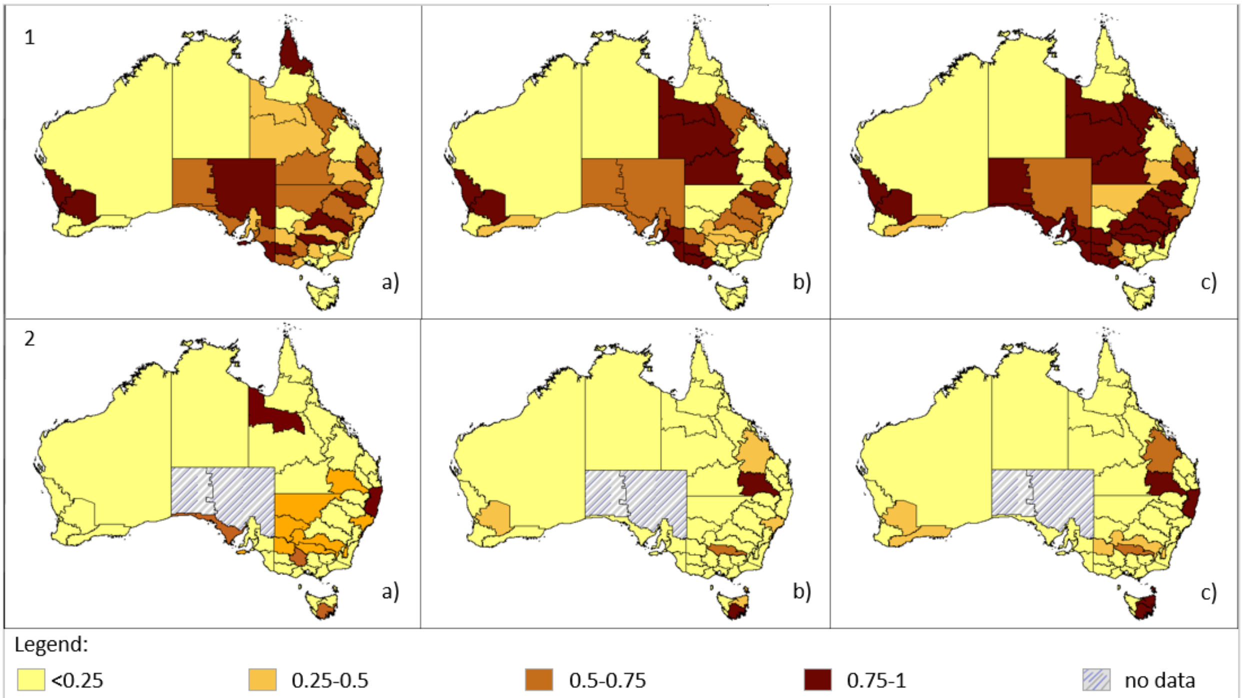


Figure 5. Maps of criteria of performance on the environmental (1) and socio-economic (2) aspects. The figure shows, the Overuse/Overexpenditure (a), the non-Reliability (b), and the non-Resilience (c) of groundwater use for irrigation purposes in the Australian continent.

In Figure 6, the environmental and socio-economic deficits are compared to determine conditions of weak and strong sustainability of groundwater use for irrigation across Australia. The consistent pattern of weak sustainability indicates that groundwater use contributes to groundwater depletion in most of the areas irrigated with groundwater. Only few provinces situated in the northern part of the country exhibit strong sustainability in groundwater use for irrigation. Murray in New South Wales, Border Rivers Maranoa-Balonne in Queensland, and South in Tasmania show an economically unsustainable groundwater use for irrigation. In those provinces, the marginal revenues of irrigation do not fully cover the irrigation costs.

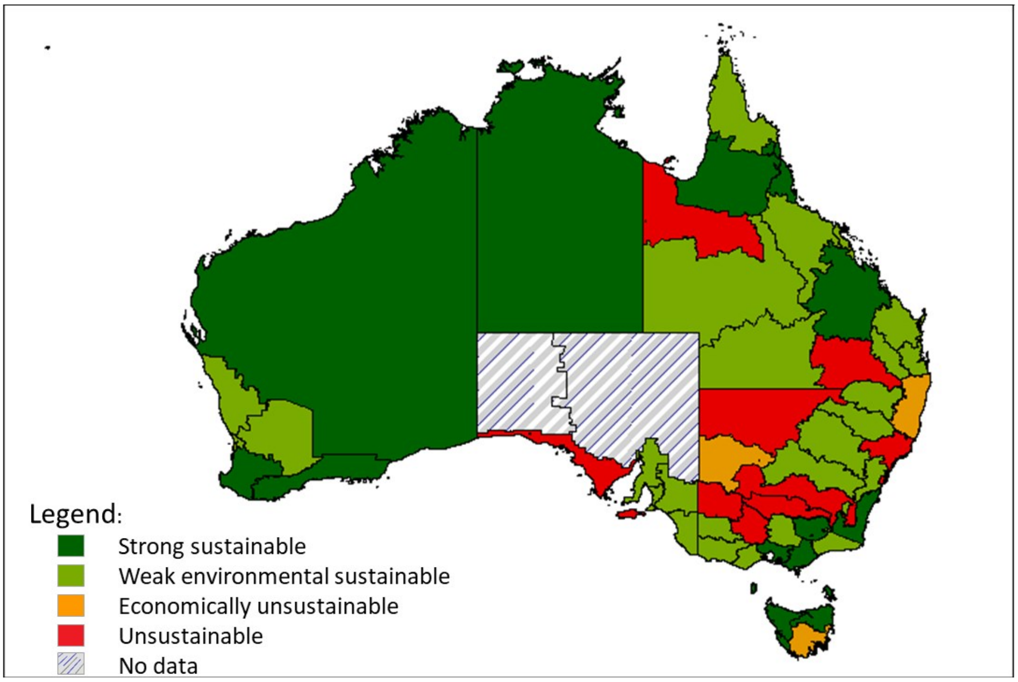


Figure 6. Map of sustainability paradigm of the irrigation practice using groundwater.

4.4.4 Sustainable for what? A different point of view

Previous studies have analysed the use of water for agricultural purposes under scarcity conditions focusing on the sustainability of irrigation from the standpoint of the environment (Gerten et al., 2013; Hazbavi and Sadeghi, 2017; Khan et al., 2006). When the focus is on the environment, an irrigation practice is said to be sustainable when water resources are not depleted and their use does not compromise the availability of ecosystem goods and services in the future (Butler et al., 2017; Papas, 2018; Unver et al., 2017). From the standpoint of farmers, however, the analysis of sustainability can also evaluate to what extent water for irrigation can be consistently available in a certain region (reliability) and whether water deficit years may persist in time (Juwana et al., 2012; van Dijk et al., 2013) (Van Dijk et al., 2013). This requires an analysis of the magnitude and frequency of water deficit, and the evaluation of the sustainability of irrigation using a set of indicators of performance (Sandoval-Solis et al., 2010) instead of using an aggregated index. In addition, farmers are also interested in the economic aspect of sustainability to evaluate whether and to what extent the cost of irrigation may exceed the marginal revenues it can generate. This study addresses the environmental and economic sustainability of irrigation looking at water use under water scarce conditions. Irrigation typically meets farmers' expectation of a greater income because the cost of water is often very small and most of the irrigation costs are associated with infrastructure, equipment and operation (Grafton and Wheeler, 2018). This type of analysis, however, has often neglected the environmental consequences of irrigation.

The intensification of food production allows for the attainment of higher yields within smaller areas. However, it may entail overuse and depletion of water resources and prevent their preservation (Rockström et al., 2017). The development of a sustainable intensive agriculture needs to focus on ways to enhance agricultural productivity while reducing the environmental impacts of irrigation. In many cases, water is used inefficiently, and action needs to be taken to promote policies that integrate the dimensions of technology, economics, and governance (Unver et al., 2017).

A common approach to enhance water sustainability focuses on the implementation of new irrigation technologies that improve efficiency (i.e. *IE*). In fact, the study of a scenario of increased efficiency (i.e., 25% increase in *IE*, see methods section; Figure 7) shows that an increase in IE tends to enhance water sustainability. As shown in Table 7 the increase in efficiency allows for both water and economic savings. Thus, regions that in the current scenario exhibit both hydrologic and economic deficits (e.g., Eyre peninsula, Fitzroy, and Murrumbidgee), can become sustainable in the scenario of increased efficiency. Likewise, areas that are presently weakly sustainable (e.g., Southern Rivers and East Gippsland) can become strongly sustainable (Figure 7).

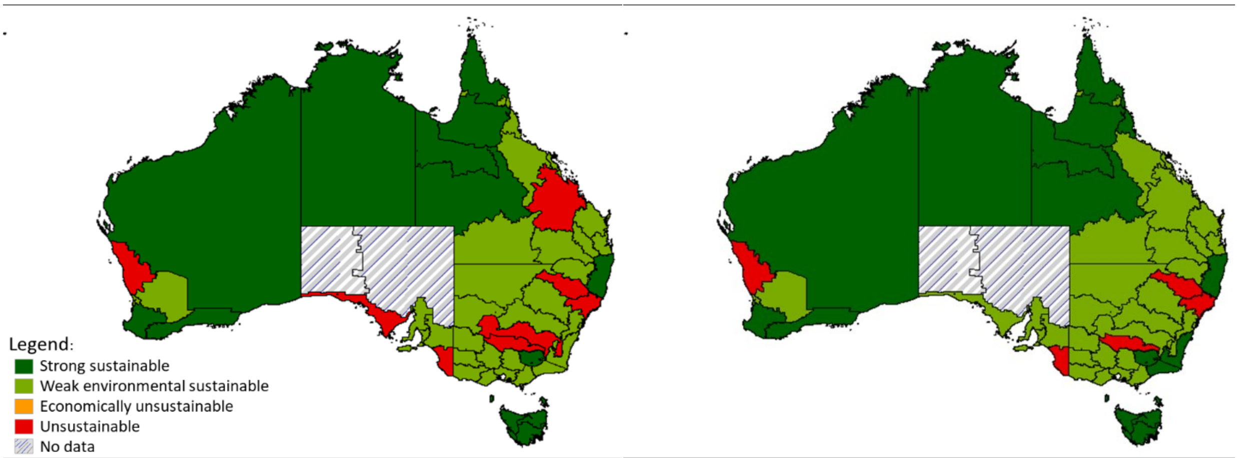


Figure 7 Weak and Strong paradox calculation in case irrigation efficiency rise up of 25%. On the left, the actual scenario A; on the right, scenario B where irrigation efficiency increases.

The 25% increase in *IE* allows for water savings ranging from 182 m^3 ha^{-1} in Northern Territory up to 520 m^3 ha^{-1} in Queensland (Table 7), which correspond to an increase in profits of 2.3% and 14%, respectively (Table7). These results show that the enhanced efficiency of irrigation systems has a positive impact on water productivity However, it has been reported that the farm-scale increase in irrigation efficiency may increase water consumption (the so called "irrigation paradox") if farmers use the water savings resulting from the increased irrigation efficiency to plant crops that are more water-intense or irrigate more land (Grafton et al., 2018).

Table 7. Water Saving and Economic Saving when irrigation efficiency rises up by 25% (scenario B). The amount is resulted from the difference between deficit within the present scenario (A) and scenario B averaged on the time period 2002-2017.

	Name Territory	*Australia*	*New South Wales*	*Victoria*	*Queensland*	*South Australia*	*Western Australia*	*Tasmania*	*Northern Territory*	*Australian Capital Territory*
Water saving	$m^3 \cdot 10^3$/ha	0.485	0.494	0.455	0.520	0.380	0.508	0.446	0.182	0.288
	STD	0.086	0.129	0.075	0.070	0.070	0.170	0.041	0.102	0.134
Economic Saving	AUD/ha	29.256	26.309	40.767	16.106	24.018	17.861	14.353	2.931	1.377
	STD	5.203	6.845	6.698	2.171	4.410	5.979	1.318	1.650	0.639
	%	18	13	18	14	11	12	10	2.3	4.6
	STD	1.5	2.7	1.7	1	1.2	1.4	0.9	1.2	3.1

We have also invoked the notions of weak and strong sustainability (Dietz and Neumayer, 2007; Gowdy and McDaniel, 1999; Nasrollahi et al., 2018) to relate biophysical conditions of hydrologic deficit to economic deficit between revenues and costs. In this way, the analysis of hydrologic and economic deficit shows whether irrigation is strongly sustainable, unsustainable or weakly sustainable. In the case of weakly sustainable irrigation, the marginal income generated by irrigation is obtained at the cost of sacrificing groundwater stocks or aquatic habitat and associated ecosystem function in surface water bodies. In this sense, an increase in water withdrawal causes a depletion of the environmental flows. We also performed a sensitivity analysis with a scenario where only 40% (instead of 20%) of available water is left for environmental flow preservation. Table 8 shows how the framework performs an increase in water exploitation. There is a decrease of the magnitude of the irrigation water deficit (d_E) and an increase of the resilience and reliability of the system but at the expense of Environmental Flows. The overuse decreases by 25% in the overall Australian continent, which can reach even the 72% decrease, for example, in Tasmania.

Table 8. Sensitivity analysis of using the 60% of EFR for irrigation from surface water.

	EFR=Runoff ×80%			***EFR=Runoff ×40%***			***Variance (σ^2)***			***∆% (80=>40)***		
Name of Territory	**Resilience**	**Reliability**	**Overuse**	**Resilience**	**Reliability**	**Overuse**	**Resilience**	**Reliability**	**Overuse**	**Resilience**	**Reliability**	**Overuse**
Australia	0.00	0.00	0.87	0.07	0.07	0.65	0.0026	0.002	0.024	7%	7%	-25%
New South Wales	0.00	0.00	0.96	0.00	0.00	0.88	0.00	0.00	0.003	0%	0%	-9%
Victoria	0.00	0.00	0.97	0.00	0.00	0.90	0.00	0.00	0.002	0%	0%	-7%
Queensland	0.00	0.00	0.76	0.18	0.27	0.54	0.017	0.036	0.025	18%	27%	-29%
South Australia	0.00	0.00	0.99	0.00	0.00	0.96	0.00	0.00	0.0003	0%	0%	-3%
Western Australia	0.08	0.13	0.68	0.57	0.53	0.49	0.12	0.08	0.017	643%	300%	-27%
Tasmania	0.00	0.00	0.50	1.00	0.87	0.14	0.50	0.38	0.065	100%	87%	-72%
Northern Territory	1.00	1.00	0.00	1.00	1.00	0.00	0.00	0.00	0.00	0%	0%	0%
Australian Capital Territory	0.00	0.00	0.98	0.00	0.00	0.94	0.00	0.00	0.0007	0%	0%	-4%

4.5 Conclusions

The novelty of the study stems from the introduction of the framework evaluating the sustainability of irrigation practice both from the hydrological and socio-economic points of view. We evaluated to what extent irrigation take place without compromising environmental flows while generating net profits for farmers or agribusinesses. This approach allows us to quantify the weak and strong sustainability of irrigation.

More specifically, this framework (i) provides indicators of performance that can be used to evaluate in quantitative terms the sustainability of irrigation and its likelihood; (ii) it shows how the notions of reliability, resilience and overuse/overexpenditure can be translated into suitable indicators addressing both the environmental and socio-economic aspects of sustainability. These indicators provide a rather comprehensive picture of the sustainability of the system, from the perspective of both farmers and the environment; (iii) it can be applied at different spatial and temporal scales and used by stakeholders to support decision-making in the context of water management and water policy development.

The case study was implemented for the case of Australia. In this specific context, we found that Australian irrigation is for most part weakly sustainable. In other words, irrigation water is unsustainably used to produce commodities at the expenses of the environment. Interestingly, our analysis shows that irrigation has stronger negative impacts on environmental flows, while groundwater use for irrigation tends to be more sustainable. Overall, irrigation contributes to a more reliable and resilient crop production with a relatively high level of economic water profitability.

Acknowledgments

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5. Evaluation of the Grey Water Footprint Comparing the Indirect Effects of Different Agricultural Practices [3]

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5.1 Abstract:

Increasing global food demand and economic growth result in increasing competition over scarce freshwater resources, worsened by climate change and pollution. The agricultural sector has the largest share in the water footprint of humanity. While most studies focus on estimating water footprints (WFs) of crops through modeling, there are only few experimental field studies. The current work aims to understand the effect of supposedly better agricultural practices, particularly precision agriculture (variable rate application of fertilizers and pesticides) and conservation agriculture (minimum, strip, or no-tillage), on water deterioration and water pollution. We analyzed the results from an experimental field study in the northeast of Italy, in which four different crops are grown across three years of crops rotation. We compared minimum, strip, and no-tillage systems undergoing variable to uniform rate application. Grey WFs are assessed based on a field dataset using yield maps data, soil texture, and crop operations field. Leaching and associated grey WFs are assessed based on application rates and various environmental factors. Yields are measured in the field and recorded in a precision map. The results illustrate how precision agriculture combined with soil conservation tillage systems can reduce the grey water footprint by 10%. We assessed the grey Water Footprint for all the field operation processes during the three-year crop rotation.

Keywords: water footprint; conservation tillage systems; precision agriculture; sustainable management; agriculture soil practices; impact reduction

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5.2 Introduction

Water degradation becomes an important problem when the territory is affected by a high risk of water scarcity and water pollution. Agriculture is the greater user of water resource that causes water depletion and degradation (FAO, 2013). In this context, the impact of agriculture soil management on water resources and its effects on the environment are clearly shown by the indicator of Water Footprint (WF) (Tuninetti et al., 2017). Water Footprint is a concept introduced by Hoekstra and Hung (2002), and it is an indicator of quantitative and qualitative water use (Hoekstra and Hung, 2002). It is quantitative since it evaluates the consumption and the

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embedded water. It is qualitative since it evaluates the pollutants load into a water body as a dilution volume under the qualitative standard threshold (Chapagain and Hoekstra, 2010). Agriculture practices have direct and indirect effects on water pollution according to the amount of fertilizers and pesticides required during the process of crop production (Bossio and Geheb, 2008). In Chukalla et al. (2017) different nitrogen rates and water management were analysed for different crops comparing conventional and no-tillage systems (Chukalla et al., 2017). The proper nitrogen rate in combination with a suitable tillage system can reduce the water pollution without compromising the yield (Bacenetti et al., 2015; TerAvest et al., 2015). In particular, a variable rate application of fertilizers improves the soil productivity and the fertilization efficiency. In addition, Hirel et al., (2011) studied how alternative tillage techniques can reduce the environmental impact for different nitrogen rates (Hirel et al., 2011). The variable nitrogen application can minimize differences in soil fertility between conservation tillage systems; especially, when the soil water storage is reduced and the fertilization is applied, there is a higher crop growth (Basso et al., 2009; Fabrizzi et al., 2005). In this sense, precision farming is a technique able to indirectly minimize the water pollution because it improves the efficiency of agricultural processes without compromising the yield; therefore, a sustainable agriculture system can be reached maintaining soil productivity (Sands and Podmore, 2000). In addition, the combination of agriculture tillage practices and precision farming might lead a reduction of the impact on water resources, since it is possible to reduce inputs (pesticides and fertilizers) applications (Pezzuolo et al., 2017). In fact, the integration of precision farming with different soil tillage techniques enhances a secure farm income, including a sustainable ecosystem management (Cristina Rulli and D'Odorico, 2014; Miglietta and Morrone, 2018), and a reduction in land use pressure. Therefore, the increased pressure in global crop productivity due to the increasing food demand involves the dilemma of intensification/extensification of crop systems that somehow affects the impact on the environment (Borsato et al., 2018b; Cassman, 1999; Van Grinsven et al., 2015). A sustainable agricultural system must decrease the environmental pollution. Moreover, an intensive farming system increases the yield productivity, while the extensive system requires reduced external inputs and reducing the yield (Cassman, 1999; Matson et al., 1997; Tilman et al., 2001). In this way, a proper compromise between agriculture intensification and extensification must be found (European Commission, 2011). The Common Agricultural Policy has strongly encouraged conservation tillage practices or the set-aside of arable land (European Commission, 2011; European Court of Auditors, 2014) and maintains the food production without intensifying the land use (Matson et al., 1997). Conservation tillage system ameliorates soil properties in different aspects (Kinoshita et al., 2017; Peigné et al., 2018; Šimon et al., 2009; Tarolli et al., 2019). This soil tillage technique was introduced to increase soil organic matter and soil biodiversity (Peigné et al., 2018; Šimon et al., 2009) and to maintain the physical soil structure that determines the soil health and quality (Kinoshita et al., 2017; Tarolli et al., 2019). For example, De Vita et al. (2007) shows how important a reduced tillage system is in a semi-arid environment that guarantees a good performance in wheat grain quality thanks to an acceptable and stable production (De Vita et al., 2007). On the other hand, conventional agriculture has positive aspects on soil properties: it mixes fertilizers and manures, it includes topsoil aeration, it reduces weeds competition with effective weeds control, and it drills soil crusts (Busari et al., 2015; Husnjak et al., 2002). However, many research studies demonstrate how conventional tillage could increase soil degradation (Iocola et al., 2017; O'Sullivan et al., 1999; Tarolli et al., 2019), it contributes to soil organic matter reduction and therefore to the loss of fertility (Kladivko, 2001).

The study focuses on the use of the indicator “grey water footprint” as a preliminary analysis of direct and indirect water deterioration during soil management and tillage. The study discusses how proper soil management and tillage system can reduce the grey water footprint, achieving a more sustainable soil practice without compromising the yield. The case study analyses the grey water footprint on different tillage systems, one conventional and three conservation tillage systems, undergoing Precision and Tradition Farming of a three-year crops rotation. A solution for which sustainable soil practices might be involved within the intensification or the extensification of agriculture is also provided.

5.3 Materials and Methods

5.3.1 Description of the Study Area and the Experimental Setup

The study area is situated in the Veneto Region plain, near the Venice Lagoon, 45.63° N and 12.95° E. The location has an extreme anthropogenic pressure; it is particularly vulnerable for its geomorphological variability. The proximity to the sea influences the salt-water intrusion, forced by the height of the water table that affects crop cultivation and can compromise the yield. The area of the test considers a total surface of 23.4 ha, divided into 16 plots of about 1.5 ha each one (Figure 1a). The experimental setup is built on four soil tillage techniques, with conservation and conventional soil tillage as described:

- The Minimum tillage (MT) refers to a cultivation system that consists with few non-inversion tillage passages with the use of tine and disc implements. The techniques were conducted with a tine cultivator at 25 cm depth. The seedbed preparation and the sowing were combined with the use of a combined power harrow planter. Crop residues remain on the surface and mixed on the topsoil or landfill at deeper depth (15–20 cm);
- No-tillage (NT) is a direct drilling system combined with the sowing crop on previous crop residues with no prior cultivation. Seeding was conducted using special discs that make a narrow and slight furrow on the soil for seed deposition. The tillage technique provides fast land preparation within the optimum period. The combination of the drilling system with the sowing system in a narrow band of soil avoids soil inversion and it provides minimum soil disturbance. Weeds have been controlled with herbicides;
- Strip tillage (ST) is a modified direct drilling system. It consists of a soil tillage technique with the combination of a knife tine and disc passage on a strip land. Strip tillage creates narrow tilled strips of 10–30 cm width and 55 cm inter row. Seeds are drilled into the cultivated strips. Crop residues are removed from cultivated strips and placed between rows;
- Conventional tillage (CT) concerns a primary operation of the topsoil inversion using a mouldboard plough at 35 cm deep. A secondary cultivation prepares the seedbed with a single passage of a tine cultivator at 25 cm and a power-harrow down to 10 cm. Conventional tillage prepares a seedbed without surface residues, interrupting weeds growth, pest and disease for the optimum condition of crop germination (Morris et al., 2010).

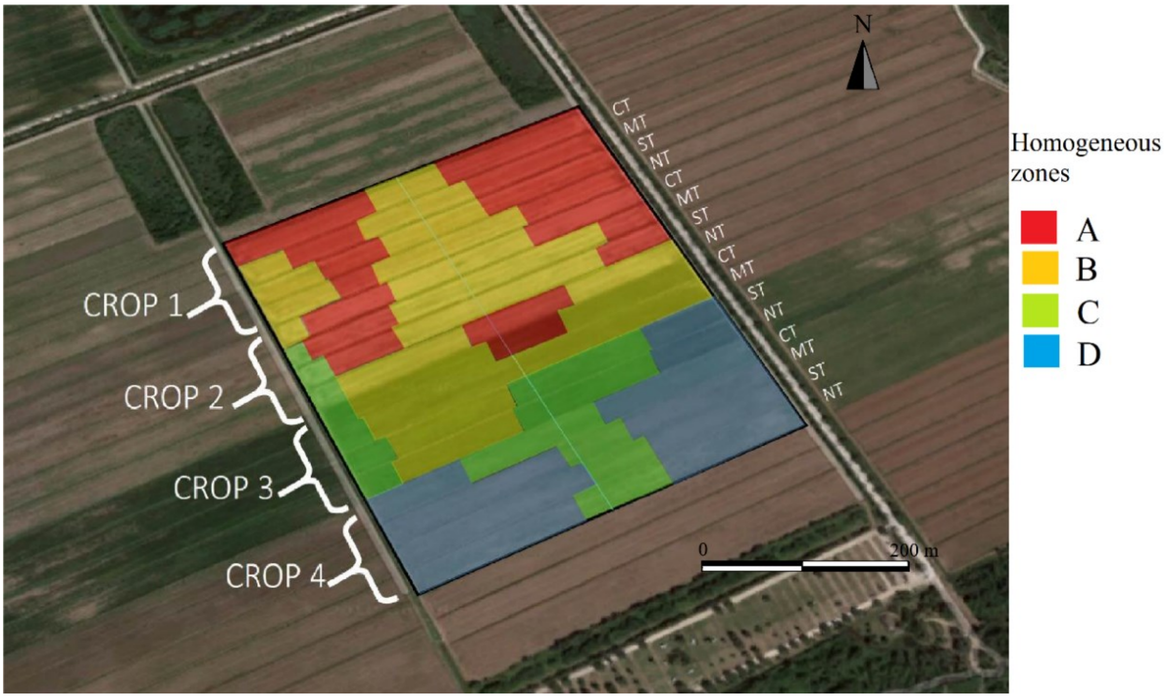
The CT was carried out only with the uniform rate technique, while the conservation soil tillage systems were applied to both the uniform and the variable application rate. Every plot was managed with variable (VRA) and uniform rate application (URA). The variable rate application consists with a variable application of nitrogen fertilization according to crop nutrient need, in a reduced overlapping and in a more efficient application of pesticides. The experimentation considers a crop rotation with the most representative crops in Veneto plane:

Maize (*Zea maize*), Soybean (*Glycine max*), Wheat (*Triticum aestivum*), and Rapeseed (*Brassica napus*) (Table 1). Data collection derives from a field dataset of a three-year crop rotation from 2014-15 to 2016-17.

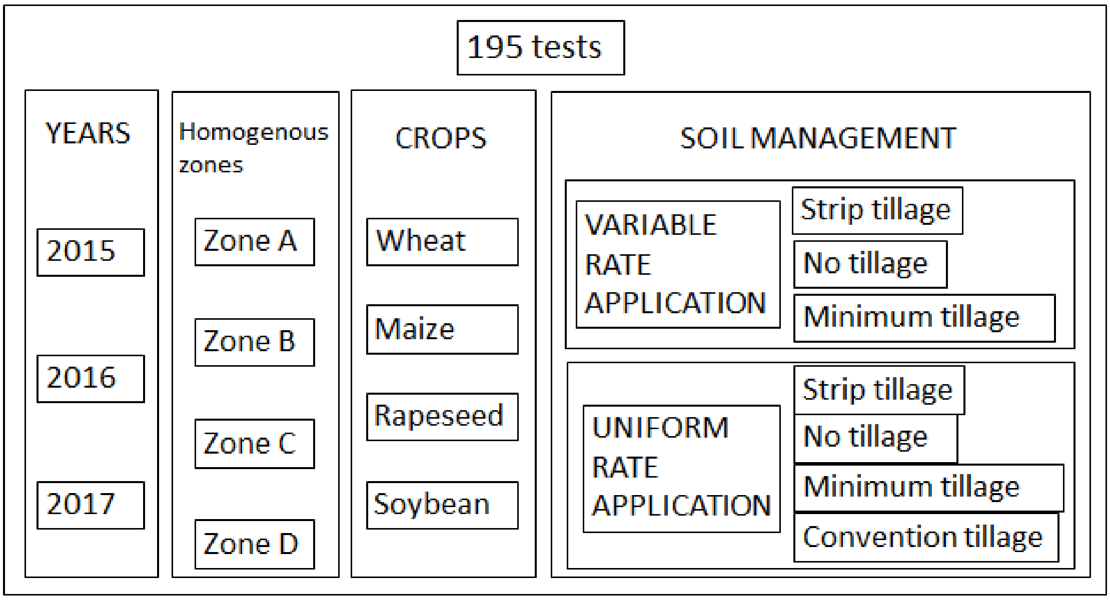
Table 1. Crop rotation schedule of maize, soybean, rapeseed, and wheat during the 3-year experimentation in 4 plots characterized by different texture (homogeneous zones).

Crop Rotation	PLOT 1	PLOT 2	PLOT 3	PLOT 4
2014–15	MAIZE	SOYBEAN	WHEAT	RAPESEED
Seeding	01 April 2015	25 May 2015	02 November 2014	06 September 2014
Harvesting	05 October 2015	28 October 2015	06 July 2015	23 June 2015
2015–16	SOYBEAN	WHEAT	RAPESEED	MAIZE
Seeding	23 May 2016	31 October 2015	28 August 2015	31 March 2016
Harvesting	24 October 2016	30 June 2016	21 June 2016	03 October 2016
2016–17	WHEAT	RAPESEED	MAIZE	SOYBEAN
Seeding	31 October 2016	30 August 2016	02 April 2017	25 May 2017
Harvesting	02 July 2017	23 June 2017	30 September 2017	27 October 2017

The crop rotation was replicated in four plots per soil tillage system, where four different patterns of crop rotation were established during the 3 years (Figure 1). A cover crop was sown after the main crop to keep a permanent soil cover. The cover crop was established on plots undergoing conservation tillage systems, which was devitalized with the soil cultivation in strip and minimum tillage or with the use of herbicides in no-tillage (Cillis et al., 2018a, 2017). The case study aims to assess the qualitative water footprint of the processes during different crop seasons and different soil tillage systems under precision (VRA) and tradition (URA) farming that are described in the experimentation pattern shown in Figure 1b.



(**a**)



(**b**)

Figure 1. Experimental field pattern. In (**a**), the map shows the annual plots setup of maize, soybean, wheat, and rapeseed under different homogenous zones; while in (**b**), the experimental setup shows the field organization and the soil management involved in 3 years of crop rotation.

The variable rate application differs among four different homogeneous zones (Figure 1a). The homogenous zones are identified using the Salus model in a map of prescription as homogenous areas of soil spatial variability of soil productive potential (Cillis et al., 2018b; Pezzuolo et al., 2017). The Salus model is designed to simulate crop production under different soil management, simulating the dynamics of nutrients with different atmospheric and soil conditions (Cillis et al., 2018a). Using the Soil Texture Triangle Hydraulic Properties Calculator from Saxton et al. (1986) (Saxton et al., 2010), three different soil textures were identified (Table 2): Sandy Loam Soil (Zones A and B), Loam Soil (Zone C), and Clay Loam Soil (Zone D).

Table 2. Soil properties of different homogeneous zones.

Soil Properties	ZONE A	ZONE B	ZONE C	ZONE D
Soil Electrical Conductivity (dS/m)	1.82	2.01	2.26	2.39
SAR (Sodium Adsorption Ratio)	0.46	0.50	0.35	0.32
pH	7.25	7.53	7.54	7.48
Total Nitrogen (%)	0.06	0.06	0.08	0.11
Organic matter (%)	1.22	1.23	1.71	2.38
P available (mg/kg)	32.83	30.0	30.9	29.5
K (mg/kg)	115.8	121.7	151.0	154.3
Clay (%)	15.17	16.33	22.14	32.00
Silt (%)	25.33	24.67	36.14	47.75
Sandy (%)	59.50	59.00	41.71	20.25

The fixed operation process for crop production are in order: ploughing, seeding, the use of herbicides, fertilizers and pesticides, and harvesting. The operation strategies applied in the field are shown in Table 3.

Table 3. Common soil management and operations during crops season.

Operations	Maize	Soybean	Wheat	Rapeseed
Ploughing and seedbed preparation	March	May	October	August
Fertilization before seeding	Fertilizer 8-20-20 (400 kg ha^{-1})	Fertilizer 0-20-20 (400 kg ha^{-1})	Fertilizer 8-20-20 (400 kg ha^{-1})	Fertilizer 8-20-20 (400 kg ha^{-1})
Herbicides pre seedling emergence	Lumax (4.21 L ha^{-1})	Corum+Harmony+Dash (947+4.2+315 mL ha^{-1})	Caliban Top (0.42 L ha^{-1})	Sultan (1.579 L ha^{-1})
Seeding	April	May	October	September
Herbicides post seedling emergence		Tuareg+Stratos ultra (947 + 4.2 L ha^{-1})		
1° fertilization during steam elevation	Urea (120 kg ha^{-1})		Ammonium nitrate (211 kg ha^{-1})	Ammonium nitrate (211 kg ha^{-1})
2° fertilization during steam elevation	Urea (230 kg ha^{-1})		Urea (211 kg ha^{-1})	Ammonium nitrate (211 kg ha^{-1})
Fungicide treatment			Prosaro (1.05 L ha^{-1})	
Insecticide treatment	Coragen (0.1053 L ha^{-1})		Karate (0.132 L ha^{-1})	Decis (0.526 L ha^{-1})
Harvesting	October	November	July	June

The common soil management and operations were replicated during the three years of experimentation. The term of soil management includes the different soil tillage systems, with either the application of variable or uniform rate. There is no nitrogen application in the case of soybean since it is assumed that the nitrogen nutrient is provided by the nitrogen fixation. The Precision Agriculture applied in the field is based on a Variable Rate Application (VRA) of nitrogen and seed density among different homogeneous zones and tillage systems (Table 4). The map of yield variability was recorded during harvesting and it discriminates the variable crop productivity across the homogeneous zones and soil tillage systems (Table 5). The yield was summarized as the cumulative crop rotation yield under the common field plot. The cumulative yield also represents the yield from different homogeneous zones, which was previously normalized on the respective area (Cillis et al., 2017). The harvesting operation was made with a harvesting machine that recorded and integrated the entire points yield in a precision map. The experimental pattern of variable application is kept constant along the three years according to the prescription map. In Table 4, different rates of inputs are recorded.

Table 4. Variable rate of nitrogen during crop season and for each homogeneous zone.

Tillage System	Homogeneous Zone	Seed Density (seeds m^{-2})				Fertilization (Kg N ha^{-1})			
		MAIZE	SOYBEAN	WHEAT	RAPESEED	MAIZE	SOYBEAN	WHEAT	RAPESEED
CT	-	7.5	45	500	50	193	-	178	128
MT	A	6.0	55	500	50	180	-	150	140
	B	7.0	50	500	50	190	-	190	120
	C	8.5	40	500	50	180	-	140	100
	D	9.5	35	500	50	200	-	180	110
ST	A	6.5	55	260	55	190	-	150	140
	B	7.5	50	260	55	200	-	190	120
	C	8.5	40	260	55	170	-	130	110
	D	9.5	45	260	55	190	-	190	120
NT	A	6.5	55	550	55	200	-	150	150
	B	7.5	50	550	55	210	-	190	130
	C	8.5	40	550	55	200	-	130	140
	D	9.5	35	550	55	220	-	170	120

Table 5. Average yield (tons ha^{-1}) of three-year crop rotation of different crops, zones, and soil practices.

		MAIZE (tons ha^{-1})		SOYBEAN (tons ha^{-1})		WHEAT (tons ha^{-1})		RAPESEED (tons ha^{-1})	
		URA	VRA	URA	VRA	URA	VRA	URA	VRA
CT	A	8.77		1.43		4.35		2.47	
	B	9.60		2.91		5.00		2.48	
	C	10.26		3.84		6.18		3.02	
	D	10.06		3.73		7.47		3.03	
MT	A	8.48	7.44	1.52	1.64	4.78	4.38	2.70	2.58
	B	9.73	7.73	2.72	3.44	5.09	5.18	2.67	2.37
	C	10.38	**10.63**	3.85	**4.20**	5.74	5.93	2.95	2.85
	D	9.86	10.22	2.91	3.73	**8.37**	8.03	2.22	2.19
NT	A	7.79	9.12	2.64	2.89	4.32	5.77	1.82	2.04
	B	10.12	10.31	3.42	3.51	4.84	4.56	1.35	2.38
	C	7.53	9.47	3.80	3.40	6.70	6.11	2.26	2.20
	D	7.22	9.85	3.14	3.00	7.85	6.94	-	-
ST	A	6.18	6.87	1.71	2.06	3.84	3.88	2.03	2.06
	B	6.17	7.42	2.94	3.11	4.44	4.52	1.95	2.28
	C	8.78	9.36	3.62	3.85	5.36	6.03	2.14	**3.22**
	D	7.18	9.31	3.54	3.66	7.16	7.80	-	-

5.3.2 Description of the Methodology Applied to the Water Footprint Assessment

The Water Footprint Assessment ideated by Hoekstra et al. (2011) focuses on two components for the consumptive water use and one component for the water quality aspect (Hoekstra et al., 2011). In this study, we focus on the grey water footprint index. It is an environmental indicator of water degradation and pollution. It becomes useful when assessing the impact on water resource due to human activities. In the case of agriculture processes, the water quality index relies on the grey water that is the freshwater volume needed to assimilate pollutants under a standard threshold of water body quality. The grey WF considers the use of pesticides and fertilizers applied in the fields. The study compares the grey WF of the process of crop production within different soil tillage systems and soil management. The Water Footprint is the relation between the volume of water to dilute pollutants, and the yield, in terms of m^3 per tons, or the surface, in terms of m^3 per hectares. In order to compare the effect of soil tillage systems across a crop rotation, we considered the grey Water Footprint (grey WF) of soil management as a sum of the process during the three years of crop rotation in a specific field plot. The grey WF is then compared undergoing variable or uniform soil management. In addition, the grey WF in terms of m^3 ha^{-1} is put in relation to the grey WF in terms of m^3 $tons^{-1}$. The relation can explain which soil practice reduces the water pollution in a context of intensive or extensive agriculture.

5.3.3 Assessment of the Grey Water Footprint

The Grey water is the volumetric amount of water to dilute pollutants under an acceptable level of water quality. Agriculture production is the main actor in water body pollution. Since The Grey Water Footprint, Grey $WF_{[i,p]}$, is a qualitative indicator of water pollution, we assess the volume of water that dilutes the most critical pollutant (p) under a sufficient level of water quality, as given by Franke et al. (2013) (Franke et al., 2013), and for each crop production (i):

$$\mathrm{GreyWF}_{[i,p]} = \frac{\propto * \mathrm{Appl}_{[t]}}{(\mathrm{C}_{max} - \mathrm{C}_{nat})} = \left[\frac{\mathrm{Volume}}{\mathrm{Area}}\right] \quad (1)$$

where $Appl_{[t]}$ is the application rate at time [t], and α is the leaching-runoff coefficient improved by Franke et al. (2013) (Franke et al., 2013) as

$$\alpha = \alpha_{min} + \left[\frac{\sum_i (s_i * w_i)}{\sum_i (w_i)}\right] * (\alpha_{max} - \alpha_{min}) \quad (2)$$

where the α_{max} and α_{min} are respectively the maximum and the minimum leaching-runoff factors, while s_i is the leaching-runoff potential and w_i is a weighting factor, and they vary according to the properties of the chemical substance. The α_{max} for nitrogen is fixed at 0.25, at 0.05 for phosphorus, and 0.01 for pesticides, while the α_{min} is fixed at 0.01 for nitrogen, at 0.0001 for phosphorus and pesticides. The toxicity of the pesticide to the selected non-target organisms was assessed using data of eco-toxicological and toxicological parameters derived from databases (FOOTPRINT, 2006) (University of Herthfordshire, 2013b). The maximum acceptable concentration (C_{max}) is an ambient water quality standard for pollutants. We considered a C_{max} of 50 mg nitrate-N L^{-1}, or 11.3 mg N L^{-1} for nitrogen (Chukalla et al., 2017), the value of 0.02 mg P2O5 L^{-1} for Phosphorus (Mekonnen and Hoekstra, 2018). We also considered a C_{max} of 0.1 µg L^{-1} for single application and 0.5 µg L^{-1} for total residues of pesticides, and according to the EU limits for pesticides in drinking water for individual substances (European Union Council, 1998; Hamilton et al., 2003). For the specific chemicals included into the Franke et al. (2013) and Hamilton et al., (2003) guidelines we considered the reported C_{max} threshold (Table 6). The natural concentration (C_{nat}) applied following the guidelines concerns a value of 0.1 mg L^{-1} for nitrogen, 0.01 mg L^{-1} for Phosphorus, and zero for pesticides. In addition, we calculated the N Grey WF (equation 1) for the only impact of nitrogen on water quality.

Table 6. Maximum acceptable concentration (µ L^{-1}) for pesticides in water and used in the field.

Chemicals	**Type of Chemical**	**C_{max} (µ L^{-1})**	**Source**
Bentazon	Herbicide	30	(Hamilton et al., 2003)
Chlorsulfuron	Herbicide	100	(Hamilton et al., 2003)
Deltamethrin	Insecticide	$0.4 \cdot 10^{-3}$	(Franke et al., 2013)
Glyphosate	Herbicide	800	(Franke et al., 2013)
Metolachlor	Herbicide	7.8	(Franke et al., 2013)
Metribuzin	Herbicide	1	(Franke et al., 2013)
Terbuthylazine	Herbicide	7	(Hamilton et al., 2003)

The Grey Water Footprint, Grey $WF_{[i,p]}$, can be assessed according to the quantitative production of the crop process in every management package (y):

$$\mathrm{Grey\ WF}_{[i,p,y]} = \frac{\mathrm{Grey\ Water\ Footprint}_{p,i}}{\mathrm{Yield}_i} = \left[\frac{\mathrm{Volume}}{\mathrm{Weight}}\right] \quad (3)$$

The Grey Water Footprint stands on the concept of the critical load for water pollution, which concerns the assimilation capacity of a water body and the acceptable concentration of pollutants receiving in the water body. The resulting grey WF is the maximum value of the critical pollutant on each process under different methods of crop production and soil management. The present methodology considers only the indirect effect of using different soil tillage systems can have on the environment. Some limitation can be found on the interpretation of the volumetric value of grey WF, which is a volumetric indication of the water pollution magnitude of the different soil management. The main limit on the analysis of soil management effects on water pollution is the limited studies on the soil management effects on chemicals fate, and the scarcity of soil models availability linking soil tillage effects. Another limitation during the field activities was to conduct a groundwater survey able to detect chemical residues for the different methods of soil management since the groundwater quality might be affected by the leaching from one plot to another. Therefore, we implement the aforementioned methodology of the grey WF assessment. The absolute value is to be considered as a mere indication of water pollution and not a real volume. A further improvement of the case study can analyze not only the grey WF, but also different indicators of water quality or the environmental effects of using different tillage systems on soil moisture increase and on irrigation water saving.

5.4 Results

The study analyses the grey WFs of crop rotation under different soil tillage systems with the variable and uniform rate of input applied. For that reason, the grey WF of crop rotation under the same soil variability was considered. The yield of crop rotation under different soil tillage systems differs among the soil productivity. The soil productivity is the result of three years of crops rotation under the same soil variability. Yield variability considers different soil type, different plant density, and different fertilizers application. The dataset presents a high statistical variability among the tests; therefore, a crop rotation analysis was assessed. The main outcomes show an of yield variation from uniform rate application (URA) to the variable rate application (VRA), where, generally, the conservation soil practices benefit from a greater yield under VRA. In the case of Minimum Tillage (MT), the production is reduced within the use of precision agriculture, where only 2% yield reduction is recorded from URA to VRA. In the others conservation tillage systems, the yield records a better trend and it gains a positive effect by an increase of 6% and 8% under Strip (ST) and No-tillage (NT) respectively.

The indirect effects of soil management under different tillage systems on water pollution are, most of the time, the indirect consequences of the applied inputs during the operational phases. The effect of precision agriculture with VRA is more effective and it shows a grey WF reduction for every soil management practice. In Figure 2, the grey WF of the process of crop rotation is shown in terms of m^3 ton^{-1}. ST VRA and NT VRA reduce the grey WF by the 10% and 11% from ST URA and NT URA respectively, while MT VRA reduces only by the 1.3%. Nevertheless, MT VRA has a 6% reduction compared to CT. Generally, the higher level of water pollution in ST and NT, with respect to MT, is generally due to the higher number of treatments for weeds control. The use of VRA has a higher reduction in the grey WF in relation to the crop yield. The technique of precision agriculture permits to reach a lower grey WF with a more efficient application of inputs, thanks to a smaller amount of pesticides and fertilizers and the higher effective rate spread in the field that reduces the impact on water pollution. In other terms, the water pollution can be reduced by the use of suitable soil management without compromising the yield.

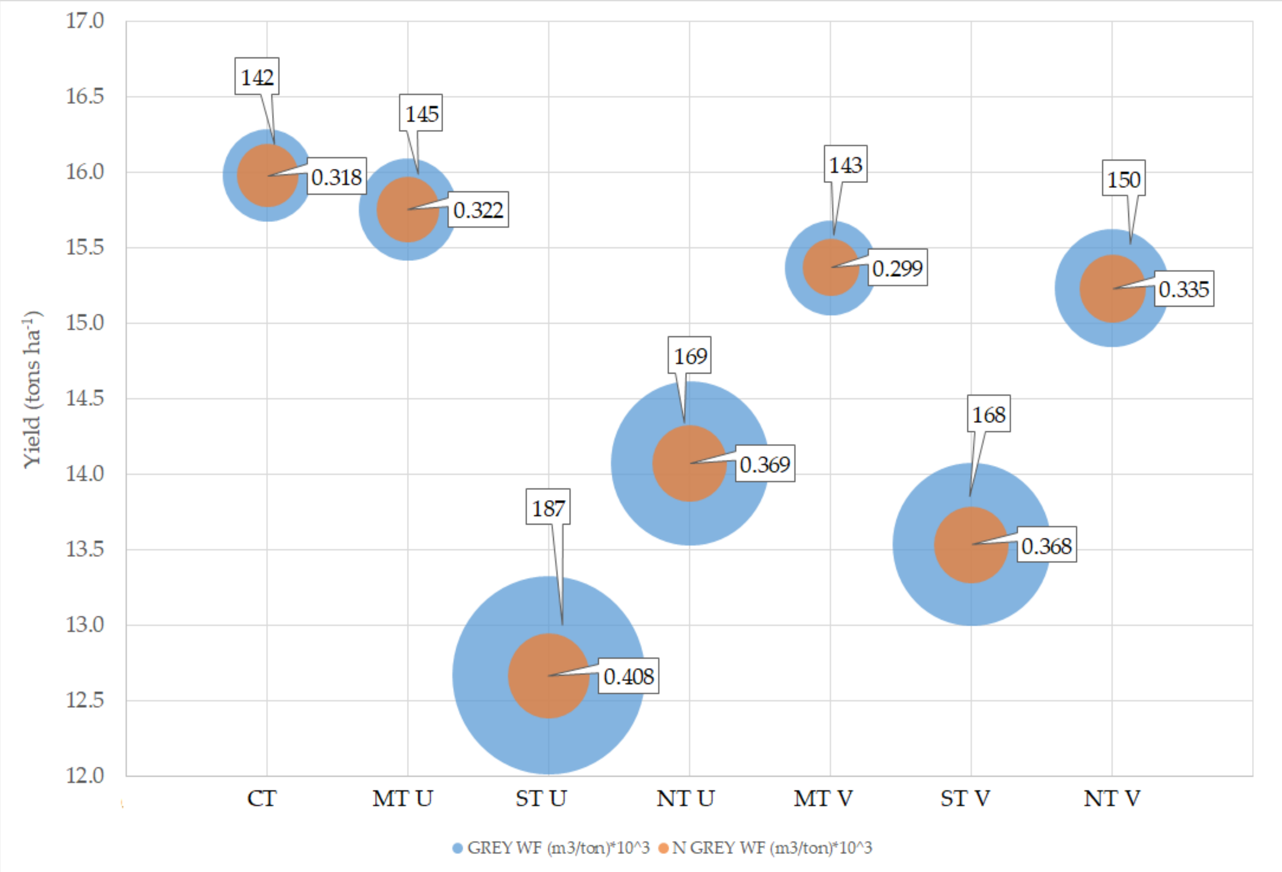


Figure 2. Relation of the three-year yield of crop rotation under different soil tillage systems (tons ha^{-1}), with the Grey Water Footprint (WF) and N Grey WF (m^3 ton^{-1}) $\cdot 10^3$ of soil tillage systems undergoing variable (VRA) and uniform application rate (URA) per unit of production.

In order to understand which soil tillage can reduce or minimize the grey WF in certain circumstances; we address the evaluation of the grey WF for the nitrogen (N) application (Figure 2). The use of VRA decreases the grey WF by the 7% in MT, 10% in ST, and 9% in NT. Also, in the case of Nitrogen, the water pollution can be reduced by the use of suitable soil practices. Comparing different soil tillage systems under VRA, ST and NT have a higher grey WF value, in terms of m^3 ton^{-1}, by 19% and 11% than MT respectively. The use of suitable soil tillage can indirectly reduce the water pollution from nitrogen losses; especially, MT minimizes the grey WF without compromising the yield. In Figure 2, MT VRA reduces the N grey WF (m^3 ton^{-1}) by 6% more than CT. The VRA practices decrease the N leaching loss and increase the crop N uptake. ST VRA and NT VRA, on the other hand, have a greater grey WF reduction than MT VRA.

The effect of soil management on the grey WF of the process for the crop rotation per unit of area is described in Figure 3. The graph shows the variability between soil tillage systems in terms of volume of grey WF per hectare (m^3 ha^{-1}). There is a clear picture that VRA reduces the grey WF of field process for every soil tillage system. The reduction changes between 3.7% and 3.8% undergoing the use of VRA, where the MT VRA has the lower grey WF. The URA has the highest grey WF in all the soil tillage practices, and every soil tillage benefit from a reduction in water pollution using the VRA. Looking in detail at the VRA management, the comparison between tillage practices shows that the MT technique has the lowest grey WF with a decrease of 3.4% more than CT. Therefore, the use of VRA reduces the grey WF on conservation tillage systems in terms of water pollution per unit of surface. The MT VRA performed a better grey WF (m^3 ha^{-1}) reduction comparing the others conservation tillage by the 3.6% and 4% from ST VRA and NT VRA respectively. In Figure 3, soil tillage systems can affect the grey N WF in terms of m^3 ha^{-1}. In fact, the use of the VRA management reduces the grey WF by the 9.5% under MT, 3.6% under ST and NT, with 1.8% from uniform rate application. MT VRA reduces the grey WF by 8% and 10% more than ST VRA and NT VRA respectively. MT VRA reduces also the N grey WF by 9.6% more than CT. In this circumstance, MT VRA represents the most sustainable soil practice with the lowest N grey WF (m^3 ha^{-1}) value and the greater grey WF reduction.

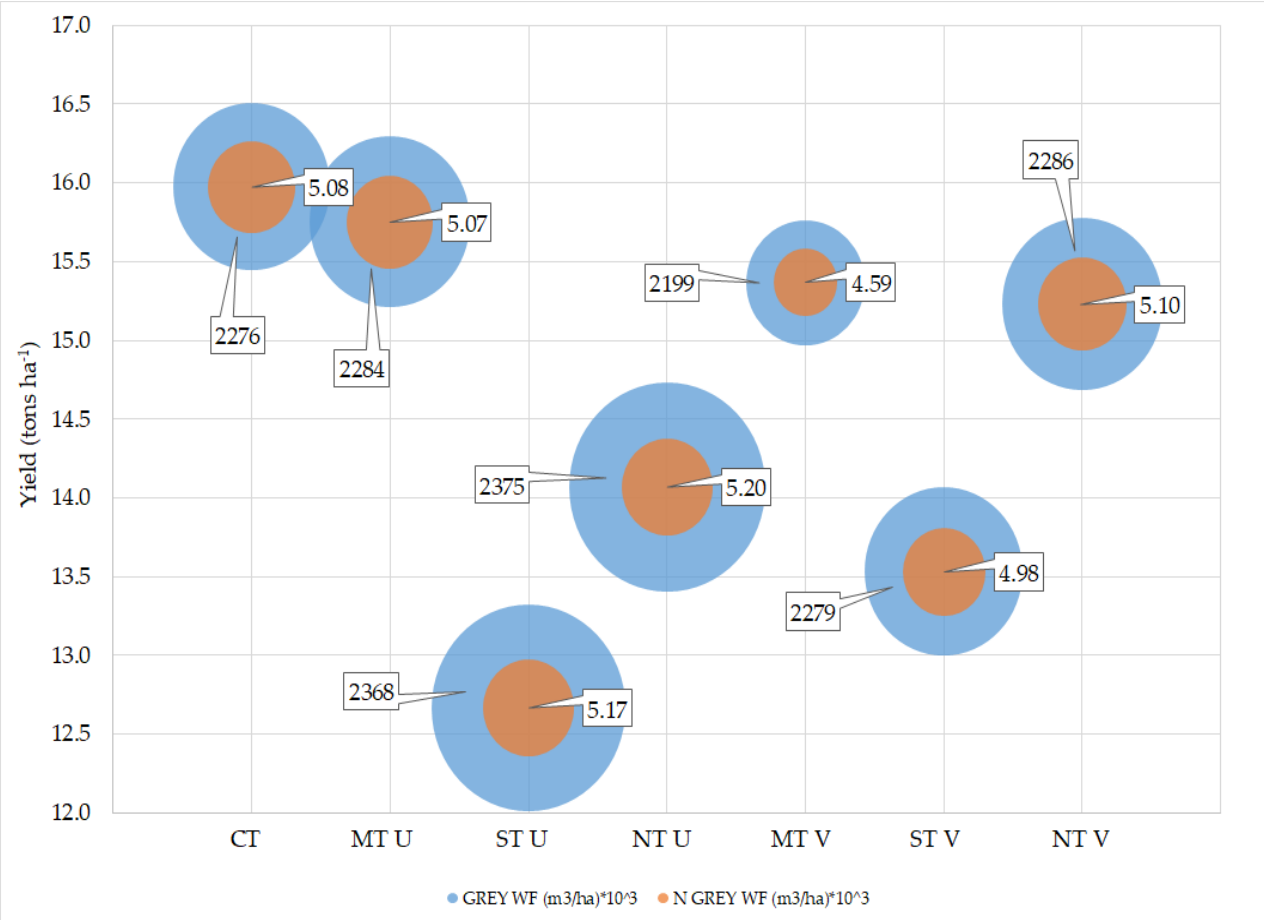


Figure 3. Relation of three-year yield of crop rotation under different soil tillage systems (tons ha^{-1}), with the Grey WF and N Grey WF (m^3 ha^{-1}) $\cdot 10^3$ of soil tillage systems undergoing variable (VRA) and uniform application rate (URA) per unit of surface.

5.5 Discussion

The use of one or the other soil tillage systems might have an indirect benefit on the reduction of water pollution (Ghaley et al., 2018). Different rate applications affect the grey WF that provides a trade-off on the decision-making level (Ibarrola-Rivas and Nonhebel, 2016). Conventional Tillage (CT) is generally the common technique applied in the fields. In this case study, CT was compared with conservation tillage systems. Minimum Tillage (MT) minimizes and decreases the grey WF in comparison either with CT both with Strip Tillage (ST) and with No-Tillage (NT). Conservation tillage could have some environmental benefits and indirect effects on water pollution (Ghaley et al., 2018; Tarolli et al., 2019). The increase of soil organic content permits decreasing the carbon oxide emissions and having a greater energetic efficiency (Cillis et al., 2018b). In this study, a preliminary grey WF assessment on the field dataset is made to understand the effects of conservation tillage and precision agriculture on water pollution.

5.5.1 Effects of Soil Management to the Grey Water Footprint Reduction

Conservation tillage provides better outcomes if combined with precision agriculture. The MT VRA technique delivers the greater trend on grey WF reduction even if the yield decreases by 2%. The operation phases in MT VRA positively affects the environmental performance compared to the others tillage systems. Especially in the case of N grey WF, the MT VRA system gives a better reduction both in terms of m^3 $tons^{-1}$ or m^3 ha^{-1}. In any case, MT VRA minimizes and reduce the water pollution under the level of the referenced CT. Similarly, ST VRA and NT VRA have a greater reduction of water pollution combining with the URA management in terms of m^3 $tons^{-1}$. The low reduction of water degradation per unit of area is due to change in inputs application during the operations phases. In detail, a different application of fertilizers and pesticides affects the value of water pollution according to soil parameters and the chemical features of the substances. In order to understand which active substance mostly affects the grey WF, a list of chemicals applied in the field was made for the four herbaceous annual crops. In Table 7, different values of the grey WF were considered. The interval of grey WF refers to the minimum and the maximum amount of chemicals suggested in the label indications, while the field value is the grey WF for chemicals applied in the field study. The table ranks the substances with a decreasing order of the grey WF of the interval. The value is considered as the sum of the three-year crop rotation. The assumption of having a three-year value is due mostly to the magnification of the grey WF variability among crops and soil management. It is interesting to have a look to which insecticides and herbicides present the higher grey WF value and which could be used with caution. In the ranking list, herbicides are the most critical group of substances for water pollution. The grey WF and therefore the water degradation might be decreased by the use of different chemical substances with a lower impact on water pollution, or with the use of the lower application rate.

Table 7. Grey WF of chemicals applied in the field. The minimum and the maximum grey WF rely on the minimum and maximum amount of chemicals applicable according to the label.

Type of Substance	Crop Concerned	Chemical or Active Substance	Field Grey WF (m^3 ha^{-1})	INTERVAL of Grey WF (m^3 ha^{-1})
Insecticides	Rapeseed	Deltamethrin	$14.5 \cdot 10^6$	$17 \cdot 10^6$–$27 \cdot 10^6$
Herbicides	Rapeseed	Metazachlor	$3.56 \cdot 10^5$	$4.75 \cdot 10^5$–$9.95 \cdot 10^5$
Fertilizers	All crops	P2O5 (phosphorus)	$3.08 \cdot 10^5$	$1.59 \cdot 10^5$–$3.82 \cdot 10^5$
Herbicides	Soybean	Imazamox	$2.75 \cdot 10^5$	$3.61 \cdot 10^5$–$7.77 \cdot 10^5$
Herbicides	Soybean	Cycloxydim	$2.48 \cdot 10^5$	$3.54 \cdot 10^5$–$4.51 \cdot 10^5$
Herbicides	Soybean	Flufenacet	$2.26 \cdot 10^5$	$2.71 \cdot 10^5$–$6.03 \cdot 10^5$
Herbicides	Maize	Mesotrione	$0.86 \cdot 10^5$	$0.86 \cdot 10^5$–$1.78 \cdot 10^5$
Fungicides	Wheat	Tebuconazole	$0.61 \cdot 10^5$	$0.63 \cdot 10^5$–$1.54 \cdot 10^5$
Herbicides	Maize	Terbuthylazine	$0.61 \cdot 10^5$	$0.61 \cdot 10^5$–$1.27 \cdot 10^5$
Herbicides	Rapeseed	Propaquizafop	$0.03 \cdot 10^5$	$0.57 \cdot 10^5$–$0.91 \cdot 10^5$
Herbicides	Wheat	Propoxycarbadone sodium	$0.39 \cdot 10^5$	$0.39 \cdot 10^5$–$0.81 \cdot 10^5$
Fungicides	Wheat	Prothioconazole	$0.38 \cdot 10^5$	$0.38 \cdot 10^5$–$0.99 \cdot 10^5$
Insecticides	Maize	Chlorantraniliprole	$0.13 \cdot 10^5$	$0.19 \cdot 10^5$–$0.38 \cdot 10^5$
Herbicides	Wheat	Amidosulfuron	$0.15 \cdot 10^5$	$0.15 \cdot 10^5$–$0.30 \cdot 10^5$
Herbicides	Wheat	Mefenpir-diethyl	$0.12 \cdot 10^5$	$0.12 \cdot 10^5$–$0.26 \cdot 10^5$
Herbicides	Soybean	Metribuzin	$0.10 \cdot 10^5$	$0.12 \cdot 10^5$–$0.26 \cdot 10^5$
Insecticides	Wheat	Lambda-cialotrina	$0.07 \cdot 10^5$	$0.11 \cdot 10^5$–$0.18 \cdot 10^5$
Herbicides	Maize	S-metolachlor	$0.07 \cdot 10^5$	$0.07 \cdot 10^5$–$0.14 \cdot 10^5$
Herbicides	Wheat	Tribenuron methyl	$0.06 \cdot 10^5$	$0.06 \cdot 10^5$–$0.14 \cdot 10^5$
Herbicides	Soybean	Tifensulfuron methyl	$0.01 \cdot 10^5$	$0.03 \cdot 10^5$–$0.06 \cdot 10^5$
Fertilizers	All crops	N (nitrogen)	$0.04 \cdot 10^5$	$0.02 \cdot 10^5$–$0.06 \cdot 10^5$
Herbicides	Wheat	Iodosulfuron-methyl-sodium	$0.02 \cdot 10^5$	$0.01 \cdot 10^5$–$0.03 \cdot 10^5$
Herbicides	Soybean	Bentazon	$0.007 \cdot 10^5$	$0.01 \cdot 10^5$–$0.02 \cdot 10^5$
Herbicides	All crops	Glyphosate	$0.0006 \cdot 10^5$	$0.0016 \cdot 10^5$–$0.0017 \cdot 10^5$
Herbicides	Wheat	Chlorsulfuron	$0.0001 \cdot 10^5$	$0.0001 \cdot 105$–$0.0002 \cdot 10^5$

The most critical substance in the field is Deltamethrin, an insecticide applied to Rapeseed that is highly affecting the water pollution according to its chemical features. Table 7 shows the Rapeseed is the crop having the highest grey WF value during the crop rotation due to the chemical substances required in the field.

5.5.2 Soil Tillage Solutions to Reduce the Impact on Water Pollution throughout the Dilemma of Intensification or Extensification in Agriculture

Soil tillage systems have different effects on water pollution in terms of intensive or extensive agriculture (Ghaley et al., 2018). As mentioned in the results part, only the MT VRA technique provides a good compromise between grey WF reduction in terms of m^3 $tons^{-1}$ or m^3 ha^{-1}. The further use of VRA management delivers a better choice in grey WF reduction. ST VRA and NT VRA, especially, have a high grey WF reduction for an intensive agriculture. This is more visible in Figure 4, where the graph describes a different pattern of grey WF undergoing precision or no-precision agriculture and regarding different possible solutions to achieve yield productivity without increasing the water pollution. The direct and indirect water pollution of a crop rotation under different tillage practices can be considered for agricultural extensification or intensification proposal. Different soil tillage techniques are addressed in different perspectives for the water pollution impact across an intensive or an extensive agriculture (Crews and Rumsey, 2018). As shown in Figure 4, the MT has the lower grey WF in terms of m^3 ton^{-1} and m^3 ha^{-1}. This means that MT practice can minimize both in an extensive and in an intensive agriculture. MT can be assumed as the referenced tillage system for further case studies like the more sustainable tillage system in terms of indirect effects on water pollution. MT compromises the perspective of an intensive or an extensive agriculture reducing the grey WF in both of the cases. When precision agriculture is addressed on ST and NT, the impact is reduced either on the extensification or on the intensification perspective. Precision agriculture decreases the grey WF of MT on the only extensification perspective. In other hands, the use of a more efficient soil management like the precision agriculture with a variable rate application can reduce the impact of water pollution. The VRA management decreases the effects of extensification of water pollution for all the tillage practices, while it decreases also the effects of intensification on water pollution for the ST and NT practices.

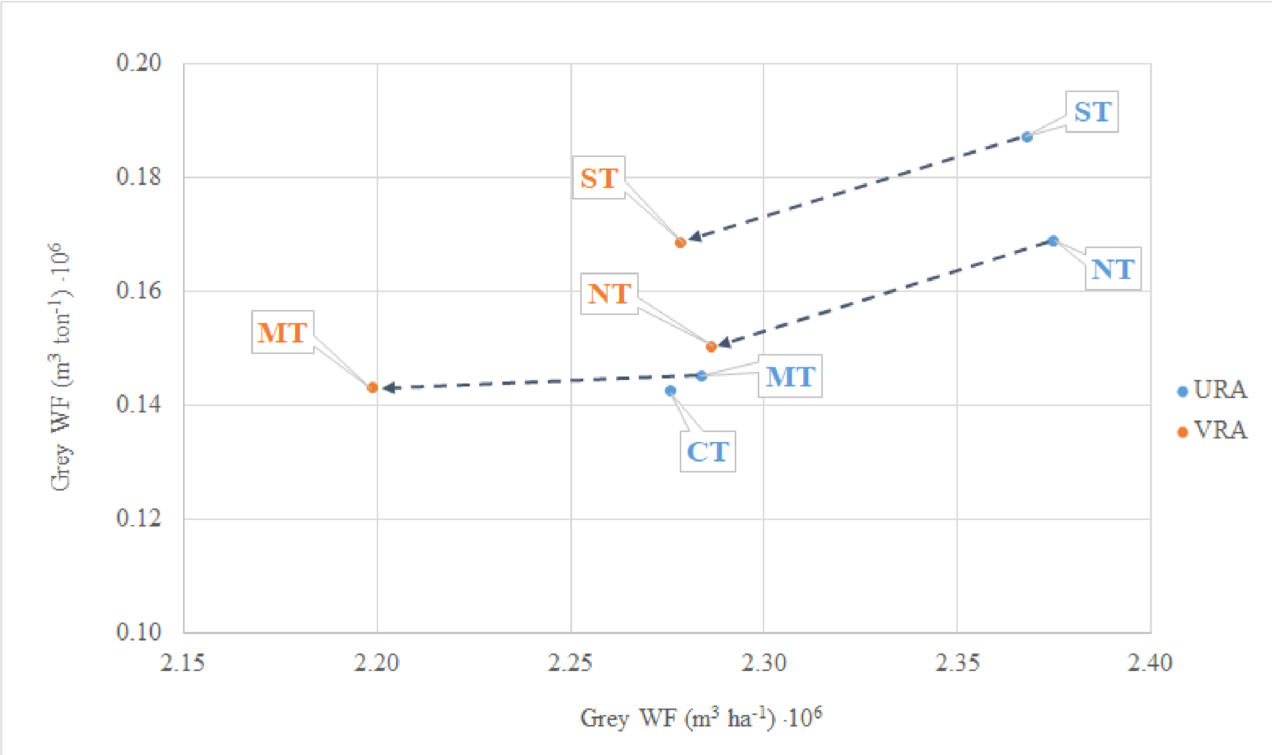


Figure 4. The pattern for suitable soil tillage practices under precision agriculture and over a three-year crop rotation, regarding the grey WF in terms of m^3 ton^{-1} and m^3 ha^{-1}.

The grey WF estimated in this paper does not consider more than three years of crop rotation. A further study might be done for a future scenario for more than ten years in order to understand the long-term effects of tillage systems for multiple crop rotation. The field dataset could calibrate the models for the long-term scenario in order to replicate the experiment in other location, climate, and different soil condition. A statistical analysis of variability is not shown since there is not a clear trend between variables in all the tests. The number of tests does not permit to having a clear picture of the variance.

5.6 Conclusions

The case study describes how different methods of soil management enhance a grey water footprint reduction. The study focuses on the synergetic effects of different soil practices as different soil tillage systems and the use of variable rate application, which have a positive grey WF reduction. Accordingly, the grey water footprint pattern is shown considering an extensive or intensive farming system. The results highlight which interaction between soil tillage systems and soil management reduces the grey WF of soil practices. The variable rate application consistently decreases the impact on water in terms of water pollution and chemical soil degradation. The Minimum Tillage presents a lower WF, both in terms of m^3 ton^{-1} and in m^3 ha^{-1} with the use of Precision Farming. ST VRA and NT VRA have a higher grey WF reduction in terms of m^3 ton^{-1} with a 10% and 11% of reduction respectively. The grey WF of Nitrogen application is better reduced within MT VRA in terms of m^3 ha^{-1} by the 9.5%. In order to decrease the water pollution and following agronomical best practices, we should prioritize the reduction, for example, of the amount of insecticides and herbicides more than fertilizers, or choose chemicals with a lower grey WF, or use an aforementioned sustainable soil management, or the interaction of all of these solutions. The case study provides suggestions for suitable soil management in order to reduce the water pollution. Further improvements of the study should consider the consumptive impact on water resource using different soil practices, in order to analyze the water saving through the soil moisture available for the crop. Furthermore, the study considers a three-year crop rotation; an additional work could consider a longer-term effect of tillage systems.

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6. Use of Multiple Indicators to compare Sustainability Performance of Organic vs Conventional Vineyard Management [4]

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6.1 Abstract:

The wine sector is paying more attention to sustainable wine production practices, but this topic is highly debated because organic viticulture aims to a reduction of environmental impacts, while conventional viticulture ensures an increase of yield. This work provides an economic and environmental comparison using different indicators whereas no previous studies on viticulture have faced on both aspects of sustainability. Two distinct vineyards within the same case study farm were considered, where conventional and organic viticulture practices were applied for 5 years. For each type of production, we calculated the economic benefit and environmental indicators such as the Water Footprint, Carbon Footprint, and an indicator of environmental performance associated with the vineyard phase ("Vineyard Management" or "Vigneto" indicator part of the Italian VIVA certification framework). This latter considers six sub-indicators investigating pesticides management, fertilizers management, organic matter content, soil compaction, soil erosion, and landscape quality. The multi criteria approach is a novel framework assessing sustainability on vineyard management using environmental indicators from VIVA calculator and the economic aspect. Main results showed that organic management in viticulture can be applied without having economic losses and with the benefit of better preserving the natural capital.

Keyword: viticulture, organic, conventional, sustainability, management, innovations

6.2 Introduction

The wine sector is increasingly paying more attention to sustainable practices promoting environmental impact reduction in wine production (Rinaldi et al., 2016). The public opinion debates about sustainability of

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conventional management, which is generally considered less sustainable than the organic management (Aschemann-Witzel and Zielke, 2017). Generally, producers claim that an organic production has a lower yield, which might become an obstacle to reach food security at global level (Kavargiris et al., 2009). Although it is recognized that organic production reduces the impacts on the environment, conventional agriculture typically ensures greater incomes thanks to the higher yield (Dal Ferro et al., 2017; Schrama et al., 2018). On the other hand, organic production is recognized to increase biodiversity and to promote the maintenance or improvement of ecosystem services (Rennings and Wiggering, 1997; Sandhu et al., 2010). Organic farming has a great environmental benefit easily transferable to small scale farms. For example, biodiversity is enhanced by biological control of pests using deterrents and by attracting their natural predators or pathogens providing them food and habitat (Patinha et al., 2018). Moreover, Caprio et al., (2015) discussed about the fact that organic vineyard management enhances biodiversity improving habitats for carabids and other arthropods predators; the preference on choosing organic vineyard rather than a conventional one depends on small-scale landscape variables that can promote rather one or another population. Organic farming can provide further ecosystem services, as for example the increase of carbon stock on soil and related fauna, which enhances benefits for plant growing, and it reduces soil erosion and compaction (Brunori et al., 2016; Puig-Montserrat et al., 2017). In fact, carbon sink is generally higher under organic management where soil respiration rate (controlled by physical-chemical factors such as organic carbon content, fine roots' density and microbial biomass, but also by abiotic factors like soil temperature and moisture) is generally higher than in conventional management (Brunori et al., 2016). In addition, an organic management can reduce the environmental impact, both in terms of Greenhouse Gas (GHG) emissions and water depletion (Abbona et al., 2007). In the first case, a lower energy input with a higher energy efficiency is responsible of the lower GHG emission under organic management (Chiriacò et al., 2019; Kavargiris et al., 2009). In the second case, the use of agro-ecological activities within organic management increases water security and soil water retention saving water and reducing the vulnerability to water scarcity (Costa et al., 2016; Wheeler and Marning, 2019). Finally, organic management seems to be fully environmentally sustainable, while on the other hand, economic sustainability might represent an issue for the producers. Those different conditions hamper the decisional process towards the identification of a fully sustainable solution (Meier et al., 2015). Furthermore, organic food is considered safer than conventional food and this is an important driver for consumers' perception and related commercial choices. However, recent studies demonstrated that organic food may have contaminants residues as well (González et al., 2019). In the case of wine production, recent studies detected more pesticides residues in conventional products, while comparable amount of sulphite and other residues were found in organic wine (Vitali Čepo et al., 2018). Beside the safety and quality of organic products, consumers may be discouraged from choosing organic foods or wine due to their higher price (Abraben et al., 2017). Generally, organic foods are usually reworded by 30% more than conventional foods (Aschemann-Witzel and Zielke, 2017). This trend may change when consumers purchase a bottle of wine; the choice between organic or conventional wine depends on the consumer perception about wine quality and their capability to award the product with a higher price that depends on consumer segments (societal class of origin) (Costa et al., 2016). Finally, the willingness to pay is generally higher for organic wine or food due to the recognition of a product with a supposed better quality and higher attention to the environment (Sandhu et al., 2010).

This study provides a comparison of the environmental and economic performances of organic and conventional vineyard management based on a set of environmental and economic indicators (namely, Water Footprint, Carbon Footprint, an indicator related to vineyard management practices, and an economic indicator based on the net income) supported by a case study in the North-East of Italy. Moreover, the main research questions, on which the manuscript focuses on, are: i) is an organic vineyard management the only sustainable vineyard management? ii) Is an organic vineyard management less economically convenient?
The final purpose of this study is to evaluate sustainability quantitatively on both environmental and economic aspects for an organic and conventional vineyard management. Previous literature studies typically focused only one aspect of wine sustainability. For example, in Kavargiris et al., (2009) only the GHG emissions assessment and an energetic analysis were performed, while Brunori et al., (2016) focused on the assessment of carbon storage in the case of organic vineyard management. Costa et al., (2016) individuated multiple indicators for sustainable water use in vineyard, stressing the fact that innovations on new drought resistant genotypes, field techniques and innovative technologies reduce the risk of water scarcity and enhance sustainability in vineyard. Flores (2018) revised different frameworks for wine sustainability and pointed out how organic viticulture is not usually a synonym of sustainable viticulture; sustainable frameworks use indicators assessment for certification or labels. In Corbo et al., (2015), the review reported how literature lack on indicators assessment, where the environmental and economic aspects are conceptually included into the framework. Climate change consequences require different mitigation strategies as well as suitable vineyard management able to cope with the increasing temperature (Eccel et al., 2016). Nowadays, there is a lack of consensus about the benefits of applying organic instead of conventional vineyard management and it is still difficult to draw a single picture about what sustainable vineyard management means and how to measure it. Based also on the results of previous studies, we can state that a combination of criteria encompassing both economic and environmental aspects of wine production can effectively address a comparative analysis of the sustainability of the two targeted management systems (conventional and organic practices) (Borsato et al., 2019b, 2018a). The proposed multi-criteria approach for the sustainability assessment is implemented along 5 years of time series (2014-2018). The novelty of this study lies in the robust analysis of the sustainability, while keeping at a minimum the amount of primary inventory data required to perform the assessment. The environmental indicators selected for this study consist in the Water Footprint (Borsato et al., 2019a; Lamastra et al., 2014), Carbon Footprint (FAO, 2014) and Vineyard Management Indicator (Lamastra et al., 2016) based on the VIVA calculator, a novel approach for Italian wine sustainability assessment, complemented by an economic balance to compare the two vineyard management systems.

6.3 Materials & Methods

The case study is located in a winery in the Nord-East of Italy (45°27’15’’ N; 11°13’10’’ E), which has South-West exposure to solar radiation and a slope of nearly 10%. In this study, two vineyards, 1.5 ha each, were assessed, one managed with organic practice while the second with conventional practice. The area is characterized by 900 mm of annual precipitation and a sub-continental temperate climate affected by the proximity of the Alps on the North and the Garda Lake on the West side. The soil texture is a silt loam soil with 20% sand, 5% clay and 5% of organic carbon content in soil. In order to implement this study, different agronomic operations were considered and included in the assessment, such as fertilization, plant protection treatments, irrigation, soil cultivation and canopy management.

Agronomic management

The field operations differed between organic and conventional management. The inventory dataset was collected for field operations and material used in vineyard during the time period 2014-2018. In fact, different frequency and amount of material inputs were applied during soil cultivation, canopy management, irrigation, plant protection and fertilization. Field operations differed for the number of treatments (Table 1); soil cultivations consisted in chopping grass and pruning residues, soil softening using a chisel plow positioned on the lines of wheeled tractors, and in an inter-row arrow with discs for weeds control. Canopy management consisted in cutting the top of the vine-grape crown (topping), and on a shoot pruning from the vie-grape trunk. The main difference in plant protection between organic and conventional management was that organic management mainly used copper and sulfur as fungicides, while conventional management uses different chemicals with different mechanisms of action. Characteristics of the field operations under organic and conventional management practices along the 5 assessed years are reported in Table 1.

Table 1. Field operations under organic and conventional management practices along the time series 2014-2018.

Vineyard	Operation	h/treatment	n° treatment/ha 2014	2015	2016	2017	2018
Organic	Leaf trimmer	2	2	1	1	2	1
	Chopping	2	3	2	1	2	1
	Soil loosening	2	1	0	0	0	0
	Fertilization	3	1	0	0	1	0
	Plant protection	2.5	12	9	12	14	10
	Irrigation	10	1	2	1	2	1
Conventional	Leaf trimmer	3	2	2	2	2	2
	Chopping	2	3	3	3	3	3
	Soil loosening	2	1	1	1	1	1
	Shoot pruning	3	2	1	2	1	1
	Inter-row arrowing	3	1	2	1	1	2
	Fertilization	3	1	1	1	1	1
	Plant protection	2.5	14	12	12	10	11
	Irrigation	10	3	2	3	2	2

Fertilization was applied using manure under organic vineyard plot for only 2 years and compost under conventional vineyard plot for 5 years (Table 2).

Table 2. Fertilization treatments under organic and conventional field management practices along the time series 2014-2018.

	Year	Fertilizer	Application rate kg/ha
Organic	2014	Manure	200
	2015		
	2016		
	2017	Manure	200
	2018		
Conventional	2014	Compost	500
	2015	Compost	500
	2016	Compost	500
	2017	Compost	500
	2018	Compost	500

Irrigation volume differed along the time series due to different irrigation schedule and decisions made according to the irrigation strategies and the vine-grape crop requirements. Most of the volume has been applied from flowering to veraison. Each irrigation treatment furnished 30 mm of water (Table 3).

Table 3. Irrigation treatments for organic and conventional vineyard management along the time series 2014-2018.

Vineyard Plot	Unit	2014	2015	2016	2017	2018
Organic	mm	30	60	30	60	30
Conventional	mm	90	60	90	60	60

Organic management on plant protection used mainly Copper and Sulfur as chemicals, while conventional plant protection needed to use several and more specific chemicals due to their modes of action and lower application rate (Table 4).

Table 4. Chemical products used during plant protection for organic and conventional vineyard management over the time period 2014-2018.

field management	chemical product	Average frequency #	Cumulative pesticide application kg/ha	Cumulative chemical application kg/ha
Organic	copper - sulphite	11	12.12	12.13
	sulfur	11	139	139
Conventional	cyflufenamid	1	1	0.0500
	cymoxanil	2	1.9	0.0827
	dimethomorph	1	0.5	0.0300
	fenamidone	3	3.6	0.1440
	folpet	6	7.1	5.6800
	fosetyl aluminum	5	6.5	2.7420
	iprovalicarb	3	3.6	0.1728
	mancozeb	15	12	9.1000
	metalaxyl	1	0.7	0.0252
	copper - hydroxide	2	2.3	0.3220
	copper - oxychloride	14	12.3	2.4994
	copper - sulphite	1	0.5	0.1200
	spiroxamine	1	1.2	0.6000
	tebuconazole	4	4	0.1800
	tebufenozide	2	1.3	0.2925
	sulfur	20	23.1	18.0800
	zoxamide	4	2.4	0.5232

The yield between different vineyard management registered different production quantities along the time series. Generally, conventional field management reached a greater average yield of 14.7 ton ha^{-1}, while organic field management had a lower average yield of 13.2 ton ha^{-1} (Figure 1).

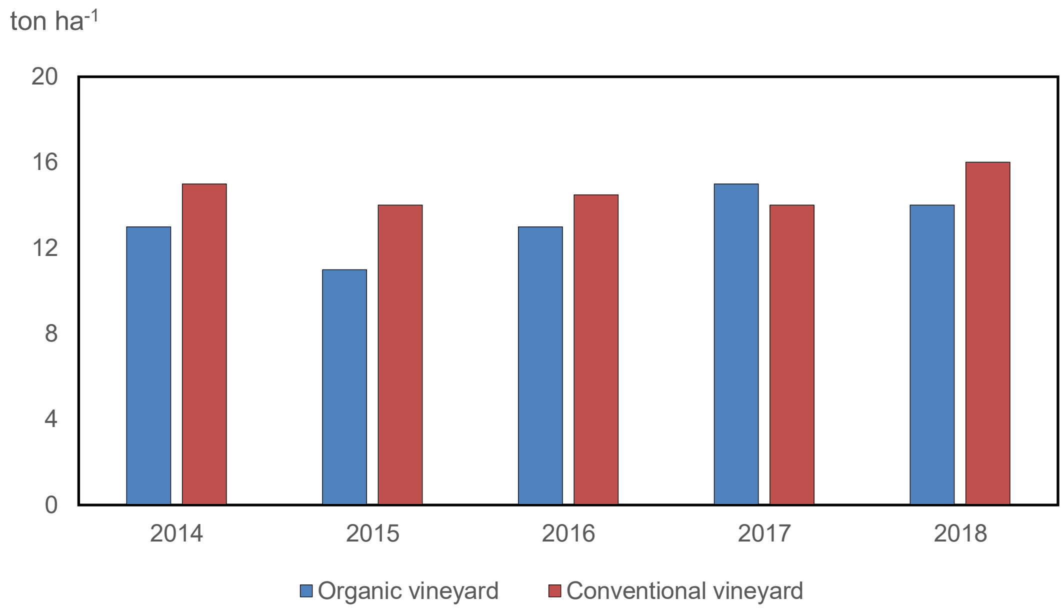

Figure 1. Yield of organic and conventional vineyard management practices along the time-series 2014-2018.

Organic vine grape production was generally lower; nevertheless, in 2017 it reached almost 7% higher production than the conventional production. The lower productive performance on organic vineyard management is due to the higher risk of contracting diseases that is one the main factors for yield loss. The lowest recorded yield under organic management with respect to conventional management was in 2015, where organic production was almost 20% less than conventional production (Figure 1). Generally, organic production had a lower yield from 10% to 13% less than conventional production over the remaining years.

Multiple indicator approach for sustainability assessment

We estimated three environmental indicators of Water Footprint, Carbon Footprint, and Vineyard Management Indicators as defined in the VIVA certification (Bonamente et al., 2017; Lamastra et al., 2016) using the VIVA calculator (VIVA, n.d.) as part of the analysis; the fourth indicator of "Landscape" ("Territorio" in Italian) was not considered thanks to its flimsiness on this analysis. The VIVA Sustainable Wine (Valutazione dell'Impatto in Vitivinicoltura sull'Ambiente) project, launched in 2011 by the Italian Ministry of Environment Land and Sea, developed a new framework to assess the sustainability of wine production based on four indicators, two of which, the "WATER" indicator (or the "ACQUA" indicator in Italian (Corbo et al., 2015; Lamastra et al., 2016; Merli et al., 2018)) and the "VIGNETO" indicator reflect the environmental performances of vineyard management, considering also the biodiversity and the quality of territory. These will be further described in the present paper. We, then, considered the economic aspect of sustainability as the marginal benefit within the economic balance. Those indicators provide useful insights assessing sustainability, where a comparison between organic and conventional was implemented for a first time; indicators were evaluated relating impact to a ton of grape production or a unit of surface (functional unit is 1 hectare). Finally, the two management were compared according to their performance and a statistical analysis was implemented using the t-test and Whitney-Mann tests for groups difference, and the Kolmogorov-Smirnov testing normality distribution (Borsato et al., 2018b). In the next sections, the description of the environmental indicators (Water Footprint, Carbon Footprint, and Vineyard Management) and of the economic indicator (marginal benefit) are presented.

6.3.1 Water Footprint Assessment

The water footprint indicator assesses the water consumption due to evapotranspiration from vinegrape and to the water used in the field for the irrigation, for the dilution of actives ingredients and for the volume of water used to clean all the equipment. Moreover, the calculation of water pollution is derived from the concept of 'critical load' using an indicator of the level of pollution of water expressed in terms of freshwater volume required to dilute the existing load of pollutants below the selected threshold value or eco-toxicological end-point (Borsato et al., 2018a; Lamastra et al., 2014). The total water footprint, it is the sum of three components such as green, blue and grey water footprint expressed as volume per unit of surface (m^3 ha^{-1}) or vine grape yield (m^3 ton^{-1}).

The green water footprint (Green WF) considers the volume of water from precipitation consumed by plants through evapotranspiration. It is affected by the soil moisture and its equation is reported in Eq.1, where Etc (mm/day) is the crop evapotranspiration which is estimated by multiplying the referenced evapotranspiration (Et_0 mm/day) derived from Penmann-Montheit equation proposed by FAO (Allen et al., 1998) and the crop coefficient Kc. When water depletion from root zone exceeds the readily available water for crop, a soil water stress coefficient Ks is introduced:

$$Et_c = Et_0 \cdot K_c \cdot K_s \left[{}^{mm}/_{day}\right] \qquad \text{Eq. 1}$$

The blue water footprint (Blue WF) includes all the water use sources for irrigation, for dilution during pesticides treatments, and for water consumed for cleaning the equipment. The water consumption is evaluated in both the assessed management practices during the growing period and considers only the direct water consumption. Irrigation water use is the predominant source of blue water use and impact on the resource.

The grey water footprint (grey WF) is the sub-indicator of freshwater pollution as a consequence of pesticides and fertilizer use. In the present study, the methodology used it is partially based on the indicator EPRIP 2 (Trevisan et al., 2009) adopted by the VIVA methodology: an estimation of the pollutant load that reaches the water body is determined for each compartment (surface water and groundwater) and accounted considering the three contamination mechanism such as drift, runoff and leaching. The main difference in comparison to EPRIP 2 is that no dilution of active ingredients concentration in the receiving water body is considered, according to the method proposed by Hoekstra et al., (2011). The grey water footprint is generally calculated with Eq.2 presented in the WFN manual (Hoekstra et al., 2011):

$$\text{Grey WP} = \frac{\alpha * \text{Appl}}{\text{C}_{\text{max}} - \text{C}_{\text{nat}}} \left[{}^{m^3}/_{functional\ unit}\right] \qquad \text{Eq. 2}$$

Where Grey WP is the grey water footprint (m^3 / functional unit), Appl is the applied chemical rate; C_{max} is the environmental water quality standard, while C_{nat} is the natural concentration in receiving water body (kg m^{-3}) usually assumed to be 0.The innovation of the method here presented concerns the way the leaching-runoff fraction α (constant), has been calculated: a Tier III approach is used to determine a more accurate prediction of the pollutant load (PL), through the following formulas (Eq. 3-6):

$$PL_{runoff} = RATE(1 - fint)(1 - fdrift)f1f2fw10^3 \quad \left[{}^{mg}/_{ha}\right] \qquad \text{Eq. 3}$$

$$PL_{drift} = DRIFT\ RATE 10^3 [mg/ha] \qquad \text{Eq. 4}$$

$$PL_{gw} = 2.739 \times AF \times RATE \times (1 - fint) \times (1 - fdrift) \times 10^3\ [\mu g/ha] \qquad \text{Eq. 5}$$

$$PL_{fertilizer} = 0.06\,RATE\,(N\%)10^6[mg/ha] \qquad \text{Eq. 6}$$

Detailed description of each parameters can be found in the publication of Trevisan et al., (2009).

Once calculated the pollutant load, to determinate the dilution volume, PL is divided by Cmax assumed to be equal to the threshold value in groundwater for pesticides and nitrates, whereas, for surface water contamination, the minimum value between no observed effect concentration (NOEC) of Daphnia, Algae and Fish was used. Daily values of GWP for each different pollutant are added over the year, and the critical contaminant is determined by the pesticide or fertilizer with the largest grey water footprint.

6.3.2 Carbon Footprint Assessment

The carbon footprint (CF) indicator evaluates the direct and indirect greenhouse gas emissions as unit of carbon dioxide equivalent emissions (kg CO_2-eq) (Bonamente et al., 2016; Rinaldi et al., 2016). In this study, the carbon footprint methodology is referred to the VIVA calculation by Bonamente et al., (2016) considering the processes of field management, and pesticides and fertilizers production as system boundaries. The metric (functional unit) is expressed as mass per unit of surface (kg CO_2-eq ha^{-1}) and mass per unit of vine grape yield (kg CO_2-eq ton^{-1}). Direct and indirect emissions from field operations were analyzed by including the emissions due to fuel consumption and the use of fertilizers and pesticides. The indicator quantifies the several GHG emissions (Eq. 7) multiplied by the Global Warming potential (GWP) with a time horizon of 100 years as proposed by FAO (FAO, 2014):

$$CF = \sum e_{i,j} \cdot GWP_j \qquad \text{Eq.7}$$

Where e_i is the GHG emission of the activity *i* (fertilizers, pesticides, field operation) for the GHG *j*, and CF is the total carbon footprint as the sum of the GHG emissions equivalent to CO_2. The GHG emissions e_i are calculated multiplying the emission factor by the amount of input (Eq. 8) involved in the activity:

$$e_{i,j} = EF_j \cdot CA_i \qquad \text{Eq. 8}$$

where, the Emission Factor (EF) depends on the input materials of the process and on their consumption (CA). The EF gives the value of GHG emission per unit of input used (e.g. Kg CH_4 per unit of MJ energy consumed)(IPCC, 2006). In table 5, EF and GWP used for CF calculation are shown (FAO, 2014; ISPRA, 2018).

Table 5. Global Warming potential (GWP) and Emission Factors (EF) factors used in the CF analysis (FAO, 2014).

GHG	GWP	Unit	EF electricity	Unit	EF Diesel consumption	Unit	EF Manure applied	Unit
CH4	21	kg CO_2eq/kgCH_4	1.2	kg/TJ	4.15	kg/TJ		
N2O	310	kg CO_2eq/kg N_2O	0.01	kg/TJ	28.6	kg/TJ	0.0143	kg N_2O-N/kg N
CO2	1	kg CO_2eq/kgCO_2	55900	kg/TJ	74100	kg/TJ		

6.3.3 *Vineyard Management Performance indicator*

The Vineyard Management Indicator is based on the ‘Vigneto’ indicator in VIVA calculator (Lamastra et al., 2016). The indicator consists in 6 sub-indicators for pest management, fertilization management, soil organic matter, soil compaction, soil erosion, and landscape quality. The indicator ‘Vigneto’ is based on a logic-based knowledge-based model. The model was elicited by experts in viticulture, who weighted the six sub-indicators according to their impact on the environment. Finally, the total score is calculated with a fuzzy logic approach that transforms output values in degree of belonging to the sustainability set by a membership degree. The different sustainability sets are aggregated by a hierarchical-structure based algorithm and a final judgment was calculated.
The Pesticides Management Indicator evaluates the probability of a predicted concentration of contaminants in the environment to overcome the supposed threshold or legal limit as the reference toxicity parameters for Daphnia, fish and algae in water, earthworms into soil, and the EC50 for rat inhalation in air. The Fertilization Management Indicator evaluates the impact of organic and inorganic fertilizers analyzing single components of nitrogen, potassium and phosphorous. The Soil Organic Matter Indicator evaluates the effects of soil management practices on soil organic matter. Vineyard soil management is expected to maintain the soil organic matter. This capacity depends on the soil texture, especially on the clay and loam presence. The Soil Compaction Indicator evaluates the degree of soil compaction due to passages of tractor or heavy machines, to the soil features and to the effect of rain intensity. The indicator analyses the susceptibility to soil compaction, which depends also to the soil cover, its percentage and length of presence during vine grape growing season. The Soil Erosion Indicator evaluates the magnitude of the soil erosion at local level due to a specific soil management and percentage of grass cover. The Landscape Quality Indicator evaluates the landscape structure and management. First, an analysis on the complexity of the landscape is done by looking at the presence of natural areas in the farm. Second, the indicator considers the time spent by the farm workers for landscape maintenance.

6.3.4 *Economic Balance*

The economic balance analyses the marginal benefit as the difference between the gross marketable income and the costs, resulting on the net income for both the organic and conventional vineyard management (Table 6). The gross marketable income was calculated by multiplying the grape price by the annual yield. The market price for grape was fixed at 0.95 € kg^{-1} for organic production and at 0.7 € kg^{-1} for conventional, as established by the trade contract with the winery that awards the grape from an eco-friendlier production. The yield is the main parameter affecting the gross marketable income. Costs of production are mainly based on operation costs, costs of materials for vineyard management, cost for maintenance of the rural ecosystem (hedges maintenance, cutting grass, pruning), and amortization costs of machinery used for vineyard operations and wages (table 6). The cost for the different vineyard operations described in Table 1 consists in costs for the energy consumption for tractor, oil fuel, the maintenance of the natural areas on the perimeter (border) of the vineyard and the maintenance of machinery used during field operation. The cost of materials considers the costs for purchasing fertilizers and pesticides, and other tools used during vine grape growing season. Amortization costs is the annual costs for replacing machineries due to their depreciation and are fixed costs that do not differ between management practices. The market value of each machine was linearly divided by

10 years (common amortization period for agricultural machineries) to calculate the amortization cost. The analysis of costs includes salaries and wages. The main difference is that wages are expenditures for field workers who are paid hourly 15 €, while salaries are expenditures for professional counselling paid 30 € per hour.

Table 6. Cost analysis for the two vineyard management practices (organic and conventional) along the time-series 2014-18 (€ ha^{-1}).

Management	€ ha^{-1}	2014	2015	2016	2017	2018	Average
Organic	Wages	675	427.5	510	690	435	548
	Salaries	60	120	120	120	60	96
	Cost for purchasing fertilizers	80	0	0	80	0	32
	Cost for purchasing pesticides	190	147	143	195	119	159
	Operation Costs	831	548	638	842	554	683
	Amortization	311	311	311	311	311	311
	Total Costs	2146	1553	1721	2238	1479	1827
	Gross marketable Income	12350	10450	12350	14250	13300	12540
Conventional	Wages	915	840	840	720	803	824
	Salaries	135	135	180	135	90	135
	Cost for purchasing fertilizers	80	80	80	80	80	80
	Cost for purchasing pesticides	136	134	115	114	124	125
	Operation Costs	1157	1073	1110	941	993	1055
	Amortization	418	418	418	418	418	418
	Total Costs	2841	2680	2743	2408	2508	2636
	Gross marketable Income	10500	9800	10150	9800	11200	10290

Net income (Eq. 9) was calculated for each vineyard management practice as the difference between the gross marketable income and total costs (Falcone et al., 2016; Naglova and Vlasicova, 2016):

$$Net\ Income = Gross\ marketable\ Income - Total\ costs \qquad \text{Eq. 9}$$

The net income is reported as the marginal benefit in terms of € ha^{-1} or € ton^{-1}.

6.4 Results and Discussion

6.4.1 Water Footprint results

The water footprint was computed for the two quantitative indicators such as green and blue water footprint as well as for the qualitative indicator of the grey water footprint. Figure 2 illustrates the average water footprint calculated with respect to the 5 years and investigating the relative contribution of the different sub-indicators for both the organic and conventional production.

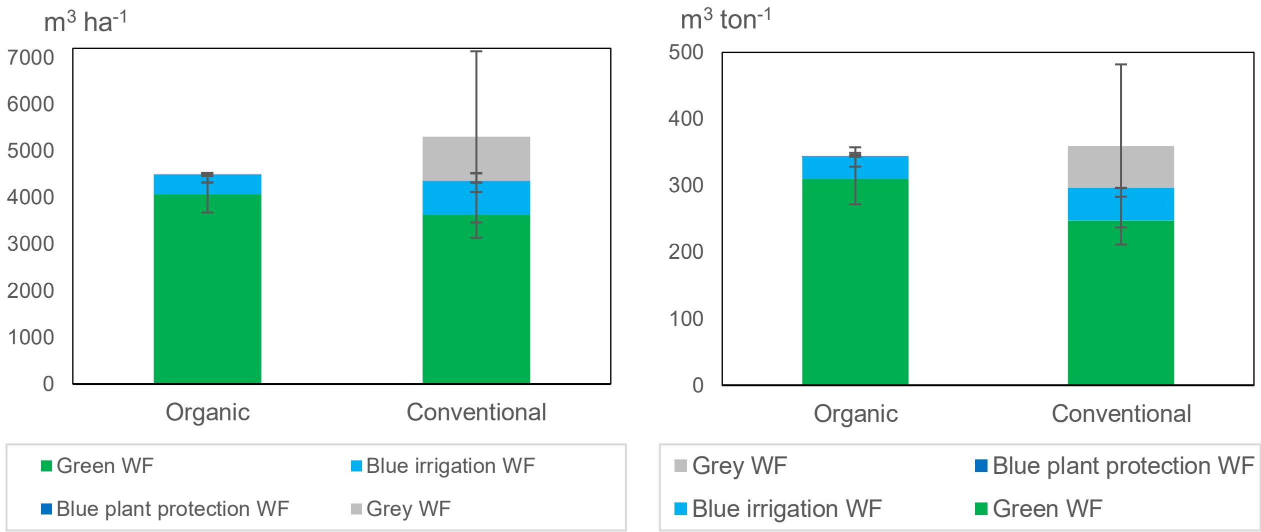

Figure 2. Annual average Water Footprint and related sub-indicators under organic and conventional vineyard management practices in terms of m^3 ha^{-1} (on the left) and m^3 ton^{-1}(on the right).

The results showed that total water footprint was greater under the conventional vineyard management due to a higher contribution of blue and grey water footprint (720 m^3 ha^{-1} and 947 m^3 ha^{-1}, respectively). In both scenarios, green water footprint accounted for the largest share, approximately 68% for conventional management and 90% for the organic one, followed by blue water footprint with a contribution of 13% and 8% accordingly. Finally, the grey water footprint was greater in the conventional one. In particular, the green water footprint was generally higher for organic vineyard management thanks to different agronomic and microclimate features (sunlight exposure, different canopy; higher evaporation). There are not substantial mitigation strategies applicable to reduce green water footprint, unless keeping the ground moist mulching the soil or increasing the amount of organic matter in the soil.

Moreover, the blue water footprint was lower under organic vineyard management, indeed the average blue water footprint due to irrigation water was 420 m^3 ha^{-1} for the organic instead of 720 m^3 ha^{-1} of water consumed for conventional vineyard management.

A similar trend is described when looking at the annual average results in terms of m^3 ton^{-1}. The green water footprint was higher under organic management with 311 m^3 ton^{-1} with respect to the 247 m^3 ton^{-1} of water consumed for the conventional vineyard management. As mentioned above, the blue water footprint was lower under organic rather than under conventional management with 33 m^3 ton^{-1} and 50 m^3 ton^{-1}. The grey water footprint, as expected, was much higher under conventional management with 947 m^3 ha^{-1} comparing to a value of 17 m^3 ha^{-1} under organic vineyard management, corresponding to 63 m^3 ton^{-1} and 1.3 m^3 ton^{-1}, respectively.

A statistical analysis to compare sub-indicators among different vineyard managements was done both in terms of m^3 ha^{-1} and m^3 ton^{-1} (Table 7).

Table 7. Statistical analysis of the significant difference between the WF scores under the different vineyard management. Table performed the Kolmogorov-Smirnov test (normality), the t-test and the Mann-Whitney test (significance).

	Kolmogorov-Smirnov organic		conventional		t-test Organic vs Conventional		Mann-Whitney	
$m^3 ha^{-1}$	K-S test statistic	K-S critical value	K-S test statistic	K-S critical value	t	p	U-value	p<0.5
green WF	0.225	0.912	0.248	0.850	0.813	0.462	6	4
Blue irrigation WF	0.393	0.328	0.393	0.328	1.361	0.245	3	4
Blue plant protection WF	0.253	0.837	0.253	0.837	0.011	0.992	11	4
Grey WF	0.373	0.388	0.468	0.161	1.131	0.321	0	4
$m^3 ton^{-1}$	K-S test statistic	K-S critical value	K-S test statistic	K-S critical value	t	p	U-value	p<0.5
green WF	0.241	0.871	0.196	0.969	1.229	0.287	3	4
Blue WF irrigation	0.361	0.429	0.336	0.523	0.874	0.431	4	4
Blue WF plant protection	0.238	0.879	0.239	0.877	0.554	0.609	6	4
Grey WF	0.357	0.444	0.471	0.155	1.130	0.322	0	4

The *p* value shows that water footprint values are not statistically different between organic and conventional management beside the fact there is a slightly difference in terms of m^3 ha^{-1} or in terms of m^3 ton^{-1}.

6.4.2 Carbon Footprint results

GHG emissions were calculated for pesticides' and fertilizers' applications, as well as for fuel consumption according to the two different field management practices. Figure 3 shows a greater GHG emissions for conventional vineyard management (about 2534 kg CO_2eq ha^{-1}) than the 1827 kg CO_2eq ha^{-1} of organic vineyard management. Conventional management had a higher contribution to air pollution mainly due to higher CF on the fertilizers and fuel consumption, with 904 kg CO_2eq ha^{-1} and 1526 kg CO_2eq ha^{-1}, respectively. On the other hand, the contribution of pesticides application was lower under conventional management with 105 kg CO_2eq ha^{-1}. The same proportion is described in terms of kg CO_2eq ton^{-1}, where the medium total score value of 173 CO_2eq ton^{-1} was found under conventional vineyard management and 138 CO_2eq ton^{-1} under organic vineyard management. The GHG reduction is around 75 % for fertilizers application from conventional to organic both in terms of CO_2eq ha^{-1} and CO_2eq ton^{-1}. GHG reduction for pesticides application from organic to conventional is around 66% in term of CO_2eq ha^{-1} and 69% in terms of CO_2eq ton^{-1}. Finally, organic management reduces GHG emission by 15% in terms CO_2eq ha^{-1} and 6% in terms of CO_2eq $tons^{-1}$ comparing to conventional vineyard management.

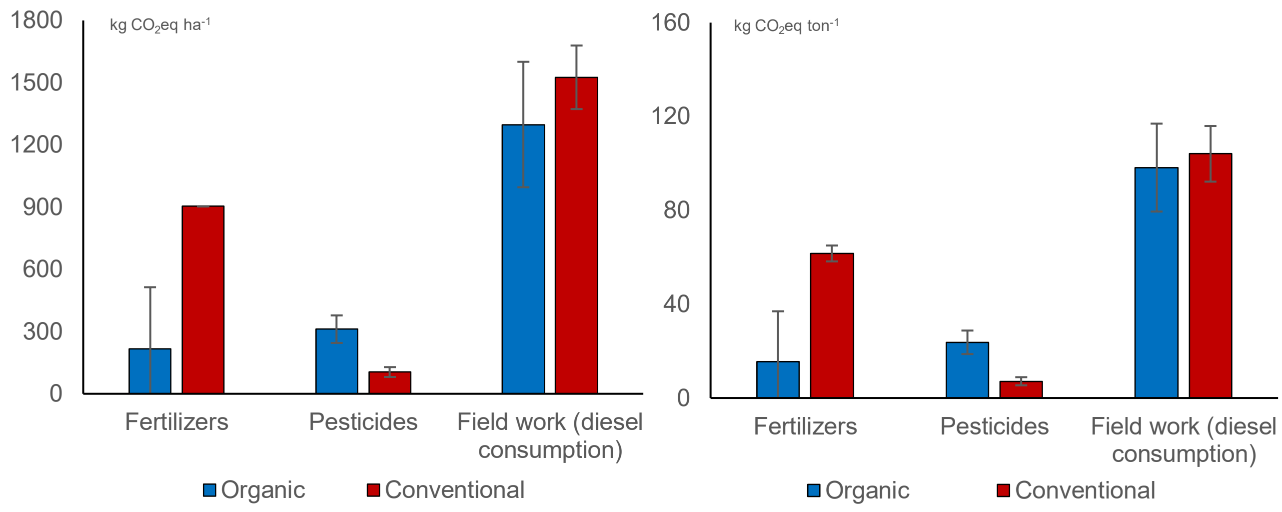

Figure 3. Annual average Carbon Footprint of principal variables involved in GHG emissions under organic and conventional vineyard management in terms of m^3 ha^{-1} (chart on the left) and m^3 ton^{-1} (chart on the right).

Table 8. Statistical analysis of the significant difference between the CF scores under the two different vineyard management. Table performed the Kolmogorov-Smirnov test (normality), the t-test and the Mann-Whitney test (significance).

	Kolmogorov-Smirnov				t-test		Mann-Whitney	
	organic		conventional		Organic vs Conventional			
kg CO_2eq ha^{-1}	K-S test statistic	K-S critical value	K-S test statistic	K-S critical value	t	p	U-value	p<0.5
Fertilizers	0.393	0.328	-	-	1.828	0.142	0	4
Pesticides	0.276	0.756	0.351	0.466	2.435	0.072	0	4
Field work (diesel consumption)	0.268	0.784	0.244	0.862	0.586	0.589	8	4
kg CO_2eq ton^{-1}	K-S test statistic	K-S critical value	K-S test statistic	K-S critical value	t	p	U-value	p<0.5
Fertilizers	0.392	0.331	0.232	0.895	1.698	0.165	0	4
Pesticides	0.241	0.871	0.317	0.597	2.595	0.060	0	4
Field work (diesel consumption)	0.222	0.921	0.242	0.870	0.242	0.821	10	4

In Table 8, a statistical analysis compares the differences of CF performances between organic and conventional vineyard management. There is a high statistical difference between organic and conventional practice for fertilizers and pesticides application. The *p* value shows that field work GHG emissions are not statistically different between organic and conventional management, beside the different number of field operation along the growing season, which actually makes the difference on fuel consumption.

6.4.3 Vineyard Management Indicator results

The indicator of Vineyard Management consists in the integration of multiple sub-indicators. In table 9, different sub-indicators are expressed with a value from 0 to 1, where 0 is the target score to reach according to VIVA methodology. As described in the table 9, the main result from this indicator is that there was not a significant difference between the two management practices. Moreover, the Pest Management Indicator was null because pesticides applied in the field had a low risk of toxicity for aquatic ecosystem and they were applied at a low rate for both management practices. Fertilizers Management Indicator was null in conventional and 0.09 (still low) in organic vineyard management because the manure was applied only two years in organic, while compost was applied over the time series for conventional vineyard management. This fact also affected the Soil Organic Matter Indicator, which was lower, and therefore better, under organic vineyard management that benefited also from a larger surface of grass cover, 100% with respect to 70% for conventional vineyard management. The Soil Compaction Indicator had an unfavourable score due to the soil sensitivity to compaction resulted from soil features and texture. In this case, the narrowest differences between the two vineyard managements on Soil Compaction Indicator was the consequence of different field operations and frequency. Furthermore, conventional management considered extra field operations such as the shoot pruning and the inter-row arrowing in comparison with organic management which did not. Another important sub-indicator is the Soil Erosion Indicator, which considered the surface covered along the year, the soil tillage practices and the presence of grass. The Soil Erosion Indicator was higher under conventional vineyard management because the percentage of soil coverage was lower than in organic management, and a further operation field of inter-row arrowing was done along the year. The last indicator is the Landscape Quality Indicator, which was equal for both the field management practices because, in this case, farmer managed and took care of natural areas in the same way. Finally, the integration of the above described sub-indicators showed that organic vineyard management gained a lower total score for Vineyard Management Indicator, 11% less than conventional vineyard management (Table 9).

Table 9. Vineyard Management Indicators scores for Organic and Conventional vineyard management along the time series 2014-2018.

	year	Pest Management	Fertilizers Management	Soil Organic Matter	Soil Compaction	Soil Erosion	Landscape Quality	Final score
Conventional vineyard management	2014	0	0	0.33	0.63	0.18	0.5	0.29
	2015	0	0	0.33	0.61	0.18	0.5	0.28
	2016	0	0	0.33	0.62	0.18	0.5	0.28
	2017	0	0	0.33	0.61	0.18	0.5	0.28
	2018	0	0	0.33	0.61	0.18	0.5	0.28
Organic vineyard management	2014	0	0.09	0.18	0.61	0	0.5	0.25
	2015	0	0.09	0.2	0.6	0	0.5	0.25
	2016	0	0.09	0.2	0.6	0	0.5	0.25
	2017	0	0.09	0.18	0.61	0	0.5	0.25
	2018	0	0.09	0.2	0.6	0	0.5	0.25

6.4.4 Results of the economic analysis

The analysis of the economic balance focused on the operation field practices applied. The Net Income has been calculated as difference between the Gross Income and Total Costs. In Figure 4, the costs are listed and compared between the two vineyard management practices both in terms of € ha^{-1} and € ton^{-1}. The major costs during field management were costs for wages and for other field operations during field treatments. Conventional vineyard management generally had higher costs; especially 33% and 27% higher wages cost than organic field management in terms of € ha^{-1} and € ton^{-1}, respectively, and 35% and 28% higher field operation costs than organic management in terms of € ha^{-1} and € ton^{-1}, respectively. Organic vineyard management had a higher cost for pesticides purchase, almost 21% and 33% more than conventional in terms of € ha^{-1} and € ton^{-1}, respectively.

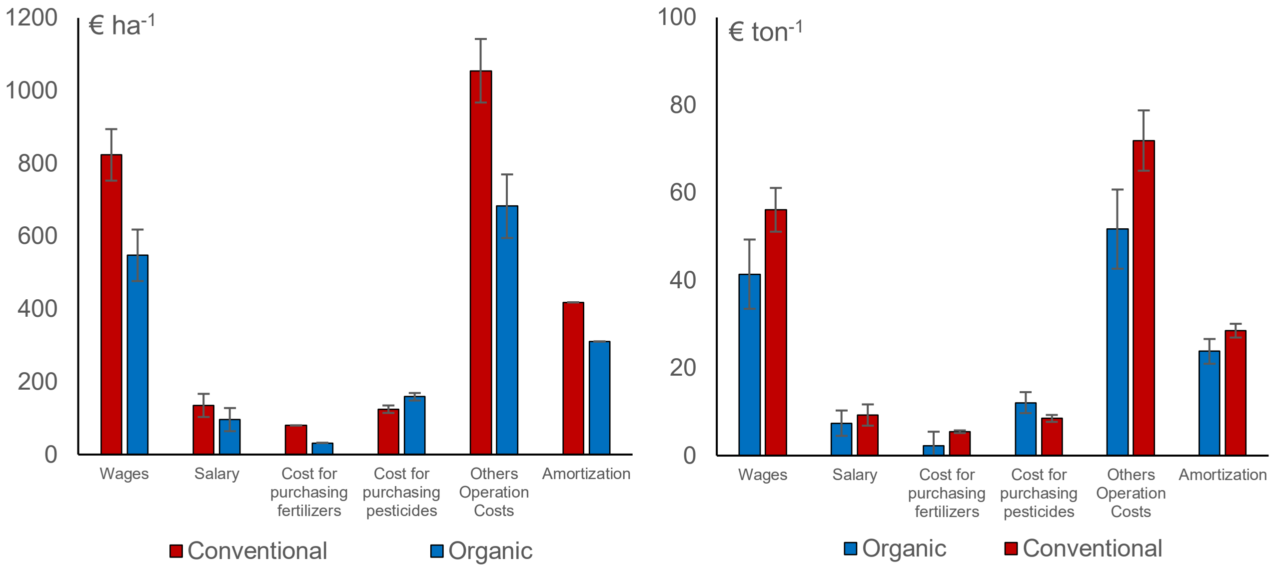

Figure 4. Cost comparison between conventional and organic vineyard management in terms of m^3 ha^{-1} (on the left) and m^3 ton^{-1} (on the right).

The graphs reported in Figure 5 describe the Total Costs, Gross Marketable Income, and Net Income from conventional and organic vineyard management expressed as m^3 ha^{-1} and m^3 ton^{-1} (marginal benefit). Conventional field had greater Total Costs for vineyard management of about 31%, when expressed as m^3 ha^{-1} and 23%, as m^3 ton^{-1}, respectively, while organic field had greater Gross and Net Income of about 18% and 29%, when expressed as m^3 ha^{-1}, than conventional, and about 26% and 36% than conventional, respectively, when expressed as m^3 ton^{-1}.

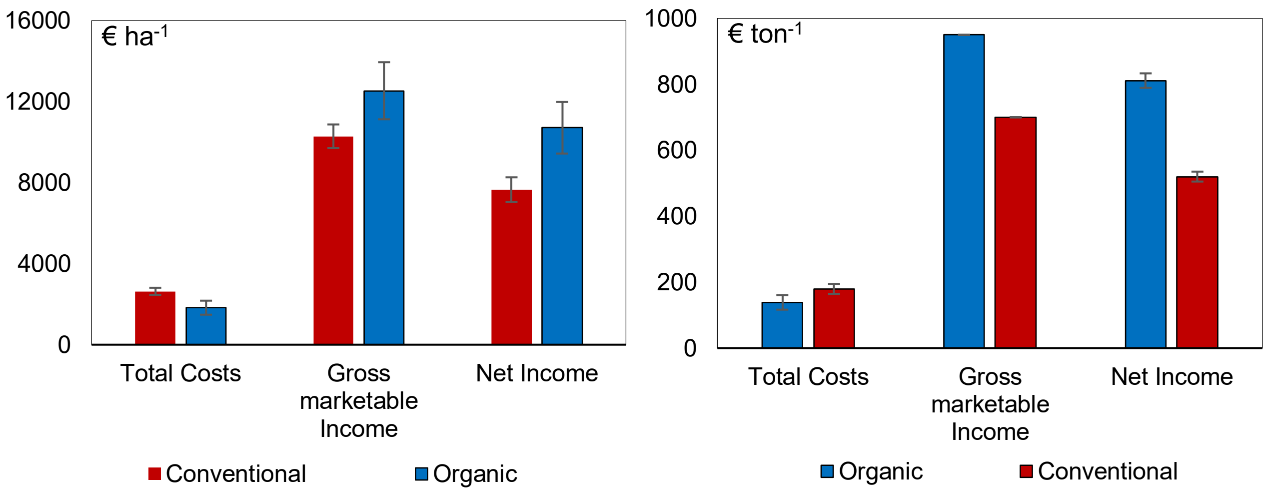

Figure 5. Comparison of the marginal benefit between conventional and organic vineyard management in terms of m^3 ha^{-1} (chart on the left side) and m^3 ton^{-1} (chart on the right side).

The results of statistical analysis are shown in Table 10. Generally, there was a strong statistical difference between the two management practices. Only the salaries and the costs for purchasing fertilizers did not seem statistically different between the two options, while in the others case there was a strong statistical difference.

Table 10. Statistical analysis of the significant difference for the economic parameters for the two vineyard management practices. Table performed the Kolmogorov-Smirnov test (normality), the t-test and the Mann-Whitney test (significance).

	Kolmogorov-Smirnov				t-test		Mann-Whitney	
	organic		conventional		Organic vs Conventional			
€ ha^{-1}	K-S test statistic	K-S critical value	K-S test statistic	K-S critical value	t	p	U-value	p<0.5
Wages	0.281	0.638	0.376	0.288	1.679	0.168	0	4
Salary	0.393	0.328	0.300	0.664	0.888	0.425	3	4
Cost for purchasing fertilizers	0.393	0.328	-	-	0.866	0.435	5	4
Cost for purchasing pesticides	0.259	0.816	0.253	0.834	0.820	0.458	3	4
Others Operation Costs	0.273	0.765	0.191	0.976	1.985	0.118	0	4
Amortization	-	-	-	-	3.78E+15	2.93E-62	0	4
Total Costs	0.249	0.849	0.209	0.946	1.818	0.143	0	4
Gross marketable Income	0.240	0.874	0.225	0.912	1.249	0.280	2	4
Net Income	0.235	0.888	0.296	0.678	1.873	0.134	0	4
	0.281	0.638	0.376	0.288	1.679	0.168	0	4
€ ton^{-1}	K-S test statistic	K-S critical value	K-S test statistic	K-S critical value	t	p	U-value	p<0.5
Wages	0.222	0.921	0.259	0.816	1.434	0.225	2	4
Salary	0.260	0.813	0.251	0.840	0.486	0.652	7	4
Cost for purchasing fertilizers	0.392	0.331	0.232	0.895	0.790	0.474	6.5	4
Cost for purchasing pesticides	0.268	0.786	0.279	0.744	1.180	0.303	2	4
Others Operation Costs	0.192	0.975	0.375	0.384	1.708	0.163	1	4
Amortization	0.283	0.731	0.232	0.895	1.300	0.264	2	4
Total Costs	0.175	0.991	0.358	0.440	1.429	0.226	1	4
Gross marketable Income	-	-	-	-	2.97E+15	7.76E-62	0	4
Net Income	0.175	0.991	0.358	0.440	10.133	0.001	0	4

6.5 Discussion

This study compared the environmental and economic performances of organic and conventional vineyard management systems through the application of a multi criteria approach on a case study in northern Italy. Comparing the obtained results with available literature on this topic, we can notice that Kavargiris et al. (2009) also mentioned that organic vineyard management implies less energy consumption and therefore less GHG emissions. Michos et al., (2018) concluded that intensive farming leads to higher environmental impacts at the expenses of sensitive ecosystems. Wheeler and Marning (2019) recorded an increased conversion from conventional to organic vineyard during the Millennium Drought in Australia because organic management was perceived as characterized by a higher water security and smaller water vulnerability due to higher soil water retention. Furthermore, many authors supported the thesis that organic management increases soil carbon stock and soil biodiversity (Brunori et al., 2016; Niccolucci et al., 2008). On the other hand, only few studies concluded that organic vineyard management had a better economic benefit, while most part of the research community considers that, finally, conventional management gains a higher net income and marginal benefit due to the lower cost of maintenance (Falcone et al., 2016; Wheeler and Crisp, 2010). The results of this study showed that organic management practices decrease the Water Footprint and Carbon Footprint thanks to the lower amount of fertilizers applied and field passages. The Vineyard Management indicator obtained a better performance under organic management, as well as the marginal economic benefit, that is greater thanks to the higher marketable price and lower management costs. The results are summarized in the table below (Table 11).

Table 11. Summary of the results of environmental and economic indicators assessment for both organic and conventional vineyard management.

Indicator	Sub-Indicator	Conventional	Organic	Conventional	Organic
		U.M. ha^{-1}	U.M. ha^{-1}	U.M. ton^{-1}	U.M. ton^{-1}
Water Footprint (m^3 U.M. $^{-1}$)	Green Water Footprint	3627	4067	247	311
	Blue Water Footprint	726	426	49	22
	Grey Water Footprint	947	17	63	1.3
Carbon Footprint (CO_2eq U.M.$^{-1}$)	Fertilizers	904	217	62	16
	Pesticides	105	312	7	24
	Field works	1526	1298	104	98
Vineyard Management Indicator (from 0 to 1)	Pest Management	0	0		
	Fertilizers Management	0	0.9		
	Soil Organic Matter	0.33	0.19		
	Soil Compaction	0.62	0.60		
	Soil Erosion	0.18	0		
	Landscape Quality	0.5	0.5		
Economic Balance (€ U.M.$^{-1}$)	Total Costs	2636	1827	180	139
	Gross marketable Income	10290	12540	700	950
	Net Income	7654	10713	520	811

The reason why the blue WF was low under organic management lies to some assumption made for vine grape quality. In this sense, one more irrigation during growing season could enhance organoleptic composition which depends mostly to soil characteristics that are leached and diluted by irrigation water applied. In fact, the soil features affect vinification, wine aromas and taste (Antonio et al., 2014). Finally, a lower number of irrigations were applied for organic management. The way round, in conventional vineyard management a different agronomic strategy was implemented because irrigation gives different organoleptic features to vine grape, as for example a higher acidity and better “bouquet” of polyphenols (Cavaliere et al., 2010). In this way, the different irrigation practice along the vine grape ripening season led to a higher blue water footprint for conventional vineyard management to reach specific organoleptic requirements. Grey WF was mostly affected by pesticides leaching into groundwater, which turned out to be the predominant environmental mechanism responsible of the grey water footprint in the conventional vineyard, while leaching of nitrates due to the use of compost is the main cause of pollution under the organic management. The approach taken in grey water footprint accounting is the same as the so-called critical-load approach. A grey water footprint larger than zero does not automatically imply that ambient water quality standards are violated: a further investigation would be needed to determine if the available water flow is fully required to dilute the active ingredients down to acceptable concentration, or vice versa, if the grey water footprint just shows that only a part of the assimilation capacity has been consumed. The number and type of field intervention and operations affected the final score of Carbon Footprint and Vineyard Management, especially fertilizers and field work operations are dominant factors on Carbon Footprint for conventional management, while the variability of field work operations under organic management affect in particular the Vineyard Management Indicator. Different factors on the economic balance affect the final total costs. Moreover, the net revenue is considerably dependent according to the grape market price.

6.6 Conclusions

In conclusion, the choice to an organic vineyard management does not compromise the economic productivity of vine grape production, and moreover it improves mitigations on the environmental impacts. According to the results of the case study presented in this work, the organic vineyard management is a suitable model for viticulture that farmers can follow to reach their economic objectives while preserving the environment from water depletion, or GHG emissions, or soil compaction enhancing biodiversity. In general, we noticed in literature a lack of comprehensive studies that considers multiple criteria and indicators to evaluate the overall sustainability of conventional and organic vineyard management practices. Therefore, the present work can help filling this gap because it explored the possibility to combine environmental indicators of impacts on diverse environmental compartments (air, water and soil) with the assessment of the economic aspect (marginal economic benefit). Future improvements might consider the assessment of social sustainability so that all the three pillars of sustainability assessment (environment, economy and society) according to the classical Triple Bottom Line approach (Elkington, 1999) would be considered. For instance, in the assessment of social sustainability the difference in risk perception by workers or bystanders between a vineyard managed under an organic or conventional system need also to be considered and performed. Moreover, the integration of environmental, economic and social indicators into a comprehensive sustainability index could be pursued through the application of Multi-Criteria Decision Analysis (MCDA) methodologies (Linkov and Moberg, 2012), since these approaches already proved to be useful and effective in the implementation of sustainability assessment in other sectors.

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7. Environmental and Economic Sustainability Assessment for Two Different Sprinkler and a Drip Irrigation Systems: A Case Study on Maize Cropping [5]

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7.1 Abstract

Water scarcity is worsened by climate change. Water savings can be reached by improving irrigation efficiency both on farm and on water supply. To do that, the choice of the best irrigation technology is not always straightforward, because farmers need to renew and implement farm infrastructures for irrigation. This study compares three irrigation systems, one drip irrigation and two sprinkler (center pivot and hose-reel) systems, on environmental, economic, and energetic performance under irrigated and non-irrigated maize cropping. The study combines impact and efficiency indicators, addressing a sustainability analysis for the irrigation practice under the three different irrigation systems. The sustainability for the irrigation systems was assessed using water-related indicators (water use efficiency, irrigation water use efficiency, and water footprint), biomass (crop growth rate, relative growth rate, harvest index, and yield response factor), and energy indicators (energy footprint, performance, and energy cost footprint) for the environmental aspect; and the economic-based indicators (water productivity and economic water footprint) for the economic aspect. Main results address the center pivot system as the best solution for irrigation practice since it demonstrated higher economic and environmental performance. Moreover, maize under the pivot system allowed a higher biomass production, economic benefits, and water use efficiency.

Keywords: efficiency, footprint, indicators, hose-reel, center pivot, water consumption

7.2 Introduction

Agriculture sector uses the 70% of the global freshwater resource (Rockström et al., 2017). Although only 20% of global croplands is equipped for irrigation, 40% of global food production is attributable to irrigated lands (Chartzoulakis and Bertaki, 2015). Actually, 24% of world river basin area suffers from a severe water scarcity (Alcamo et al., 2003), and irrigation water use is in conflict with the ecological flows (European Commission, 2016) in 15 % of the global lands (Smakhtin et al., 2004). The economic welfare of a country is also linked with the water depletion; in fact, the water poverty increases following an increase of ecosystem degradation (Sullivan, 2002). The use of irrigation water enhances a higher crop production, which is usually correlated to higher economic benefits (Sullivan, 2002). Hence, sustainable irrigation practice is a strategic enhancements to rise food production and enabling the Earth system to operate within planetary boundaries (Rockström et al., 2017; Rosa et al., 2018). A common opinion for increasing sustainability is given from the implementation of water saving irrigation technologies (D'Odorico et al., 2018). Although an increase of irrigation efficiency at farm scale reduces water consumption at farm scale, it fails to increase water availability at watershed and

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basin scale (Unver et al., 2017), leading to the irrigation paradox exactly because to an increase of local beneficial water consumption due to a more efficient irrigation system. Moreover, previous non-consumed water losses at farm scale are recovered and reused at basin scale, and even more, the increase of irrigation efficiency at farm scale might increase also the water consumption once the farm when switching to a more water-intense crops (Grafton et al., 2018). Therefore, sustainable implications of water use refers to integrate environmental preconditions while satisfied the societal demand (Falkenmark, 1997). In other words, a sort of "semi-sustainability" criterion need to combine the best possible approach taken by human needs against the resulting environmental depletion (Falkenmark, 1997). In addition, the human interaction with the water cycle makes some unavoidable changes on water resource (Falkenmark, 1997; Tuninetti et al., 2019). For that reason, decision-makers must make better decisions using appropriate tools to support policies and investments (Borsato et al., 2018a; Mejía et al., 2012). Thus, the sustainability of irrigation practice could be improved by the governance with appropriate water policies (Gómez-Limón and Riesgo, 2009). Water policies enhance, for example, the use of water price and subsides to improve sustainability use of water resource (Bubb et al., 2016; Pellegrini et al., 2019). As defined for the 6.4 Sustainable Development Goals target, water use indicators should analyse the inter-annual and intra-annual variability of water stress (Vanham et al., 2018), and analyse changes on water management integrating the several aspects of sustainability (Chaves and Alipaz, 2007; Galli et al., 2012). In fact, a core of indicators, better analyse the overall sustainability (Čuček et al., 2012; Rennings and Wiggering, 1997). Especially, the relation among the biomass production, the economic productivity related to irrigation water, the energetic costs and the emissions on the environment permit to fully understand the sustainability of water use (Galindo et al., 2018; Herva et al., 2011), and was used in this case study to compare the different irrigation practices.

This study determines the agro-environmental and economic sustainability for supposed suitable irrigation systems, comparing two sprinkler irrigation (center pivot and hose-reel) and one drip irrigation systems under maize cropping. This study focuses on different irrigation managements comparing irrigation systems and a non-irrigated test using several water-based indicators, analysing the production of the biomass and yield, and looking into their economic and energetic performance.

7.3 Material and Methods

7.3.1 Study area and Field management

This study compares the management of three-irrigation systems during the maize cropping season. The study area was located in the high plain on the Northeast of Italy (45°49'44.6"N; 12°16'35.6"E) (Figure 1) in proximity to rivers, the mountain area and the coast. The area is characterized by a sub-continental climate with 1100 mm of annual rainfall, while during summer season, the area is characterized by warm temperatures and a high humidity.

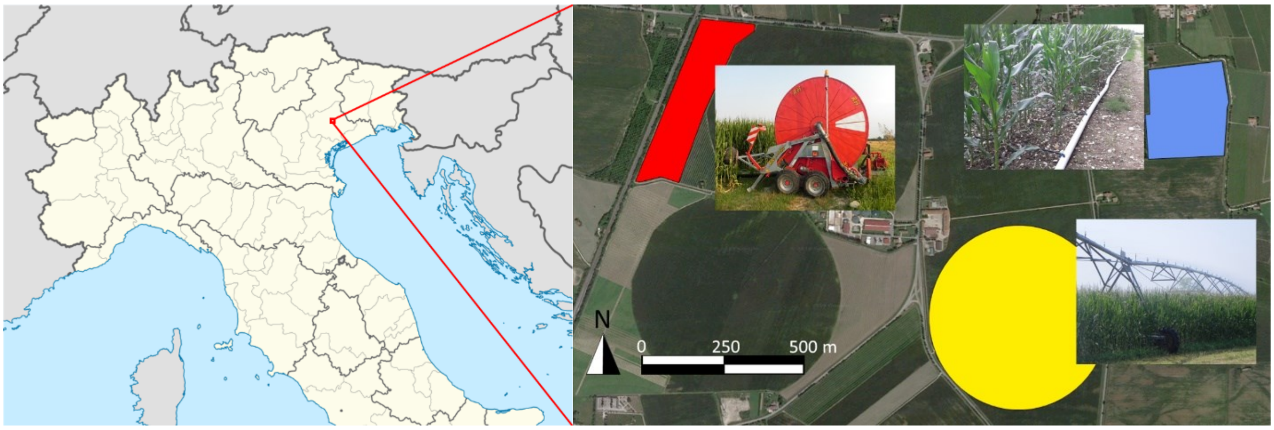

Figure 1. Illustration of the study area on the northeast of Italy (left), and the zoomed image with the three fields with the three irrigation systems identified with different colors (right): center pivot in yellow, micro irrigation in blue, and the hose-reel sprinkler machine in red.

Soil texture is silty-clay-loam textured (sand 12%, loam 61%, clay 27%). The area is characterized by the presence of 9% of gravel**.** The soil is sub-alkaline (pH 7.4) with a 2.2% organic matter and 1.3% of organic carbon. Soil nitrogen is about 1.37 grams kg^{-1}; usable phosphorous (P_2O_5) is 55 ppm and potassium is 275 ppm of K_2O.

Agronomical management has foreseen a 35 cm deep ploughing in autumn, after a fertilization with 50 t ha^{-1} of manure from biogas digestion. Prior to the seedbed preparation, a fertilization with 250 kg ha^{-1} of potassium chloride (KCl) and 200 kg ha^{-1} of calcium perphosphate $Ca_3(PO_4)_2$ (P 46%) was applied in the fields. Maize was sown on the 20th march in the Pivot plot and on the 1st of April in the other plots at 7-8 plants m^{-2} density. A second fertilization with 500 kg ha^{-1} of urea N 46% (230 kg ha^{-1} of nitrogen) was applied during maize growing season in both fields under sprinkler irrigation. In the microirrigation plot, on the other hand, a fertigation with urea was done in five times during the early growing season. Maize was harvested on the 1 of September and maize yield was recorded to 11.5 tons ha^{-1} under the drip irrigation system, and to 10.3 and 9.7 tons ha^{-1} under center pivot and hose-reel systems, respectively, while the non-irrigated test performed 4.6 tons ha^{-1}.

7.3.2 Irrigation

The irrigation timing was scheduled according to the Irrigation Board calendar compensating the evapotranspiration before water stress occurred. Irrigation season lasted from half May to half August. Rainfall and temperature data were recorded by a local weather station located in proximity of the farm. The non-irrigated test was located on each plot and kept willingly without water application for a total surface of 0.5 ha. The three irrigation systems included two sprinklers and a microirrigation system (Figure 1). The sprinkler irrigation systems were a center pivot system and a hose-reel sprinkler. The microirrigation system was a drip line system located in an area of 10 ha. Water was firstly supplied from an irrigation channel, then pumped at a pressure of 1.5 bar by an electro pump of 37 kW and filtered by three automatic 120 mesh filters. The main line combined a 101.6 mm line that lay flat and a drip line with 22 mm of diameter. The drip line was characterized by drippers with a nominal flow rate 1.05 L h^{-1} at 0.8 bar pressure (water applied was 1.4 mm h^{-1}) and spaced 0.5 m each other. The drip line was positioned 1.5 m apart or every two planting rows. The irrigation season counted 12 irrigations for a total of 232 mm of water volume.

The center pivot system was settled on an area of 40 ha. The pivot was 325 m long with a terminal section of 25 meters, with the sprinklers positioned every 3 meters. The pressure was 2.2 bar, with a flow rate of 47 L s^{-1}. Water was pumped from the irrigation channel with an electro pump of 50 kW power. The irrigation volume was 25 mm each irrigation for a total of 10 irrigations and a cumulative irrigation volume of 240 mm (the last irrigation was 15 mm).

The hose-reel machine was settled in 10 ha. Water was pumped from the irrigation channel with a three-impeller motor pump with a 176 kW diesel engine. The pump pressure was 10 bar. The sprinkler gun had a flow rate of 35 L s^{-1} at 6–6.5 bar and about 70 m of radius of throw. Four irrigations were applied with a total irrigation volume of 200 mm during the whole irrigation season.

7.3.3 Water-based indicators

The environmental performance of the three irrigation systems was calculated applying the indicators of water use efficiency by crop (WUE), the Irrigation Water Use Efficiency (IWUE) under the irrigation system, and the Relative Irrigation Supply (RIS) indicator, while the impact on water resource were computed using the Water Footprint (Marino et al., 2019). In order to calculate all those indicators of environmental performance, the crop water requirements were previously assessed. The adjusted crop evapotranspiration ($ET_{c\,adj}$) was assessed on a daily basis according to FAO (Allen et al., 1998) multiplying the crop coefficient (k_c) by ET_0 and by a water stress coefficient (k_s) if a water deficit happen:

$$ET_{c\,adj} = ET_0 * k_c * k_s \quad 1)$$

The term “efficiency” can be associated to irrigation systems or be referred to water-use on a broader sense (Unver et al., 2017). In this study, the indicator of Water Use Efficiency (WUE) was calculated as the ratio between the yield and the actual crop evapotranspiration in kg m^{-3} (Lovelli et al., 2007) for each irrigation system *(i)*.

$$WUE_i = \frac{B_{a,i}}{Et_{c\,adj,i}} \quad 2)$$

where the actual maize biomass (B_a) under a certain irrigation system i or the biomass produced on the non-irrigated test (B_{test}) is related to the adjusted crop evapotranspiration ($ET_{c\,adj}$) for irrigated and the non-irrigated maize(Steduto and Albrizio, 2005). In addition, the Irrigation Water Use Efficiency (IWUE) was assessed. This indicator evaluates the marginal productivity per cubic meter of water provided by the irrigation. The IWUE (kg m^{-3}) was calculated as following (Molden et al., 2010):

$$IWUE_i = \frac{(Y_a - Y_{test})}{I_r} \quad 3)$$

Where the Y_a (kg ha^{-1}) is the actual yield under irrigation, the Y_{test} (kg ha^{-1}) is the non-irrigated yield of the test in the field, and I_r (m^3 ha^{-1}) is the irrigation volume applied. The IWUE corresponds to the increase production (in dry matter) in comparison to the non-irrigated sample yield in terms of the volume of water used for irrigation. Irrigation is one of the most important factors for increasing yield crop. In order to inform how the management of the irrigation systems matched the water requirement, the Relative Irrigation Supply (RIS) indicator was assessed (Morillo et al., 2015):

$$RIS_i = \sum_{ug=1}^{n} \frac{I_r}{10 \cdot (ET_{c\,adj} - P_{eff})} \qquad 4)$$

Basically, RIS evaluates if irrigation under an irrigation system *i* match with the irrigation water deficit relating irrigation water volumes with the difference between $ET_{c\ adj}$ and the effective rainfall (P_{eff}). The RIS indicator shows the irrigation systems performance, which is not merely the irrigation efficiency, but it indicates if supply meets the water requirement (Playán and Mateos, 2006). RIS values less than 1 indicates a deficit of water application, while larger values indicate a surplus.

In addition to the indicators of efficiency, the impact on water resource was calculated. The Water Footprint (WF) indicator measures the impact on water consumption in relation to the surface or the yield (Hoekstra et al., 2011). The study performs the WF by calculating the water consumption as the green water, which is the rainwater used by maize trough evapotranspiration (Chukalla et al., 2015; M.M. Mekonnen A.Y. Hoekstra, 2011), and the blue water, which is the irrigation volume applied to the plants (Bonamente et al., 2016). The green and blue water are different crop water use over the crop growing season, which are calculated for each irrigation system *i* as following (Borsato et al., 2019; Lamastra et al., 2014; Zhuo et al., 2014):

$$CWU_{green,i} = \sum_{d=1}^{lgp} ET_{c\ adj,i} \qquad 5)$$

$$CWU_{blue,i} = I_r \cdot I_e \qquad 6)$$

The green crop water use ($CWU_{green,i}$) represents the total rainwater evapotranspired from the planting day (day 1) to the day of harvest (lgp, length of growing period). The blue crop water use ($CWU_{blue,i}$)represents the total irrigation water evapotranspired from the crop. The blue Crop Water Use (CWU_{blue}) estimated from equation (6) considers the Irrigation efficiency (I_e), which depends on the type of irrigation system used by the farmer: micro or drip irrigation is the most efficient system with a 0.9 coefficient, and sprinkler irrigation (center pivot and hose-reel system) with a 0.7 coefficient (Castellanos et al., 2016). The WF is then calculated as the ratio between the CWU and the yield in terms of $m^3\ ton^{-1}$ of dry matter or the surface in terms of $m^3\ ha^{-1}$:

$$WFgreen_i = \frac{CWU_{green,i}}{Y} \quad \text{or} \quad \frac{CWU_{green,i}}{Surface} \qquad 7)$$

$$WFblue_i = \frac{CWU_{blue,i}}{Y} \quad \text{or} \quad \frac{CWU_{blue,i}}{Surface} \qquad 8)$$

7.3.4 Biomass and yield analysis

The analysis of maize productivity concerns the field measurements of plant biomass and harvesting mass (yield). The crop growth was determined by the measure of the maize biomass on two steps survey. The surveys collected maize plants for one square meter in six points of the field. The first biomass survey was done on the 22nd of June during stem elongation and pre-flowering phase, and the second was done on the 28th of July during full ripening stage of maize grain. The surveys were done for all the fields, and both in the irrigated and in the non-irrigated area. The aerial biomass was collected, then dried in an oven at 105°C for 36 hours, and finally weighted. A statistical analysis was implemented to observe the repeatability of the survey using the standard deviation from the mean of samples. The biomass was used to analyze different indicators related to the crop growing rate and productivity. The crop growth rate (*CGR*) is the absolute amount of biomass which increase in a certain period, and it was assessed by the equation (Toniolo et al., 1985):

$$CGR_i = \frac{Biomass_{t2} - Biomass_{t1}}{time} = \tag{9}$$

where the mass increase (g day^{-1}) is calculated as the difference of the biomass measured at the second survey ($Biomass_{t2}$) minus the biomass measured at the first survey ($Biomass_{t1}$) by the interval time (*days*). The *CGR* is also applied to the ear growing rate as in equation 10, while the biomass is referred to the grain production. Furthermore, an additional indicator of crop increase biomass is the relative growing rate (*RGR*), which is the efficiency to growth with respect to mass (Toniolo et al., 1985; Hunt et al., 2002):

$$RGR_i = \frac{ln_{Biomass\,(t2)} - ln_{Biomass\,(t1)}}{time} \tag{10}$$

The yield data collection was implemented with a field survey on the 28th of August collecting grain samples (ears) in 8 points corresponding to 1 m^2 with a replication both on the irrigated and non-irrigated area and during harvesting on the 2nd of September, with a harvester machine recording the yield. The yield performance was analysed through the index of crop productivity in terms of tons ha^{-1} of dry matter. Then, the Harvest Index (HI) was assessed computing the ratio between the yield and the total biomass collected during the hard dough phase:

$$HI = \frac{Corn\ mass}{Total\ biomass} \tag{11}$$

An additional indicator useful to analyse the crop response to yield is the *crop yield response factor to water* (ky), which is the ratio between the relative evapotranspiration decrement and the relative yield decrement (Haghverdi et al., 2019). The coefficient is useful to analyse how crop can be tolerant to water stress according to different irrigation treatment *i*. The yield response factor was assessed as following (Doorenbos and Pruitt, 1977):

$$k_{y,i} = \frac{\frac{(ET_{ir} - ET_{test})}{ET_{ir}}}{\frac{(Ya - Y_{test})}{Ya}} \tag{12}$$

where the ET_{ir} (mm) is the adjusted crop evapotranspiration within irrigation, and ET_{test} (mm) is the adjusted crop evapotranspiration of the non-irrigated test. Accordingly, Y_a is the crop yield within irrigation, and Y_{test} is the yield of the non-irrigated test. In a second step, ky was calculated substituting the yield with the CGR to implement a further analysis of yield response factor.

7.3.5 Economic balance and related indexes of performance

The comparison of the economic benefit of irrigation between the three-irrigation systems concerns the analysis the gross marketable output and the costs. The gross marketable output was performed multiplying the yield by the corn price on trademark established to 172 € ton^{-1}. The corn price is referred to the price of Bologna trademark on a national hybrid corn during season 2015. The net income is expressed as the gross income minus the irrigation costs (fix and variable). Table 1 lists the inventory of expenditures (total Irrigation Costs), the total Gross income and the Net Income for the three irrigation systems.

Table 1. Total gross income, costs and net income for the three different irrigation systems. All the items are expressed in € ha^{-1}.

€ ha^{-1}	Center pivot system	Drip Irrigation	Hose reel system	TEST
Gross marketable output	2054	2300	1949	917
Amortisation value	131	62	111	-
Labour and maintenance	30	125	75	-
Energy	99	104	171	-
Cost for drip line	-	320	-	-
Total Irrigation costs	260	610	357	-
Net income	1794	1690	1592	917

Costs are divided into costs for labour and maintenance, cost for energy, cost for purchasing the equipment and costs of linear depreciation (amortisation). The cost of labour is 15 € h^{-1}, that is the average among the dependent and temporary job. It considered energetic costs for gasoline (0.65 € L^{-1}) and electricity (0.18 € kWh^{-1}) and the price of new machines while calculating the amortisation value (higher cost of investment for sprinkler systems) depreciated over their obsolescence time period (10, 15 and 20 years for drip, hose-reel, and center pivot, respectively). The water price was intentionally not mentioned, because it was considered negligible, since it refers to merely the service cost of water supply by the Water Board, which is fixed, not volumetric, and computed based on the benefit of growing on irrigated or non-irrigated land.

The economic sustainability analysis consider the indicator of the "Water Productivity" (€ m^{-3}) that is the ratio between the difference of the net income between the irrigated (Net income_{ir}) and non-irrigated maize $(\text{Net income}_{test})$ and the volume of water used for irrigation:

$$\text{Water productivity} = \frac{\text{Net income}_{ir}\ (\text{€ ha}^{-1}) - \text{Net income}_{test}\ (\text{€ ha}^{-1})}{\text{I}_r\ (\text{m}^3\text{ha}^{-1})} = \quad (13)$$

The water productivity highlights the positive expected return of irrigation from an economic point of view. The volumetric impact on water resource is given by the ratio of the WF blue or green (m^3 ha^{-1}) and the marginal economic benefit from irrigation (€ ha^{-1}):

$$economic\ WFgreen = \frac{\text{WF}_{green}\ (\text{m}^3\ \text{ha}^{-1})}{\text{Net income}_{ir}\ (\text{€ ha}^{-1}) - \text{Net income}_{test}\ (\text{€ ha}^{-1})} = \quad (14)$$

$$economic\ WFblue = \frac{\text{CWU}_{blue}\ (\text{m}^3\ \text{ha}^{-1})}{\text{Net income}_{ir}\ (\text{€ ha}^{-1}) - \text{Net income}_{test}\ (\text{€ ha}^{-1})} = \quad (15)$$

The blue and green WF have a different cost of opportunity and they need to be compared with the marginal Net Income for a full understanding of their economic performance.

7.3.6 Energetic balance and related indexes

The irrigation systems were compared in terms of their energetic consumption and Green House Gas (GHG) emissions. The energetic analysis considers the type of energy used for pumping water and irrigation itself (Table 2). The energy consumption is referred to a unit of irrigated surface.

Table 2. Inventory list of annual energy consumption and costs for the three different irrigation systems considered per hectare.

Inventory list	Metric	Drip Irrigation	Center Pivot	Hose reel
Number of Irrigations	n°	12	10	5
Electricity consumption single irrigation per hectare	kWh (*)	48	55	52.5
Energetic cost	€ ha⁻¹	104	99	171
Electricity price	€ kWh⁻¹ (**)	0.18	0.18	0.65

*Diesel (L) in case of hose-reel system

**Diesel price (€ L^{-1}) in case of hose-reel system

The energy consumption is the product of the electricity or the diesel consumed per hectare by the number of irrigations. Thereby, the conversion factor was used for electricity (1 kWh = 3.6MJ) and diesel (1L diesel = 42.7 MJ). The energetic costs were assessed as the product of the energy consumption and the unit cost (Handa et al., 2019). The energy-related indictors are described by the impact of energy consumption per unit of irrigation water consumed:

$$Energetic\ Footprint = \frac{\mathrm{CWU}_{blue}}{Energy\ consumption} = \quad (16)$$

Similarly, the Energetic Cost Footprint is calculated as followed:

$$Energetic\ Cost\ Footprint = \frac{\mathrm{I}_r}{Energy\ cost} = \quad (17)$$

The energetic performance is evaluated as the ratio between the energy consumption and the GHG emission with the blue water consumption (Borsato et al., 2018b):

$$Energetic\ Performance = \frac{Energy\ content\ (MJ\ ha^{-1}) / GHG\ (kg\ CO_{2-eq}\ ha^{-1})}{\mathrm{CWU}_{blue}\ (m^3\ ha^{-1})} \quad (18)$$

The energy content is the energetic nutritional content in maize production, where 1 kg of maize contains 14.75 MJ (Carnovale and Marletta, 1987). The GHG emissions are implemented using the characterization factor of diesel (1L diesel corresponds to 0.544 kg $CO_{2\text{-eq}}$) and electricity (1 kWh electricity medium voltage at Italian level corresponds to 0.534 kg $CO_{2\text{-eq}}$) for Carbon Footprint provided from the Ecoinvent 3.3 database (Ecoinvent, Zurich, Switzerland).

7.4 Results and Discussion

7.4.1 Water based indicators

The crop season was characterized by 300 mm of rainfall. Maize had a different evapotranspiration according to the plant vigour under the different irrigation systems plots. The cumulative water supply from rainfall and irrigation was 534 mm for drip irrigation, 542 mm for center pivot system, and 502 mm for the hose reel system

(Figure 2). The $ET_{c\ adj}$ of maize for the entire season was 560 mm under the drip irrigation, 586 mm under center pivot, and 524 mm under the hose reel system.

During the growing season, the maize under center pivot gains a greater water supply than all the other irrigation systems. The water supplied with irrigation and rainfall met the crop water requirement all the time. In drip irrigation, the irrigation met the crop water requirement with no water stress, while in the maize plot under the hose reel system, the maize suffered from a water stress only in the last period during the phase of corn ripening (Figure 2).

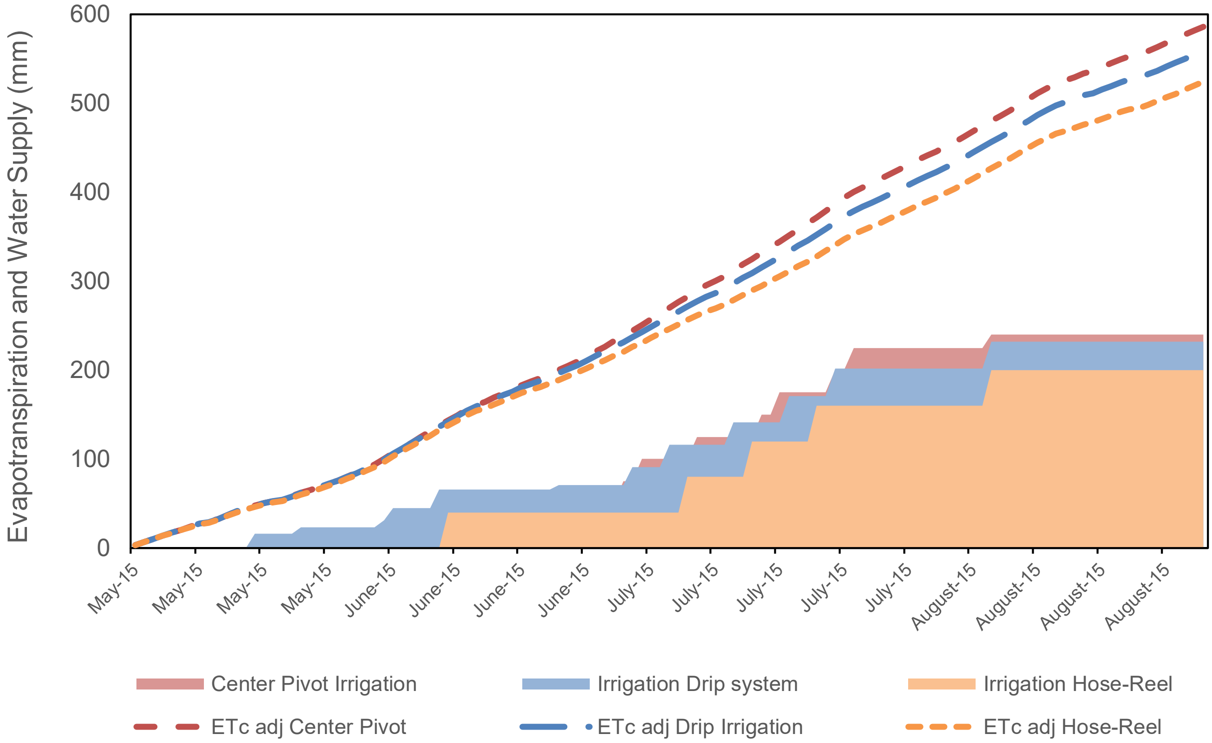

Figure 2. Cumulative water supplied from irrigation and crop evapotranspiration ($ET_{c\ adj}$) over the growing season for each irrigation system.

The WUE shows the efficiency of water consumption and biomass production, which is related with the crop growing stage (Table 3). The WUE was determined in two different periods of the growing season during which time there were slight differences between irrigated and non-irrigated tests visible when looking on the WUE from the 22 of June to the 28 of June (Table 3). It is curious to note how the non-irrigated test had a higher WUE (1.317 kg m^{-3}) than the hose-reel system (0.972 kg m^{-3}), although the lower evapotranspiration and biomass production of the test. This might be explained by the fact that the non-irrigated test benefitted from the rainfall water supply at the begin of the season and suffered later from the water stress, while the difference with a lower WUE within the hose-reel system might be due to other factors of loss beside irrigation (diseases). The difference in WUE started with the late season, where drip irrigation (1.875 kg m^{-3}) and center pivot systems (2.04 kg m^{-3}) reached higher values.

In addition, the IWUE was higher in drip irrigation (2.981 kg m^{-3}) and hose-reel irrigation systems (2.58 kg m^{-3}), which meant that the productivity obtained from a unit of water supplied with the practice of irrigation provided a better response than with the center pivot (2.37 kg m^{-3}). This was true because the hose-reel system furnished a lower amount of water with the irrigation, and drip irrigation gained a better performance combining irrigation volume and crop yield. The RIS was very similar over the three irrigation systems and it stood between 0.75 for the center pivot and 0.79 for drip irrigation. In other words, drip irrigation expressed better the efficiency on supplying water compared to the other irrigation systems because of its superior management.

Table 3. Comparison of water use efficiency (WUE), irrigation water use efficiency (IWUE), and relative irrigation supply (RIS) under different irrigation systems. The dates of WUE are reported where the non-irrigated test is included.

Irrigation plot	WUE 22-June	WUE 28-July	IWUE	RIS
	Kg m^{-3}	Kg m^{-3}	kg m^{-3}	
Drip irrigation	1.251	1.875	2.981	0.79
Center pivot	1.349	2.040	2.370	0.75
Hose-Reel system	0.972	1.541	2.580	0.78
Non-irrigated test	1.319	1.627		

The WUE and the IWUE are two indicators that have similar metric, but they are conceptually different. According to the Sustainable Development Goals, a sustainable irrigation practice can produce more food with less water. In this sense, drip irrigation is the one that results in a greater ear production to the detriment of a lower total biomass production per m^3.

In addition, the impact on the water resource from water consumption is expressed as the blue and green water footprint. Figure 3 expresses blue WF and green WF as the water consumed over the edible biomass produced. The blue WF was lower under the hose-reel system with 144 m^3 ton^{-1}. The reason is due to the lower irrigation volume applied, while drip irrigation and center pivot presented a blue WF of 182 m^3 ton^{-1} and 164 m^3 ton^{-1}, respectively. In addition, the green WF under the hose-reel system gained the higher value of 274 m^3 ton^{-1}, and only 232 m^3 ton^{-1} and 260 m^3 ton^{-1} for drip irrigation and center pivot, respectively (Figure 3). In addition, the green WF of the non-irrigated test had the greatest value of 583 m^3 ton^{-1}.

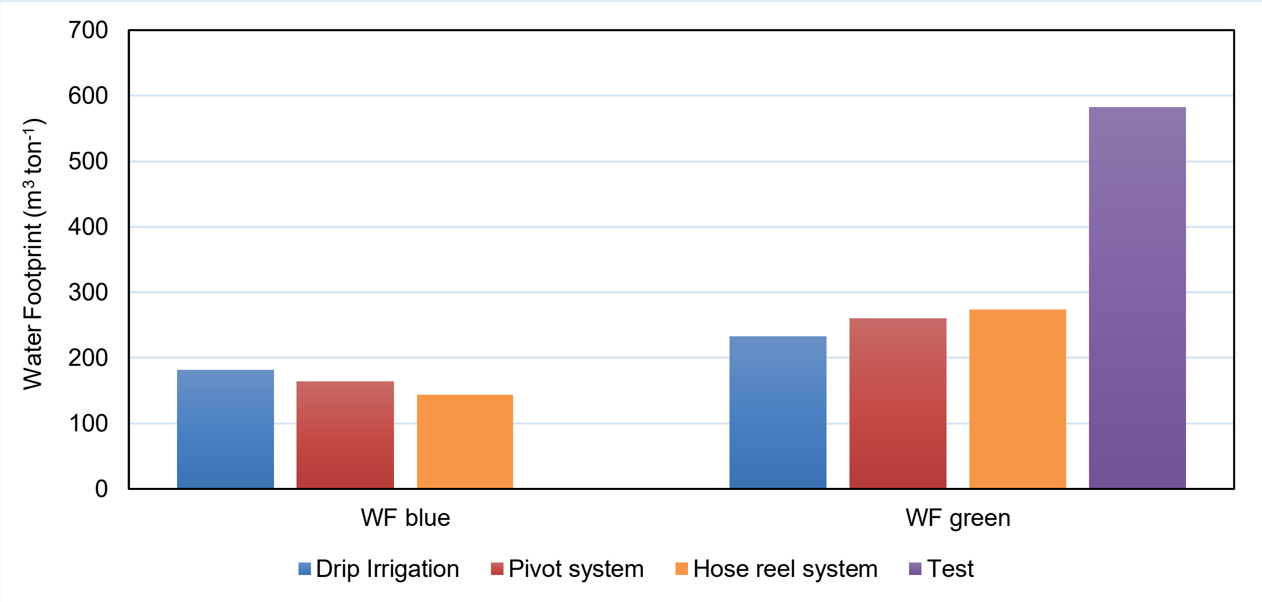

Figure 3. Water Footprint (WF) blue and green of each irrigation system in terms of m^3 ton^{-1}. For Water Footprint (WF) green, the non-irrigated test was included.

7.4.2 Indicator for Biomass and Yield Evaluation

The crop yield registered during the harvesting stage was higher in center pivot with 11.5 tons ha^{-1}, followed by maize under drip irrigation system with 10.3 tons ha^{-1} and hose reel with 9.7 tons ha^{-1}. The biomass was measured in two phenological stages (BBCH-scale, Hess et al., 1997): end of stem elongation (BBCH 39) and ripening-early dough (BBCH 83). The HI measures the relation between the harvest product and the biomass produced under the related irrigation system, which is higher (0.533) for hose reel irrigation machine (Table 4). Center pivot shows a relatively low HI (0.478) due to the higher biomass production, while the rain-fed (non-irrigated test) has the lowest HI in relation with the lower yield production (0.416). Table 4 reports both the CGR that shows the difference in maize vigour under the different irrigation systems, and the RGR that determines the mass productivity and the efficiency of growth from the end stage of stem elongation to the early dough of corn kernel (Table 4).

Table 4. Indicators of performance of biomass and harvest production such as the Harvest Index (HI), Crop Growth Rate (CGR) and Relative Growth Rate (RGR) are shown for drip, center pivot and hose-reel irrigation systems.

Irrigation system	HI at hard dough phase	CGR maize biomass	CGR maize ear	RGR maize biomass	RGR maize ear
	$kg_{harvest} \cdot kg_{biomass}^{-1}$	g day^{-1}	g day^{-1}	g g^{-1} day^{-1}	g g^{-1} day^{-1}
Drip irrigation	0.521	13.5	21.5	0.031	0.031
Center pivot	0.478	15.9	17.5	0.033	0.026
Hose-reel	0.533	10.4	18.9	0.032	0.033
Test	0.416	6.4	7.3	0.019	0.024

The CGR was different between irrigation systems if we consider the total biomass, where the center pivot had the greater value of 15.9 g day^{-1} (Table 4). Moreover, looking to the CGR for the edible part of corn, the drip irrigation system had the greater value (21.5 g day^{-1}). The maize growing speed was described by the RGR indicator (g g–1 day^{-1}), which was higher under the center pivot in terms of total biomass (0.033 g g^{-1} day^{-1}), and greater under the hose-reel in terms of ear production (0.033 g g^{-1} day^{-1}). The RGR under drip irrigation showed that this type of system allows one to reduce water stress and plants can grow at the same rate both for the edible part of the corn ear and for the vegetal part of steam and leaves. A further indicator of biomass productivity and crop performance is the dimensionless crop yield response factor (ky). This indicator helps to understand the contribution of irrigation on crop productivity and the occurrence of a potential water deficit. As seen in Figure 4, different maize conditions under different irrigation management showed also that the yield response factor was greater within the center pivot system due to the higher evapotranspiration in relation to the yield. This is different if the biomass productivity is considered; the ky CGR describes a different trend of the crop response on biomass production in relation with irrigation. The hose-reel irrigator machine has the higher value of 0.57 in comparison with the drip irrigation that gains a value of 0.41. A general definition of ky factor defines that the higher the value is, the more sensible is the crop to water deficit. Figure 4 shows how ky might change between different irrigation practices looking either to the biomass or to the grains production. In Figure 4, the error bars are added to describe the standard deviation from between the irrigated plots and the rain-fed test.

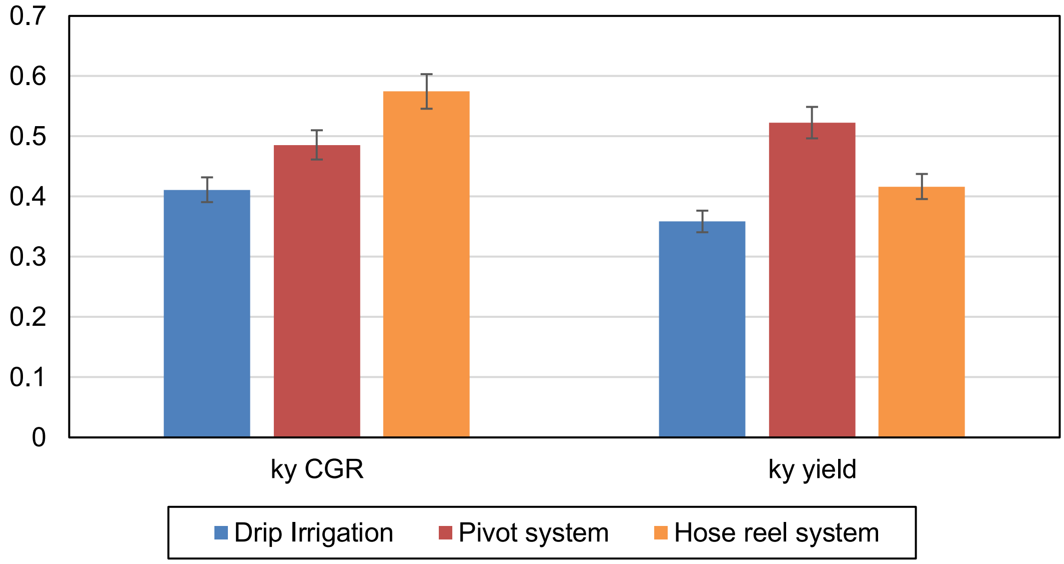

Figure 4. Crop yield response factor (k_y yield) of harvesting product and the k_y from biomass production (k_y CGR) of maize under different irrigation systems.

7.4.3 Economic indicators

The economic aspect of sustainability of irrigation systems deals with the use of the indicator of water productivity and the economic water footprint. The water productivity is the economic benefit of an additional unit of water. Looking to the water productivity, the greater value was observed for center pivot (0.37 € m^{-3}), which presented a higher irrigation volume in relation with a higher net income by 12% and 8% than drip irrigation (0.33 € m^{-3}) and hose-reel machine (0.34 € m^{-3}), respectively. Figure 5 shows the relation between the water consumed to produce a unit of income explained by the economic water footprint (WF) of different irrigation systems. In fact, blue and green water use had different opportunity costs. This was especially more evident under the hose-reel system where a lower economic WF blue (2.07 m^3 $€^{-1}$) corresponding to a higher economic WF green (3.95 m^3 $€^{-1}$). The center pivot machine gained a great water productivity and the lowest impact on water resource with 1.91 m^3 $€^{-1}$ and 3.04 m^3 $€^{-1}$ for WF blue and WF green, respectively. In this paper, a general consideration was needed when looking to economic indicators, because a small variation in the market price can vary consistently the gross marketable income, and in the meantime could tip the economic balance in favor of one or another irrigation system due to the high weight of this economic component compared to the rest of the balance.

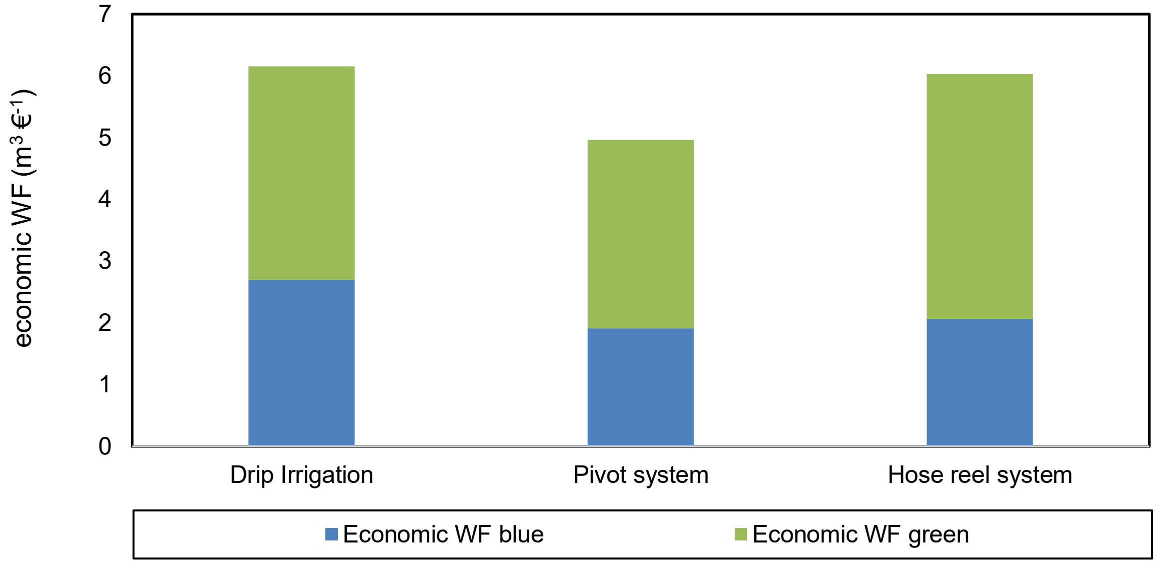

Figure 5. Water productivity and economic Water Footprint of different irrigation systems.

7.4.4 Energetic indicators

The energy consumption for irrigation purposes, which corresponds to the electricity and diesel consumption for the engine pumping water, involves GHG emissions. GHG emissions held from drip irrigation correspond to 308 kg $CO_{2\text{-}eq}$ ha^{-1}, while becomes lower to 26 kg $CO_{2\text{-}eq}$ ha^{-1} if looking to the single irrigation stage. The total GHG emissions for center pivot and hose-reel systems are 294 and 143 kg $CO_{2\text{-}eq}$ ha^{-1}, respectively. Those values change when we consider the GHG emissions of the single irrigation to 29.4 kg $CO_{2\text{-}eq}$ ha^{-1} for center pivot and 28.5 kg $CO_{2\text{-}eq}$ ha^{-1} in case of hose-reel system (Table 5). The Energetic Footprint shows the water consumption per unit of energy consumed per irrigation. The drip irrigation gains a greater value (1 m^3 MJ^{-1}), while the center pivot consumes 0.85 m^3 per MJ consumed, and the hose-reel spends 0.12 m^3 MJ^{-1}. The Energetic Cost Footprint is the water consumption supplied with irrigation per unit of energy cost. center pivot is the more efficient system in terms of water supplied per unit of cost energy spent (24.2 m^3 €$^{-1}$). Drip irrigation and hose-reel system show an Energetic Cost Footprint of 22.3 m^3 €$^{-1}$ and 11.7 m^3 €$^{-1}$, respectively (Table 5). Differently, the energy performances are given from the relation between the ratio of energy content in maize, the GHG emissions, and the blue WF. The hose reel system shows the greater value of energy performance with 0.35 MJ m^3 $kgCO_{2\text{-}eq}^{-1}$. Drip irrigation and center pivot show a lower value of 0.21 MJ m^3 $kgCO_{2\text{-}eq}^{-1}$ and 0.15 MJ m^3 $kgCO_{2\text{-}eq}^{-1}$, respectively. Energy indicators related to environmental impact for the different irrigation systems studied allow to understand energy performance, the environmental impact from the energetic point of view and in relation to water consumption.

Table 5. Energy indicators of environmental impact for irrigation practice comparing different irrigation systems.

	Energetic Footprint	Energetic Cost Footprint	Energy performance	GHG emission per irrigation
	m^3 MJ^{-1}	m^3 €$^{-1}$	MJ m^3 $kgCO_{2\text{-}eq}^{-1}$	$kgCO_{2\text{-}eq}$ ha^{-1}
Drip irrigation	1.00	22.3	0.21	26
Center pivot system	0.85	24.2	0.15	29
Hose reel system	0.12	11.7	0.35	29

7.5 Conclusion

This study provides different environmental indicators suited to analyze the sustainability of different irrigation systems. The study includes indicators of water balance, economic balance, and energetic analysis. Those indicators help users to detect the most performant irrigation system from environmental, energetic, and economic points of view. The main results indicate the center pivot system has generally higher performance among the irrigation systems in this case study. In fact, the CGR, RGR, and ky gained a better response under the center pivot. At the same time, indicators related to the economic balance showed that the center pivot was the better irrigation system with higher water productivity and lower economic water footprint. Indicators related to water balance generally performed better under drip irrigation. However, the hose-reel system supplied a lower amount of water (lower WF blue), while the center pivot had a better WUE but a worse IWUE. Finally, drip irrigation showed better performance on IWUE and RIS, and with a lower WF green. Looking to the energetic performance, there is no clear picture of the best performant irrigation system. However, drip irrigation gained a higher energetic footprint, while the center pivot had a higher energetic cost footprint, and the hose-reel system showed a better energetic performance. Comparing the overall contribution on the environmental sustainability, the center pivot system combines good performance and presents an overall good solution for irrigation sustainability, especially from an economic point of view.

This study enhances the choice of the most appropriate irrigation system held under maize cropping. Further analysis under different crops must be implemented. In fact, the three systems are not always alternatives. For example, a pivot is not feasible on very small plots or on irregularly shaped plots; drip is not applicable to pasture crops (if crops rotate), while sprinkler systems may be; and hose-reel is inefficient in very windy areas. Investment in irrigation should consider crop succession throughout the year. Moreover, a one-year study was carried out; a sensitive analysis should be implemented to analysis what variable might affect the choice of the users to a suitable sustainable irrigation system. Besides that, the use of the correct irrigation system needs to combine a proper irrigation water management to take advantage of its performance.

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8. Conclusions

The thesis uses multiple indicators and frameworks to compare the sustainability of agronomic practices on water resource management. This manuscript focuses on the quantification of water footprint and related indexes to analyze water sustainability, to understand which approach is more suitable and robust to quantify the impact on water resource, and it gives indications of the most appropriate agricultural practices for a better water management and impact reduction on a specific environmental and climate context. Although literature provides several studies on WF for different research fields, not many of them deal with agricultural field management. Moreover, while the International standard ISO 14046 furnishes a guideline for Water Footprint, the research community is still debating what methodology can better represent the whole impact on water resource. This thesis overcomes this gap and suggests a combined approach between the Life Cycle Assessment and Water footprint Assessment comparing both methodologies in a single report. Going into detail, this study supports the hypothesis of reducing and mitigating water footprint applying an alternative and sustainable agricultural system looking for: precision agriculture, different crop choice, variable rate application of inputs (ex. chemicals), conservative soil management, irrigation system applications, organic field management, and efficient improvements of the water use in a rural system.

In the first paper (chapter 2), 70 main agricultural products were analyzed for their impact on water and energy resources. For each product, a global average score for the Water Footprint (WF) determined the impact of water consumption. The Energetic Ratio (ER) was evaluated in order to combine the input of energy, as calories of energy content, and the output as the greenhouse gases emitted during food production. The aim of the paper was to show the scheme of sustainability for several agricultural products in a graph of food clusters according to their WF and ER values. The common trend of the impact degree for each product showed a high value of water consumption related to a high value of the energetic parameter, and on the other hand, a low water consumption is related to a low energetic ratio. In other words, oil crops, pulses and dry fruits provided a good energy trend but with a wasteful water trend, while vegetables had a low water consumption linked with a low energetic ratio. However, in the cluster of animal products, a low energy ratio was related to a higher water use. Although there were no eco-friendly clusters, vegetable products represented the closest cluster highly sustainable.

In the second paper, two WF methods were compared to implement their results interpretation. Basically, the two approaches for the WF evaluation need to be complementary although methodological differences are present. The WFA based indicators (VIVA WATER indicator) and the water-focused LCA method were compared evaluating the same case study of the WF of a bottle of wine. The main outcome from this work was that VIVA framework provided recommendations for the optimal management of direct water use during vineyard phase, while LCA approach addressed the sustainability assessment from a Life Cycle Thinking perspective. VIVA WATER indicator mainly focused on water management aspects based on the agronomic and ecological knowledge needed at the vineyard stage, while other stages of wine production play a less significant role. The LCA approach allowed the assessor to investigate in more detail the cellar and bottling stages and to understand the hotspot impacts of chemicals on water quality indicators. The synergic interpretation of the two methodologies can reinforce the scope of the evaluation of environmental sustainability for reducing freshwater use and water pollution. The assessment of different impact categories in the water-oriented LCA methodology allowed to integrate VIVA WATER indicator especially in the vineyard

stage, where a set of water-related impact indicators can give more information in terms of direct and indirect water depletion and water quality impact.
The third paper provided a framework for a sustainable water use. The novelty of the study stems from the introduction of the framework evaluating the sustainability of irrigation practice both from the hydrological and economic points of view. In the paper, we evaluated trade-offs between the loss of natural capital, while preserving human made capital and vice versa. This approach allowed to quantify the weak and strong sustainability of irrigation. More specifically, the framework provided quantitative performance in terms of sustainability of irrigation and its likelihood; it showed how the notions of reliability, resilience and overuse/overexpenditure can be translated into suitable indicators addressing both the environmental and socio-economic aspects of sustainability. These indicators provided a rather comprehensive picture of the sustainability of the system; it can be applied at different spatial and temporal scales and used by stakeholders to support decision-making in the context of water management and water policy development. The case study was implemented for the case of Australia. In this specific context, we found that Australian irrigation is for most part weakly sustainable. In other words, irrigation water is unsustainably used to produce commodities at the expenses of the environment. Interestingly, our analysis showed that irrigation had stronger negative impacts on environmental flows within river catchments, while groundwater use for irrigation was more sustainable. Overall, irrigation contributed to a more reliable and resilient crop production with a relatively high level of economic water profitability.
The following research paper discusses about suitable agricultural practice to reduce the WF. In particular, the fourth paper (chapter 5) evaluated the grey WF of different soil management. The case study described how different methods of soil management enhanced a reduction on grey water footprint. The study focuses on the synergetic effects of different soil practices as different soil tillage systems and the use of variable rate application, which had a positive grey WF reduction. Accordingly, the results highlight which interaction between soil tillage systems and soil management reduced the grey WF of soil practices. The variable rate application consistently decreased the impact on water in terms of water pollution and chemical soil degradation. The Minimum Tillage presented a lower WF, both in terms of $m^3\ ton^{-1}$ and in $m^3\ ha^{-1}$ with the use of Precision Farming. Strip Tillage under Variable Rate Application and No Tillage under Variable Rate Application have a higher grey WF reduction in terms of $m^3\ ton^{-1}$ with a 10% and 11% of reduction respectively. The grey WF of Nitrogen application was better reduced within Minimum Tillage under Variable Rate Application in terms of $m^3\ ha^{-1}$ by the 9.5%. The case study provided suggestions for suitable soil management in order to reduce the water pollution. In the fifth paper (chapter 6), we implemented the comparison between a conventional and organic vineyard management using WF and other environmental indicators and showing that organic management decreased the Water Footprint and Carbon Footprint thanks to lower field passages and amount of fertilizers applied. The Vineyard management Indicator presented a better performance under organic management and the economic benefit was greater thanks to the higher marketable price and lower management costs. Finally, an organic vineyard management did not compromise the economic productivity for vine grape production, and moreover it can enhance a mitigation on the environmental impact. Considering the case study, the organic vineyard management is a suitable model that farmers can follow to reach their objectives while respecting the environment. In the sixth paper (chapter 7), we compared three different irrigation systems under maize cropping, one drip irrigation and two sprinkler irrigation (center pivot and hose

reel machine). The study performed indicators to detect the most performant irrigation system from environmental, energetic, and economic points of view. The main results addressed the pivot system as the righteous choice for irrigation practice. In fact, the Crop Growth Rate, Relative Growth Rate and ky gained a better response under centre pivot. On the same time, indicators related to the economic balance showed that center pivot was the better irrigation system with higher income with respect to production. Indicators related to water balance, generally performed better under drip irrigation. However, hose-reel system supplied less amount of water (lower WF blue), while center pivot had a better Water Use Efficiency. Finally, drip irrigation showed better performance on Irrigation Water Use Efficiency and Relative Irrigation Supply, and with the lower WF green. Looking to the energetic analysis, there is no clear picture of the best performant irrigation system; drip irrigation gained a higher Energetic Footprint, while pivot had a higher Energetic Cost Footprint and hose reel system showed a better Energetic Performance. Comparing the overall contribution, the pivot system presents an intermediate trend, which represents a generally good solution most of the time, especially from the economic point of view with a great potential on biomass production and on the environmental aspect of sustainability.

Finally, future studies can focus on the research of tools able to provide a comparable value of water footprint, which can be combined or grouped within a family of footprints, in order to better analyze sustainability. Innovations should be studied as strategies which can enhance mitigations on water footprint or enhance a possible water footprint detection integrating for example a remote sensing tool with field measurements (sensors and instruments) at local or regional level.

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