Scuola di Dottorato per il Sistema Agro-alimentare

Doctoral School on the Agro-Food System

cycle XXIX

S.S.D: AGR/13

Towards a sustainable food system

Life Cycle Assessment applied to agro-food products

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Academic Year 2015/2016

Summary

Providing energy and nutrients, food production is essential for life. However it represents also an important environmental concern. Indeed, the rapidly growing world population is requiring an increased food production which is one of the greatest causes of environmental degradation throughout the world. Emitting a large amount of powerful greenhouse gases (GHG), such as methane (CH_4) and nitrous oxide (N_2O), agricultural production is one of the main contribution to climate change. Furthermore, agricultural production contributes to water pollution, land use and biodiversity loss.

The agricultural system is based on complex relations that link agricultural productivity, environmental functions and environmental conditions. Therefore, moving towards less polluting production systems is of the utmost importance to satisfy the current demand for food without compromising the possibility for future generations to have access to a proper amount of food of adequate quality.

Life cycle assessment (LCA) methodology is at the core of sustainability assessment. Indeed, considering the entire life cycle of a product or service, it allows to account for potential shifts of environmental impacts between environmental compartments or stages of the food supply chain. The use of LCA is widely spread both in business and in decision-making contexts as support to strategic decisions and for environmental communication.

ISO 14040 and 14044 standards are internationally recognised as general references for the application of LCA. However, being applicable to different productive sectors, these standards do not specify how to deal with all the choices that a person has to make when performing a LCA, such as, by a way of example, the functional unit and the allocation procedures.

With the aim of fostering the adoption of the same approach, guaranteeing the comparability of the results and, therefore, make easier the interpretation of environmental claims, several product category rules have been developed for different productive sectors and in different countries. However, these initiatives have contributed to the definition of a fragmented framework in which the results of LCAs on a certain product or service are hardly comparable. Moreover, the current situation can, on one hand, represent a market barrier for companies who wish to sell and communicate the environmental performance of their products in different countries and, on the other, limit consumers' trust in environmental claims. In order to foster the diffusion of the most resource efficient products, the European Commission has defined two methods for the

assessment of the environmental performance of products (PEF) and organisations (OEF) and a set of principles for their communication. In this context, a pilot phase to test and implement the PEF and OEF methods is currently ongoing, involving several stakeholders.

The general aim of the present thesis is to contribute to the ongoing debate on the harmonisation of the approaches to carry out a LCA referred to agro-food products in order to foster its reliability and effectiveness.
The research activities were carried out between January 2014 and December 2016 at Institute of Agricultural and Environmental Chemistry of Università Cattolica del Sacro Cuore in Piacenza and at Institute for Environment and Sustainability (IES) of Joint Research Centre (JRC) in Ispra.

The contents of each chapter are hereunder briefly described.

Chapter 1 introduces the general framework in which the concept of the present thesis has been developed.

Chapter 2 reports the aims of the thesis, a brief description of the chapters and the research questions discussed in Chapters 3, 4 and 5.

In Chapter 3 the analysis of secondary datasets modelling arable crops production and belonging to three databases commonly used in the agro-food sector is described. The use of secondary datasets is a common practice in LCA when primary data are not available or their collection is too much resource-intensive. However, different inventory data and modelling approaches are used to populate secondary datasets, leading to different results. The analysis identified the characterising elements of datasets and highlighted important differences among them. Therefore recommendations were drawn from the datasets comparison, supporting the selection of the datasets coherently with the goal and scope of the study and interpretation of results.

Chapter 4 reports some considerations on the development of a systematic approach to account for the burden of food loss and waste in LCA. Currently food loss and waste are rarely included in LCA studies and, when considered, different approaches are adopted leading to contrasting results. Considering the relevant impact associated with food loss and waste, their systematic inclusion in LCA studies is highly desirable. The chapter includes an analysis of the published relevant literature. It suggests a definition of food loss and waste to be adopted in LCA, it investigates the consequences of using such definition and it proposes potential paths for the

development of a common methodological framework to increase the robustness and comparability of the LCA studies.

In Chapter 5 an analysis of the GHG emissions of three balanced dietary patterns for an Italian man is reported. The consumption phase of food is often omitted in LCA studies on diets. However, as demonstrated by our analysis, it can represent a significant contribution to the overall GHG emissions caused by a diet. Therefore the analysis highlighted the need to include the consumption phase within the system boundaries, above all when the environmental burdens of different diets are compared, and emphasised the central role of consumers in the reduction of the GHG emissions of the diet.

Finally, Chapter 6 reports the general conclusions of the present thesis and potential future research proposals.

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List of Acronyms

AP: Acidification Potential

CC: Climate Change

FAO: Food and Agriculture Organization

FEP: Freshwater Eutrophication Potential

FL: Food Loss

FSC: Food Supply Chain

FW: Food Waste

FWecotox: Freshwater ecotoxicity

GHG: Greenhouse Gas

GWP: Global Warming Potential

HT, c: Human Toxicity, cancer

HT, nc: Human Toxicity, non cancer

IE: Industrial Ecology

JRC: Joint Research Centre

LCA: Life Cycle Assessment

LCI: Life Cycle Inventory

LCIA: Life Cycle Impact Assessment

LCT: Life Cycle Thinking

MEP: Marine Eutrophication Potential

OEF: Organisation Environmental Footprint

PEF: Product environmental footprint

PM: Particulate Matter

POFP: Photochemical Ozone Formation Potential

PPP: Plant Protection Product

TEP: Terrestrial Eutrophication Potential

UNEP: United Nations Environment Programme

1. Background and introduction

1.1 Outline of the environmental impacts of the food system

The scientific community agrees that we are currently exploiting resources and polluting the environment at a speed much higher than its natural restoration rate (Ewing et al., 2010; FAO, 2012). Climate change is often perceived as one of the main environmental issue to be currently faced, but it is not the only one. Rockström et al. (2009) highlighted that anthropic activities are responsible of other environmental changes – such as the loss of biodiversity and the alteration of the nitrogen cycle - that have brought our planet to an environmental state beyond the planetary boundaries which define a "non-dangerous situation for humanity". The food system, intended as the set of processes and infrastructures involved in feeding a population, is one of the main drivers of these environmental alterations.

It has been estimated that in developed countries food consumption generates between 15% and 28% of the overall national greenhouse gas (GHG) emissions (Garnett, 2011). Contributing from 17% to 32% of the global GHG emissions, agricultural production is by far the most GHG-emitting stage of the supply chain. With the mineralisation of biological carbon stocked in soil and trees, land use change associated with agricultural activities accounts for 6% to 17% of the global GHG emissions. Moreover, direct emissions of methane, mainly due to ruminants' enteric fermentation, and emissions of nitrous oxide associated with nitrogen fertilisers application represent 10% to 12% of the global GHG emissions (Figure 1) (Bellarby et al., 2008).
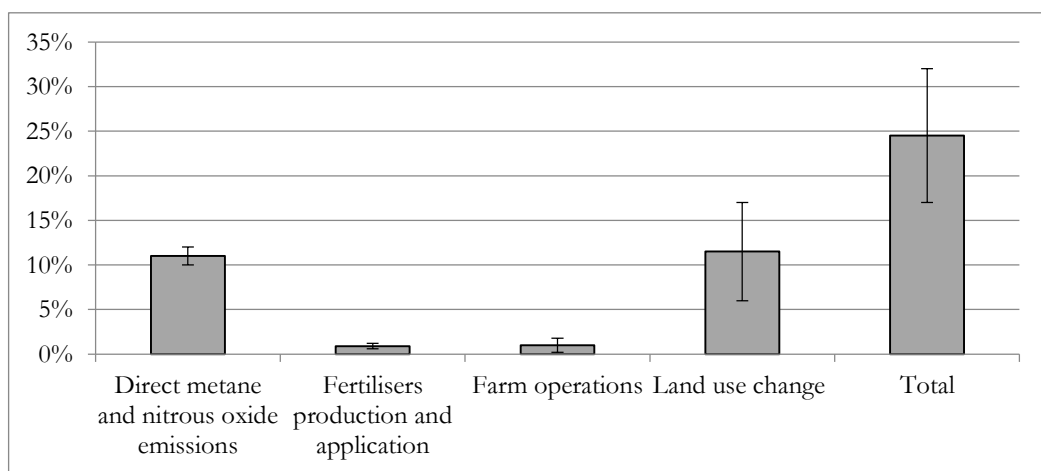

Figure 1: Contribution of the agriculture sector to global GHG emissions (Adapted from Bellarby et al., 2008)

Garnett (2011) estimated that in the UK, about 45% of the GHG emissions of the food system (excluding the contribution of land use) are due to agricultural production whereas the reaming 55% pertains to the downstream stages of the supply chain. With the share of 12% of the GHG emissions each, food manufacturing and transport are the main contributors, followed by food preparation at home (9%), packaging (7%), retail (7%), catering (6%) and waste disposal (2%).

It has been estimated that we are using about 34% of the global land area, excluding Antarctica and Greenland, for agricultural purposes (Ramankutty et al., 2008) and it represents the largest use of land on the planet (Foley et al., 2011). In addition to causing GHG emissions, changes in land use from natural ecosystems into agricultural or urban areas are among the main causes of biodiversity loss, which, being estimated to be 100 to 1000 times higher than what could be considered natural, have already reached a dangerous level (Rockström et al., 2009). Díaz and colleagues (2006) highlighted that this process represents a threat for human well-being. Indeed, we clearly benefit from the diversity of organisms for the production of medicines, food, fibres and other renewable resources, for the access to water and other basic materials and for the capability of facing environmental changes. Furthermore, unsustainable management of agricultural activities could lead to accelerated soil erosion with negative consequences for soil fertility and quality (Lal, 2008).

Through the production of fertilisers and the cultivation of pulses, agriculture is responsible of converting about 120 million tons per year of nitrogen from molecular form (N_2) into other reactive nitrogen forms, such as nitrous oxides which contributes to global warming, nitrates which pollute rivers and lakes and ammonia and nitrogen oxides that are an atmospheric pollution source. The amount of converted nitrogen because of anthropic activities is considerably above the combined effects of all Earth's terrestrial processes (Rockström et al., 2009).

In addition, with up to 70 percent of the water we take from rivers and groundwater going into irrigation, the agricultural sector is the largest and often one of the most inefficient users of water and, at the same time, a major source of water pollution. (FAO, 2012).

All these elements not only make agriculture a driver of the above mentioned environmental pressures, but also affect the viability of the agricultural sector itself, influencing, therefore, the entire food system. Indeed, it has been demonstrated that climate change affects crop yields and the environmental performance of cultivation (Lobell and Field, 2007; Niero et al., 2015), that the over-exploitation of the soil decreases its quality and fertility and, therefore, its productivity (Lal, 2008), that climate change causes more and more frequently severe draught stress which negatively affect agricultural production (Pachauri et al., 2015). These are only some examples of

the complex cause-effect relationships that involve the food system and the environment, based on a complex balance involving several factors.

1.2   The challenge of a sustainable food system

According to the United Nations projections, the global population is expected to increase importantly in the next years, reaching almost 10 billion inhabitants by 2050 (United Nations - Department of Economic and Social Affairs, 2015) and, at the same time, making the demand for food growing. Moreover, the access to food is characterised by a strong paradox: while 800 millions of people suffer from chronic undernourishment and do not get a sufficient amount of food and nutrients for an active and healthy life (FAO, 2014a), we are facing worldwide an increase in food-related health problems as cardiovascular diseases, obesity, and diabetes because of rich foods, modern diets, sedentary lifestyles, and overeating (IHME, 2016).

Policy makers are increasingly aware that they have to face the double challenge of guaranteeing food security, namely the production of a suitable quantity of food to satisfy the nutritional needs of the growing population and, at the same time, promote the adoption of adequate dietary patterns, reducing the onset of diseases related to unbalanced nutritional patterns. Furthermore, as illustrated in the previous paragraph, our food system exerts a considerable pressure on the environment which can potentially threaten the sustainability of the system itself, namely the capability of producing food for the current generation without offsetting the possibility of doing the same for future generations.

The food system is based on a complex set of relationships which involve socio-economic, nutritional, environmental and ethical elements (Figure 2). Therefore, in order to achieve the afore mentioned objectives, a holistic approach that combines the attention for all these elements should be applied (Kearney, 2010). Within the scientific literature, different actions are suggested to implement a sustainable food system. De Laurentiis et al. (2016) and Foley et al. (2011) agreed that the transition towards a sustainable food system should be based on i) the adoption of sustainable primary production methods, ii) changes in diets composition, reducing the intake of animal-origin foods and iii) a decrease of the amount of food that is lost and wasted within the supply chain.

As far as the first pathway is concerned, being responsible of land use, soil degradation, water consumption, eutrophication, biodiversity loss and the introduction of hazardous chemicals in the environment, agriculture is a major source of environmental impacts (Reisch et al., 2013). Stopping the expansion of agriculture would be beneficial for biodiversity, carbon storage and important environmental services without compromising the increasing demand for food. Indeed, Foley and colleagues (2011) argued that the production benefits of tropical deforestation

are often limited especially if compared with its environmental burdens. Therefore, to guarantee an adequate food production without expanding the agricultural soil, it would be essential to optimise the agricultural yields with innovative approaches – based on the organic systems and precision agriculture - that do not further degrade the soil and, at the same time, limit the environmental pollution. Moreover, the pathway towards a sustainable food system should encompass the increase of crop productivity and the optimisation of the use of resources, such as water, nutrients and chemicals. Indeed, despite fertilisers and chemicals have played a major role in agricultural intensification, nowadays the distribution of nutrients is not balanced, with certain areas of the world characterised by water pollution due to nutrients excess and other characterised by nutrients deficiency.
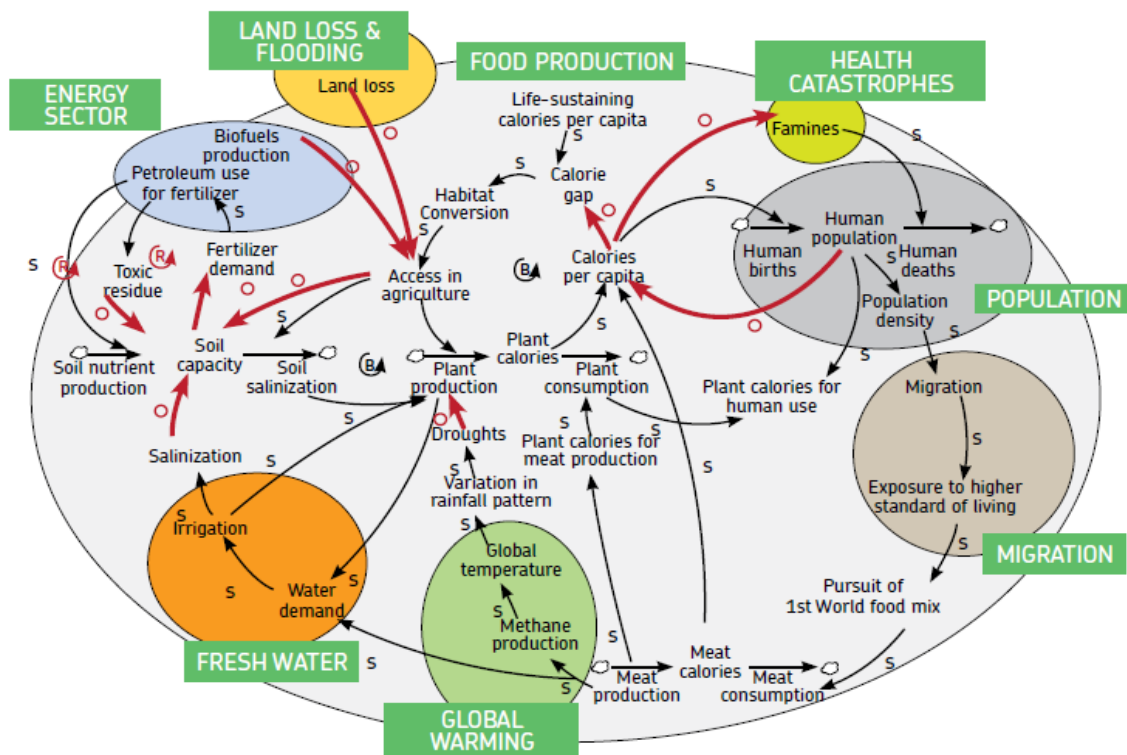
Figure 2: Model developed by IAASTD[1] on the complex relationships that characterise the agricultural system (S=Same; O=Opposite; R=Reinforcing; B=Balancing) (BCFN, 2011)

In addition, a potential increase of food availability is expected to be associated with the shifting of crop to human consumption, instead of being used for livestock feed, bioenergy crops and other non-food applications. Particularly, it has been estimated that shifting 16 major crops to 100% human consumption would increase by about 30% food availability and that other minor

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[1] International Assessment of Agricultural Knowledge, Science and Technology for Development of the United Nation

changes in diets, such as shifting grain-fed beef consumption to poultry and pork pasture-fed, could increase food availability while reducing the environmental impacts of agriculture.

Finally, the reduction of food loss and waste within the supply chain has the potentiality to enhance food availability while reducing the environmental impact of food production. Indeed, FAO (2011) has estimated that about 30% of food produced worldwide is not consumed by humans and this represent an enormous waste of resources. Although agronomists suggest that a food supply of 130% of our nutritional needs is necessary to guarantee food security, the current food production in affluent countries is by far above this threshold, representing, de facto, a threaten for food security (Papargyropoulou et al., 2014).

1.3 Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA) to improve the environmental performance of the food system

Life Cycle Assessment (LCA) – based on the life cycle thinking (LCT) approach - *addresses the environmental aspects and the potential environmental impact throughout a product's life cycle, from raw material acquisition, through production, use, and-of-life treatment, recycling and final disposal* (ISO, 2006a). The assessment of the potential impact through LCA is based on a four-steps, iterative procedure (Figure 3).
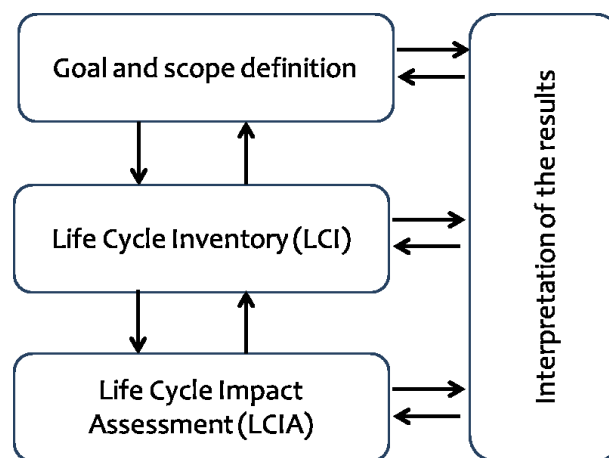
Figure 3: Steps of a LCA, according to ISO (2006b)

The definition of goal and scope consists in the description of the product system under analysis, including the definition of system boundaries and functional unit, to which the results of the study are referred and the description of the reasons why the study is carried out. Furthermore, in this first step of the LCA, other important elements, such as the allocation criteria, should be defined. The life cycle inventory (LCI) is the step in which data collection is planned and realised. It consists in the definition and quantification of all input and output flows that enter and exit the product system. When available, primary data collected directly from the actors of the supply chain should be preferred. If this is not possible, secondary data reported in databases and

literature o tertiary data, e.g. from estimates, can be used. The life cycle impact assessment (LCIA) step includes the characterisation of the results, namely the quantification of the potential environmental impacts associated with resource used and emissions generated within the supply chain. This is obtained multiplying each input and output flow by a characterisation factor that expresses the extent to which a certain substance contributes to a certain environmental impact or impact category. For each impact category, the characterisation factors are defined through the application of characterisation models that should be scientifically and technically valid and link the substances to their potential impact, on the basis of distinct, identifiable environmental mechanisms or reproducible empirical observations (EC, 2010). The final step is the interpretation of the results in order to satisfy the aim or the aims of the study.

LCT and LCA have a central role in supporting the definition of a sustainable food system. Indeed, through their holistic approach, they allow to avoid the shifting of the environmental burdens between different environmental compartments and stages of the supply chain (Sala et al., 2017). LCT and LCA are commonly applied to agro-food products in different contexts, e.g. business and policy making, and with different purposes, such as support to strategic decisions to improve their environmental performance and environmental communication.

LCA is internationally ruled by the standards ISO 14040 (ISO, 2006b) and 14044 (ISO, 2006a), which define the general principles of LCA, encompassing several fields of potential application. The principles of LCT and LCA are at the basis of numerous methods and schemes for measuring and benchmarking the environmental performance of products and services. Within these schemes the general principles of LCT and LCA are inflected taking into consideration the specificities of the field of application. By way of example, the development of product category rules (PCRs) is a requisite of the type III-environmental declarations (ISO, 2006c). The main aims of PCRs and other specific guidelines are to guarantee a transparent calculation procedure and the comparability of the results between different studies and to harmonise the communication of the environmental performance of products (Schau and Fet, 2008). However, the existence of different schemes for the assessment of the environmental performance of products and services has led to an extremely varied framework. Such situation represents a limiting element to the spread of products and services with reduced environmental impact both from companies' and consumers' points of view. Indeed, on one hand the existence of several environmental schemes can represent an additional cost and a market barrier for companies who wish to sell their products and services in different countries. On the other hand, surveys have shown that the purchase of "green products" is often limited by a lack of consumers' trust

towards environmental claims and by the absence of comparable information on the environmental performance of products (EC, 2013a).

With the aim of fostering the construction of a single market for green products, overcoming the aforementioned limiting elements, in 2010 the Council of the European Union called on the Commission to develop a harmonised method for the calculation of the environmental footprint of products. Such initiative has led to the definition of two methods for the calculation of the environmental footprint of products (PEF) and organisations (OEF) which require the definition of product category rules (EC, 2013b).

Currently, the European Commission is leading an environmental footprint pilot phase[2] which involve several stakeholders with the aims to test i) the process of developing the product and organisation category rules, ii) the approach to verification and iii) the possible communication vehicles for the environmental performance of products and organisations. This initiative encompasses several productive sectors and particularly a large focus is on the food sector. Indeed, almost half of the pilots are focused on food products or contexts related also to food distribution, such as the retail.

In addition, from 2012 to 2016 the United Nation Environmental Programme (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC) lead the third phase of the Life Cycle Initiative, whose aim was to enhance the streamlining of the LCA approaches in three areas: i) data, methods and product sustainability information; ii) capability development and implementation and iii) communication and stakeholder outreach. For each area, there were one or more working group dealing with specific different aspects and, thanks to this initiative, reference documents on the definition of a shared approach for LCA have been published.

1.4   Critical elements of LCA: an overview

In the LCA community there is a general consensus that, despite the efforts that have already been made, some elements of LCA, e.g. related to methodological choices, still need to be further defined, harmonised or improved to ensure the reliability and effectiveness of LCA in supporting strategic decisions and environmental communication both in business and policy-making contexts (Guinée et al., 2011; Hellweg and Milà i Canals, 2014; Nemecek et al., 2016; Notarnicola et al., 2016; Reap et al., 2008).

As far as food products are concerned, several authors have highlighted the need of broadening the system boundaries including all the phases of the supply chain, from cradle to grave (Nemecek et al., 2016; Sala et al., 2017). Indeed, despite the fact that agricultural production is generally the main environmental concern of the food supply chain, other stages can represent an

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[2] Internet website *ec.europa.eu/environment/eussd/smgp/policy_footprint.htm*

important share of the environmental burden and their contribution depends on the type of food (Pernollet et al., 2017). In addition, Nemecek et al. (2016) highlighted that, when assessing the environmental impact of a diet, consumer behaviour can strongly affect the results. In particular food waste generation, the mode of preparing food, the amount of food cooked and food storage modalities and time can largely influence the environmental performances of food.

The choice of the functional unit is another critical element of LCA applied to food products, since it importantly influences the results (Salou et al., 2016). The use of mass or volume functional units is a common practice in LCA (van der Werf and Salou, 2015), however some scientists claims for functional units which take into consideration the functionalities and the value of food. Particularly, Heller et al. (2013) suggested that, although food has several functions, nutrition should be considered the main one, therefore the functional unit should be nutritionally based. Sonesson et al. (2017) proposed a functional based on the digestible intake of nine amino-acids, whereas van der Werf and Salou (2015) supported the use of an economic functional unit, particularly for environmental communication through labels.

The generation of food loss and waste, namely the food that a certain point of the supply chain is diverted from human consumption, is a considerable environmental concern particularly due to the environmental impacts associated with food over-production (Nemecek et al., 2016). Its environmental burden, however, is often omitted in LCA of food products, leading to an underestimation of their impacts (Bernstad et al., 2016). Furthermore, a shared approach to account for food loss and waste generation and management in LCA is currently missing (Bernstad et al., 2016; Laurent et al., 2014). The accounting of food loss and waste is important not only when assessing the environmental performance of food, but also when considering innovative solutions to reduce packaging in which the risk is to offset the benefits associated with packaging reduction by an increase of food loss and waste (Nemecek et al., 2016).

LCA is characterised by noteworthy uncertainty and variability. Uncertainty is related to a limited knowledge of scenarios, parameters and models. Variability, instead, describes the differences between different processes or products and interests the spatial, the temporal and the technological dimensions (Huijbregts, 1998). Dealing with such uncertainty and variability is one of the hard challenges that LCA practitioners are called to face.

In particular, being able to capture the variability among different production systems is of the utmost importance in the agricultural field (Notarnicola et al., 2016). Indeed, a large part of the environmental impacts of agricultural production is associated with direct field emissions. These emissions are generally estimated through models defined to be used at a larger scale, such as the European Environment Agency (2013) and the IPCC (2006) guidelines, and therefore are not

always enough detailed to capture the differences among production systems (Cederberg et al., 2013).

Further improvements are needed also as far as the LCIA is concerned. Agricultural production is responsible of impacts on soil fertility, hydrology, structure and biodiversity. However, these impact are generally omitted in LCA and a shared approach to account for them does not currently exist (Notarnicola et al., 2016). In addition, currently used LCIA methods show some shortcomings. Impact categories referred to phenomena relevant at the regional scale, e.g. water use, are often based on general characterisation models that do not take into consideration the characteristics of the specific area where they happen, with the risk of limiting their meaningfulness (Hellweg and Milà i Canals, 2014). Therefore, spatially differentiated characterisation models need to be adopted within the LCA field.

Another critical element highlighted by Notarnicola et al. (2016) is the inconsistency between LCI modelling and LCIA that currently characterise, for example, the impacts on eutrophication and toxicity. Indeed, often the LCI includes emissions in the environment that happen after a partial degradation of the substance. Since LCIA models account for emissions, there is the potential risk to omit part of the environmental impact that takes place before the degradation of the substances.

Finally, some scientists call for broader perspectives of LCA, which on one side include the social and the economic sustainability assessment and, on the other, encompass not only the product dimension but also the productive sector and the economy level (Guinée et al., 2011; Reap et al., 2008).

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2. Outline of the thesis and research questions

The present thesis includes the results of the research activities carried out between January 2014 and December 2016 at Institute of Agricultural and Environmental Chemistry of Università Cattolica del Sacro Cuore in Piacenza and at Institute for Environment and Sustainability (IES) of Joint Research Centre (JRC) in Ispra.

The overall aim of the present thesis is to contribute to the ongoing debate on the harmonisation of the approaches to carry out a LCA of agro-food products and to find ways to improve and overcome the critical elements that currently characterise the application of LCA in the agro-food sector.

Particularly, **Chapter 3** aims to provide an overview of secondary datasets modelling arable crops production in order to support LCA practitioners in the choice of datasets and in the interpretation of the results, and, to certain extent, to provide indications to database developers on how to improve datasets modelling. It is based on the paper "Systematic analysis of secondary Life Cycle Inventories for modelling agricultural production: a case study for arable crops", accepted for publication by Journal of Cleaner production and currently in press. It consists in an analysis of the modelling approaches and underlying assumptions of twelve secondary datasets modelling four arable crops and belonging to three databases (ecoinvent®, AGRIBALYSE® and Agri-footprint®). The analysis highlighted important differences among the modelling approaches adopted within the databases and, therefore, recommendations are drawn from the datasets comparison, supporting the selection of the datasets coherently with the goal and scope of a study and interpretation of results.

The contents of **Chapter 4,** based on the paper "Modelling of food loss within life cycle assessment: From current practice towards a systematisation", published Journal of Cleaner Production, aim to contribute to the definition of a systematic approach to include food loss and waste in LCA, which is currently missing. The commonly adopted practices to account for food loss and waste in LCA studies have been analysed and some considerations and recommendations for LCA practitioners have been derived accordingly.

In **Chapter 5**, based on the paper "Influence of personal behaviour on the greenhouse gas emissions of three balanced dietary patterns", submitted for publication to the Journal of Cleaner production and currently under review, the GHG emissions of three balanced dietary patterns for an Italian man have been analysed. The study highlighted the importance of considering the consumption phase including its variability, particularly when comparing the environmental burdens of different dietary patterns and the central role of consumers in defining the overall GHG emissions generated by the diets. Moreover, it demonstrated that particularly careless choices can offset the lower GHG emissions associated with the consumption of vegetable-origin food.

Finally, the conclusions of the present thesis and potential future research proposals are reported in **Chapter 6**.

2.1 Research questions

In light of the considerable pressure exerted by the food system on the environment, the present thesis originated from the need of the scientific community and policy makers of having a robust and reliable tool to assess its environmental burdens.
The overall aim of the thesis is to contribute to the ongoing debate on the harmonisation of the approaches in LCA and, particularly, the thesis faces the following three research questions.

A. To which extent can methodological choices and modelling approaches influence the LCA results?
B. Which are the potentialities for improving the harmonisation of modelling approaches in LCA applied to the agro-food sector in order to foster its reliability and robustness?
C. Which is the influence of the consumption phase – often omitted in LCA on dietary patterns – on the GHG of dietary patterns?

Questions A is discussed in the Chapters 3, 4 and 5 from different perspectives, namely the choice of secondary datasets, the modelling of food loss and waste and the assessment of the environmental burden of different dietary patterns. Chapters 3 and 4 deal with Question A. Chapter 5, instead, regards Question C.

3. Analysis of secondary datasets modelling agricultural production

Abstract

Analysis of agricultural production with life cycle based methodologies is data demanding. To build comprehensive life cycle inventories, secondary datasets are commonly used when primary data are not available. However, different inventory data and modelling approaches are used to populate secondary datasets, leading to different results.

The present study analyses the features of twelve secondary datasets to support datasets selection and proper interpretation of results. We assess twelve datasets for arable crop production in France, as modelled in three databases often used in the LCA field (Agrifootprint, Ecoinvent and Agribalyse). First, we compared system boundaries and general assumptions. Second, we focused on foreground systems comparing, inventory data, data sources and modelling approaches. Third, we performed a contribution analysis of impact assessment results to identify modelling choices that contribute most to differences in the results. Nine relevant elements were identified and assessed: definition of system boundaries and modelling of agricultural practices, characteristics of inventory data, agricultural operations, fertiliser application and fate, plant protection products application and fate, heavy metals inputs to the agricultural system and fate, irrigation assumptions, land use and transformation. The datasets differ greatly with respect to these elements. Hence, recommendations are drawn from the dataset comparison, supporting the selection of the datasets coherently with the goal and scope of a study and interpretation of results.

1. Introduction

The assessment of the environmental profiles of food supply chain is increasingly needed in the context of sustainable production and consumption initiatives. The aim is to identify drivers of environmental impacts associated with food production and possible improvements thereof. Life Cycle Assessment (LCA) is a reference methodology for supply-chain impact assessment (ISO, 2006a). However, when the subject of the study is a manufactured product (e.g. a food product), data on agricultural stages of basic ingredients (e.g. wheat) are often not collected directly, relying instead on "secondary data" (Williams et al., 2009). This approach helps streamline estimation of the product's environmental profile (Teixeira, 2015), reducing the resources required to collect data and allowing an LCA to be performed when the necessary life cycle inventory data are not available from primary sources.

The choice of the secondary dataset to be used is considered one of the challenges for a robust LCA study (Notarnicola et al., 2016) and can influence the results of the LCA study (e.g. Peereboom et al., 1998, found out a variation of impact results from 10% to 100% when different datasets were used in a case study on PVC). Indeed, different modelling assumptions in datasets aiming to represent the same product system can lead to different results, affecting the reliability of the LCA study (Williams et al., 2009). LCA practitioners are, therefore, recommended to choose datasets carefully according to the goal and scope of their studies (Fazio et al., 2015).

Several authors have already analysed secondary data from different points of view: (i) developing criteria for assessing data quality (e.g. Garraín et al., 2015; Grabowski et al., 2015); (ii) estimating influence of dataset quality on life cycle impact assessment (LCIA) results (Peereboom et al., 1998); (iii) proposing approaches based on a descriptive and statistical analysis to assess reliability of secondary data used in LCA (Teixera et al., 2015) (iv) adopting meta-analysis to estimate average values of environmental impacts (e.g. Achten and Van Acker, 2015). However, a systematic analysis of secondary dataset aiming at identifying commonalities and differences in the underpinning modelling approach has not been performed so far, to authors' knowledge. Hence, the present study provides an analysis of secondary datasets of arable crop production, based on the approach adopted by Peereboom et al. (1998) with some adaptations for the agricultural context. It aims to understand commonalities and differences in datasets of arable crop cultivation and the extent to which the differences may affect LCIA results.

We identified and analysed the elements in the different datasets which may influence the most the LCA results, as well as the strengths and weaknesses of the modelling approaches adopted. Results of the present study could help LCA practitioners choosing secondary datasets

consistently with the goal and scope of their study and carrying out a proper interpretation of the results. Furthermore, the results may inform dataset developers about the need for potential improvements to, for example, modelling approaches and underlying assumptions on which datasets were built.

The article is organised as follows: first, a review of system boundaries and underlying assumptions adopted in secondary datasets for arable crops within three databases is presented. Second, a summary of the approaches adopted to model the foreground system, is complemented by highlighting similarities and differences among the approaches. Next, the influence that modelling approaches can have on LCIA is illustrated. Finally, combining these elements conclusions about relevant elements of datasets are provided.

2. Materials and methods

The present study is focused on analysis of secondary datasets for arable crop production as modelled in three of the most commonly used LCA databases: AGRIBALYSE® v 1.2 (Colomb et al., 2015), Agri-footprint® v 1.0 (Blonk Agri-footprint BV, 2014a) and ecoinvent® v 3.1 (Weidema et al., 2013).

Four arable crops for which the cultivation in France was included in all three databases were selected for analysis: wheat, barley, rapeseed and pea (Table 1). As AGRIBALYSE includes both spring pea and winter pea, the average of the two was considered in the analysis.

Table 1: Databases assessed in the study., FR = France, U= unit process, Alloc def= ecoinvent default allocation

Database	Dataset
AGRIBALYSE v 1.2	Soft wheat grain, conventional, national average, at farm gate/FR U Barley, conventional, malting quality, national average, at farm gate/FR S Rapeseed, conventional, 9% moisture, national average, at farm gate/FR U Winter pea, conventional, 15% moisture, at farm gate/FR U Spring pea, conventional, 15% moisture, at farm gate/FR U
Agri-footprint v 1.0	Wheat grain, at farm/FR Barley grain, at farm/FR Rapeseed, at farm/FR Pea, at farm/FR
ecoinvent v 3.1	Wheat grain {FR} | wheat production | Alloc Def, U Barley grain {FR}| barley production | Alloc Def, U Rape seed {FR}| production | Alloc Def, U Protein pea {FR}| production | Alloc Def, U

The selection of the country of production and the crops was based on the highest number of available comparable datasets. As ecoinvent includes not only attributional datasets but consequential datasets and datasets based on the so-called "cut-off system model" approach, whose underlying philosophy is that primary production of materials is always allocated to the primary user of a material (ecoinvent, 2016), differences in the LCIA results of these three

modelling approaches were screened. As few differences were found (supplementary material, Figures S6-S9) and inclusion of consequential and "cut-off" datasets would have rendered the comparison too complex, due to the use of different modelling approaches, only ecoinvent's attributional datasets (the default) were analysed.

Datasets were analysed based on information reported in dataset documentation, data provided in the databases as implemented in the software SimaPro v 8.0.5 and certain other relevant publications (Frischknecht and Rebitzer, 2005; Nemecek et al., 2014).

We considered a generic representation of an agricultural production system and distinguished foreground and background systems when analysing datasets (Figure 1). System boundaries and underlying assumptions of each dataset were compared referring to this diagram.

* Plant Protection Products
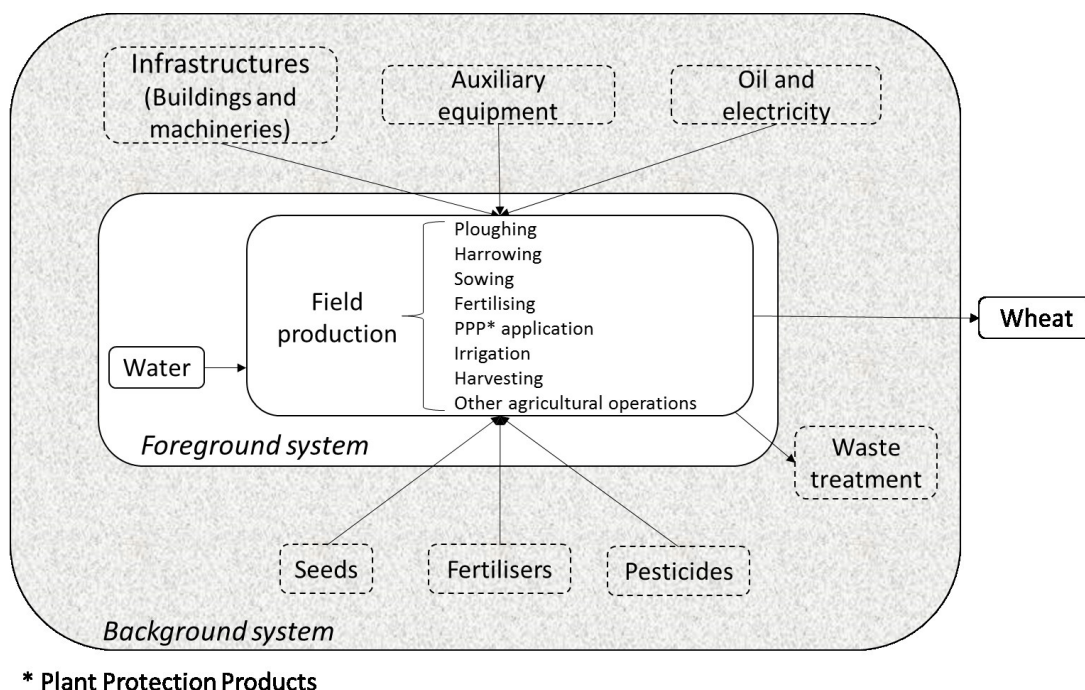
Figure 1. Representation of agricultural production systems. Adapted from Hayashi et al. (2006).

The foreground system was examined by describing the assumptions adopted to model it, highlighting similarities and differences among datasets per hectare of cultivated land. For wheat and barley, all impacts of cropping were allocated to the grain (none to the straw) to allow results to be compared.

LCIA was conducted for the three datasets using the ILCD Midpoint v 1.06 characterisation method (EC-JRC, 2012) as implemented in the software SimaPro v. 8.0.5, used for calculations and analysis. We assessed the potential impacts of 1 kg of product at the farm gate using ILCD v 1.06 (EC-JRC, 2012). As for the foreground system analysis, the effect of allocation of the potential impacts between co-products was removed from the inventories. Furthermore, since

Agri-footprint includes datasets modelling the same product with different allocation approaches, a screening of the effect of allocation on LCIA was performed.

We performed three types of analysis:

- comparison of system boundaries and underlying assumptions

- analysis of how the foreground system is modelled, focusing on agricultural operations, fertiliser application and nutrient fate; plant protection product (PPP) application and fate; heavy metal (HM) input, mass balance and fate; irrigation; land occupation and transformation

- comparison of LCIA results of the foreground system, including the relative contribution of the background system.

When comparing inventory data, uncertainty data provided within the datasets were taken into account, and differences among data were considered statistically significant when 95% confidence intervals did not overlap.

The 95% confidence intervals were estimated using Monte-Carlo simulation with 500 replicates.

3. Results

One substantial difference among the datasets is the source of activity data from which the inventories were developed. AGRIBALYSE and Agri-footprint data were derived from average information for the French context, collected respectively using questionnaires distributed to technical institutes and from available statistics or other specific data (e.g. the literature). In contrast, ecoinvent datasets were built from data collected for a single French region, Barrois, in the GL-Pro project (Nemecek and Baumgartner, 2006).

Database providers checked the quality of activity data. For AGRIBALYSE and Agri-footprint, the quality check was performed by experts not directly involved in defining the inventory data, and quality was analysed at two levels: plausibility of activity data and presence of data gaps or errors in the LCIA and LCA results (Blonk Agri-footprint BV, 2014a; Koch and Salou, 2013). Ecoinvent, datasets, instead, were independently reviewed before integration in the ecoinvent database. Data quality was assessed based on the pedigree-matrix approach (Frischknecht and Rebitzer, 2005, Weidema et al., 2013), which, by qualitatively assessing data quality indicators, is applicable when only a single mean value for activity data is available (Frischknecht and Jungbluth, 2007). The pedigree matrix considers information about the quality of each primary input and output datum in terms of reliability, completeness, temporal correlation, geographical correlation and further technological correlation. The scores obtained with the pedigree matrix in ecoinvent are reported also in AGRIBALYSE, when datasets from ecoinvent are used. Uncertainty in activity and inventory data cannot be avoided due to variability and stochastic

errors in activity data, appropriateness of input and output flows, model uncertainty and the exclusion of important flows (Frischknecht and Jungbluth, 2007). Therefore, a basic uncertainty is reported in ecoinvent for "unit process" datasets.

In AGRIBALYSE, barley production is reported only as a "system process" with no information about uncertainty. In Agri-footprint, uncertainty is estimated only for certain categories of background data, mainly according to expert knowledge (Blonk Agrifootprint BV, 2014a).

The lack of information about uncertainty partially influenced analysis of each dataset's foreground.

3.1 Analysis of system boundaries and underlying assumptions

System boundaries and the main underlying assumptions adopted to model arable crop production differ somewhat among the three databases (Figure 2, Table 2.).
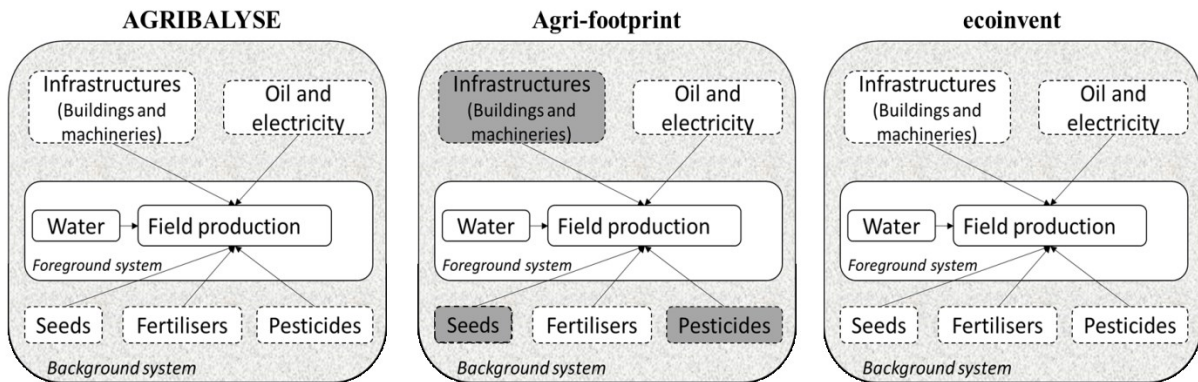

Figure 2: System boundaries of the datasets considered in the study. White boxes represent processes included in inventories, whereas grey ones represent processes not included in inventories.

It is evident that, within a given database, the same modelling approach is adopted for the arable crops analysed. Furthermore, databases have several similarities in how they model agricultural systems, probably because they were developed with knowledge of one another. In particular, AGRIBALYSE background data were taken from ecoinvent v 2.2, which explains some similarities between the two databases. However, differences among the databases were observed. Concerning system boundaries, production and maintenance of infrastructure and machines are excluded from Agri-footprint datasets because they generally represent a negligible contribution to the LCA results, and seed and pesticide production are excluded due to a lack of data at the time when the database was released (grey boxes in Figure 2), (Blonk Agri-footprint BV, 2014a) whereas they are included in AGRIBALYSE and ecoinvent. Concerning modelling assumptions, the main differences are related to allocation of co-products and allocation of emissions from nutrient input (with reference to crop rotation). The way in which co-products (grain and straw)

are allocated (or not) can influence results of the LCIA phase (supplementary material, Figures S10 and S11).

Table 2:. Underlying assumptions and modelling approaches adopted to build the datasets analysed, as described in their documentation.

		AGRIBALYSE	Agri-footprint	Ecoinvent
Data source		Provided by technical institutes (e.g. ARVALIS – Institut du Végétal)	Multiple sources (e.g. scientific literature, official statistics such as FAOstat, Eurostat)	GL-Pro project – Barrois region, France (Nemecek and Baumgartner, 2006)
Straw management (when applicable)		Partly removed from the field	Completely removed from the field	Left on the field
Allocation of co-products (grain and straw)		Not performed because the straw market was not well organised when the datasets were developed	Economic, mass and energy allocation	Not applicable because straw is assumed to be left on the field
Nutrients from straw left on the field		Fertilising effects of crop residues and emissions from the residues are allocated to the crop that generated the residues	Not applicable	Fertilising effects of crop residues are allocated to the crop that generated them (only for P and K). The amount of fertilisers is corrected for the amount of nutrients in crop residues. A description of the allocation of the emissions from the crop residues is lacking in the database report, therefore they are probably allocated to the crop that generated them.
Crop rotation modelling	**Phosphorus (P) and potassium (K) input and emission allocation**	P and K fertiliser production and emissions due to their application are allocated to each crop proportional to crop exports	Not reported	P and K supplied to the field in residues are allocated to the crop that generated them
	Nitrogen (N) input and emission allocation	Organic N available for the crop to which the fertiliser is applied is allocated to that crop. The remaining fraction that increases the stock of organic matter is allocated to all crops in the rotation. Mineral N is allocated completely to the crop to which it is applied.	Not reported	Allocation not performed

Foreground system		
Fertilisers application	→	NH_3, NO_x, N_2O to atmosphere NO_3^-, PO_4^{3-}, P, HM[a] to water HM to soil P to soil
Seeds use	→	HM to soil HM to water
Plant Protection Products (PPPs) application	→	Active ingredient (AI)[b] to soil HM to water
Agricultural activities	→	CO_2[c], NMVOC, PM, NO_x […][d] to atmosphere Cd, Pb, Zn to soil Land occupation and land transformation Water use

[a] HM = Heavy metals (cadmium (Cd), copper (Cu), zinc (Zn), lead (Pb), nickel (Ni), chromium (Cr) and mercury (Hg))

[b] In all the three datasets AI are assumed to end up entirely in the soil

[c] CO_2 emissions are due to fuel combustion and land transformation

[d] Other atmospheric emissions from combustion are considered within the datasets, only the most contributing to the impact categories considered are reported
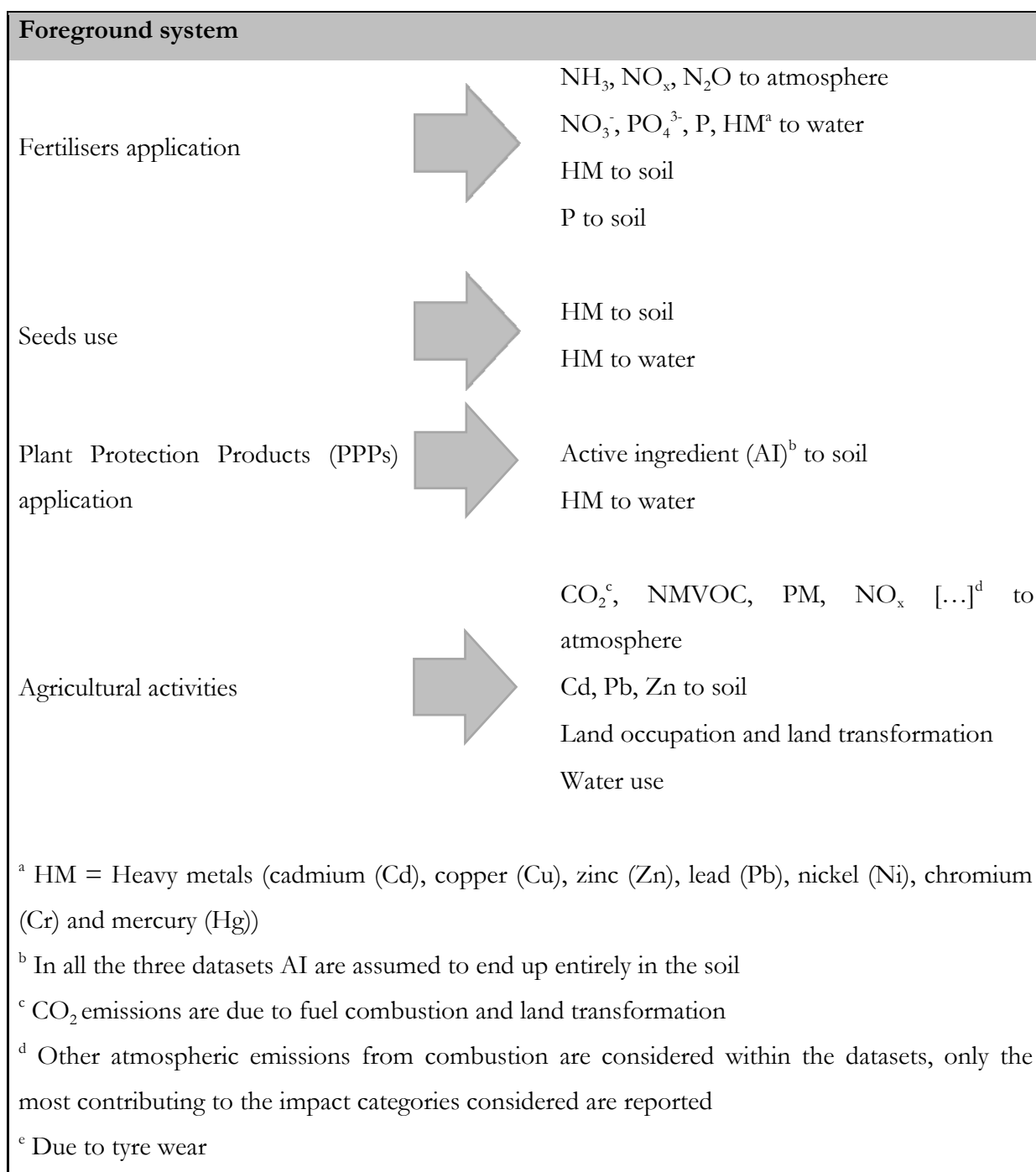
[e] Due to tyre wear

Figure 3. Relation between field activities and environmental emissions and use of resources deduced from analysis of all datasets. Not all activities are considered in all datasets

3.2 Foreground system analysis

Mean crop yields differed slightly among databases (supplementary material, Figure S1). Only Agri-footprint defines 95% confidence intervals of yields. Relations between field activities and environmental emissions and use of resources were deduced from analysing the datasets (Figure 3).

The databases modelled agricultural operations for arable crops differently (Table 3).

Table 3:. Decisions made to model agricultural operations in the three databases

	AGRIBALYSE	Agri-footprint	ecoinvent
Source of data	Operating time and fuel consumption: technical institutes	Not reported	Use of machinery from the GL-Pro project (Nemecek and Baumgartner, 2006)
Temporal and geographical representativeness	2005-2009, France	Not reported	1991-2014 or 2011-2014, World average (GLO)
Reference unit for agricultural operations	Hours of work	Energy content of the fuel consumed	Area
Inclusion of emissions to soil due to tyre wear	Yes	No	Yes

Different reference units for agricultural operations did not allow them to be compared directly; however, we estimated databases' emissions factors (kg pollutant/kg diesel) and found differences among them due to differences in the modelling approaches adopted (supplementary material, Table S1). According to Koch and Salou (2015), describing agricultural operations as a number of hours of work in AGRIBALYSE is more flexible than the approach adopted in ecoinvent (operations described as areas) because it takes into account different amounts of time required to perform the same process (e.g. tilling different types of soil).

Temporal representativeness may strongly influence emissions factors of technology-related pollutants, such as particulate matter (EMEP/EEA, 2013); if machinery is assumed to be older than it really is, emissions of air pollutants due to fuel combustion can be overestimated.

Foreground emissions due to agricultural operations are related to fuel combustion and tyre wear. Fuel combustion includes the compounds emitted in the atmosphere during combustion, considered in the three databases: carbon dioxide (CO_2), carbon monoxide (CO), particulate matter (PM), ammonia (NH_3), nitrogen oxides (NO_x), methane (CH_4), non-methane volatile compounds (NMVOC) and sulphur dioxide (SO_2). Tyre wear, which emits cadmium (Cd), lead (Pb) and zinc (Zn) to the soil, is included only in AGRIBALYSE and ecoinvent.

3.2.2 Modelling fertiliser application and nutrient fate

Three elements characterise the modelling approach adopted for fertiliser application and nutrient fate: (i) amount of nutrients provided to the field, (ii) type of fertilisers used ("fertiliser mix") and (iii) models adopted for nutrient loss to the environment. Data sources for amounts of nutrients applied to the soil varied (Table 2.). Specifically, data used for Agrifootprint are derived from Feedprint reports (Vellinga et al., 2013), mainly based on personal communications (Blonk

Agri-footprint BV, 2014b). Amounts of phosphorus and potassium applied vary, and no nitrogen fertiliser is applied to pea in ecoinvent (supplementary material, Figure S2) Different kinds and amounts of fertilisers are included in the analysed datasets. AGRIBALYSE allocates fertilisers applied within a crop rotation, whereas Agri-footprint and ecoinvent do not appear to do so (Table 2).

Different kinds and amounts of fertilisers are included in the analysed datasets,. The "fertiliser mix" used in AGRIBALYSE reflects French statistics on fertiliser use from 2005- 2009 from UNIFA (French fertiliser industry association). Data in Agri-footprint come from international statistics from the International Fertilizer Industry Association (IFA) for 2012, and those in ecoinvent come from the GL-Pro project (Nemecek and Baumgartner, 2006). Fertiliser application emits nutrients to the environment in the form of nitrogen, phosphorus and potassium compounds and may also emit HMs (Brentrup et al., 2004). Furthermore, application of urea and lime generates emissions of CO_2 (IPCC, 2006).

The three databases do not consider emissions of potassium compounds; however, emissions of nitrogen and phosphorus compounds are included and estimated using different approaches (supplementary material, Table S2).

Emission factors for NH_3 emissions are similar for AGRIBALYSE and ecoinvent and are lower than those of Agri-footprint. Emission factors for N_2O and NO_3^- emissions are similar for all databases, except for pea in AGRIBALYSE, for which inexplicable higher emissions were observed. NO_x emissions differed significantly among the datasets (supplementary material, Figure S3).

NH_3 emissions are estimated for AGRIBALYSE and ecoinvent by considering characteristics of fertilisers, as indicated respectively by EMEP/EEA (2009) and the Agrammon model (Agrammon Group, 2009), which is also based on EMEP/EEA (2009) methodology (Nemecek et al., 2014). In contrast, for Agri-footprint, a rougher estimate is made, considering the emission factors for NH_3 that volatilises after mineral and organic nitrogen fertiliser application reported by IPCC (2006). Estimates for Agri-footprint are more conservative than those of AGRIBALYSE and ecoinvent (Figure S3a).

All databases estimate direct and indirect emissions of N_2O according to the same method (IPCC, 2006). Emissions factors per unit of nitrogen applied are the same for all databases, except for pea in AGRIBALYSE, for which the emission factor is 5% higher than the others (Figure S3b).

NO_x emissions are considered only in AGRIBALYSE and ecoinvent, each using a different modelling approach (respectively EMEP/EEA, 2009 and NOx emissions = 0.21×N2O emissions), which leads to higher emissions factors for AGRIBALYSE. However, in both

databases, NO_x emissions represent only a minor loss of the nitrogen applied to the field, lower than 1% (Figure S3c).

NO_3^- emissions are estimated in AGRIBALYSE according to a model specific to France that considers information about farming practices (e.g. residue management, use of intermediate crops, application of nitrogen fertilisers), crops in the rotation, soil properties and climatic conditions (Koch and Salou, 2015). In Agri-footprint, the average emissions factor reported for NO3 – emissions by IPCC (2006) is applied, whereas in ecoinvent the SALCA-NO3 model (Richner et al., 2014) is used. Anomalous emissions of NO_3^- were observed for pea datasets in AGRIBALYSE and ecoinvent: NO_3^- emissions are higher than the amount of nitrogen applied to the field. For the other crops, Agri-footprint had higher NO_3^- emissions factors than AGRIBALYSE, whereas no significant differences were observed between ecoinvent and AGRIBALYSE and ecoinvent and Agri-footprint (Figure S3d).

Three pathways are considered for phosphorus emissions: (i) leaching to groundwater, (ii) runoff and (iii) emission to surface water due to soil erosion.

AGRIBALYSE and ecoinvent estimate leaching and runoff using the SALCA-P model (Prasuhn, 2006), validated for Switzerland but not for France, that takes into consideration different parameters, such as soil characteristics and type of fertilisers (Nemecek, 2013), whereas Agri-footprint uses a fixed emissions factor (Blonk Agrifootprint BV, 2014b). AGRIBALYSE and ecoinvent include soil erosion as considered by Prasuhn (2006), whereas Agrifootprint does not include it due to limited data availability (Blonk Agri-footprint BV, 2014a).

CO_2 emissions from urea and lime application are included in all three databases, which use the same emission factors (IPCC, 2006).

3.2.3 Modelling PPP application and environmental fate

PPP use is modelled according to several data sources (Table 2). PPP application and fate modelling have large uncertainties. Indeed, Agri-footprint documentation emphasises using default data and suggests that dataset users modify the inventory with primary data whenever possible.

Many PPPs are included in the datasets, even though AGRIBALYSE and ecoinvent lack specific production process for some of them, for which they use average PPP production inventories.

Regarding PPP fate, all databases assume that 100% of PPP end up in agricultural soil after application; this is considered a highly controversial assumption when estimating the contribution of PPPs to toxicity impacts (Rosenbaum et al., 2015). Another problem is the representativeness of the PPPs included in the datasets. PPP use in France is subject to European Union legislation requiring that active ingredients be approved before being sold on the market (EU, 2009).

However, some active ingredients in the datasets, such as bitertanol (AGRIBALYSE and ecoinvent) and metolachlor (Agri-footprint), are no longer authorised in France (EC, 2015) (supplementary material, Table S3) and should be excluded from the inventory of any crop cultivation in France.

3.2.4 Modelling HM input, mass balance and fate

HM mass balance is performed in all datasets following the same principle (Freiermuth, 2006).

HMs emitted to the soil are calculated as the sum of all HMs that enter the agricultural system minus the sum of all HMs that leave it.

Estimates of HM flows to and from the soil are highly uncertain. In fact, some datasets for crop production estimate that more HMs leave the system than enter it, resulting in a net decrease of HMs in the soil. As emphasised by Koch and Salou (2013), these figures should not be interpreted as true removal of HMs from the soil, but rather as an effect of uncertainty in input and output data.

Atmospheric deposition and application of mineral and organic fertilisers represents the major sources of HM inputs to agricultural soil (Nicholson et al., 2003) and are considered in all datasets. In contrast, other HM sources, such as seeds and PPPs, are considered only in AGRIBALYSE and ecoinvent. Leaching and exportation in biomass are considered as removal mechanisms for HMs in the soil in all datasets, whereas runoff of HMs in eroded soil particles is included only in AGRIBALYSE and ecoinvent.

Among the three databases, different literature data are used to estimate amounts of HMs input to and removed from soil. HM removal due to leaching is estimated according to available average data, and all three databases use the same values.

Since specific data for France were not available, data for Switzerland were used in AGRIBALYSE (Koch and Salou, 2013). AGRIBALYSE and ecoinvent estimate soil erosion using the same equation (i.e. RUSLE); however, estimated soil HM content and amount of soil eroded differ (supplementary material, Table S5).

Another source of emissions of HMs to soil is tyre wear due to agricultural operations (see section 3.2.1).

3.2.5 Modelling irrigation

Irrigation volumes for a given crop vary greatly among the datasets (Figure 4), according to the data source: AGRIBALYSE data were collected from technical institutes, Agri-footprint data were taken from the "blue water footprint" of Mekonnen and Hoekstra (2010), and ecoinvent

data came from the literature (Doll and Zhang, 2010), with calculation performed at ETH, Zurich.

Ecoinvent has the highest irrigation volumes for each crop. Furthermore, different elementary flows are used to model irrigation: "Water, river" in AGRIBALYSE, "Water, unspecified natural origin, FR" in Agri-footprint, and "55% water, river FR 45% Water, well, in ground, FR" in ecoinvent. Differences in the types and locations of water sources included in the inventory may lead to large differences in predictions of water depletion after characterisation, especially if characterisation factors are spatially explicit.
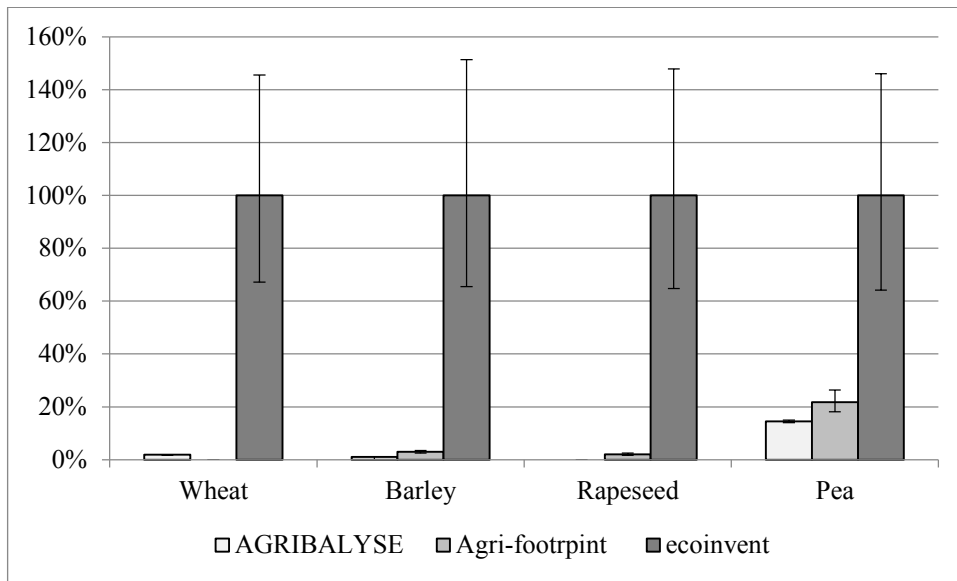

Figure 4. Relative mean water irrigation volumes per hectare of cultivated land and 95% confidence intervals. For each arable crop, the maximum irrigation volume is reported as 100% and the others are expressed as percentages of the maximum.

3.2.6 Modelling land occupation and transformation

All databases consider agricultural land occupation as $m^2 * y$, taking into account the duration of cultivation. Land transformation is modelled according to different approaches (Table 4). Ecoinvent assumes zero net land transformation because it considers that no land is transformed for arable crops in France (Nemecek et al., 2014). In contrast, AGRIBALYSE and, for wheat and rapeseed, Agri-footprint, include land transformation from natural areas, such as pasture and forests, or from permanent crops to agricultural land uses. Furthermore, AGRIBALYSE considers transformation from "discontinuously built urban" land uses to agricultural land uses. Land transformation may emit CO_2 due to organic carbon mineralisation. These emissions are included only in Agri-footprint, which estimated them using the Direct Land Use Change assessment tool, compliant with the PAS 2050-1 and European PEF methods (Blonk

Consultants, 2014), assuming that the previous land use was not known. AGRIBALYSE excludes GHG emissions from land use change due to lack of data about land occupation over time (Koch and Salou, 2013).

Table 1. Models and sources of data for land transformation

	AGRIBALYSE	**Agri-footprint**	**Ecoinvent**
Model	Frischknecht and Niels, 2007	Direct Land Use Change Assessment Tool (Blonk Consultants, 2014)	Milà i Canals et al., 2012 (see Nemecek et al., 2014)
Source of data	Teruti-Lucas, 2006	Direct land use change assessment tool based on data from FAOstat (FAO, 2012)	FAOstat (FAO, 2012) (see Nemecek et al., 2014)

3.3 Life cycle impact assessment results

In general, the choice of the database (and related dataset) used to model a given product can lead to different LCIA results (Figures 5 and S4). In some cases (e.g. toxicity-related impact categories and water depletion in ecoinvent), uncertainty in results from a given dataset is larger than differences in results among the databases. This high degree of uncertainty can affect interpretation of results and the ability to achieve the goal of the study. In other cases, results differ greatly even when considering the uncertainty and, for some impact categories, i.e. acidification and terrestrial eutrophication, this is particularly true when the analysis focuses only on the foreground system (Figures 6 and S5). This means that sometimes the contribution of background datasets can partly offset the differences between LCIA due to different modelling approaches. Figure 5 reports the LCIA for wheat, showing foreground and background contributions. LCIAs for the other arable crops are reported in the supplementary material (Figure S4). Only the impact categories which had a contribution of the foreground system are reported in Figure 5. and Figure 6. The impact categories ionising radiation; mineral, fossil and renewable resource depletion; and ozone depletion potential were excluded from the analysis because they were influenced only by the background system. The impact of the foreground system on land use is reported in Figure 7.
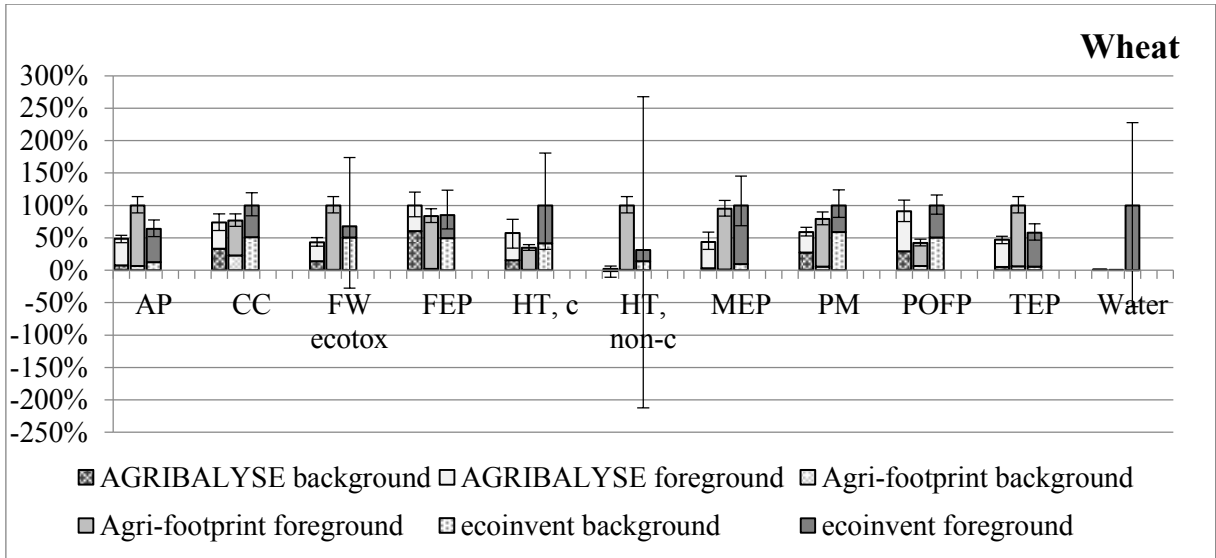
Figure 5: Relative mean foreground and background system contributions to LCIA and 95% confidence intervals per kg of wheat grain. For each impact category, the largest value is reported as 100% and the others are expressed as percentages of the maximum. The following impact categories were considered: acidification (AP), climate change (CC), freshwater ecotoxicity (FW ecotox), freshwater eutrophication (FEP), human toxicity-cancer (HT, c), human toxicity- non cancer (HT, non-c), marine eutrophication (MEP), particulate matter (PM), photochemical ozone formation (POFP), terrestrial eutrophication (TEP), water resource depletion (Water)

Figure 6: Relative mean foreground system contributions to LCIA and 95% confidence intervals per kg of wheat grain. For each impact category, the largest value is reported as 100% and the others are expressed as percentages of the maximum. The following impact categories were considered: acidification (AP), climate change (CC), freshwater ecotoxicity (FW ecotox), freshwater eutrophication (FEP), human toxicity-cancer (HT, c), human toxicity- non cancer (HT, non-c), marine eutrophication (MEP), particulate matter (PM), photochemical ozone formation (POFP), terrestrial eutrophication (TEP), water resource depletion (Water)
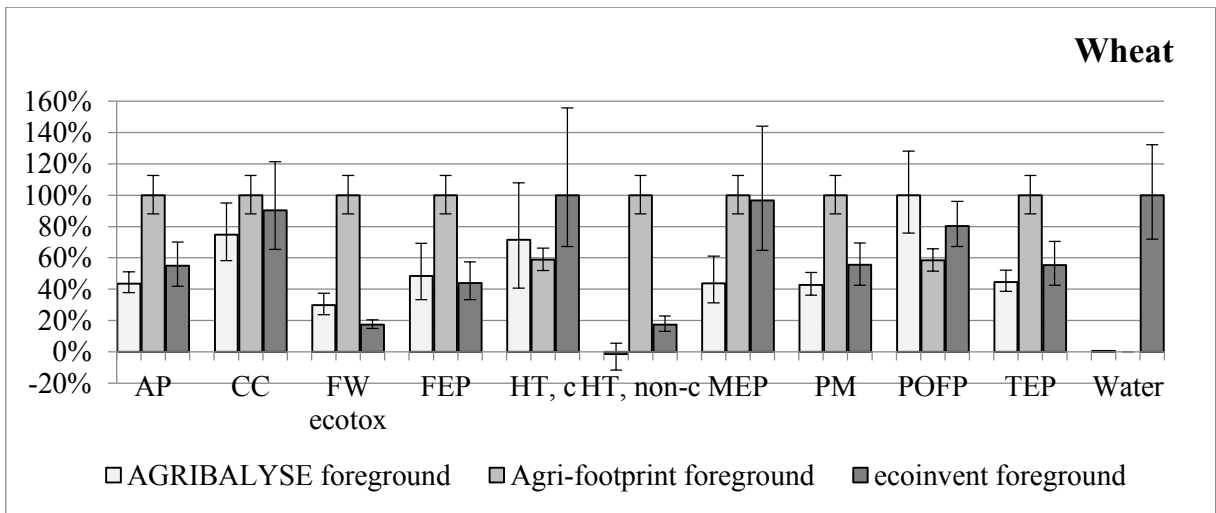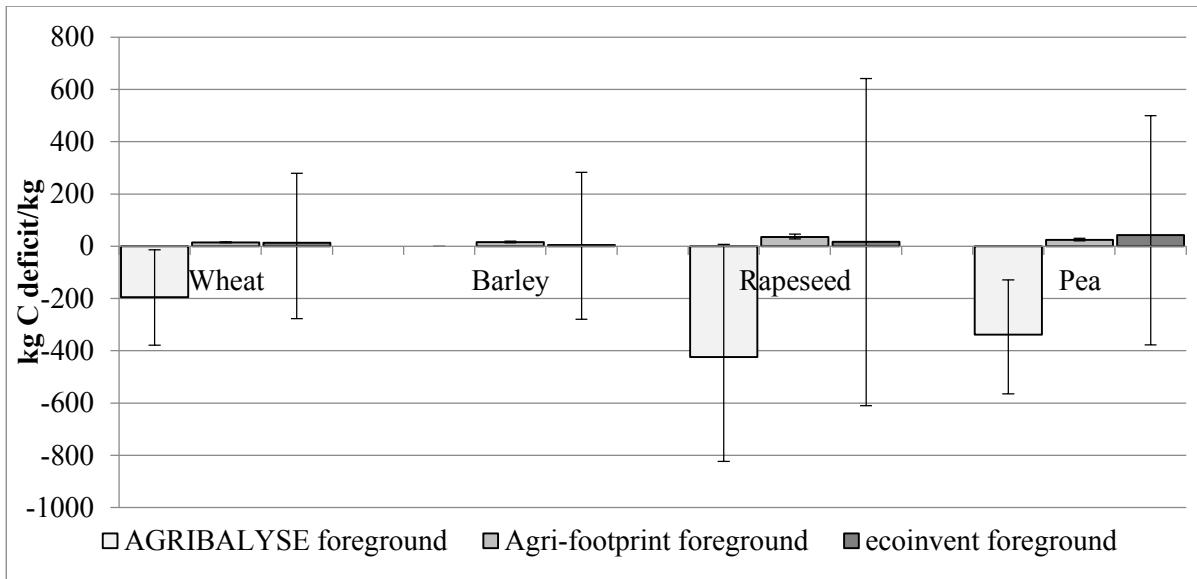
Figure 7. Mean potential land use impacts of the foreground system and 95% confidence intervals per kg of arable crop.

Agri-footprint generally estimated lower contributions from the background system than the other databases, which could be explained by its smaller system boundaries for background systems than those of AGRIBALYSE and ecoinvent (Figure 2). When considering the effects of allocation in Agri-footprint, important differences were observed between datasets with no allocation and those with allocation (supplementary material, Figures S10 and S11).

Concerning the foreground system (Figures 6. and S5), a significant difference in climate change impact was observed only for rapeseed, due to Agri-footprint's inclusion of CO_2 emissions from land transformation and use of more nitrogen fertilisers, which increase N_2O emissions.

The main contributions in the foreground system to the impact categories particulate matter, photochemical ozone formation, acidification and terrestrial eutrophication were NH_3 and NO_x emissions. In contrast, marine eutrophication was influenced mainly by emissions to water of NO_3^- and phosphorus compounds, respectively. Agri-footprint predicted significantly higher foreground contributions than AGRIBALYSE and ecoinvent to acidification, particulate matter and terrestrial eutrophication, mainly due to NH_3 emissions. AGRIBALYSE and ecoinvent showed no significant differences between acidification, particulate matter and terrestrial eutrophication of wheat and rapeseed, but did so for pea, due to significant differences in nitrogen fertiliser application and, therefore, NH_3 emissions.

Photochemical ozone formation was caused mainly by NO_x emitted by application of nitrogen fertilisers and combustion of diesel for agricultural machinery. Agri-footprint datasets for wheat, rapeseed and pea had significantly lower photochemical ozone formation because they excluded NO_x emissions from nitrogen fertiliser application. Agri-footprint and ecoinvent had significantly higher marine eutrophication for wheat than AGRIBALYSE, and Agri-footprint had higher

marine eutrophication than AGRIBALYSE for rapeseed, due to higher NO_3^- emissions per unit of nitrogen fertiliser applied to the field.

Emission of HMs and PPPs were the main contributors to the human-toxicity-related impact category in the foreground system. Human toxicity was mainly influenced by HM emissions. As in the inventory, negative contributions were predicted due to uncertainty in the modelling rather than a positive potential impact on human health (Koch and Salou, 2013). The impact category "human toxicity, cancer" (influenced mainly by chromium emissions to water) for wheat was higher in ecoinvent than in Agri-footprint, whereas for pea it was highest in Agri-footprint. Significant differences in "human toxicity, non-cancer" were also observed for wheat, barley and pea. PPP emissions to the soil had a large influence on freshwater ecotoxicity; however, characterisation factors for some PPP assumed to be released to the soil were missing in the chosen characterisation method (supplementary material, Table S4), which may underestimate PPP impacts.

For irrigation, differences in irrigation volumes and water sources explained significant differences in potential water depletion among the databases. Since the ILCD characterisation method was spatially explicit, characterisation factors in Agri-footprint and ecoinvent was nearly four times as high as that in AGRIBALYSE.

AGRIBALYSE predicted a negative land use impact because it assumed transformation from discontinuously built urban soil to agricultural soil, which resulted in a strong negative contribution of the foreground system (Figure 7.). Indeed, the ILCD characterisation method associated a highly negative characterisation factor with this transformation. In contrast, Agri-footprint and ecoinvent predicted a similar average land use impact, although land use and land transformation are modelled differently.

4. Discussion

Analysis of secondary datasets for arable crops highlighted significant differences among the LCIA that are influenced by sources of activity data and modelling approaches adopted to estimate environmental emissions and use of resources, such as land.

Generally, different arable crops are modelled in a similar way within a given database, whereas greater differences are observed for the same arable crop modelled in different databases. Here we provide an overview of main features of the databases analysed and considerations about their accuracy.

Data characteristics

Databases differ in their sources of data: AGRIBALYSE and Agri-footprint use average data for France, whereas ecoinvent uses data from one French region to represent all of France. In AGRIBALYSE and ecoinvent, data uncertainty is assessed for each input from and output to the foreground system, except for crop yields, using the pedigree matrix approach. In contrast, Agrifootprint estimates uncertainty in crop yields and derives uncertainty in input flows and emissions accordingly. For this reason, the range of the 95% confidence interval of LCIA varied among impact categories for AGRIBALYSE and ecoinvent but not for Agri-footprint.

System boundary definition

The choice of system boundaries influences contributions of the background system to results. Indeed, Agri-footprint datasets – for which agricultural machinery, infrastructures and PPP and seeds production are excluded from system boundaries – had generally lower contribution from the background system than the others for nearly all impact categories analysed. System boundaries of secondary data should be consistent with the goal and scope of the study and, when pertinent, with LCA guidelines, which sometimes have instructions for including or excluding specific processes. Indeed, the inclusion of infrastructures, for example, can result to be important in light of the impact categories analysed (Frischknecht et al., 2007). Moreover, if the aim of the LCA is to assess the environmental burden of a certain product in compliance, for example, with the PEF, then, according to the PEF guide (EC, 2016), machinery and infrastructure should be included within system boundaries.

Agricultural practice modelling

Management of an agricultural system includes complex dynamics that should be considered when performing a LCA. Indeed, management of agricultural residues and composition of a crop rotation can affect field productivity and the inputs required (Cherubini and Ulgiati, 2010; Nemecek et al., 2015). The three databases modelled these effects using different approaches (Table 2). Furthermore, allocating the impact to co-products and the choice of allocation method can influence the LCIA strongly (supplementary material, Figure S10, Figure S11). Although the most appropriate way to model agricultural practices remains under debate (e.g. Cherubini and Ulgiati, 2010; Nemecek et al., 2015), it is important that LCA practitioners verify that modelling of crop rotation and co-product management are consistent with the goal and scope of their studies and, if applicable, product category rules.

Agricultural operation modelling

Agricultural operation modelling differs in the number of operations, the number of passes and the time-related representativeness. Other elements that influence impacts of agricultural operations substantially, such as machine power and soil texture (Lovarelli et al., 2016; Van linden and Herman, 2014), are not explicitly considered in the datasets analysed, and Lovarelli et al. (2016) found that this can lead to misleading results.

Use of agricultural machinery causes airborne emissions due to fuel combustion and emissions of HMs due to tyre wear (Hjortenkrans et al., 2007), whose emissions are included only in AGRIBALYSE and ecoinvent.

Fertiliser application and nutrient fate modelling

Significant differences were found in the amount of fertilisers applied to the field among the datasets. Official statistics on amounts of fertilisers per crop were not available for France; therefore, it was not possible to check which database contains the most accurate data.

Nutrient fate is greatly influenced by site-specific conditions, such as environmental conditions, soil type, agricultural management practices and fertiliser type (Brentrup et al., 2000), and spatially explicit modelling of emissions from agricultural systems is considered of paramount importance (Basset-Mens et al., 2006; Biswas et al., 2008; Cederberg et al., 2013). A spatially-explicit approach was partially applied in AGRIBALYSE and ecoinvent but not in Agri-footprint. Agri-footprint estimates of NH_3 emissions, based on IPCC guidelines (IPCC, 2006), lead to significantly higher emissions per unit of nitrogen applied to the field than in AGRIBALYSE and ecoinvent, explaining Agri-footprint's higher acidification and terrestrial eutrophication impacts. In contrast, NO_3^- emissions factors were equal for all crops except peas.

Emissions factors for NO_x from nitrogen fertiliser application were higher in AGRIBALYSE than in ecoinvent and were not considered in Agri-footprint.

Databases expressed phosphorus compound emissions using different flows, limiting the ability to compare inventory data. Scherer and Pfister (2015) found that estimates of emissions of phosphorus compounds in ecoinvent were up to one order of magnitude lower than results of their model. Phosphorus compound emissions represent the main contribution to freshwater eutrophication, which, except for peas, did not significantly differ among the databases, despite significant differences in phosphorus fertiliser application.

PPP application and environmental fate modelling

Estimation of PPP emission and fate is a topic of intense discussion both in the LCA community and beyond (Rosenbaum et al., 2015). In the three databases, active ingredients were assumed to

end up completely in the soil after application. However, depending on the active ingredient, application method, weather and soil conditions, crop characteristics and irrigation, PPP fate can change, and using a pre-determined fate factor can lead to extremely high uncertainty (Rosenbaum et al., 2015). Moreover, the databases analysed included active ingredients no longer authorised in France, of which one (carbendazim) contributes most to the freshwater ecotoxicity impact of wheat in Agrifootprint.

When choosing a dataset, it is therefore recommended to verify that modelling of PPP application follows legislation of the country in question.

Regarding the LCIA, PPP emissions influenced freshwater ecotoxicity and, to a lesser extent, human toxicity. However, estimated impacts can be influenced by assumptions about PPP fate and the type of PPP used. Some PPP emissions did not have associated characterisation factors in the ILCD characterisation method, which may have caused toxicity-related impacts to be underestimated. Therefore, since no of the characterisation methods currently available has characterisation factors for all the possible emissions of PPPs, LCA practitioners should be aware that the combination between characterisation method and dataset can influence the results of the toxicity-related impact categories.

Heavy metal inputs and environmental fate modelling

Mass balance and fate of HMs is affected by several uncertainties and limitations. For example, in the datasets analysed, uncertainty in HM inputs to the agricultural system and in fate modelling led to misleading negative emissions to the agricultural soil (Koch and Salou, 2013) that resulted in negative contributions to human toxicity and freshwater ecotoxicity impacts. Furthermore, the choice of characterisation method can influence assessment of impacts on human health greatly (Pizzol et al., 2011).

Irrigation modelling

Databases differed in both irrigation volumes and water flows, with different spatially-explicit characterisation factors. Those differences contribute to different LCIA results. Temporal variability of irrigation, instead, was not taken into consideration in any of the datasets analysed. Indeed, despite Pfister and Bayer (2014) highlighted the importance of taking into consideration the temporal variability when assessing the impact on water stress, nowadays databases implemented in commonly used LCA softwares do not report temporally-explicit flows. Therefore, LCA practitioners are recommended to prefer spatially-explicit water flows to assess water resource depletion, whereas LCA databases developers should focus on the inclusion of temporal variability of water withdrawals within their datasets.

Land transformation modelling

The three databases model land transformation differently. In Agri-footprint and AGRIBALYSE, in which average data are considered, land transformation is reported, but different amounts of transformed land are considered. In contrast, ecoinvent excludes net land transformation. Transformation from a discontinuously built urban area to an agricultural one gives a relevant negative contribution to land use impact in AGRIBALYSE.

CO_2 emissions due to land transformation are considered relevant and are included only in Agri-footprint. In addition, aspects of land management associated with impacts not yet predicted well by LCIA (e.g. biodiversity) are still not represented sufficiently, and differences in land management are difficult to assess.

5. Conclusions

Datasets from different databases that model the same crop have methodological differences that can lead to significantly different LCIA results.

In the present study nine relevant elements that characterise datasets modelling arable crop cultivation were analysed, in order to highlight similarities and differences and investigate the extent to which they affect the results.

The nine elements are: data, system boundary definition, agricultural practice modelling, agricultural operation modelling, fertiliser application and nutrient fate modelling, PPP application and environmental fate modelling, HM input and environmental fate modelling, irrigation modelling and land transformation modelling.

Results of the present study provide LCA practitioners with elements according to which evaluate the characteristics of datasets that they use for modelling (not necessarily belonging to the analysed databases), to choose the most appropriate one, depending on the aim and scope of the study, and to interpret results. Furthermore, to a certain extent, they can provide information to database developers to improve dataset quality. For instance, the exclusion of infrastructures and machineries, and PPP and seeds production from system boundaries can significantly influence contribution of the background system to nearly all impact categories.

Activity data from which datasets were built differ greatly because datasets rely on different data sources. Since official statistics on arable crop production in France are currently not available for most activity data, it was not possible to identify the most accurate datasets; however, a check of activity data by a pool of experts may yield a higher level of accuracy.

Concerning the LCIA, the foreground system contributed more to overall impact for most impact categories and nearly all of the datasets analysed. Impacts of the foreground system were associated mainly with field emissions, most of which are estimated with models. Since field

emissions are influenced largely by site-specific conditions, including site-specific parameters in the modelling may lead to more accurate estimates.

Although the present study examined only 12 datasets modelling arable crops, we consider the present work as a basis from which to start analysing and interpreting other datasets of agricultural products. Furthermore, as the study highlighted that much of the LCIA is associated with estimated emissions, we ask other researchers to explore the pertinence of models used to estimate field emissions and to provide more details about the representativeness of and uncertainty in the results.

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Supplementary materials

Average crop yields

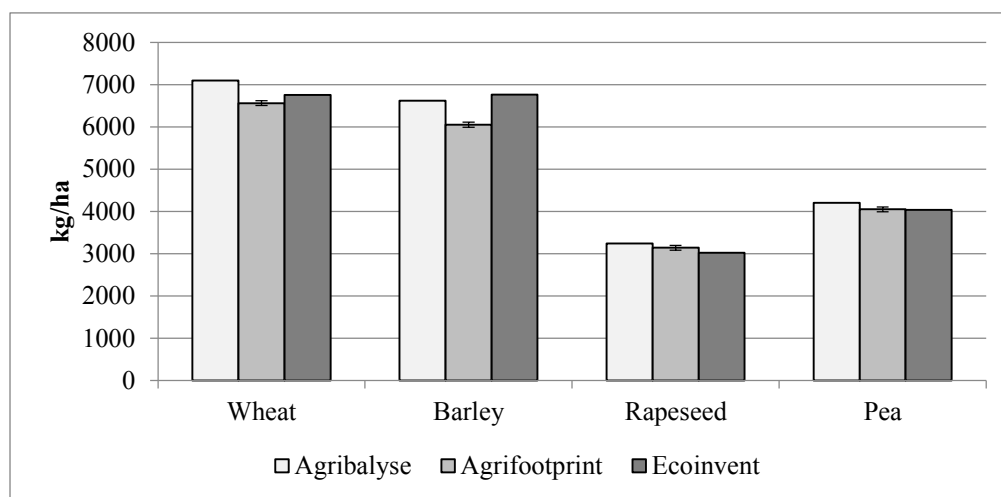Figure S1 reports the crop yield assumed in each dataset.

Figure S1: Average crop yields with 95% confidence interval. The 95% confidence interval of the yields is defined only in Agri-footprint datasets, not in AGRIBALYSE and Ecoinvent datasets

Agricultural operations

Table S1 reports the average emissions factors for agricultural machineries expressed in terms of mass of pollutant emitted by the combustion of 1 kg of diesel.

Table S1: Emissions factors expressed in term of mass of pollutant emitted by a unit of mass of diesel (kg/ kg_{diesel})

	AGRIBALYSE	Agri-footprint	Ecoinvent
NH₃	2.0E-05	1.0E-05	2.0E-05
PM < 2.5 um	4.3E-03	1.4E-03	4.7E-03
CO₂	3.1	3.2	3.1
N₂O	1.2E-04	2.6E-05	1.2E-04
CH₄	1.3E-04	1.1E-04	1.3E-04
CO	5.4E-03	4.9E-03	8.1E-03
NOx	4.2E-02	3.0E-02	4.5E-02
NMVOC	2.7E-03	2.8E-03	2.7E-03

Modelling of fertilisers application and nutrients fate

In Figure S1 the amount of nitrogen, phosphorus and potassium fertilisers applied to the field for the arable crops cultivation is reported.

Table S2 reports the modelling approaches adopted in the three datasets to estimate the nutrients fate and Figure S2 represents the resulting emissions of nitrogen compounds expressed as percentage of nitrogen applied to the field in each dataset.
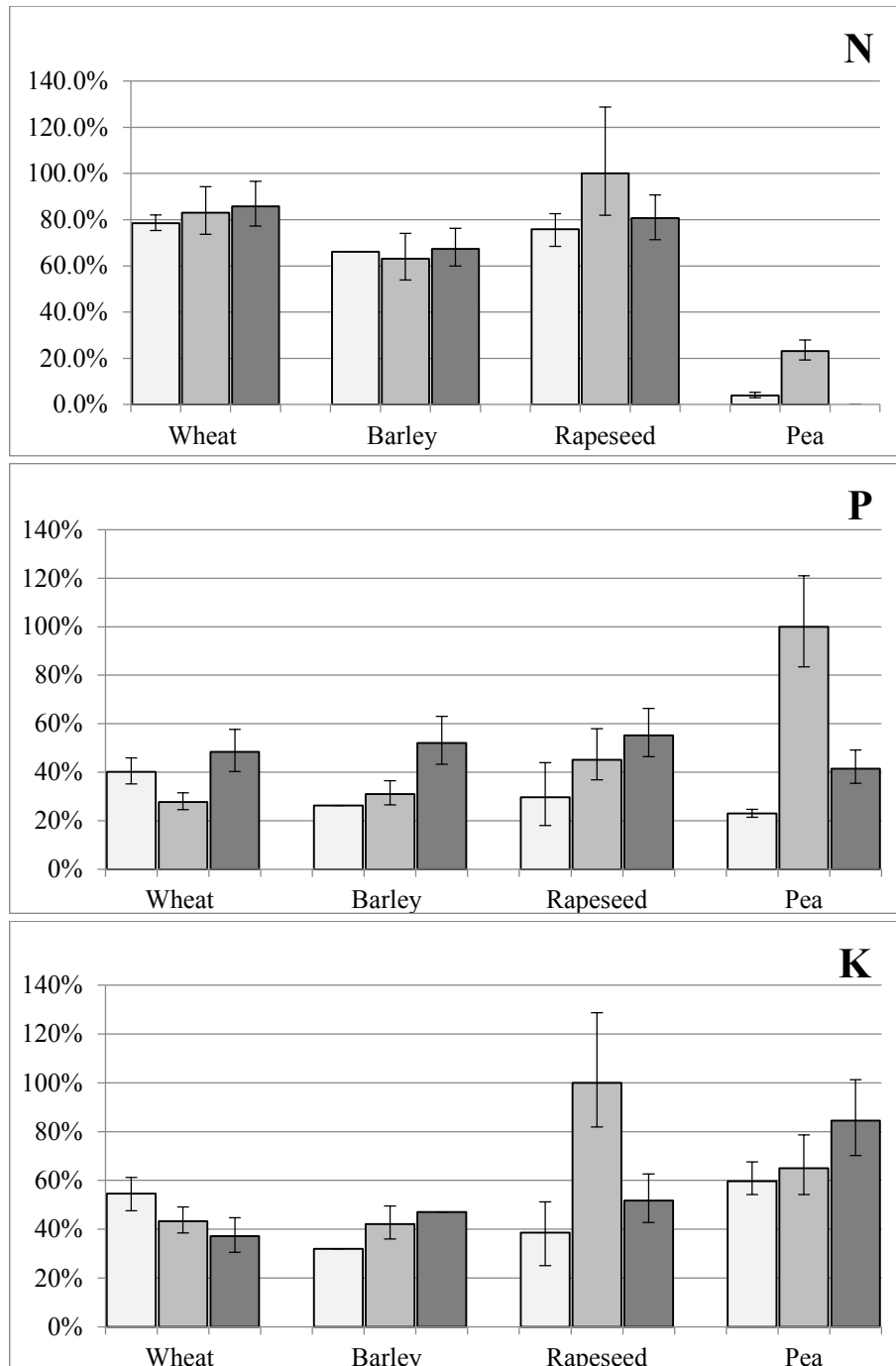
Figure S2: Average fertilisers application and 95% conference intervals per ha of field. The 95% confidence interval is not reported in the AGRIBALYSE barley dataset, N refers to nitrogen fertilisers, P to phosphorus fertilizers and K to potassium fertilisers. For each nutrient, the highest result among the three datasets is reported as 100% and the others are expressed as percentage of the maximum amount.

	AGRIBALYSE	Agri-footprint	Ecoinvent
Ammonia (NH_3)	(EMEP/EEA, 2009) Tier 2 (for organic fert.); EMEP/CORINAIR 2006 Tier 2 (see (Koch and Salou, 2013) (for mineral fert.)	(IPCC, 2006)	(Asman, 1992) (for mineral fertiliser)
Nitrogen oxides (NO_x)	(EMEP/EEA, 2009) Tier1	Not considered	Personal communication in (Nemecek and Schnetzer, 2011)
Nitrate (NO_3^-)	COMIFER 2001 adjusted (see (Koch and Salou, 2013)	(IPCC, 2006) (all the N leached is emitted as NO3)	SALCA-NO3 (Richner et al., 2014)
Dinitrogen oxide (N_2O) (direct + indirect emissions)	(IPCC, 2006) Tier 1	(IPCC, 2006) Tier 1	(IPCC, 2006) Tier 1
Phosphorus (P) (leaching)	SALCA-P (Prasuhn, 2006)	0,05 of P in fertilisers and manure reaches freshwater	SALCA-P (Prasuhn, 2006)
Phosphorus (P) (runoff)	SALCA-P (Prasuhn, 2006)		SALCA-P (Prasuhn, 2006)
Phosphorus (P) (erosion)	SALCA – P (Prasuhn, 2006)	Not included due to limited data availability	SALCA-P (Prasuhn, 2006)

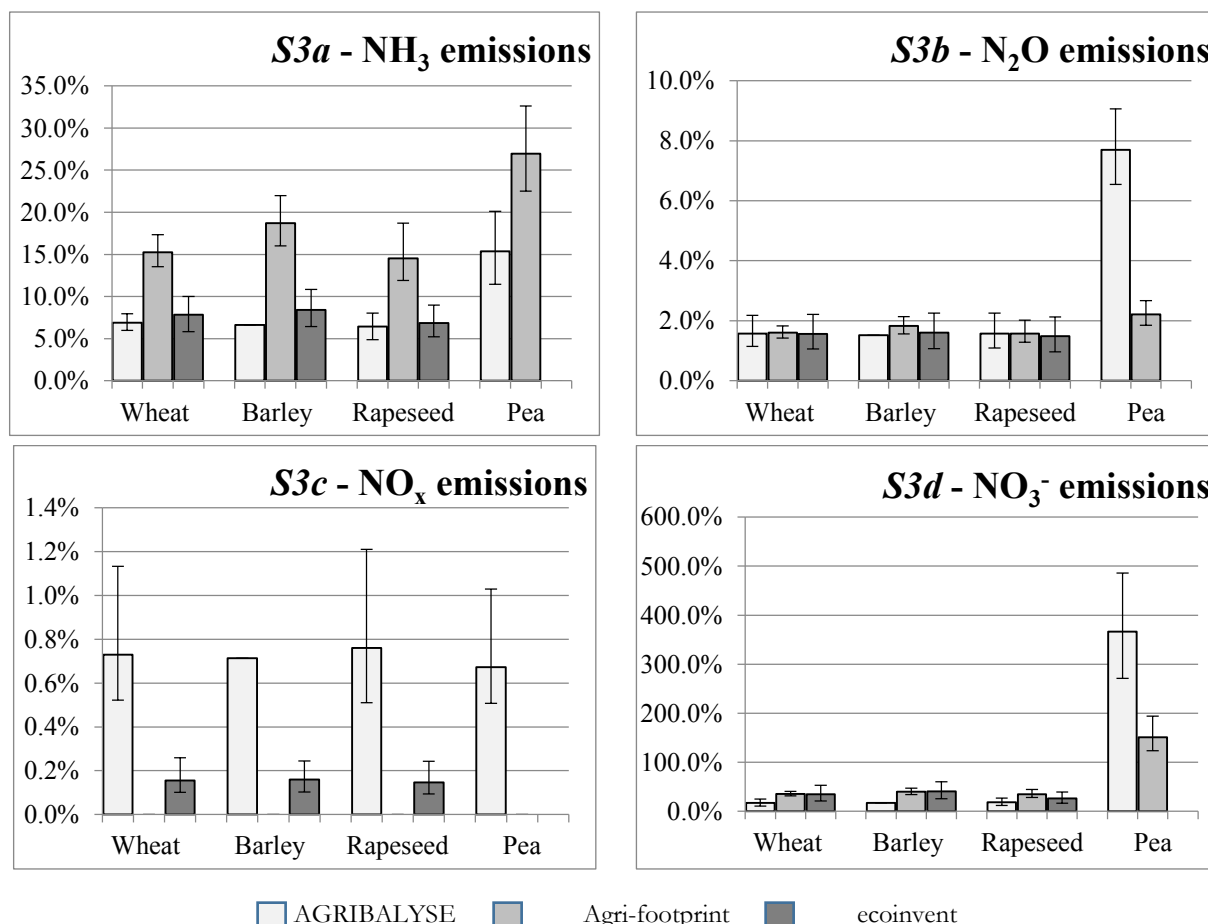*Table S2: Models adopted for nutrients fate*

Figure S1: Average emissions of N compounds and 95% confidence interval, expressed as percentage ratio of the N emitted in the environment to the N applied to the field through fertilisers

Modelling of plant protection products and environmental fate

In Table S3 a list of the active ingredient that are included in datasets, whose use was not authorised in France anymore, according to European legislation, is reported.

Active ingredient	Database
Anthraquinone	ecoinvent
Bitertanol	AGRIBALYSE, ecoinvent
Carbendazim	Agri-footprint, ecoinvent
Choline chloride	AGRIBALYSE, ecoinvent
Cyfluthrin	AGRIBALYSE
Flusilazole	ecoinvent
Ioxynil	AGRIBALYSE
Metolachlor	Agri-footprint
Oxydemeton methyl	AGRIBALYSE
Procymidone	ecoinvent
Trifluralin	AGRIBALYSE, ecoinvent
Vinclozolin	Agri-footprint, ecoinvent

Table S3: Active ingredients included in the analysed datasets whose use is not authorised in France

In Table S4 are reported the active ingredients emitted in the environment for which a characterisation did not exist for any of the impact categories considered.

	AGRIBALYSE	Agri-footprint	ecoinvent
Boscalid	X		
Fenpropidin			X
Fenpropimorph			X
Florasulam	X		X
Fluoxastrobin	X		
Flupyrsulfuron-methyl			X
Fluquinconazole	X		
Iodosulfuron	X		
Iodosulfuron-methyl-sodium			X
Mefenpyr-diethyl	X		
Mesosulfuron-methyl (prop)	X		X
Metaldehyde	X		X
Metconazole	X		X
Metosulam			X
Picoxystrobin	X		X
Prohexadione-calcium			X
Propoxycarbazone-sodium (prop)			X
Prothioconazol	X		
Pyraclostrobin (prop)	X		X
Silthiofam			X
Spiroxamine			X
Trifloxystrobin			X

Table S4: List of emitted active ingredients of PPP for which a characterisation factor was not reported for any of the impact categories considered. X indicates that the emissions of the specific active ingredient were considered in the dataset

Modelling of soil erosion

Table S5 reports the references used to model soil erosion in the AGRIBALYSE and ecoinvent datasets.

$M_{erosion}$ (heavy metals emissions through erosion)	=	C_{tot} (HM content in the soil)	*	B (soil erosion)	*	A (accumulation factor)	*	F (fraction of soil the reaches rivers)
Value/sources of data								
AGRIBALYSE v1.2		BDAT database (RMQS, 2013)		RUSLE soil loss equation (USDA – Agricultural Research Service, 2005)		1.86 (Freiermuth, 2006)		0.2 (Freiermuth, 2006)
ecoinvent v3.1		(Keller and A., 2001)		(Oberholzer et al., 2001)		1.86		0.2

Table S5: Equation used to estimate the heavy metal and phosphorus emissions due to erosion and sources of data and parameters

LCIA

In this section the LCIA for 1 kg of the analysed arable crops is reported. In each figure, the result for land use is not displayed because it is out of scale. Figure S4 represents the LCIA by showing the foreground and background contributions, whereas Figure S5 represents only the foreground contribution. In both Figure S4 and Figure S5 the 95% confidence interval is reported.
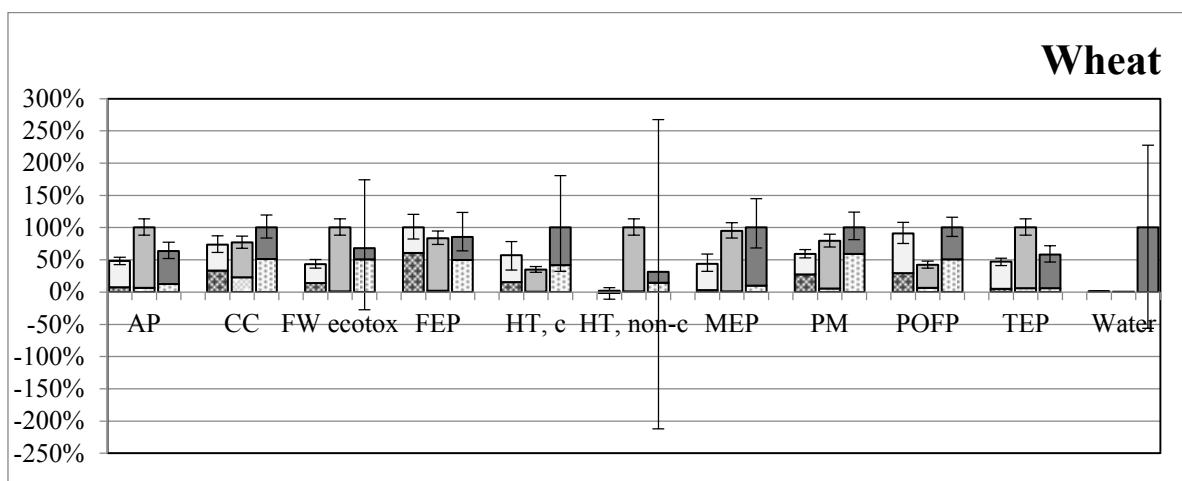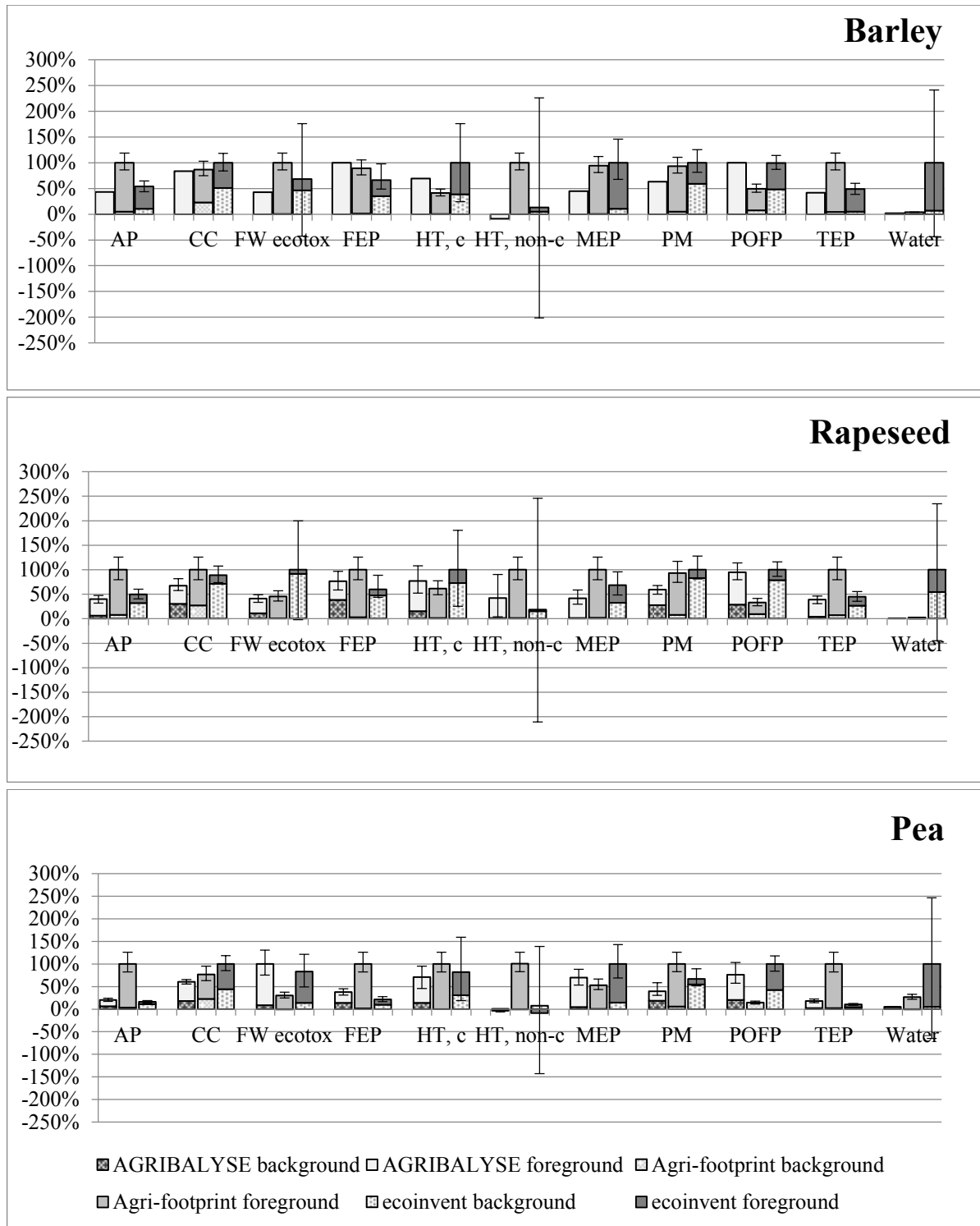
Figure S4: *Average foreground and background systems contributions to LCIA and 95% confidence interval for 1 kg of the analysed arable crops. For each impact category, the higher result is reported as 100% and the other are expressed as percentage of the maximum amount. The following impact categories were considered: acidification (AP), climate change (CC), freshwater ecotoxicity (FW ecotox), freshwater eutrophication (FEP), human toxicity-cancer (HT,c), human toxicity-non cancer (HT, non-c), marine eutrophication (MEP), particulate matter (PM), photochemical ozone formation (POFP), terrestrial eutrophication (TEP), water resource depletion (Water)*
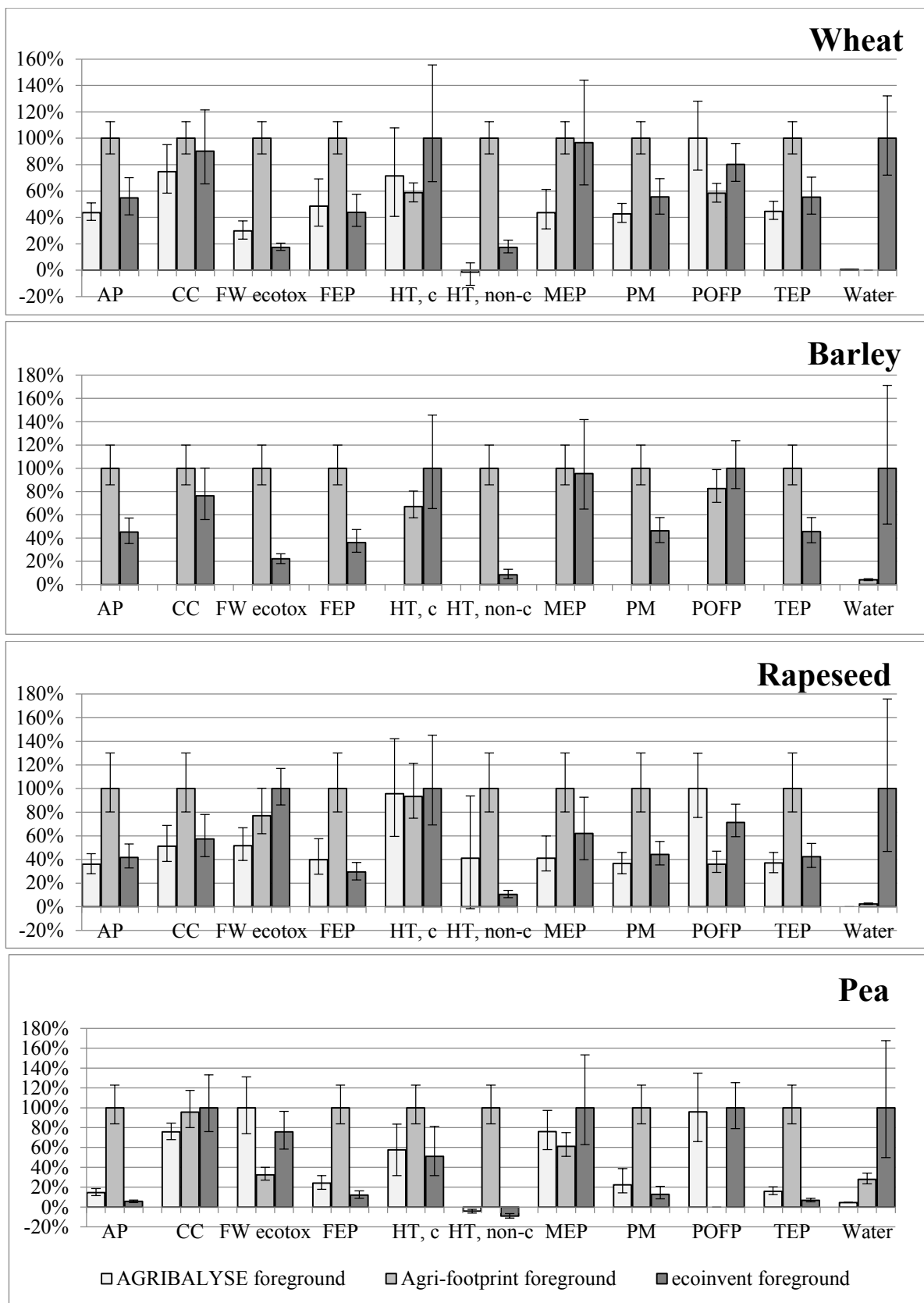
Figure S5: Average foreground system contributions to LCIA and 95% confidence interval for 1 kg of the the analysed arable crops. For each impact category, the higher result is reported as 100% and the other are expressed as percentage of the maximum amount. The following impact categories were considered: acidification (AP), climate change (CC), freshwater ecotoxicity (FW ecotox), freshwater eutrophication (FEP), human toxicity-cancer (HT,c), human toxicity- non cancer (HT, non-c), marine eutrophication (MEP), particulate matter (PM), photochemical ozone formation (POFP), terrestrial eutrophication (TEP), water resource depletion (Water).

Effects of modelling approach on the LCIA

Figure S6, Figure S7, Figure S8 and Figure S9 report a comparison of average potential impact of 1 kg of the analysed crop at farm gate as modelled in ecoinvent datasets following three modelling approaches: attributional (also called "Alloc def" in ecoinvent database), cut-off system model (also called "Alloc rec" in ecoinvent database) and consequential (also called "Conseq" in ecoinvent database).
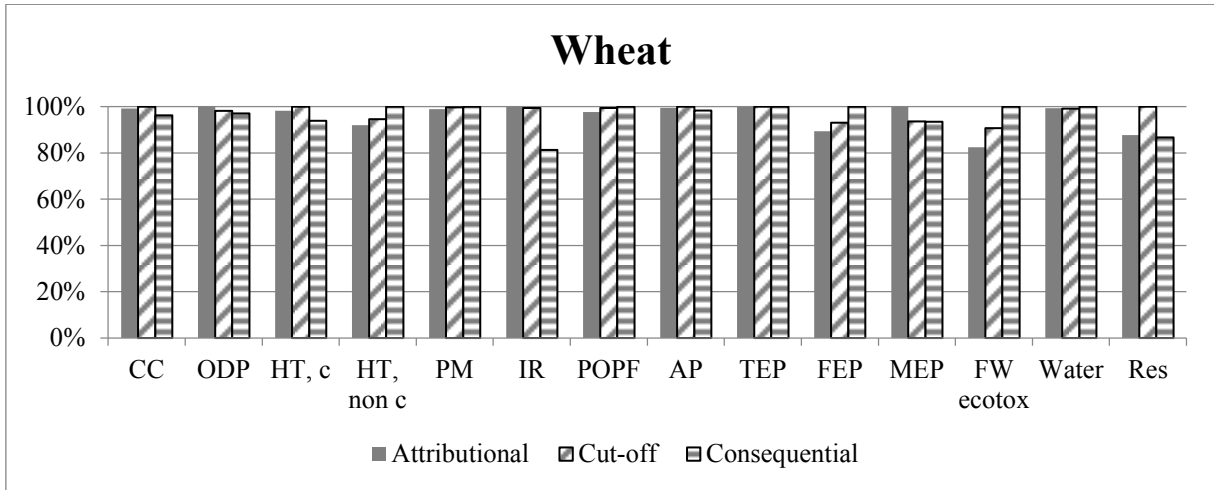

Figure S6: Average potential impact of 1 kg of wheat at farm gate as modelled in ecoinvent datasets with different approaches. For each impact category, the higher result is reported as 100% and the other are expressed as percentage of the maximum amount. The following impact categories were considered: climate change (CC), ozone depletion potential (ODP), human toxicity-cancer (HT,c), human toxicity- non cancer (HT, non-c), particulate matter (PM), ionising radiation (IR), photochemical ozone formation (POFP), , acidification (AP), terrestrial eutrophication (TEP), freshwater eutrophication (FEP), marine eutrophication (MEP), freshwater ecotoxicity (FW ecotox), water resource depletion (Water), Mineral, fossil and ren resource depletion (Res)
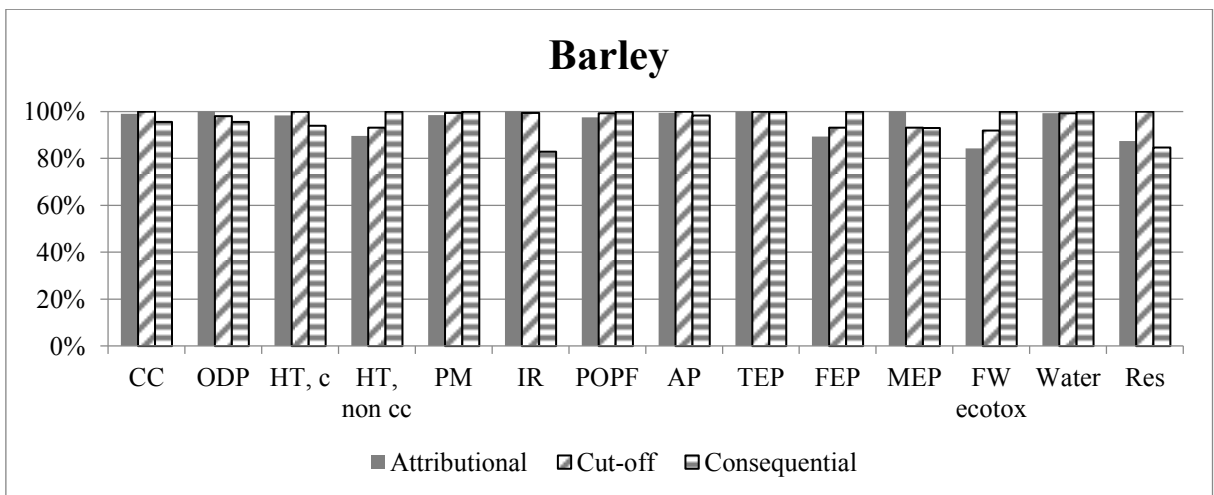

Figure S7: Average potential impact of 1 kg of barley at farm gate as modelled in ecoinvent datasets with different approaches. For each impact category, the higher result is reported as 100% and the other are expressed as percentage of the maximum amount. The following impact categories were considered: climate change (CC), ozone depletion potential (ODP), human toxicity-cancer (HT,c), human toxicity- non cancer (HT, non-c), particulate matter (PM), ionising radiation (IR), photochemical ozone formation (POFP), , acidification (AP), terrestrial eutrophication (TEP), freshwater eutrophication (FEP), marine eutrophication (MEP), freshwater ecotoxicity (FW ecotox), water resource depletion (Water), Mineral, fossil & ren resource depletion (Res).
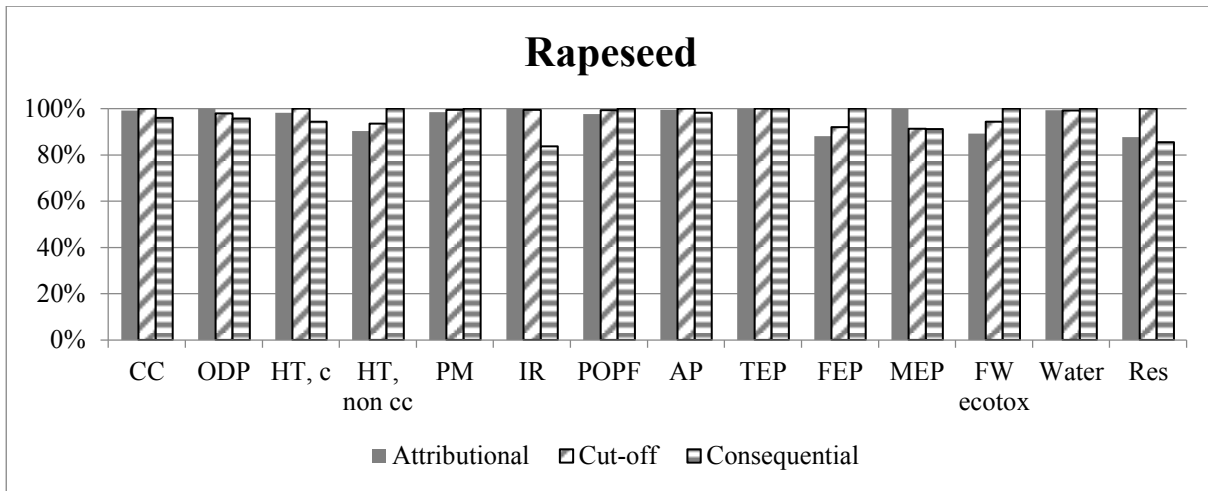
Figure S8: Average potential impact of 1 kg of rapeseed at farm gate as modelled in ecoinvent datasets with different approaches. For each impact category, the higher result is reported as 100% and the other are expressed as percentage of the maximum amount. The following impact categories were considered: climate change (CC), ozone depletion potential (ODP), human toxicity-cancer (HT,c), human toxicity- non cancer (HT, non-c), particulate matter (PM), ionising radiation (IR), photochemical ozone formation (POFP), , acidification (AP), terrestrial eutrophication (TEP), freshwater eutrophication (FEP), marine eutrophication (MEP), freshwater ecotoxicity (FW ecotox), water resource depletion (Water), Mineral, fossil and ren resource depletion (Res)

Figure S2: Average potential impact of 1 kg of wheat at farm gate as modelled in ecoinvent datasets with different approaches. For each impact category, the higher result is reported as 100% and the other are expressed as percentage of the maximum amount. The following impact categories were considered: climate change (CC), ozone depletion potential (ODP), human toxicity-cancer (HT,c), human toxicity- non cancer (HT, non-c), particulate matter (PM), ionising radiation (IR), photochemical ozone formation (POFP), , acidification (AP), terrestrial eutrophication (TEP), freshwater eutrophication (FEP), marine eutrophication (MEP), freshwater ecotoxicity (FW ecotox), water resource depletion (Water), Mineral, fossil and ren resource depletion (Res)
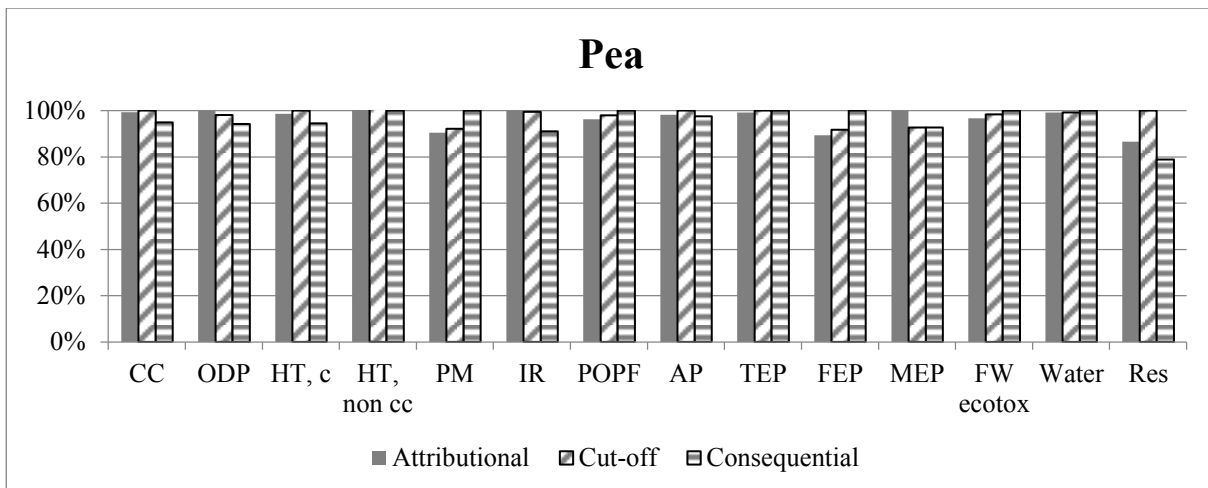
Effects of allocation on the LCIA

Figure S10 and Figure S11 report the LCIA of the average potential impact of 1 kg of wheat and barley at the farm gate, considering the effect of allocation of the impact to co-products for the Agri-footprint datasets. Particularly, four allocation criteria are reported for Agri-footprint datasets: impact allocated entirely to the grains (no alloc) and allocation according to the economic value, the energy content and the mass.
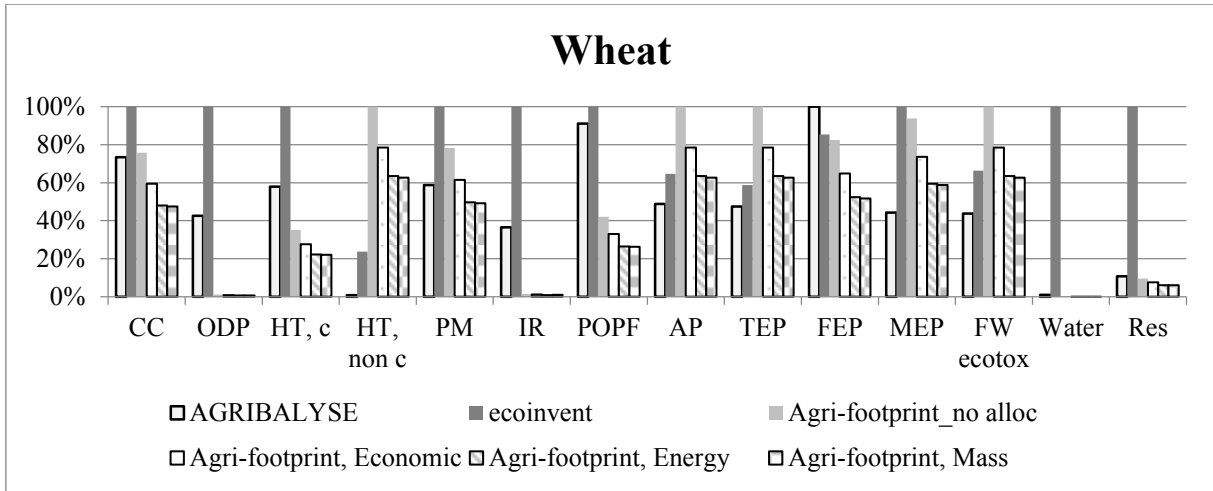
Figure S10: LCIA of 1 kg of wheat at farm gate. In AGRIBALYSE and ecoinvent the impact is entirely allocated to the grains, in Agri-footprint different allocation criteria are considered: economic, energy and mass. Agri-footprint_no alloc refers to the datasets considered in the present study, in which the entire impact of cultivation is allocated to grains. For each impact category, the higher result is reported as 100% and the other are expressed as percentage of the maximum amount. The following impact categories were considered: climate change (CC), ozone depletion potential (ODP), human toxicity-cancer (HT,c), human toxicity- non cancer (HT, non-c), particulate matter (PM), ionising radiation (IR), photochemical ozone formation (POFP), , acidification (AP), terrestrial eutrophication (TEP), freshwater eutrophication (FEP), marine eutrophication (MEP), freshwater ecotoxicity (FW ecotox), water resource depletion (Water), Mineral, fossil and ren resource depletion (Res)

Figure S11: LCIA of 1 kg of barley at farm gate. In AGRIBALYSE and ecoinvent the impact is entirely allocated to the grains, in Agri-footprint different allocation criteria are considered: economic, energy and mass. Agri-footprint_no alloc refers to the datasets considered in the present study, in which the entire impact of cultivation is allocated to grains. For each impact category, the higher result is reported as 100% and the other are expressed as percentage of the maximum amount. The following impact categories were considered: climate change (CC), ozone depletion potential (ODP), human toxicity-cancer (HT,c), human toxicity- non cancer (HT, non-c), particulate matter (PM), ionising radiation (IR), photochemical ozone formation (POFP), , acidification (AP), terrestrial eutrophication (TEP), freshwater eutrophication (FEP), marine eutrophication (MEP), freshwater ecotoxicity (FW ecotox), water resource depletion (Water), Mineral, fossil and ren resource depletion (Res)
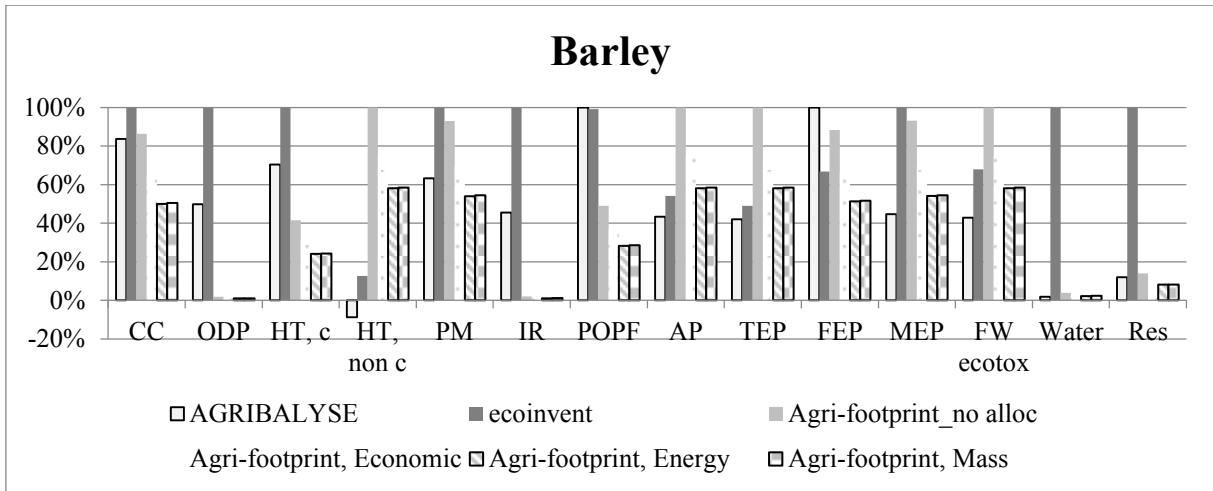
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4. Accounting for food loss and waste in life cycle assessment

Abstract

Food loss is a major concern from both environmental and social point of view. Life Cycle Assessment (LCA) has been largely applied to quantify the environmental impact of food and to identify pros and cons of different options for optimisation of food systems management, including the recovery of potential waste occurring along the supply chain. However, within LCA case studies, there is still a general lack of proper accounting of food losses. A discrepancy both in food loss definition and in the approaches adopted to model the environmental burden of food loss has been observed. These aspects can lead to misleading and, sometimes, contrasting results, limiting the reliability of LCA as a decision support tool for assessing food production systems. This article aims, firstly, at providing a preliminary analysis on how the modelling of food loss has been conducted so far in LCA studies. Secondly, it suggests a definition for food loss to be adopted. Finally, the article investigates the consequence of using such definition and it proposes potential paths for the development of a common methodological framework to increase the robustness and comparability of the LCA studies. It discusses the strengths and weaknesses of the different approaches adopted to account for food loss along the food supply chain: primary production, transport and storage, food processing, distribution, consumption and end of life. It is also proposes to account separately between avoidable, possibly avoidable and unavoidable food loss by means of specific indicators. Finally, some recommendations for LCA practitioners are provided on how to deal with food loss in LCA studies focused on food products. The most relevant recommendations concern: i) the systematic accounting of food loss generated along the food supply chain; ii) the modelling of waste treatments according to the specific characteristics of food; iii) the sensitivity analysis on the modelling approaches adopted to model multi-functionality; and iv) the need of transparency in describing the modelling of food loss generation and management.

Keywords

Food loss; Food waste; LCA; Allocation; Life cycle inventory; LCA modelling.

1. Introduction

The Food and Agriculture Organization of the United Nations (FAO) has estimated that each year approximately 1.3 billion tons of edible food are wasted throughout global food supply chains (FSCs), corresponding roughly to one-third of all food produced for human consumption (FAO, 2011a). Food loss (FL) represents a major concern from both an environmental and social point of view. On the one hand, by tackling FL in FSC, there is a great opportunity to reduce major environmental burdens related to FL generation and management, especially in developed countries; while on the other hand, about 800 million people on the planet are suffering from chronic undernourishment (FAO, 2014a). Wasting food means wasting all the inputs consumed along the entire food supply chain (energy, natural resources, human labour, etc.) and contributes directly to the depletion of some already scarce resources, such as phosphorous used to produce fertilisers, land and water. FAO (2013) has estimated that the total water used to produce the food currently lost within global food supply chains is equivalent to 3 times the size of the lake of Geneva (about 80,000 m^3) whereas the land use needed accounts for 1.4 billion of hectare. Food produced and not eaten at global level is responsible for the emissions of 3.3 GtCO$_{2eq}$ equal to more than 30 times the greenhouse gas emissions associated to domestic final demand in Switzerland in 2005 (Jungbluth et al., 2011). Moreover, food production is expected to increase in order to satisfy the needs of the raising world population, which may reach 9.5 billion by 2050 (United Nation - Department of Economic and Social Affairs, 2015). Reducing FL can play an important role in addressing this challenge, since - together with closing yield gaps, increasing cropping efficiency, and changing diets - it is one of the key actions to increase the availability of food for human consumption while reducing the environmental impact per unit of product (Foley et al., 2011).

In the European context, tackling FL is one of the objectives of the European Commission. The Roadmap to Resource Efficient Europe (EC, 2011) has identified food production and FL as key areas where resource efficiency can be improved. Two interventions are foreseen: setting targets for FL reduction for each EU member state and improving industrial symbiosis practices recovering waste and by-products (EC, 2014). Furthermore, the recent communication on circular economy, a system where the products, materials and resources value is maintained in the economy for as long as possible andwaste production is minimised, has identified food waste (FW) as one of the priority areas of intervention (EC, 2015; UNEP, 2006). To achieve these objectives at international as well as at lower scale of intervention, integrated assessment methodologies and a full supply chain perspective are needed. Indeed, it is crucial that the envisaged actions for a reduction of FL and its better management are assessed through a life

cycle perspective to avoid the shifting of burdens amongst different life cycle stages along the supply chain or different environmental compartments (EC-JRC, 2011). Given that FL occurs all along the supply chains, Life Cycle Assessment (LCA) represents a valuable tool for assessing: i) the environmental burdens associated with FL, ii) the benefits associated with FL reduction as well as iii) the preference among the possible recovery options.

The available scientific literature on LCA and food is rather wide (Arvanitoyannis et al., 2014; Chen et al., 2016). Currently, the most remarkable study estimating the impact of FL at global level, applying LCA, is a recent report from FAO (2013). In this report FL has been estimated in all regions of the world for both developing and developed countries. Within the published LCA studies on food, the assessment of FL along the supply chain is often performed partially or inconsistently (Cerutti et al., 2014), limiting the effectiveness of LCA as a decision support tool in this context.

In order to contribute to the current debate on FW assessment and accounting, the present article has a triple purpose. Firstly, it aims to summarise the terms related to FL currently used to address the topic and to enhance their harmonised use in the LCA context.

The use of shared terminology is, indeed, fundamental to achieve a harmonised approach (FAO, 2014b; Ostergren et al., 2014;Williams et al., 2015). Secondly, it aims to analyse and classify the different approaches observed in the scientific literature to assess the environmental burdens of FL, highlighting strengths, criticalities and possible inconsistencies. While conducting this analysis, the article discusses some relevant studies in the literature which can be considered as "exemplary" of different modelling approaches used by LCA practitioners. Finally, recommendations for the harmonization of these approaches within LCA studies have been provided, fostering the effectiveness of LCA as a decision support tool to achieve FL reduction.

2. Materials and methods

A selection of recent scientific articles, reviews and reports was analysed in order to shed light on the terminology currently adopted when referring to FL as well as to depict a classification of approaches to account for FL. The assessment of FL was performed only from an environmental perspective, whereas the economic and the social dimensions of sustainability were not taken into consideration. Relevant documents have been identified through search engines (e.g. Scopus and Google Scholar) using the key words "food loss", "food waste", "food wastage", "food + LCA", "vegetables + LCA", "fish + LCA", "meat + LCA". Furthermore, the reference list of these articles was analysed and additional references considered relevant were included in the survey. In particular, 82 articles published in peer review journals, 1 published in conference proceedings

and 17 scientific reports have been analysed. All the documents are written in English and published starting from 1998. Among these, more than 70% of the documents have been published after 2010. The selected documents cover different themes: production of vegetables food origins (25 documents), production of meat, dairy and eggs (7 documents), fish production (7 documents), the assessment of the environmental burden of dietary choices and meals (10 documents), waste treatments (5 documents), industrial ecology (14 documents), methodological aspects related to the application of LCA (14 documents) and other themes related to the topic (18 documents).

The present work investigated the use of the terms "food loss" and "food waste" and the definitions provided. These were compared and, when necessary, combined in order to provide some recommendations about their clear application within the LCA. Furthermore, the documents were reviewed in order to analyse the approaches adopted to account for FL in LCA studies focused on food products. In order to support such analysis, some articles were taken as example. However, since the present article is not intended as an extensive literature review, the list of mentioned articles should not be considered as exhaustive.

Accordingly to FAO (2011), five stages of the FSC were considered: (1) primary production, (2) transport and storage, (3) food processing, (4) distribution and (5) consumption. Furthermore, the end of life of FL generated within all the FSC stages was also considered. Food items were classified according to their origin as: (1) fruit and vegetables; (2) meat, dairy and eggs; and (3) fish. "Primary production" includes the agricultural stage for fruit and vegetables, breeding, aquaculture or fishing for animals and animal products and, when pertinent in case of fishing, it includes also first processing on fishing boat (Vazquez-Rowe et al., 2012). "Transport and storage" includes the activities between the primary production and the processing of the food. "Processing" includes a variety of options and treatments according to the food output. The "distribution" stage refers to both wholesale and retail distribution and it involves transport and storage activities. "Consumption" represents the last stage of the FSC and it includes household consumption or consumption in restaurants or canteens. Finally, the analysis covers the "end of life" stage. This includes the treatments performed in dedicated plants for the disposal or recovery of the waste derived from FL generated along the FSC. The recovery of FL in industrial ecology (IE) applications, in which FL are used as raw materials in downstream production processes, was discussed as an alternative to waste treatment for FL. As results of the analysis performed, some recommendations for LCA practitioners were derived to foster the systematic inclusion of FL within their studies.

3. Results

The establishment of a possible common framework to account for FL in LCA should consider, among others, relevant elements, as: i) the definitions to be used; ii) accounting of FL in LCA; and iii) the modelling of FL recovery processes. An overview of these elements is presented in the following sections.

3.1. Definition of food loss and food waste: characterisation and contextualisation for LCA applications

Different definitions FL and FW are reported in the scientific literature limiting the comparability of studies and the integration of their results into a common strategy for reducing FL (FAO, 2014b; Ostergren et al., 2014; Williams et al., 2015). Parfitt et al. (2010) and Papargyropoulou et al. (2014) agreed that three main definitions of FW could be found in the literature at the time of their studies. Firstly FAO (1981) defined FW as the wholesome edible material intended for human consumption, arising at any point of the FSC that is discarded, lost, degraded or consumed by pests. Stuart (2009) included to the cited FAO definition the fraction of edible food that is intentionally fed to animals and the by-products of food transformation that are diverted away from human consumption. Smil (2004) added to the aforementioned definition of FW the over-nutrition, intended as the gap among energetic consumption and human needs. WRAP (2008) proposed a further distinction among avoidable, possibly avoidable and unavoidable FW with the aim of analyzing FW at households in the United Kingdom.

FAO was a pioneer in proposing to harmonise the definitions and the terms related to FL and FW within the Global initiative on food loss and waste reduction (FAO, 2011b) through a Definitional framework of food loss (FAO, 2014b). This document was intended to improve data collection, data comparability, evidence-based regulatory and policy decisions for FL prevention and reduction. According to FAO (2014b), FL is "the amount of food intended for human consumption that, for any reason is not destined to its main purpose". A considerable effort towards an harmonised definition of FW was also made by the Fusions project that aimed to improve resource efficiency of Europe by reducing FW (Ostergren et al., 2014). According to Ostergren et al. (2014) FW is food produced to be addressed to humans that is disposed or recovered, excluding the fractions that are fed to animals and sent to bio based material production or biochemical processing.

Within LCA studies, FL definition has been rarely reported, apart from studies where the focus was specifically on FL (e.g. Eberle and Fels, 2015; Heller and Keoleian, 2014).

It is suggested to adopt the FAO (2014b) as basis for LCA studies. However, this definition was conceived to be generic enough to be applied to a broad range of contexts, not only within the LCA field. Therefore, it is necessary to analyse additional aspects of FL in order to move towards a systematized use of this definition within LCA and to avoid problems of interpretation. These additional aspects are hereunder discussed.

3.1.1. Differences among "food loss" and "food waste"

FL may occur at each stage of the FSC. The non-food parts of food plants (straw, leaves, roots, branches, etc.) and animals (bones, horns, etc.) are not included in the FL definition. In a LCA context, these parts can be, for example, considered as farming residues and left on the field or processed by established waste treatments (i.e. aerobic or anaerobic digestion, landfill, etc.) (FAO, 2014b). The terms FL and FW have been used to reference different kind of losses generated along the FSC (Parfitt et al., 2010). FL is used to describe the losses that occur in the production, post-harvest, processing and distribution stages of the FSC. Main drivers of FL generation, depending where in the world FL is generated, could be: i) poor storage infrastructure and logistics; ii) lack of technology; iii) insufficient skills, weak knowledge and management capacity of FSC actors; iv) no access to markets; and v) bad weather conditions. FW, instead, describes the losses that take place at retail and consumers stages, mainly due to: i) marketing consideration; ii) economic forces; iii) regulatory measures ("best before" or expiration date); iv) poor stock management; and v) consumer attitudes (FAO, 2011a; Parfitt et al., 2010). In the framework proposed by FAO, all kinds of food that is lost along the FSC are named "food loss", considering FW as part of FL (Fig. 1). To improve consistency, it is suggested to LCA practitioners to be compliant with the differentiation adopted by FAO (2014b), as reported in Fig. 1.

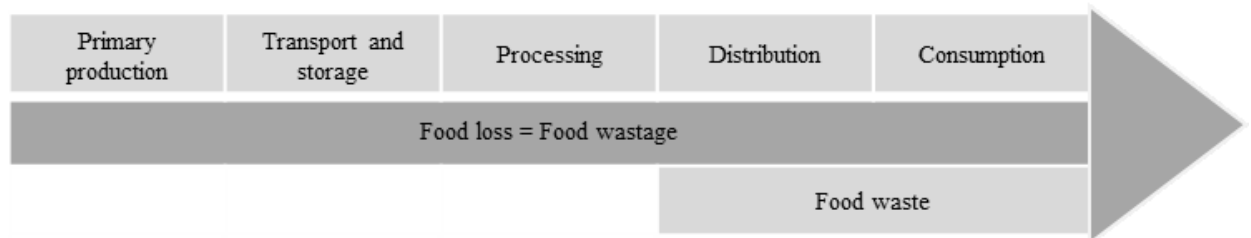*Fig. 1: Correspondence between the FSC stages and the definitions of "food loss", "food waste" and "food wastage" according to (FAO, 2013) and (FAO, 2014b)*

3.1.2. "Avoidable", "unavoidable" and "possibly avoidable" food loss

Many food products have parts which are not edible (e.g. egg shell, some fruits skin, animal bones). These correspond to what is called "unavoidable FL". In contrast, "avoidable FL" is the

amount of food thrown away because it is no longer wanted or has been allowed to go past its "best before" or "expiration" date (Papargyropoulou et al., 2014). The distinction between avoidable and unavoidable FL is not always sharp and the subjectivity in food use as well as cultural specificity may play an important role in setting the boundaries. In some countries, for example, animal hide can be eaten while in others it is a by-product used in the leather industry or just considered as waste (The Daily Meal, 2015). Therefore, the definition of what is considered edible and what is not in the specific context is essential in LCA studies trying to account for impacts within the food supply chains. A further distinction between avoidable and unavoidable FL has been proposed in the report "Household food and drink waste in the UK" (WRAP, 2009). The concept of "possibly avoidable" FL is put forward as the amount of food that some people eat and others do not, or food that can be eaten when it is prepared in some particular ways. Although the distinction was initially thought only for FL at household level, this can also be applied to food processing in which an edible part of food is discarded due to specific process characteristics. For example, the production of olive oil generates pomace (Fantozzi et al., 2015), a possibly avoidable loss that would have not been generated if the olives were consumed fresh. Hence, possibly avoidable FL is within the scope of the present work. It is recommended to make the distinction among 'avoidable', 'not avoidable' and 'possibly avoidable' FL in LCA studies, especially when results are used to analyse decisions about a decrease in FL and FW. Indeed, the reduction of the three kinds of losses should be obtained by different kind of interventions. The 'avoidable' FL, for example can be reduced by increasing consumer awareness, whereas the decrease of 'possibly avoidable' FL for a given product can be realised by improving the efficiency of the transformation process and gastronomical habits. Furthermore, this classification is crucial when analysing FW prevention scenarios (Bernstad and Canovas, 2015). Different components of FL are summarized in Fig. 2.

3.1.3. "Prevented" food loss

Within the European Waste Framework Directive (EU, 2008), waste prevention is the most preferable option for waste management. In LCA studies, very few examples included an assessment of the impacts and benefits of waste prevention (e.g. Gentil et al., 2011; Nessi et al., 2012). Cleary (2010) proposed a model to include waste prevention in the LCA of municipal solid waste management systems, but there is still no consolidated approach to include waste prevention in LCA studies on products. A possible way to account for FL prevention at a product level could be to compare different scenarios for FL prevention with a baseline (see e.g. Nessi et al., 2012). However, the inclusion of waste prevention in LCA is still at an embryonic phase and it implies the adoption of a different approach compared to "generated" FL.

Furthermore, FL prevention was not considered as part of the FL definition, therefore, it was out of the scope of the present analysis.
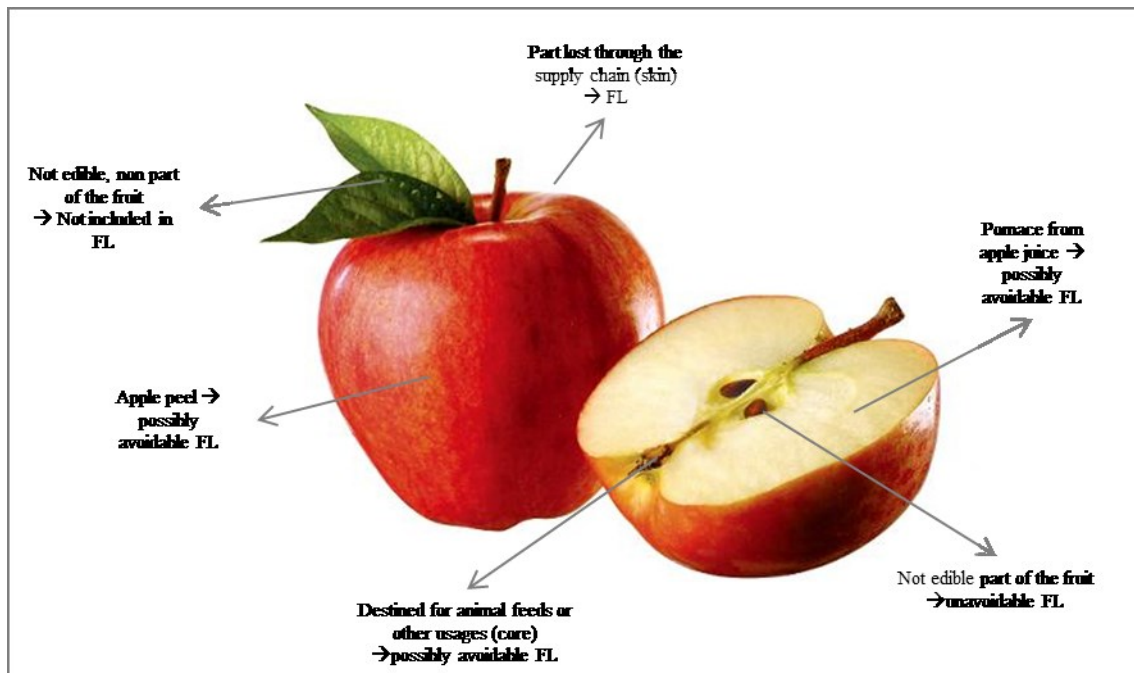

Fig. 2: Representation of different types of FL applied, as example, to an apple. Each food category will have a different split. Splits may also change based on local cultural and/ or consumer habits

3.1.4. "Over-eaten" food

Smil (2004) reported that in high-income countries part of the food produced in excess is consumed beyond human needs. If not combined with a proper physical activity, it can lead to obesity, already known as an important social and health concern. Over nutrition, i.e. food eaten beyond nutritional needs, is a rather controversial subject and FAO (2014b) decided not to retain this possibility in its accounting. No methodological consolidated approach currently exists to include over nutrition in LCA applications. As for waste prevention, over-nutrition was considered outside of definition of FL and therefore out of scope of the present study.

3.1.5. "Qualitative" food loss

Qualitative FL consists in a decrease of food attributes such as nutritional value, economic value, food safety and consumers' appreciation. According to (FAO, 2014b), "qualitative FL" should be considered when accounting for the total FL. From a LCA perspective, the quality of the food can be related to the function of the system analysed and to the choice of the functional unit. However, not all the food attributes can be measured objectively and there is a vivid discussion for the choice of the most appropriate functional unit for food products (e.g. Sonesson et al.,

2015). For these reasons the assessment of qualitative FL was excluded from the present analysis. However, it is highlighted that some qualitative aspects of food can be relevant in the LCA for the definition of the functional unit or in the modelling of co-products (e.g. via system expansion).

3.2. Accounting of food loss in LCA

The generation of FL can be considered as an "inherent" component of the FSC. Indeed, over-production is a current practice since producers have to cope with adverse weather conditions or with fluctuant market demand. Up to 30% overproduction contributes to guarantee food security, however the current level of food overproduction in high-income countries is far more higher, threatening in fact global food security (Papargyropoulou et al., 2014). FL happens in all life cycle stages and varies greatly according to different elements, e.g. the type of food, the specific socio-cultural and economic contexts, the technological availability, the geographical location etc (FAO, 2011a). Table 1 reports a summary of the main FL that can occur within the FSC. The table can be used by LCA practitioners in the identification of the most important FL according to the specific context of their study.

The generation of FL within the FSC influences the potential impact of a food product for two reasons: the increase of food production in order to deliver the same amount of food and the generation of an additional environmental burden due to FL treatments (FAO, 2013). Different elements characterise the inclusion of FL in LCA and can lead to the adoption of inhomogeneous methodological approaches among LCA studies. In the next sections possible approaches to account for FL occurring at the different stages of the FSC are presented. The modelling of FL recovery processes will be discussed separately (section 3.3) since this transversally affects different stages of the FSC.

3.2.1. Food loss at the primary production stage

In conventional open-field agriculture, a part of marketable (intended over production to cope with market fluctuation) and non-marketable food (e.g. not fitting marketing standards) can be left on the field (Strid and Eriksson, 2014) or incorporated into the soil (e.g. Romero-Gamez et al., 2014). This practice is not common for crops cultivated into greenhouses, in which the excess of food has to be removed from the soil (Battistel, 2014; Cellura et al., 2012).

Table 1: Possible FL per FSC stage. Built from (FAO, 2013; Parfitt et al., 2010) and complemented with other information in literature

	Crops	Animals and animal products
Primary production	- Not-harvested edible products - Edible products left in the field - Edible product harvested but not sold - Rotten fruit or vegetables - Product damaged by machines	- Dead animals during breeding - Milk lost due to animal diseases - Discarded fishes
Transport and storage	- Spilled product - Product damaged due to bad handling - Product damaged by machineries - Product store at a wrong temperature	- Food lost during transport to slaughterhouse - Food lost due to bad storage
Processing	- Process FL (e.g. inefficiencies, contaminations…) - Possibly avoidable FL - Unavoidable FL (e.g. skins, seeds, etc.) - Food damaged by inappropriate packaging	- Process FL (e.g. inefficiencies, contaminations, etc.) - Possibly avoidable FL - Unavoidable process FL (e.g. bones, leather, etc) - Food damaged due to inappropriate packaging
Distribution	- Food damaged due to lack of cooling, storage facilities, - Expired food - Unsold food - Rejected food after quality controls	*As for Crops*
Consumption	- Food damaged due to the lack of storage facilities - Due not eaten due to the preparation of excess of food - Food not eaten due to passed expiration date - Food not eaten due to inappropriate packaging size (more food than the quantity wanted) - Food not eaten due to low consumers' appreciation - Unavoidable FL (e.g. fruit kernels, bones etc.)	*As for Crops*

It was observed that in some LCA studies on agricultural products the environmental burden of discarded rotten fruit and vegetables was charged to the functional unit, referring to the net yield (e.g. Mogensen et al., 2015) or to the marketable yield (e.g. Romero- Gamez et al., 2014). Over-production was discussed just in a few studies (e.g. Romero-Gamez et al., 2014; Strid and Eriksson, 2014). As highlighted by Lal (2008) crop residues can contribute to cycle nutrients and enhance the soil quality. These elements can be relevant from a LCA perspective, in terms, for example, of additional inputs that has to be provided to the field. However, only few evidence of this accounting in LCA studies was found. For example, Cerutti et al. (2014) confirmed in their literature review on LCA applied to the fruit sector that FL at the agricultural stage was not addressed in the papers they analysed. Blengini and Busto (2009) reported that benefits associated with the incorporation of agricultural residues into the soil were indirectly taken into account

since the crop under study was cultivated on a soil with better properties. A reduction of the input of nutrients to be provided to the soil due to residues left on the field, instead, was considered in the datasets referred to European agricultural production systems of the database Ecoinvent (Nemecek and Schnetzer, 2011a). Furthermore in the databases Ecoinvent and Agrifootprint the emissions due to crop residues decompositionwas assessed (Blonk Agri-footprint BV, 2014; Nemecek and Schnetzer, 2011b). Alternative destinations for FL at the primary production stage can be the composting or the anaerobic digestion, especially for FL generated into greenhouses (Cellura et al., 2012). This will be discussed in detail in section 3.3.2. Concerning the manufacturing of meat and livestock-derived products, no evidence was found on the inclusion of FL at the primary stage in LCA. However, the amount of this FL can potentially become significant. FL could be associated with animal's mortality and diseases and refuse of animals' products due to quality standards. The world organisation for animal health estimated that mortality and morbidity due to animal diseases caused the loss of at least 20% of livestock and livestock-derived production globally (World Organisation for Animal Health, 2015). Therefore, the exclusion of animal loss from the breeding system could lead to an underestimation of its environmental burden. In case of fisheries in open sea, by-catch may represent a significant cause of FL. By-catch is catch that is either unused or unmanaged and it is therefore discarded after sorting. It includes fishes that are fit for human consumption and could be sold, but also fishes that, for regulatory or economic reasons, are not sold (Davies et al., 2009). Different options exist to account for by-catch, affecting the comparability of data (Davies et al., 2009; FAO, 2013). Furthermore, the amount of discard is dependent from the context, namely: the season, the type of fishing method, the target species and the fisherman behaviour (Hornborg et al., 2012). These aspects make it difficult to obtain detailed data (Vazquez-Rowe et al., 2012). Discarded by-catch fish is an important environmental concern for fisheries since, together with the overfishing of a target specie, represents a threat for the equilibrium of aquatic ecosystems (Davies et al., 2009; Emanuelsson et al., 2014; Eyjolfsdottir et al., 2003). So far, different LCA studies have included the by-catch (e.g. Almeida et al., 2014; Ziegler et al., 2003). Besides, commonly used life cycle impact assessment methods are not addressing comprehensively the impact on the environment of fishing activities. Hence, LCA practitioners have developed some specific indicators to account for the impacts of discards during fishing. An example of an indicator is the amount of discarded by-catch (Davies et al., 2009). However, this indicator can underestimate the real impact on the marine biotic resources. For example, juveniles, often discarded after being by-caught, have a small mass but may have a large ecological relevance. Other indicators have been developed to capture the complexity of this aspect, taking into account specific geographical and temporal aspects (e.g. Emanuelsson et al., 2014. Hornborg et

al., 2013). Table 2 provides a description of these indicators, including a description of their strengths and weaknesses.

Table 2: Indicators used to account for the impact of discards in LCA studies on fisheries

Indicator	Description	Strengths	Weakness
Total Discard (TD)	Ratio between the mass of the discarded fishes and the functional unit	Gives a general idea of the amount of discarded fish	Mass is not representative of the ecological value of discarded fishes. Juveniles or rare species, for example, could represent a small contribution in term of mass but play a fundamental role in the function of the ecosystem. (Davies et al., 2009)
Primary Production Required (PPR) of discards	Fraction of carbon, used by photosynthesis to produce a kilogram of biomass in the population of a species at a certain tropic level, associated with the discarded fish,	Representative of the amount of nutrients wasted	In highly eutrophic ecosystems it could be not very significant (Emanuelsson et al., 2009). Furthermore it does not account for the ecological value of the discards due to its trophic level (the lower is the trophic level, the lower is the PPR, but the higher may be the ecological value) (Hornborg et al., 2013)
Threatened fish species in discards (VEC)	Amount of threatened fish species in discards	Proxy of the impact on the ecosystem	Difficult to have primary information on the composition of discards (Emanuelsson et al., 2009)

3.2.2. Food loss at transport and storage stage

In the analysed LCA studies, there was no evidence of accounting for FL during transport of food from the production place to the storage and during storage. However, FAO (2011a) reported that this contribution can potentially be relevant - especially in developing countries - and that it depends from the food categories. For example, FL during postharvest handling and storage of roots and tubers in South and Southeast Asia was estimated to be 19% of the food produced. FL of meat in the same FSC stage and in the same geographical area, instead, was estimated to be equal to 0.3% (FAO, 2011a). Consequently, the exclusion of FL at the transport and storage stage, in some contexts, could lead to an underestimation of the environmental burden of food products.

3.2.3. Food loss at the food processing stage

The processing stage can potentially generate three kinds of FL, mainly due to: (1) inefficiencies of the processing stage or overproduction (avoidable FL); (2) specific production processes of the commodity (possibly avoidable FL); (3) parts discarded because not edible (unavoidable FL). Avoidable FL was explicitly reported only in a few studies. Koroneos et al. (2005), for example,

reported beer losses during bottling and Kim et al. (2013) accounted for food loss at each stage of the FSC of cheese. Possibly avoidable and unavoidable FL, instead, were reported in a higher amount of studies in which they implied a relevant reduction of the output compared to the raw ingredient used (e.g. Coltro et al., 2006; Manfredi and Vignali, 2014; Rajaeifar et al., 2014; Röös et al., 2011). These kinds of losses are strictly related with the type of food and the type of processing and are less dependent from the efficiency of the process. Depending on the process, the amount of losses can be relevant and the modelling approach adopted to account for the environmental burden can considerably influence the LCA results, as highlighted in section 3.3. Indeed, according to the specific process, different destinations can be planned for FL at the processing stage: FL may undergo a recovery in another industrial process or may be treated as a waste with the potential recovery of resources or energy. A common recovery option for process losses is animal feeding (e.g. Grönroos et al., 2006; Jensen and Arlbjørn, 2014; Koroneos et al., 2005). Other possible destinations are fertilisation (e.g. Coltro et al., 2006) or other industrial ecology (IE) applications (e.g. Nucci et al., 2014). FL can be also recovered in downstream with human feeding purposes (e.g. Svanes and Aronsson, 2013). In some cases, FL at the processing stage is disposed without any recovery (e.g. Gonzalez-García et al., 2013).

3.2.4. *Food loss at distribution stage*

FL at the distribution stage can be generated both at the wholesale, due to handling and rejections after quality controls, and at the retail, due to unsold products (Strid and Eriksson, 2014). As for previous stages of the FSC, FAO (2011) highlighted that the type of food and the country where it is distributed have a relevant influence on the amount of FL generated. A large number of the analysed LCA case studies adopted an approach from cradle to gate, therefore FL generated in the distribution stage was not considered (e.g. Cordella et al., 2008; Fantozzi et al., 2015; Humbert et al., 2009; Röös et al., 2011). Others, instead, accounted for FL at the distribution stage: primary data (e.g. Svanes and Aronsson, 2013), specific assumptions (e.g. Andersson et al., 1998) and national statistics (e.g. Meier and Christen, 2013) were the sources of data used for the amount of FL at distribution. Adopting a cradle to grave perspective allows LCA practitioners to have a complete overview of possible consequences of choice taken within the FSC. In this stage of the FSC, FL was generally assumed to be managed as waste and, consequently, to be sent to waste management treatments (e.g. De Menna et al., 2014; Jensen and Arlbjørn, 2014).

3.2.5. Food loss at the consumption stage

FL at consumption stage is a major environmental issue in industrialised countries whereas is relatively limited in developing ones (FAO, 2013). Vanham et al. (2015), for example, showed that in Europe the quantity of food wasted is directly correlated with the total expenditure of the household: rich countries waste more food (e.g. UK with 190 kg/cap/year) while poorer countries waste less (e.g. Romania with 55 kg/cap/year). Besides, FL generation at consumption is also influenced by cultural aspects, due to e.g. different preparations and different eating habits of consumers (Parfitt et al., 2010). Among analysed studies, some LCA focused on diets and meals considered the generation of FL at the consumption stage (e.g. Davis and Sonesson, 2008; Meier and Christen, 2013). Sometimes FL is estimated as difference between per capita agricultural supply data and consumption data of actual intake level (e.g. Hallström et al., 2015). However, this approach does not distinguish among the contribution of the different FSC stages. Studies focused on single food products, instead, seldom considered the consumption stage within the system boundaries (e.g. Andersson et al., 1998; Jensen and Arlbjørn, 2014). Data for waste generation in the LCA studies analysed were mainly derived from national data (e.g. Meier and Christen, 2013; Schmidt Rivera et al., 2014; Svanes and Aronsson, 2013). WRAP reports were frequently cited (WRAP, 2013, 2009). Although they reported FW generation per category of food commodity in the UK, data therein were also used in studies that considered consumption elsewhere (e.g. Svanes and Aronsson, 2013). Other sources of data were national statistics (e.g. Meier et al., 2014) or assumptions, when specific local data were not available (e.g. De Menna et al., 2014). At EU level, a recent study (Vanham et al., 2015) has accounted for both total and avoidable waste per country - based on data of some representative countries - as well as the water and nitrogen footprint associated with the consumer FW. It could be an interesting source of data for LCA practitioners to account for FL. At this stage of the FSC, FL are generally managed as organic waste, collected separately or with municipal solid waste according to the specific waste management systems. Analogously to other previously considered stages, FL at the consumption stage can be addressed to different processes, such as composting or incineration (e.g. Berlin, 2002).

3.3. Modelling of food loss recovery in LCA

FL generated at different stages of the FSC can be processed for different purposes, depending on the type of loss and the context. FL can be recovered in other production processes, generally defined as IE applications, or it can be disposed or recovered through waste treatment technologies (e.g. composting, incineration, anaerobic digestion or landfilling) (FUSIONS EU Project, 2015). From a LCA perspective, the modelling of IE applications can be considered

analogous to the modelling of waste treatments. Indeed, both these systems treat FL and produce useful outputs. FL represents therefore a co-product of the system and this has to be modelled with the common approaches dealing with multi-functionality, namely system expansion and substitution, and allocation (Pelletier et al., 2015).

Fig. 3 illustrate a summary of the approaches adopted in the analysed studies.

3.3.1. Recovery of food loss in industrial ecology applications

IE is a set of principles, tools, and perspectives derived from ecology and adapted to industrial systems (Lowenthal and Kastenberg, 1998). The principles of IE are applied to design or redesign industrial systems to create more efficient interactions both within industrial systems and between industrial systems and natural systems (Leigh and Li, 2015). IE applications are generally based on the interrelationships of firms that exchange a variety of materials - including residues and waste - and energy flows to feed different production processes (Ardente et al., 2009; Niutanen and Korhonen, 2003).
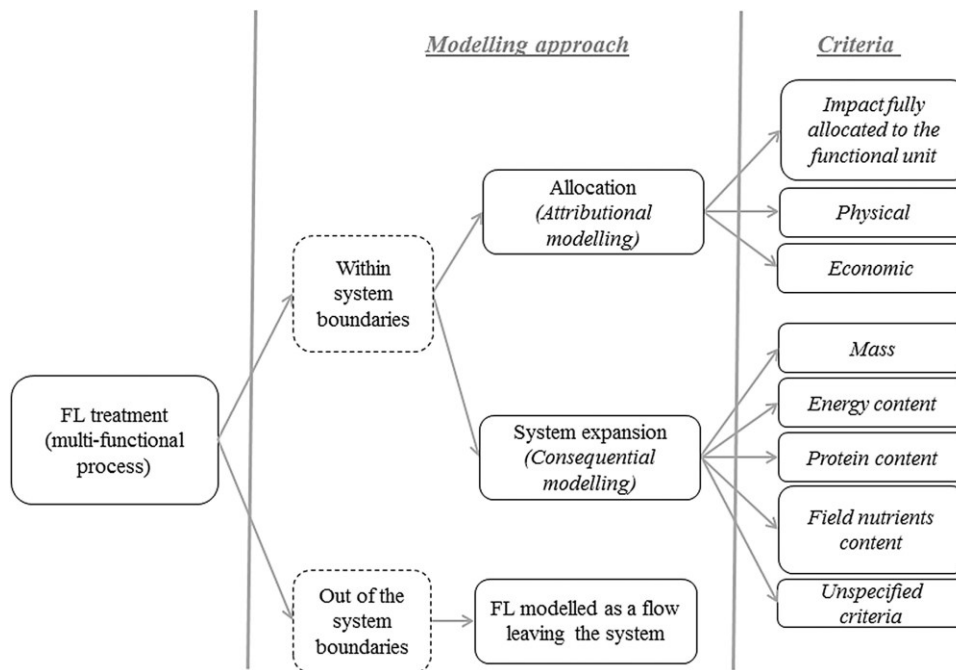
Fig. 3: Approaches adopted in the analysed studies to model the treatments of the FL. The graph refers to both industrial ecology applications and waste treatments.

The quantification and characterisation of FL and FW along the FSC have been proved to be crucial for the identification of potential new IE applications (Mirabella et al., 2014). Moreover, Svanes and Aronsson (2013) illustrated that IE applications can be used to recover FL into innovative food productions, e.g. baby food. In this case, recovered materials do not represent

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anymore a FL since destined to human consumption. However, the benefits of FL recovery should not be undermined by the environmental impact caused by IE production processes (Mirabella et al., 2014). To such purpose, LCA can be applied with different aims, for example to (Mattila et al., 2012): assess the benefits of realising IE applications (e.g. Chiusano et al., 2015; San Martin et al., 2016; Simboli et al., 2015); assess existing IE applications to improve them (e.g. Contreras et al., 2009); communicate to third party the performance of IE systems (e.g. Schau and Fet, 2008); compare IE applications with traditional industrial processes (e.g. Duchin, 2005; Iribarren et al., 2010). Several LCA practitioners analysed the recovery of FL in different industrial sectors, mainly: animal feeding (e.g. Cordella et al., 2008; Koroneos et al., 2005; San Martin et al., 2016); cosmetics production (Nucci et al., 2014; Secchi et al., 2016); fertilisation (e.g. Fantozzi et al., 2015; Notarnicola et al., 2011; Salomone and Ioppolo, 2012). Examples of IE applications are however very wide, including that some authors discussed some applications without specifically mentioning these as IE (e.g. Secchi et al., 2016). As mentioned in section 3.3., critical aspects concerning the modelling of FL in IE applications are: i) the definition of the system boundaries; and ii) the modelling of multi-functionality. The definition of the system boundaries is crucial to assess what is included or excluded from the LCA. This is particularly the case of assessment of IE applications, since two or more industrial subjects, generally very different in processes and characteristics, are involved. In turn, these industrial subjects could have other byproducts utilised by other industries, in a complex network that have to be truncated at a certain point. According to Mattila et al. (2012) supply chain impacts are usually excluded from the analysis of IE, hence introducing the risk of transferring impacts from the studied system to elsewhere in the supply chain. On the other hand, the enlargement of the system boundaries implies higher uncertainties, data availability and data quality issues. For example, this is the case of industrial symbiosis application in which a system of two or more entities exchanges energy and materials for the mutual benefit (Chertow, 2000). Few examples of LCA applied to industrial symbiosis systems have been discussed in the literature (e.g. Eckelman and Chertow, 2009.; Mattila et al., 2010; Sokka et al., 2011) but none specifically focused on food industries has been identified in our analysis. Applications of hybrid and Input-Output LCA have been proposed as worth of note to capture the complexity of industrial symbiosis systems (Mattila et al., 2012, 2010). The application of system expansion to solve multi-functionality problems implies the selection of a "reference case" for the substitution, in which emission credits are given from the substitution of alternative production processes than those in the IE application (Mattila et al., 2012). Criteria for substitution are not always univocal, meaning that different approaches can be applied for the same case-study. For example, apple residues can be used for different IE applications, such as fuel production, pectin extraction, cattle feed,

biotransformation and sources of fibres (Mirabella et al., 2014). Substitution criteria should be carefully investigated and discussed considering all the possible applications. However, the description of the approach used for the substitution is sometimes not sufficiently detailed or lacking (Mattila et al., 2012; Pelletier et al., 2015). The selection of a not representative "reference case" for the substitution implies the risk of overestimating the benefits of byproduct exchange (Mattila et al., 2012). Moreover, substitution could be improperly applied to lower 'artificially' the impacts of the studied product. On the other hand, it is recognised that the application of system expansion implies some advantages, as being this able to assess indirect land use changes due to some avoided agricultural production (Schmidt et al., 2015).

The allocation of impacts among co-products can be performed according to different approaches: physical allocation (e.g. Gonzalez-García et al., 2013; Rajaeifar et al., 2014), economic allocation (e.g. Ayer et al., 2007; Hospido et al., 2003), or impact allocated entirely to the functional unit (e.g. Mila i Canals et al., 2006). The allocation procedures can have a relevant influence on the results of the study (Cederberg and Stadig, 2003). Despite the ISO 14040 (ISO, 2006a) hierarchy suggests the selection of physical criteria as the preferred option for allocation, economic allocation is often applied to LCA, especially for those related to the agro-food sector where a large quantity of low-value by-products are generated (Ardente and Cellura, 2012). For example, cow slaughtering produces meat and animal by-products (e.g. innards, fat, skin), the latter normally utilised in IE applications for various productions. By applying physical allocation (e.g. with criteria as mass or energy content) these by-products could have a high impact. On the contrary, the application of economic allocation would imply byproducts to have a low share of the impacts due to their limited economic value. Recently the FAO (2016) suggested to perform economic allocation to partition the environmental burden between meat and animal by-products. In this sense, the application of economic values to allocate impacts has been recognised as a driving force for the promotion of new IE applications for the recovery of FL (Weinzettel et al., 2012). On the other hands economic allocation is affected by limitations, mainly that it produces results that reflect existing market relationships that can potentially change (via price ratios) rather than the physical relationships (Pelletier et al., 2015) and that economic values are affected by a multitude of factors not strictly related to the effective emission of the studied system (Ardente and Cellura, 2012).

3.3.2. Treatment of food loss in waste plants

Several articles accurately analysed the environmental performance of different options for the waste treatment (e.g. Laurent et al., 2014a; Bernstad and la Cour Jansen, 2011). In some articles with the focus on food, instead, it was observed a low detail provided for the modelling of FL

and FW recovery and/or disposal (e.g. Gonzalez-García et al., 2014; Meier and Christen, 2013). This can be explained by the prejudice that the end-of-life stage is of relatively low relevance compared to the environmental impacts generated along the FSC. However Manfredi et al. (2015) suggested that decisions, choices and assumptions related to the waste treatment (e.g. the decision context and the choice of the impact assessment indicators) can exert an important influence on the results of the LCA.

FL occurring at the different stages of the FSC can be treated by incineration, composting, anaerobic digestion and landfill. According to the ILCD Handbook for LCA (EC-JRC, 2010) waste are part of the 'technosphere' and, therefore, they should not be considered as elementary flows leaving the analysed product system. This means that the system boundaries of the studied system should include the waste treatment, accounting all the processes until elementary flows cross the system boundaries as emissions to the ecosphere. However, not all LCA practitioners followed these recommendations. Some authors did not account for the environmental burden of FL management treatments either because they excluded them from the system boundaries (e.g. Ardente et al., 2006; Gonzalez-García et al., 2014) or because they considered FL management treatments as a negligible source of emissions (e.g. Saarinen et al., 2012). Other studies accounted for the environmental burdens of waste treatments, however they adopted different modelling approaches. For example, Svanes and Aronsson (2013) referred to IPCC to account for emissions of methane from the landfilling of banana FL, whereas Jensen and Arlbjørn (2014) referred to a combination of information derived from different sources to model the incineration of uneaten food. A high detail in the characterisation of the waste it is also necessary for a precise modelling of the waste treatments. Waste composition may greatly influence the performance of the waste plant regarding, for example, the quality and quantity of nutrients recovered through anaerobic digestion or the amount of energy recovered by the incinerators (Bernstad and la Cour Jansen, 2012). The use of generic or unspecified data for the modelling of waste treatments can lead to misleading results. For example, Gruber et al. (2014) modelled the incineration of unconsumed with data concerning the incineration of mixed municipal solid waste. Successively Gruber et al. (2014) concluded that incineration was preferable than composting, concerning the eutrophication, acidification and primary energy demand impact categories. However, this result is in contrast to other specific studies, as in Arafat et al. (2015), which reported that incineration was not the best environmental option for FW management. Conclusions by Gruber et al. (2014) could be affected by the assumption concerning the modelling of waste with not representative data. As for IE applications, the modelling of FL treated in waste plants implies multi-functionality problems to be solved through allocation or system expansion. Laurent et al. (2014) highlighted a general confusion about this

distinction and found several inconsistencies among LCA studies on waste management systems. This applies also to LCA of food products, which did not model the multifunctionalities consistently with the overall modelling approach (i.e. attributional or consequential). It has been also observed that the modelling approach adopted for the waste treatment is not always explicitly reported (e.g. in Fantozzi et al., 2015). Moreover the present analysis of the literature did not identify any application of allocation criteria to the modelling of FL, with the exception of the environmental burdens of the waste treatment entirely allocated to the functional (e.g. Svanes and Aronsson, 2013). On the other hand, system expansion was the modelling option most commonly observed in the literature. These applications accounted the impacts of waste treatments together with credits due to the avoided production of certain substituted commodities. For example, energy outputs from incineration or anaerobic digestion plants were credited as energy from fossil fuels (e.g. Davis and Sonesson, 2008; De Menna et al., 2014); nutrients from anaerobic digestion or composting were credited as fertilisers from conventional production plant (e.g. De Menna et al., 2014; Salomone and Ioppolo, 2012). However, the reasons for the avoided production and the detail of credited impacts are sometimes lacking or not sufficiently discussed (e.g. how credits are assigned for avoided production associated with the use of by-products as fertilisers). More importantly, assumptions related to the system expansion can largely affect the LCA results. Indeed, secondary datasets modelling the same products can lead to highly different environmental burdens (Peereboom et al.,1998). On the other hand, it is recognised that in some cases it is difficult or even impossible to provide a detailed analysis of the substituted system, since it is not known in advance where and how waste will be treated. This is recognised as a limit of the system expansion approach.

4. Discussion

The analysis of the relevant literature on the inclusion of FL and FW within the LCA studies highlighted some shortcomings, which can potentially affect the LCA results. Indeed, Manfredi et al. (2015) reported that the lack of homogeneity among key factors and assumptions can justify differences among LCA results, rather than differences among the environmental performance of waste treatments. In order to strengthen the use of LCA for the assessment of initiatives aimed at FL minimisation and sustainable management, it is necessary to have a shared framework on how to account for FL.

Based on the analysis of the relevant literature, some recommendations for LCA practitioners were derived to open the way towards a harmonisation of the approaches to account for FL in LCA.

A first general recommendation is to use a transparent reporting of the key assumptions for the modelling. Indeed, the lack of a clear description generally represented a limit for the studies analysed, nevertheless of their robustness. This recommendation can be seen as general enough to be applied to all type of LCA studies and to all phases. However, according to the present analysis this is particularly critical for the modelling of FL and for the correct interpretation of results. The lack of transparency, in fact, negatively affects the reproducibility and comparability of the presented results.

Moreover, it is suggested to LCA practitioners to consider aspects related to FL within all the FSC, starting from the preliminary phases of the LCA study, e.g. already during the definition of the system boundaries and the product system to be analysed. Also this can be seen as a general LCA recommendation, since cut-offs should be avoided or, at least, clearly motivated. However, a general tendency of LCA practitioners to underestimate the potential burdens of FL was observed. The discussion of FL aspects in the LCA and the explicit accounting of FL generated at each stage of the FSC would allow a more transparent picture of the impact of the analysed product.

The environmental burden of FL generation and management, especially in the primary production stage, can only partially be considered through the analysis of the commonly considered impact categories. Indeed, elements such as the enhancement of soil quality due to residues left on the field and by-catch during fishing are only partially captured by "traditional" impact categories. Therefore, LCA practitioners are recommended to identify and select indicators and impact categories that can be important according to the specific context, also trying to go beyond common LCA categories.

The distinction between avoidable, possibly avoidable and unavoidable FL can help in defining a comprehensive overview of all the FL that happen within the FSC and can be useful to support actions aimed at FL reduction and prevention. LCA practitioners are therefore recommended to systematically account in their LCA studies on food, three additional indicators as:

1) $Avoidable\ FL = \sum_i Avoidable\ FL_i$ (with 'i' lifecycle stage)
2) $Unavoidable\ FL = \sum_i Unavoidable\ FL_i$ (with 'i' lifecycle stage)
3) $Total\ FL = Avoidable\ FL + Unavoidable\ FL$

It is suggested to report transparently the amount of each indicator, the sources of data and the related assumptions. These indicators do not represent per se an index of the potential impact of FL generated along the supply chain. However, since LCA aims to provide exhaustive information on impacts along the life cycle, this information could be crucial for decision-makers

in taking informed choices to optimise FSC and finding sustainable solutions to "feed the planet". In order to calculate the aforementioned indicators, it is essential to clearly define which part of food has to be considered edible, according to the specific context. It is suggested that LCA practitioners specify the amount of edible food and indicate whether it is included or not in the functional unit. Indeed, a certain amount of some kind of food, such as melon, bananas, or cheese with crust, can include a large inedible fraction that will become unavoidable FL or possibly avoidable FL in the processing or consumption stage. The information on the edible parts can be particularly relevant for comparative studies among different kinds of food, or among different studies of the same food product but with different characteristics.

A cradle to grave approach should always be preferred since studies limited to the company gate can miss some important aspects (e.g. choice of packaging), which can influence the FL generation and their consequent impacts in the following FSC stages. Despite the destination of FL, LCA practitioners are recommended to set the system boundaries in such a way that emissions from FL treatments are accounted within the environmental burden of the functional unit.

Multi-functionalities should be modelled coherently with the specific decision context (attributional or consequential). If primary data on the waste destinations are not available, the most representative data should be considered, according to the specific geographical and technological context. Moreover, impacts of the waste treatment plants generally refer to processes where heterogeneous waste is treated. As discussed for the modelling of the waste management treatments by Bernstad and la Cour Jansen (2012) and Laurent et al. (2014), the characteristics of FW can importantly influence the performance of the waste treatment in terms of potential nutrients or energy recovery and in terms of environmental emissions and are sources of uncertainty in LCA results (Mendoza Beltran et al., 2016). Consequently, LCA practitioners are recommended to check the representativeness of secondary inventory data used to model the treatment of FL and to model waste treatments coherently with the characteristics of the specific FL they are considering. This can be particularly relevant, for example, when using average data about incineration or anaerobic digestion. If the waste treatment delivers more co-products, the way in which multi-functionality is dealt should be transparently described (allocation or system expansion). In particular, when allocation is applied, practitioners should clearly state i) the allocation criteria and ii) the allocation factors; when the system expansion is applied, practitioners should report i) the substitution criteria, ii) the amount of product substituted, and iii) the accurate description of the product system substituted (e.g. sources of data). This recommendation can be seen as very general, since applicable to all LCA applications. However, it was observed that this is particularly crucial for food products, since these generally

have a large number of outputs, including FL. Since the modelling of multi-functionality has a relevant influence on the results and a single criteria is generally not representative of all the complex characteristics of the co-products (Ardente and Cellura, 2012), it is suggested to LCA practitioners to perform a detailed analysis of the representativeness of the adopted substitution criteria. Although ISO standards (ISO, 2006a, 2006b) on LCA recommend the sensitivity analysis of allocation procedures, it was observed that this was generally missing for studies on food. Therefore, it is suggested that LCA practitioners should consider in their study at least a "pessimistic" scenario for the sensitivity analysis of the FL modelling. In this scenario the burdens of the waste treatments could be entirely allocated to the functional unit, without accounting for any potential credits due to substituted coproduct. A final recommendation on the waste treatment modelling is, whenever possible, to model multi-functional processes with commonly agreed procedures, as for example, procedures adopted by the large majority of studies in the literature, or as recommended by product category rules, as those recommended by the EU Product Environmental Footprint (EC, 2013). This would largely improve the comparability among several studies about the same product.

All the recommendations here illustrated have been summarized in Table 3. This table firstly introduces the critical methodological aspects observed in the present analysis, i.e. aspects that can generate mistakes or problems of interpretation and comparability of the results. For each identified critical aspect the rationale for it being critical is clarified. Successively, for each critical aspect, some recommendations for LCA practitioners and the reasons why those recommendations are considered important to move towards the definition of a common framework to account for FL in LCA are listed.

As final remark, it is highlighted that over-eating aspects were not considered in the present study due to lack of inclusion in LCA studies. However, over-eating can represent a hotspot from an economic and social point of view. Therefore, it is suggested to further explore this aspect, especially in studies dealing with the evaluation of the economic and social sustainability of food systems.

Table 3: Summary of the critical aspects observed in the present study and of potential recommendations for LCA practitioners to handle them (* GS = goal and scope definition; I = inventory or data collection; M = modelling approach; R = reporting)

LCA stage*	Critical aspects	Rationale for criticality	Recommendation(s)	Strengths of recommendation(s)
GS, R	Systematic exclusion of FL from LCA	Partial assessment of the environmental burden of food	Include FL in LCA studies	Comprehensive analysis of the product system analysed
GS, I, R	Exclusion of some FL generated within the FSC	- Possible exclusion of relevant losses - Limited knowledge of the relevance of FL generated at different FSC stages	Introduce in the LCA framework three indicators for each stage of the FSC including: "avoidable FL", "unavoidable FL" and "total FL"	- Comprehensive analysis of the product system - Possible to perform a detailed contribution analysis (interesting e.g. when LCA is used as a decision support tool for food production strategies)
GS, R	"Traditional" impact categories capture only partially the effects of FL generation and management overall in primary stage production	Possible exclusion of relevant environmental impact of FL	Choose impact categories according to the specific context	Comprehensive analysis of the potential environmental consequences of FL generation and management
GS, R	Definition of edible part of food is strictly context –specific	The distinction of edible and inedible part of food is at the basis of the distinction among the different categories of FL	Clearly define which parts of food are considered inedible in the specific study	Allows possible comparison among product systems delivering the same function
GS, R	Approach from cradle to gate	Possible exclusion of correlations between the generation of FL and the products design (e.g. choice of packaging)	Prefer a cradle to grave approach	- Holistic analysis of the product system analysed - Wider knowledge of the FL generation dynamics
GS, R	Exclusion of waste treatment from the system boundary	Exclusion of potentially relevant burdens	Include waste treatments within system boundaries	Holistic analysis of the product system analysed
I, R	Use of secondary data to model waste treatments	Different characteristics of the waste can influence relevantly	Check the representativeness of the data used to model the waste treatment	Avoidance of having misleading results related to improper waste modelling

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LCA stage*	Critical aspects	Rationale for criticality	Recommendation(s)	Strengths of recommendation(s)
		the performance of the waste treatment		

M, R	Unclear description of the allocation procedure adopted to model FL and outputs of waste treatments	- Limited reproducibility of the study (ISO requirement)
		- Allocation procedures can have a strong influence on LCA results
			- Report clearly allocation procedures. Particularly, in case of allocation:
			  • Allocation criteria
			  • Allocations factors
			In case of system expansion:
			  • Substitution criteria
			  • Amount of product substituted
			  • Accurate description of the product system substituted
			- Assess the representativeness of the substitution criteria
			- Sensitivity analysis (including also the "pessimistic scenario" without any credits from the waste treatments)
				- Improved transparency of the study and reproducibility of the results
				- Better understanding of the influence of modelling choices on the results of the study

5. Conclusions

The clear definition and transparent accounting and the modelling of FL within LCA are essential for a comprehensive and detailed assessment of the environmental burden associated with the production of food products. This clarification is crucial especially when results of LCA studies are used to define policies and initiatives aiming at reducing the environmental impact of the agro-food system and, finally, aiming at achieving a sustainable supply of food.

According to the present analysis, so far FL has not been defined nor included systematically in LCA studies. When included, different approaches have been adopted, leading to potentially misleading consideration or non-comparable results. Therefore, in order to reinforce the reliability of LCA as a decision support tool, there is the need to develop a common modelling framework to account for FL within LCA. The analysis of the relevant literature was firstly intended to identify some shortcomings in the modelling of FL and to draw some recommendations to foster the systematic inclusion of FL generation and management within the boundaries of LCA studies and to move towards a common approach to account for FL. LCA practitioners are recommended to account for all the FL generated along the FSC stages. Other recommendations include: the definition of what is considered edible for the studied product, the inclusion of the waste treatments within the system boundaries and their modelling to be coherent with the specific composition of waste. It is highly recommended to perform a sensitivity analysis of the different approaches to model multi-functionalities derived from waste treatment, since these approaches can have a relevant influence on the LCA results. Moreover, a transparent description and discussion of the FL generated along the food FSC and of the related modelling approaches adopted is recommended, especially for the modelling of multi-functionalities. A systematic assessment of FL and FW is crucial also in light of identifying and applying IE principles and improving resource efficiency among different production chains and life cycle stages.

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5. Greenhouse gas emissions of three balanced dietary patterns

Abstract

In light of the considerable pressure exerted by food production on the environment, the assessment of the environmental burdens of dietary choices has recently gained interest among the scientific community. Several studies based on life cycle thinking approach agreed that a transition from an omnivorous to either a vegan or vegetarian diets would reduce the environmental impact associated with food consumption. The majority of the these studies set the system boundaries up to the retail, excluding the consumption phase and generally do not account for uncertainties. The aim of the present study was to assess the greenhouse gas emissions generated by three balanced dietary patterns (omnivorous, vegetarian and vegan), defined on the basis of nutritional recommendations for an average Italian man, including the consumption stage. It took into consideration the uncertainties associated with three elements, namely the greenhouse gas emissions due to the production of the food items, the emissions associated with cooking and the food wasted by consumers. The results of the study highlighted that, despite the higher share of greenhouse gas emissions of the supply chain stages prior to consumption, cooking and food waste generation, have an important influence of the total greenhouse gas emissions of the diet, which can offset the lower greenhouse gas emissions due to the choice of vegetable-origin foods. Therefore this study remarks the importance of adopting a cradle to grave perspective when assessing the environmental burden of dietary patterns and emphasises the central role of consumers in the definition of low GHG-emitting dietary patterns.

1. Introduction

Accounting for about 30% of the total greenhouse gas (GHG) emissions at the global level (Garnett, 2011), the entire food chain represents a considerable contribution to the global warming. Several institutions and researchers, indeed, have recognised that moving towards a less GHG-emitting food production and consumption system is of the utmost importance to preserve the capability of the Earth to produce food for the future generations (EC, 2011; Pachauri et al., 2015; Reisch et al., 2013). The analysis of the environmental burden of food production and consumption through life cycle assessment (LCA) - a methodology that addresses the environmental aspects and the potential environmental impacts of a product or service, throughout their life cycle (ISO, 2006) - has been rapidly raising interest in the last years. In particular, the growing availability of data on the environmental performance of single food items has allowed to adopt a wider perspective, assessing and comparing the impacts of a combination of foods consumed within a meal or a diet (Nemecek et al., 2016).

Modelling consumers' behaviour is a complex task due to the huge number of variables that characterises it, such as the type of food, the mode of preparation and the amount of food cooked (Nemecek et al., 2016).

The choice of the dietary patterns, e.g. omnivorous, vegetarian, vegan or other "healthy" diets, is the most investigated variable. The literature reviews realised by Hallström et al. (2015) and Heller et al. (2013) highlighted that the majority of the studies agreed that a transition from an omnivorous to a vegetarian or vegan diet can be beneficial for the environment in terms of GHG emissions and land use reduction, thanks to the lower impact of vegetable-origin foods compared to animal-origin ones. Another variable analysed is the environmental burden associated with the choice of seasonal or non-seasonal fruits and vegetables. Despite the lack of a shared definition of "seasonal food", it was generally intended as the food produced in the same country where it is consumed and, in some cases, grown in open field (Foster et al., 2014; Röös and Karlsson, 2013). Choosing seasonal fruit and vegetables was found to reduce the carbon footprint of consumptions patterns (Hospido et al., 2009; Röös and Karlsson, 2013; Stoessel et al., 2012), except for raspberries consumed in the UK (Foster et al., 2014). However, Röös and Karlsson (2013) and Hospido et al. (2009) highlighted that the consumption of only local fruit and vegetables could be economically not feasible and it would imply a substantial change in consumers' consumption habits. Webb et al. (2013), instead, compared a wider range of food items, including animal-origin ones, produced in the United Kingdom or imported from other countries. They found higher differences for foods for which the primary stage accounted for a small part of the total impact, however imported food could result to have a lower environmental

burden than local one if it was produced in countries where the productivity is far higher than in United kingdom. Furthermore, they highlighted that prioritising the impact on global warming may lead to an increase of other impacting burdens. Gruber et al. (2016) explored the effects of consumers' behaviour in relation to food waste generation and found that it influenced by 10 to 45% the environmental performance of food products. Therefore, they underlined the need to develop a common methodology to model systematically consumers' behaviour in LCA studies. Dolci et al. (2016) compared the amount of waste and the potential impacts of loose distribution of pasta, breakfast cereals and rice. They found that choosing to buy loose cereal breakfast or rice rather than packed ones could reduce the amount of waste produced and the environmental burdens of food, whereas contrasting effects can be obtained for pasta. Yoshikawa et al. (2016) analysed the potentiality of reduction of GHG emissions thanks to changes in personal behaviours and highlighted that the consumption of local food, the seasonal production of fruit and vegetables, the reduction of the use of fertilisers and of food packaging can lead to a reduction of the GHG emissions associated with food consumption.

Despite the vast increasing interest within the scientific community on the environmental burden of different dietary patterns and food choices, still different modelling challenges need to be faced in order to reach robust and comprehensive considerations (Nemecek et al., 2016; Notarnicola et al., 2017a; Sala et al., 2017). Several studies focused on the assessment of the environmental burden of dietary patterns excluded the consumption phase from their system boundaries (Hallström et al., 2015; Heller et al., 2013). However, although agricultural production generally is the main contribution to the environmental impact of dietary patterns (Pernollet et al., 2017), there is a general agreement that consumers' behaviour con represent an important contribution when assessing the environmental burden of diets (Gruber et al., 2016; Heller et al., 2013; Nemecek et al., 2016; Notarnicola et al., 2017a). Furthermore, the uncertainty that characterises the environmental impact of dietary scenarios is generally omitted, lwith the risk of incurring in misleading conclusions (Hallström et al., 2015; Henriksson et al., 2015).

The present study is an attempt to go beyond the two aforementioned shortcomings in the analysis of the environmental impact of dietary patterns. It aims to assess the GHG emissions of three average balanced dietary patterns (omnivorous, vegetarian and vegan) of an Italian man, taking into consideration the variability associated with the GHG emissions of single food items, the GHG emissions associated with cooking and the food waste generation. The results of the study can provide useful indications for decisions-makers and for environmental communicators about effective strategies to reduce the GHG of dietary patterns from a holistic perspective.

2. Materials and methods

The GHG emissions of three balanced dietary patterns for an Italian man were assessed: omnivorous, vegetarian and vegan. For each of them, the combined effects of three variables that characterise food purchase and consumption were considered: i) the consumers' attention for the GHG emissions of food products ii) the amount of energy used for cooking and iii) the amount of food wasted at the consumption phase. Their variability was modelled as explained in the following sections.

The average GHG emissions, the standard deviation and the 95% confidence interval of each dietary pattern was estimated with the software Simapro v 8.0.5 through a 1000-run Monte Carlo analysis. Furthermore a pairwise comparison was done in order to analyse the differences in GHG emissions of the dietary patterns.

2.1 Dietary patterns definition

For the purpose of this study, we have identified three different dietary pattern (omnivorous, vegetarian and vegan menu, Table 1) which are defined according to the reference amount assumption of energy and nutrient, recommended for Italian population by Italian Society of Human Nutrition (SINU, 2014a). All the dietary patterns are referred to an average Italian man, aged between 18 and 59, 170 cm in height, with a low physical activity level (SINU, 2014a). The average daily energy, proteins, fats, carbohydrates, iron and vitamin B12 intakes were calculated for each menu, considering the nutrients content of food items reported by the European Institute of Oncology (2016) or, in few cases, when data were available for the specific food item, other publications, consulted also for the GHG emissions (see section 2.2. All the dietary patterns are compliant with the recommended nutrients and energy daily intake (LARN) released by the Italian Society of Human Nutrition (SINU, 2014a) for all the nutrients considered except for vitamin B12, lacking in the vegetarian and vegan diets, and the carbohydrates intake of the vegan diet, slightly higher than the recommendations (Table 2).

Omnivorous dietary pattern

The omnivorous menu was at first determined according to nutrient and energy recommended intake (SINU, 2014a), and in second step by following food frequency described by the omnivorous food pyramid (Unità di Ricerca di Scienza dell'Alimentazione e Nutrizione Umana, 2005). The omnivorous menu included animal origin food consumption (such as meat, milk, cheese).

Vegetarian dietary pattern

The vegetarian menu was defined in accordance to energy and nutrient intake recommended by SINU addendum to LARN (SINU, 2014b). Because of the lack of a scientifically recognised vegetarian or vegan pyramid for Italian diet, the food frequency of vegetarian dietary pattern was adapted to omnivorous food pyramid recommendations. Meat and fish were substituted with non-animal protein food, such as eggs, cheese and legumes.

Vegan dietary pattern

Even the vegan menu was defined in accordance to energy and nutrient intake reported by SINU (2014b). As aforementioned, even in this case the omnivorous food pyramid was taken into account in order to define food frequency: legumes and vegetable milk substituted animal origin food. Biscuits were not included because data on GHG emissions was lacking for vegan biscuits. In order to guarantee the recommended carbohydrates intake, bread servings were increased with respect to those recommended by omnivorous food pyramid, the same was for legumes whose serving were increased in order to guarantee the right protein intake.

Table 1: Weight and number of foods portions considered for the three dietary patterns

			Weekly number of portions		
	Portions weight (g)	Omnivorous pyramid recommendations	Omnivorous	Vegetarian	Vegan
Bread	50	≤16	16	16	23
Biscuits	20	≤7	7	7	0
Pasta	80	≤8 (pasta+rice)	4	4	5
Rice	80		3	3	3
Potatoes	200	≤2	2	3	3
Fruits	150	≤21	21	21	21
Vegetables	80(fresh)/200(cooked)	≤14	14	14	14
Butter/margarine	10	≤5	5	5	5
Extra-virgin olive oil	10	≤20	20	20	20
Milk	125	≤14 (milk+yogurt)	4	4	21*
Yogurt	125		3	3	0
Fresh cheese	100(fresh)/50(ripened)	≤4	4	4	0
Meat	100	≤5	4**	0	0
Fish	150	≥2	3	0	0
Eggs	60	≤2	2	2	0
Legumes	150	≤2	2	7	9
Cured meat	50	≤3	3	0	0
Meat substitutes	180	n.d.	0	4	6
Sugar	5	≤21	21	21	21

*Intended as vegetable drink
** 1 beef meat, 1 pork meat, 2 poultry meat

		Energy (kcal/d)	Proteins (g/d)	Fats (% of kcal)	Carbohydrates (% of kcal)	Iron (mg/d)	VitaminB12 (μg/d)
Reference	Average	2350	63			10.0	2.4
	Min	1880	50.4	20%	45%	8.0	1.9
	Max	2820*	75.6	35%	60%	**	**
Provided by the menus	Omnivorous diet	2131	73.8	26.2%	48.5%	9.6	4.9
	Vegetarian diet	2133	68.2	23.6%	51.8%	14.1	1.4
	Vegan diet	1949	69.8	23.0%	63.0%	17.8	0.0

2.2. Greenhouse gas emissions of single food items

A database containing the average GHG emissions and the standard deviation of different foods consumed in Italy was developed. For each food item or group of foods, a sample of GHG emissions were collected from peer-reviewed articles and environmental product declarations (EPDs). The research was performed through the search engines Scopus (www.scopus.com) and the EPD database (www.environdec.com), using the keywords "LCA" and "name of specific food". Only studies in which the system boundaries were clearly defined were taken into consideration. In order to reflect the average GHG emissions of food actually consumed in Italy, the selection of the studies was based on available trade statistics related to each food item or food category. A detailed description of other selection criteria per each food or group of foods is reported in the next sections. All the GHG emissions were converted to a common functional unit, equal to the a kilogram of edible food. When not specified by the study itself, the average edible fractions of food items were taken from the database developed by the European Institute of Oncology (2016). The system boundaries were uniformed to the point of consumption. When the transport from the production to the distribution sites and from the distribution to the consumption sites and data on domestic conservation were excluded from the system boundaries, these stages were modelled respectively according to the Product Environmental Footprint (PEF) guidelines on default data to be used to model distribution and storage (EC, 2015) and the product category rules for product environmental footprint (PEFCR) for specific products (EC, 2013). The cooking phase was modelled as described in section 2.4. The management of food waste and waste from packaging was excluded from the assessment because different modelling approaches, e.g. attributional and consequential, lead to important differences

among the results (see Chapter 4). After having made uniform the system boundaries, the sample mean and the standard deviation of the GHG emissions associated with each food item or food group were calculated. It was assumed that the GHG emissions of each food item or category of foods were normally distributed.

2.2.1. Cereal –based food

This category of food includes bread, pasta, rice and biscuits. Bread, pasta and biscuits are all wheat-based products. About 40% of the wheat used to produce them is imported, mainly from Canada and USA (International Trade Centre, 2015; ISTAT, 2011), however it is generally transformed in Italy. Therefore, we considered only studies on bread, pasta and biscuits produced in Italy, which, if it was the case, took into consideration the importation of wheat. As far as rice is concerned, Italy is a net exporter of rice, however the import of specific varieties of rice from Asian countries has recently importantly increased (Dow AgroSciences, 2011; International Trade Centre, 2015). Therefore, the production of rice from Asian countries was included in the database.

2.2.2. Milk, dairy products and dairy substitutes

About 25% of the milk consumed and transformed in Italy is imported, mainly from Germany and France (CLAL, 2016). Therefore the average GHG emissions of study referred to milk and dairy products produced in Italy, Germany and France were considered. Few data were available for the emissions dairy substitutes such as vegetal drinks and margarine, therefore all the studies on this kind of products were considered.

2.2.3 Meat and fish

Italy is a net importer of beef and pork meat from other European countries (Basile, 2012; Vigna et al., 2013), whereas is almost self-sufficient as far as poultry meat production is concerned (OEC, 2016). Therefore studies on beef and pork meat production within Europe were considered.

For poultry meat production, instead, both studies related to Italian and French production were considered. Indeed, Lesschen et al. (2011) found that the GHG emission of 1 kilogram poultry produced in France was similar to the emissions generated by the same amount of poultry produced in Italy.

For cured meat the same GHG emissions of pork meat were considered due to a lack of data. Italy is a net importer of fish and fish products, both from European and extra-European countries (ISMEA, 2015). The GHG emissions associated to an average kilogram of different

fish species at the regional distribution centre estimated by Clune and colleagues (2016) were considered for the present study.

2.2.4. Fruit and vegetables

The food category fruit and vegetables included a large number of food items. Due to the large variability that characterised this category, only the GHG of the fruits and vegetables mostly consumed by Italian citizens according to the Italian National Food Consumption Survey INRAN-SCAI (Leclercq et al., 2009) were considered. Among the vegetables, fruiting and leaf vegetables represented about 80% of the consumption and were considered in the present study. Accounting for 75% of the fruit consumption, stone, pome and citrus fruits resulted to be the most consumed fruits by Italian people and were, therefore, considered in the present study. Furthermore, the use of greenhouses and the transport of imported fruit and vegetables can have an important influence on GHG emissions of food (Stoessel et al., 2012). Therefore, fruit and vegetables were divided in two categories: i) seasonal and produced in the Mediterranean area without the use of greenhouses and ii) imported or produced within greenhouses. According to trade statistics (Peperkamp and Schotel, 2015), it was then assumed that fruits and vegetables consumed daily belonged respectively by 88% and 91% to the category i) and the by 12% and 9% to the category ii).

2.2.5. Pulses

More than 70% of the pulses consumed in Italy are imported, mainly from extra-European countries (Peperkamp and Salazar, 2015). Therefore the GHG emissions associated with pulses production were calculated as mean of different kind of pulses with different origins. Only data on fresh pulses were taken into consideration, whereas dried and canned ones were excluded.

2.2.6. Other foods: extra-virgin olive oil, eggs and potatoes

Italy is a net exporter of olive oil and eggs (International Egg Commission, 2013; International Olive Council, 2015), therefore only the emissions of olive oil and eggs produced in Italy were considered in the present study.

Potatoes consumed in Europe are mainly produced within the European boundaries, with some countries, namely France, the Netherlands and Germany being the main potatoes producers. Therefore, studies on potatoes produced in Europe were included in the GHG database (De Cicco and Jeanty, 2016).

2.3 Modelling personal attention for the GHG emissions of food products

Different GHG emissions can be associated with the same food product, due to different managing approaches adopted within the supply chain stages prior to consumption, such as origin of ingredients and production process. In order to account for this variability within each food group, it was assumed that the emissions of each were distributed normally, as described in the previous section.

2.4 Modelling of the cooking phase

Two possible sources of heat were considered for the domestic cooking phase: a gas cooker and an induction burner. The amount of heat needed to boil the water and cook was estimated according to the equations reported by Fusi et al. (2016). Different cooking times and functioning powers were considered according to the food item (Table 3). The GHG emissions were estimated considering respectively the GHG emission factors 2.59 kgCO2eq/m^3 for the extraction and combustion of natural gas burned in the gas cooker (JRC, 2016, assuming a complete oxidation of the fuel) and $1.98*10^{-4}$ kg CO_{2eq}/KJ for the electrical energy used for the induction burner (JRC, 2016, Italian energy mix). For each food items, the GHG emissions from the cooking stage were assumed to be uniformly distributed, ranging between the ones of a gas cooker with 56% efficiency (EC, 2014), and the ones of an induction burner with 92% efficiency (SoleinRete, 2016).

Table 3: Assumptions on the modelling of the cooking phase

	Pasta	Rice	Eggs	Meat - Tofu	Vegetables	Milk	Fish	Potatoes	Legumes
Water (kg)	1.6	0.6	-	-	0.4	0.25	-	1	1
Amount of food* (kg)	0.16	0.16	0.11	0.3	0.4	0.25	0.3	0.3	0.3
ΔT (°C)	85	85	85	-	-	66	-	85	85
Cooking time (after water boiling) (s)	600	960	5	420	600	0	1200	1200	3000
Functioning power of gas cooker (kW)	3	3	2	3	3	2	3	3	2
Min emission (kg$_{CO2eq}$/kg$_{food}$)	0.59	0.37	0.25	0.09	0.19	0.03	0.26	0.41	0.58
Max emission (kg$_{CO2eq}$/kg$_{food}$)	1.12	0.86	0.47	0.25	0.53	0.06	0.71	0.97	1.44

* It was considered that averagely two portions are cooked simultaneously

2.5 Modelling food waste generation

The amount of food waste generated at consumers level was assumed to be distributed normally (Vanham et al., 2015). The statistical normal distribution values (mean and standard deviation) on food waste generation in Europe estimated by Vanham et al. (2015) per food category were considered in the present study. The environmental burden of unavoidable waste - the part of food that cannot be eaten - was allocated to the edible fraction of food because it is considered to

be a specific property of the single food item. The GHG emissions associated with avoidable food waste were calculated multiplying the amount of food wasted by the GHG associated with the edible fraction of food, assuming that the all the food was wasted after being cooked.

2.6 Analysis of the GHG emissions

The GHG emissions were calculated with the software Simapro V 8.0.5. The results were calculated on a daily basis, dividing by 7 the average GHG associated with the weekly balanced menus. A 1000-run Monte Carlo analysis was performed to assess the combined effects of the analysed variables related to personal behaviour. It allowed to estimate the standard deviation and the 95% confidence intervals for the mean for each dietary pattern. Furthermore, through Monte Carlo simulations, pairwise comparisons were performed and differences were considered significant when at least 95% of the Monte Carlo runs were favourable for a dietary pattern.

3. Results and discussion

In Table 4 the calculated average GHG emissions per food item or food group and the standard deviations are reported. Totally 194 values on GHG emissions were collected.

Table 4: GHG emissions per food item, referred to a kilogram of edible food at consumption point, excluding the cooking phase and the disposal of packaging

| | GWP (kg CO2eq/kg edible food) | | GWP values | Sources of data |
	Mean	Standard deviation		
Bread	0.97	0.24	6	(Barilla, 2012; Espinoza-Orias et al., 2011; Jensen and Arlbjørn, 2014; Kulak et al., 2015; Notarnicola et al., 2017b)
Biscuits	1.61	0.31	16	(Barilla, 2016a, 2016b, 2016c, 2015a, 2015b, 2013a, 2013b, 2013c, 2013d, 2013e, 2013f, 2013g, 2013h, 2013i, 2013j, 2013k)
Pasta	1.49	0.58	9	(Barilla, 2014; Cerere, 2014; De Cecco, 2014; Mulino e pastificio Sgambaro, 2016)
Rice	2.72	0.40	30	(Alam et al., 2016; Arunrat et al., 2016; Blengini and Busto, 2009)
Milk (cow)	1.55	0.26	31	(Battini et al., 2014; Fantin et al., 2012; Granarolo, 2015a, 2012; Serra, 2013)
Drink of vegetal origin (soy and rice)	0.91	0.25	3	(Granarolo, 2016; Head et al., 2011)
Yogurt	2.76	0.22	5	(Granarolo, 2015b, 2015c, 2015d, 2015e)
Fresh cheese	9.79	1.54	2	(Granarolo, 2014, 2013a)

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Ripened cheese	10.32	1.83	3	(Corrado et al., 2015; González-García et al., 2013; van Middelaar et al., 2011)
Fruit (locally produced)	0.34	0.04	9	(Alaphilippe et al., 2016; Assomela, 2015; Cerutti et al., 2013; Pergola et al., 2013)
Fruit (greenhouse or imported)	1.88	0.74	5	(Cellura et al., 2012; Roibás et al., 2015; Svanes and Aronsson, 2013; Zeus, 2011)
Vegetables (locally produced)	0.47	0.24	6	(Foteinis and Chatzisymeon, 2016; Hospido et al., 2009; Martínez-Blanco et al., 2011; Theurl et al., 2014)
Vegetables (greenhouse or imported)	1.11	0.83	8	(Cellura et al., 2012; Fusi et al., 2016a; Martínez-Blanco et al., 2011; Torrellas et al., 2012)
Meat substitutes	3.37	0.69	3	(Breedveld et al., 2014; Head et al., 2011)
Beef meat	26.02	6.78	1 (average of European values)	(Clune et al., 2016)
Pork meat	5.60	1.51	1 (average of European values)	(Clune et al., 2016)
Poultry meat	4.31	1.41	6	(de Vries and de Boer, 2010; González-García et al., 2014; Lesschen et al., 2011; Prudêncio da Silva et al., 2014)
Fish	4.45	3.62	1 (average value)	(Clune et al., 2016)
Ham (assumed equal to pork meat)	5.60	1.51	1 (average of European values)	(Clune et al., 2016)
Butter	8.16	1.27	2	(Nilsson et al., 2010)
Margarine	1.69		1	(Nilsson et al., 2010)
Olive oil	2.87	1.93	10	(Assoproli Bari, 2014; De Cecco, 2012; Iraldo et al., 2014; Monini, 2014; Pattara et al., 2016; Rinaldi et al., 2014)
Eggs	3.07		1	(Granarolo, 2013b)
Potatoes	0.36	0.11	9	(González et al., 2011; Koch and Salou, 2015; Röös et al., 2010; Williams et al., 2010)
Pulses	0.70	0.42	14	(Abeliotis et al., 2013; González et al., 2011; Prudêncio da Silva et al., 2014; Romero-Gámez et al., 2012)

				(Pryor et al., 2017; Renouf et al., 2008)
Sugar	0,73	0,05	6	

3.1. GHG emissions of the analysed dietary patterns

The GHG emissions of the analysed dietary patterns and the contribution of the analysed variables characterising human behaviour are reported in Table 5. The supply chain stages "prior to consumption" included the primary agricultural production, food transformation, transports and conservation of food that was assumed to be actually eaten. "Cooking" represented the emissions associated with the cooking phase of food that was assumed to be eaten, whereas "waste" included the GHG emissions due to the agricultural production, food transformation, transport, conservation and cooking of food assumed to be wasted.

Table 5: GHG emissions of the analysed dietary patterns (mean ± standard deviation)

	Total GHG emissions	Prior to production	Cooking	Waste
Omnivorous	3.24 ± 0.34	2.64 ± 0,29	0.27 ± 0.03	0.36 ± 0.06
Vegetarian	2.81 ± 0.21	2.08 ± 0.15	0.37 ± 0.04	0.33 ± 0.09
Vegan	2.67± 0.25	1.88 ± 0.19	0.44 ± 0.05	0.34 ± 0.10

The production stage resulted by far to be the main contribution in all the dietary patterns, ranging between 70% and 81% of the total emissions and the remaining GHG emissions were associated with cooking and food waste generation.

The pairwise comparisons among the dietary patterns (Figure 1) highlighted that no significant differences were found among the GHG emissions associated with the three analysed dietary patterns. Furthermore, as far as the nutritional values of the different dietary patterns is concerned, it has to be underlined that the supply of vitamin B12 was below the recommended intake in the vegetarian and vegan menu. This is a largely acknowledged limit of this kind of dietary patterns and the assumption of supplements or fortified foods is generally recommended (Sobiecki et al., 2016). Due to a lack of data, the GHG emissions of the supplements were not included in the present study and further in-depth analysis would be useful in order to make broader considerations.
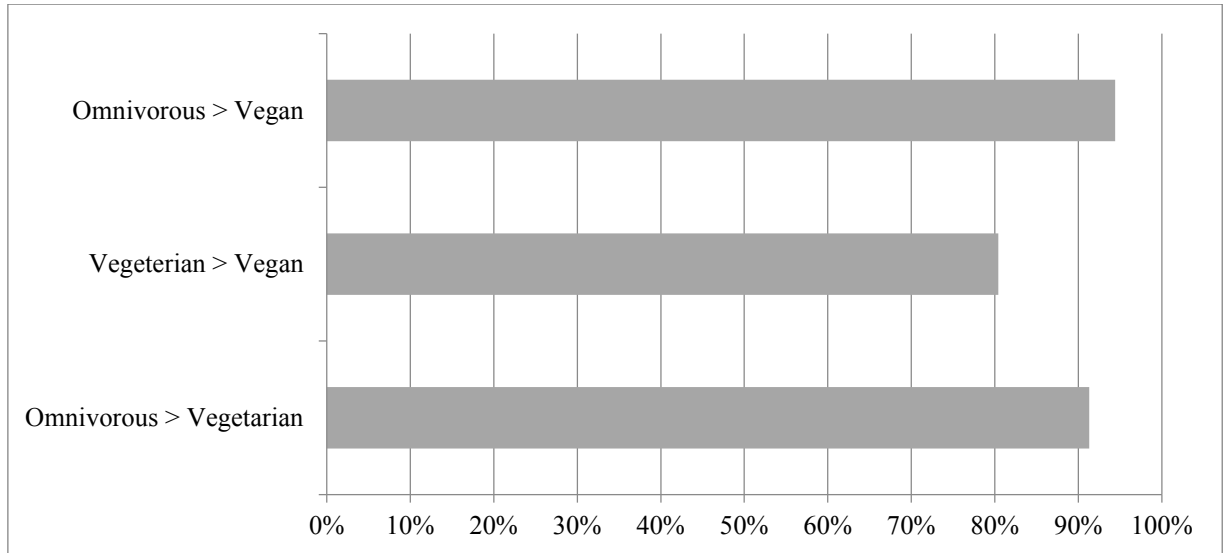
Figure 1: Results of the pairwise comparison between the total GHG emissions of the three dietary patterns done through the Monte Carlo analysis

3.2. Contributions of different types of food

According to the food pyramid recommendations, the food items were categorised as follows: fruit and vegetables, starch-based foods (bread, biscuits, pasta, rice, potatoes), dairy and substitutes (milk, cheese, yogurt/vegetal drinks), protein-based foods (meat, cured meat, fish, eggs, legumes, meat substitutes), fats (butter, margarine, oil) and sweets (sugar). The share of macro-nutrients (proteins, fats and carbohydrates) provided by each food category are reported in Table 6.

Table 6: Amount of macro-nutrients (g/d) supplied by the each food category (OMN = omnivorous diet, VEGET =vegetarian diet; VEGAN = vegan diet)

	Proteins (g/d)			Fats (g/d)			Carbohydrates (g/d)		
	OMN	VEGET	VEGAN	OMN	VEGET	VEGAN	OMN	VEGET	VEGAN
Fruit and vegetables	7	7	7	2	2	2	52	52	52
Starch-based foods	22	23	24	2	2	2	179	184	192
Dairy and substitutes	13	13	6	12	12	5	7	7	24
Protein-based	31	25	33	12	6	6	5	17	23
Fats	0	0	0	35	35	34	0	0	0
Sweets	0	0	0	0	0	0	16	16	16

3.2.1. Contribution to GHG emissions of supply chain stages prior to consumption

The share of GHG emissions due to the supply chain stages prior to consumption were the main contribution in all the dietary patterns and were significantly lower for the vegan and vegetarian diets than for the omnivorous diet (Figure 2). Indeed, as demonstrated in previous studies (De

Laurentiis et al., 2016), producing meat performs in most cases less efficiently than cultivating vegetable-origin foods. The differences among the omnivorous and the vegetarian diets were explained by the differences among the share of GHG emissions of protein-based foods (Figure 3a). Indeed, the consumption of meat increased the amount of GHG emissions associated with them. The omnivorous diet, instead, differentiated from the vegan diet mainly for the GHG of dairy and substitutes products and, to a lesser extent, for the difference in the protein-based foods. However it has to be considered that dairy substitutes have a lower nutrient concentration than dairy products and supplied about half of the proteins and fats provided by the dairy products consumed in the omnivorous diet (Supplementary material). Therefore, in order to satisfy the recommended intake of proteins within the vegan diet, an higher amount of non-animal protein-based foods is needed in comparison to the omnivorous diet. This explains the relatively low differences associated with the GHG of the protein-based foods within the omnivorous and the vegan diets. The contribution of the other food categories, namely starch based foods, fruit and vegetables and fats, were approximately the same in all the dietary patterns analysed.
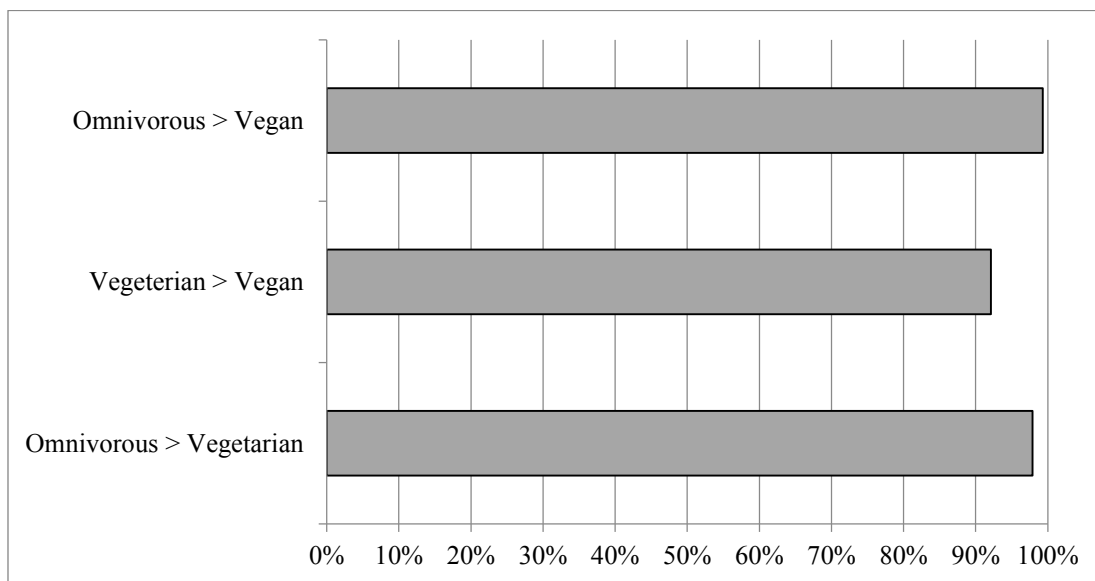

Figure 2: Results of the pairwise comparison between the GHG emissions associated with the supply chain stages prior to consumption of the three dietary patterns done through the Monte Carlo analysis

3.3. Contribution of food cooking

The contribution of cooking varied significantly among the dietary patterns and the higher share was associated with the vegan diet, whereas the lower with the omnivorous. Explaining about 45% of the GHG emissions for the vegan diet, cooking pulses was by far the highest contribution, due to the longer cooking times and the larger amount of food to be eaten to fulfil the nutritional requirements for proteins than other dietary patterns (Figure 3b). Therefore, adopting a low GHG emitting cooking system can result to be more beneficial when a vegan or a

vegetarian diets are adopted rather than an omnivorous one. The gas cooker was found to cause less GHG emissions than the induction burner, when electric energy is generated from the average Italian energy mix. However, the choice of the induction burner could turn to be less GHG-emitting whether electric energy is produced from a renewable source, such as the solar photovoltaic.

3.4. Contribution of food waste

Considering all the food categories, food waste accounted for about 15% of the total emissions of the dietary patterns analysed. The pairwise comparison between the GHG emissions associated with wasted food highlighted no significant differences among different dietary patterns. Fruit and vegetables represented the main share of the GHG emissions for the three dietary patterns considered (Figure 3c). Indeed, despite the lower GHG emissions generated for the production and consumption of fruit and vegetables than for the other foods, higher amount of fruit and vegetables are generally wasted. They were followed in all the dietary patterns by starch-based and protein based foods, respectively due to high level of waste and high GHG emissions associated with the farm stage. The share of GHG emissions of wasted dairy products and substitutes were relatively low due the facts that small amount of dairy products are generally wasted and that few GHG emissions are caused by cooking.

3.5. Take home messages deriving from the present research

In agreement with other previous researches, the present study highlighted that, when a cradle to the point of distribution approach is adopted, vegan and vegetarian diets generated less GHG emissions than an omnivorous one, thanks to lower emissions associated with the primary production of vegetable-origin compared with animal-origin foods. However, when domestic preparation and consumption of foods were taken into consideration, the differences among the three dietary patterns resulted to be not significant. Therefore, a plant-based dietary pattern is generally less GHG emitting than an omnivorous one. However, it can be argued that particularly careless cooking and food waste generation, can compensate the lower GHG emissions associated with the choice of vegetable-origin foods.

Moreover, the above mentioned results highlighted the importance of adopting a cradle-to-grave approach in order to reach comprehensive results, when assessing and estimating the GHG emissions of different dietary patterns. Indeed, although the supply chain stages prior to consumption were the main contributions to the GHG of the diets, the domestic stages, namely cooking and food waste generation, had a relevant influence on the GHG emission of dietary patterns that depended on the type of foods that were consumed. For example, cooking, a phase

generally neglected when comparing the environmental burden of different diets (Benis and Ferrão, 2017; Hallström et al., 2015), resulted to be an hotspot for pulses.

As already demonstrated by other researches (Ghvanidze et al., 2016; Polizzi di Sorrentino et al., 2016), the present study confirmed the central role of consumers in the transition towards less GHG emitting food system. It is, therefore, important to make them aware that their choices have a relevant influence on the GHG emissions of their diets not only regarding the selection of foods but also as far as other personal choices and behaviour are concerned.
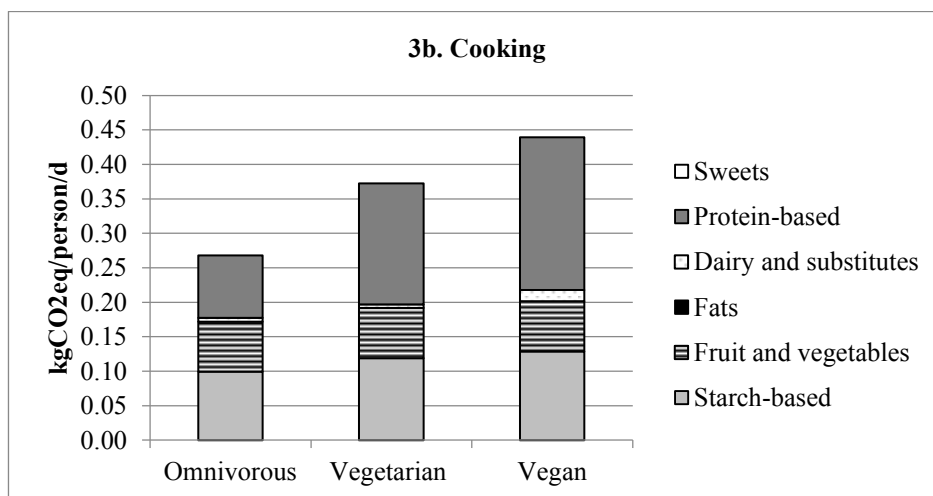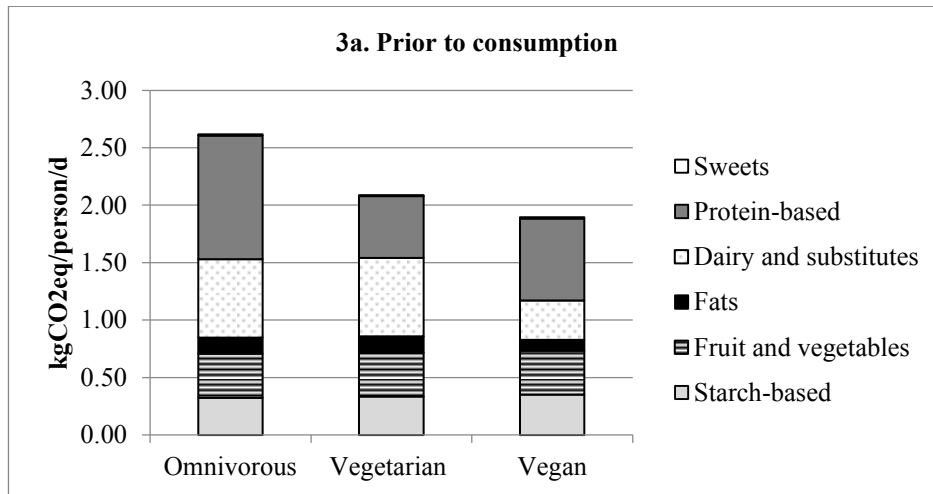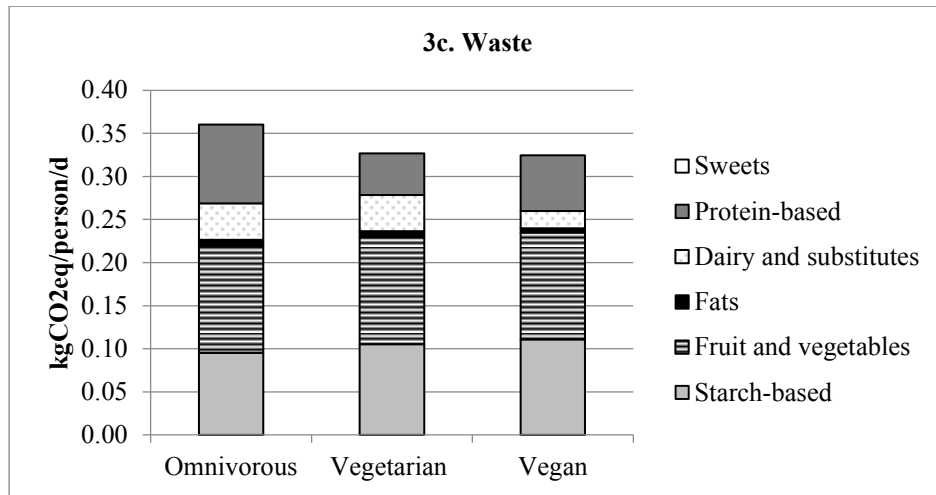
Figure 3: Share of GHG emissions associated with the supply chain stages and different food categories

3.6. Limitations and further developments

The assessment of GHG of different dietary patterns is based on GHG emissions of food items taken from published studies, which, as highlighted in Chapters 3 and 4, may be calculated with different methodological approaches and be not directly comparable.

Furthermore, we acknowledge that the present study has several limitations due to the extreme complexity of the modelled reality and the lack of data. A brief description of such shortages is hereunder reported in order to foster their exploration in future researches on the topic. The GHG of food items were representative of only a small part of the enormous number of foods that a person can potentially buy and consume in Italy. Furthermore, only three elements characterising personal behaviour were considered, whereas there is a large number of elements that can influence the GHG emissions, such as the purchase frequency and distance and the countless possible alternative combinations of foods that fulfil nutrients requirements. Finally, considering only an impact category, namely the potential impact on global warming, is not sufficient to describe the full range of the environmental impact associated with a food system (Nemecek et al., 2016).

4. Conclusions

The present analysis of the effects of personal behaviour on the GHG associated with three dietary patterns highlighted that, despite the predominance of the upstream supply chain stages, the at-home dietary behaviour of food can represent an important contribution. It indeed offsets the significant differences between the GHG emissions of the primary stages of the supply chain between the dietary patterns. Moreover, the results of the present study confirmed the important role of consumers in the transition towards a less GHG emitting food system. Increasing their

awareness on the environmental burden associated with different kinds of food is therefore of the utmost importance.

Due to the complexity of the system analysed, and the lack of data, the study showed some limitations, such as the limited number food items for which average GHG emissions were estimated, a narrow number of variables characterising personal behaviour and the assessment of the potential impact only on global warming. However, it is one of the first attempts to assess the combined effects that the personal choices and behaviour can have on the environmental performance of different dietary patterns and we think that it introduces an interesting broad view of the topic, that is often faced from a more restricted perspective. Indeed, when analysing the impact of dietary patterns, the focus is generally put on the primary phase, whereas the effects of personal choices and behaviour are often omitted. Furthermore, we believe that the present study offers valuable hints for decision-makers and environmental communicators in order to address people towards low GHG-emitting food consumption patterns.

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6. Conclusions and research outlook

6.1 *General conclusive remarks*

In light of the increasing world population and of the current paradoxical distribution of resources and food throughout the planet, guaranteeing the access to a proper amount of food of adequate quality is one of the main challenge that the world is called to face. Overlooking environmental elements is not possible in facing this challenge because food production and agricultural productivity strongly influence and, at the same time, are influenced by the environmental conditions. Therefore, the optimisation of resources and the minimisation of the environmental impacts are two of the pillars on which strategies for food production should be based.

Life cycle thinking and LCA have a central role in assessing and supporting the reduction of the environmental burdens and the optimal use of resources within the food system. Being based on an holistic approach, they avoid the shifting of the environmental burdens between different impact categories or environmental compartments. However, the application of LCA in the agro-food sector shows some critical methodological elements and a lack of homogeneity in the approaches that can limit its effectiveness and reliability.

The general aim of the present thesis is to contribute to the current debate within the LCA community on the harmonisation and improvement of the approaches for LCA of agro-food products, in response to the need to consolidate the use of LCA in business and research contexts and take full advance of its potentialities.

Each chapter focuses on different critical elements, namely the choice of secondary datasets modelling arable crops, the systematic inclusion and accounting of food loss and waste and the modelling of variable elements in food consumption patterns, and proposes some considerations and recommendations to deal with them.

In general terms, the present thesis highlights that methodological and modelling flaws of LCA of agro-food products need to be managed and overcome both through a proper modelling of the product system, compliant with the goal and scope of the study and through a cautious interpretation of the results by LCA practitioners.

Finding a balance between the standardisation of the approaches to conduct a LCA and the numerous purposes for which a LCA can be performed is nothing but straightforward. However, the definition of an harmonised approach to deal with some element of LCA applied to agro-food products is highly desirable to foster the reliability and usefulness of LCA above all in some contexts, such as, by way of example, environmental communication or support to strategic decisions. Indeed, differences in the modelling approaches or in methodological choices can bring to important differences in LCA results and, sometimes, to contrasting conclusions, increasing the confusion among non-expert persons while reducing the reliability of the LCA tool.

The choice of secondary datasets for arable crop production, as highlighted in Chapter 3, can be a critical element of LCA. Indeed, the differences in the modelling approaches on which secondary datasets are built can influence significantly the LCA results. These differences are not always straightforwardly recognisable, if not through a detailed contribution analysis. Indeed, the analysis of datasets for arable crops production demonstrated that, for some impact categories, the results can be non-significantly different, despite the important differences among modelling approaches. This implies that the differences among the LCIs are smoothened when coming to the LCIA stage.

The modelling of the average agricultural production of a certain crop is characterised by a large number of uncertainties, associated with primary activity data and with the environmental fate models used to estimated emissions of fertilisers and PPPs. If differences among secondary datasets can, therefore, be considered "physiological" and a complete harmonisation among the datasets modelling approaches is improbable, it is also true that LCA practitioners are rarely aware about the reasons behind these differences.

Hence, secondary datasets modelling crop production can be object of potential improvements. On one hand, there is the need to have a more precise and accessible information on the uncertainties characterising LCI data. It would guarantee a greater awareness of LCA practitioners in the choice of secondary datasets, according to the goal and scope of their studies. Currently, indeed, this kind of information is not always available or it is hardly accessible (e.g. extensive description of the sources of data within the database reports without a specific assessment which allow comparability). On the other hand, the development and application of more specific environmental fate models for fertilisers and PPPs instead of generic ones, would lead to a more accurate assessment of the potential environmental impact of a product. This last element is of particular importance when modelling agricultural production because a considerable share of the environmental burden is generally associated with estimated emissions.

Modelling and accounting for food loss and waste generation is another critical element of LCA of agro-food products. Indeed – despite the generation of food loss and waste represents a considerable inefficiency of our food system and an important source of environmental impacts – Chapter 4 demonstrated that often available LCA applications neglected the contribution of food loss and waste and show methodological flaws in the modelling of their generation and treatment and in the interpretation of the results.

On one hand the omission of the environmental burdens of food loss and waste can lead to an under-estimation of the environmental burden of the functional unit. On the other hand, an improper modelling of credits reachable thanks to recovery of material or energy during waste treatment can lead to an over-estimation of the environmental benefits, hiding de facto, the higher advantages achievable through food loss and waste prevention strategies.

Developing a shared framework for the systematic assessment of the environmental burdens of food loss and waste in LCA of food products would foster the application of LCA in supporting the reduction of food loss and waste challenge and to increase circular economy practices, at the core of European Commission's environmental policies.

Chapter 4 puts forward some basic principles on which such framework can be developed.

This framework should include a shared definition of the terms food loss and waste, a distinction among avoidable and unavoidable food loss and waste, considerations of the most relevant impact categories to account for the potential impacts of food loss and waste and considerations on how to model food loss and waste and treatment.

Chapter 5 focused on the use of LCA to support the assessment and the comparison of the environmental performance of different dietary patterns. It included the consumption stage,, still little investigated and assessed in LCA, when comparing different dietary patterns.

The results of Chapter 5 put forward two relevant points to reach a comprehensive assessment of the environmental burdens of dietary patterns and identify solutions for their reduction.

The first interests LCA practitioners and highlights the importance of considering the consumption phase - which represents a considerable share of the environmental burden of the diet - taking into consideration its wide range of variability.

The second point concerns the communication on the reduction of the environmental burdens of the diets. The results of Chapter 5 pointed out the central role of consumers in determining the GHG emissions of a dietary patterns, not only as far as the preference of vegetable-origin foods is concerned, but also regarding the cooking modalities and the food waste generation.

Hence, there is the need to adopt a comprehensive perspective in sensitising consumers on the combined effects that the consumption stage can have on the GHG emissions of their diets.

6.2   Research outlook

The present thesis analyses and proposes some considerations on some critical elements that characterise the application of LCA to agro-food supply chains.

It does not deal with all the potential criticalities identified within scientific literature and listed in the introductive section, which would need further investigations.

As far as the topics faced in the present thesis are concerned, hereunder some suggestions for future research are reported, on the basis of the current experience.

- Definition and application of site-specific environmental fate models for fertilisers and pesticides to be used in the definition of secondary datasets modelling arable crops cultivation.

- Testing the applicability of recommendations and principles reported in Chapter 4 for the definition of a common framework for food loss and waste accounting in LCA, including the realisation of case-studies.

- Analysis on the possible approaches to account for the environmental burdens and benefits of food loss and waste prevention practices and industrial ecology applications.

- Foster the assessment of the environmental impacts associated with food consumption taking into consideration their wide range of variability.

List of publications

Published

Corrado S., Ardente F., Sala S., Saouter E., 2017. Modelling of food loss within life cycle assessment: from current practices towards systematisation. Journal of Cleaner Production, 140, 847 -859

Corrado S., Castellani V., Zampori L., Sala S.. Systematic analysis of secondary Life Cycle Inventories for modelling agricultural production: a case study for arable crops. Journal of Cleaner Production, *in press*

Submitted

Corrado S., Lamastra L., Luzzani G., Trevisan M.. Influence of personal behaviour on the greenhouse gas emissions of three balanced dietary patterns. Submitted for publication to the Journal of Cleaner Production

Abstracts for scientific conferences

Corrado S., Lamastra L., Luzzani G., Trevisan M., 2016. Influence of Personal Consumption Behaviour on the Greenhouse Gas Emissions Caused by different Dietary Patterns. 10th International Conference on Life Cycle Assessment of Food 2016. Dublin,19th -21st October 2016.

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