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# Modelling of food loss within life cycle assessment: From current practice towards a systematisation

Sara Corrado [a], Fulvio Ardente [b], Serenella Sala [b], Erwan Saouter [b], *

[a] Università Cattolica Del Sacro Cuore, Istituto di Chimica Agraria e Ambientale, Via Emilia Parmense 84, 29122 Piacenza, PC, Italy
[b] European Commission, Joint Research Centre (JRC), Institute for Environment and Sustainability (IES), via Enrico Fermi 2749, 21027 Ispra, Va, Italy

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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 proposed 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.

## 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, or 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

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* Corresponding author.
*E-mail address:* erwan.saouter@jrc.ec.europa.eu (E. Saouter).

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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 and waste 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 to FL, ii) the benefits associated to 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 harmonisation 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 dimension 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 (Vázquez-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. As an alternative to waste treatments for FL, it was discussed the recovery of FL in industrial ecology (IE) applications, in which FL are used as raw materials in downstream production processes.

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 analysing 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. 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.

#### 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 Cànovas, 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.

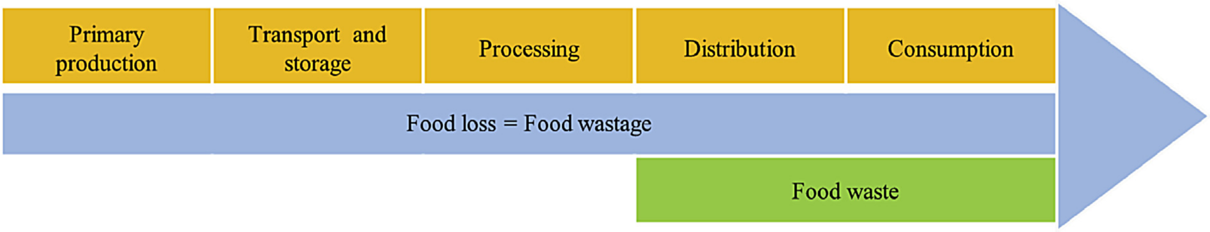

**Fig. 1.** Correspondence between the FSC stages and the definitions of "food loss", "food waste" and "food wastage" according to FAO (2013) and FAO (2014b).

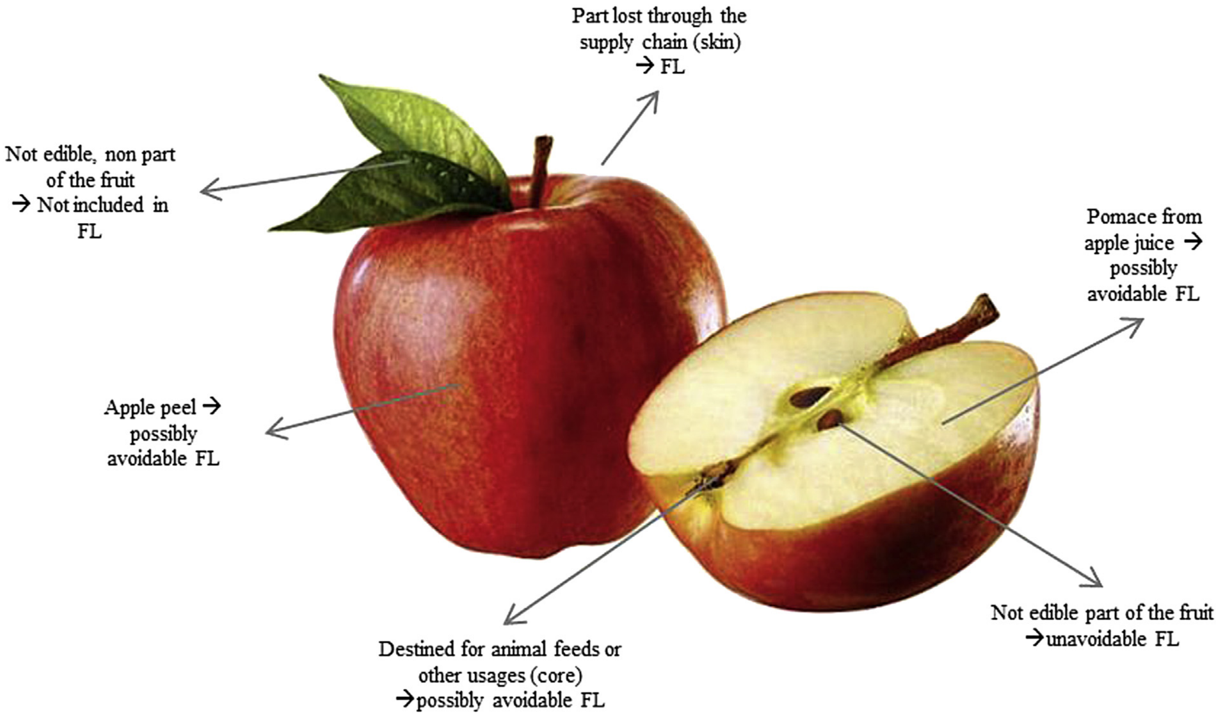

**Fig. 2.** Representation of different types of FL applied, as example, to an apple. Each food category will have a different split. Splits may also change based on local cultural and/or consumer habits.

#### 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 an 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-Gámez 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).

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-Gámez et al., 2014). Over-production was discussed just in a few studies (e.g. Romero-Gámez 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 an 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 decomposition was 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 (Vázquez-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; Eyjólfsdóttir 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

**Table 1**
Possible food loss (FL) per food supply chain (FSC) stage. Built from FAO (2013) and Parfitt et al. (2010) and complemented with other information in literature.

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

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.

### *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 context, 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. González-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

**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) |

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 an 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).

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 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 by-products 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 by-product 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.

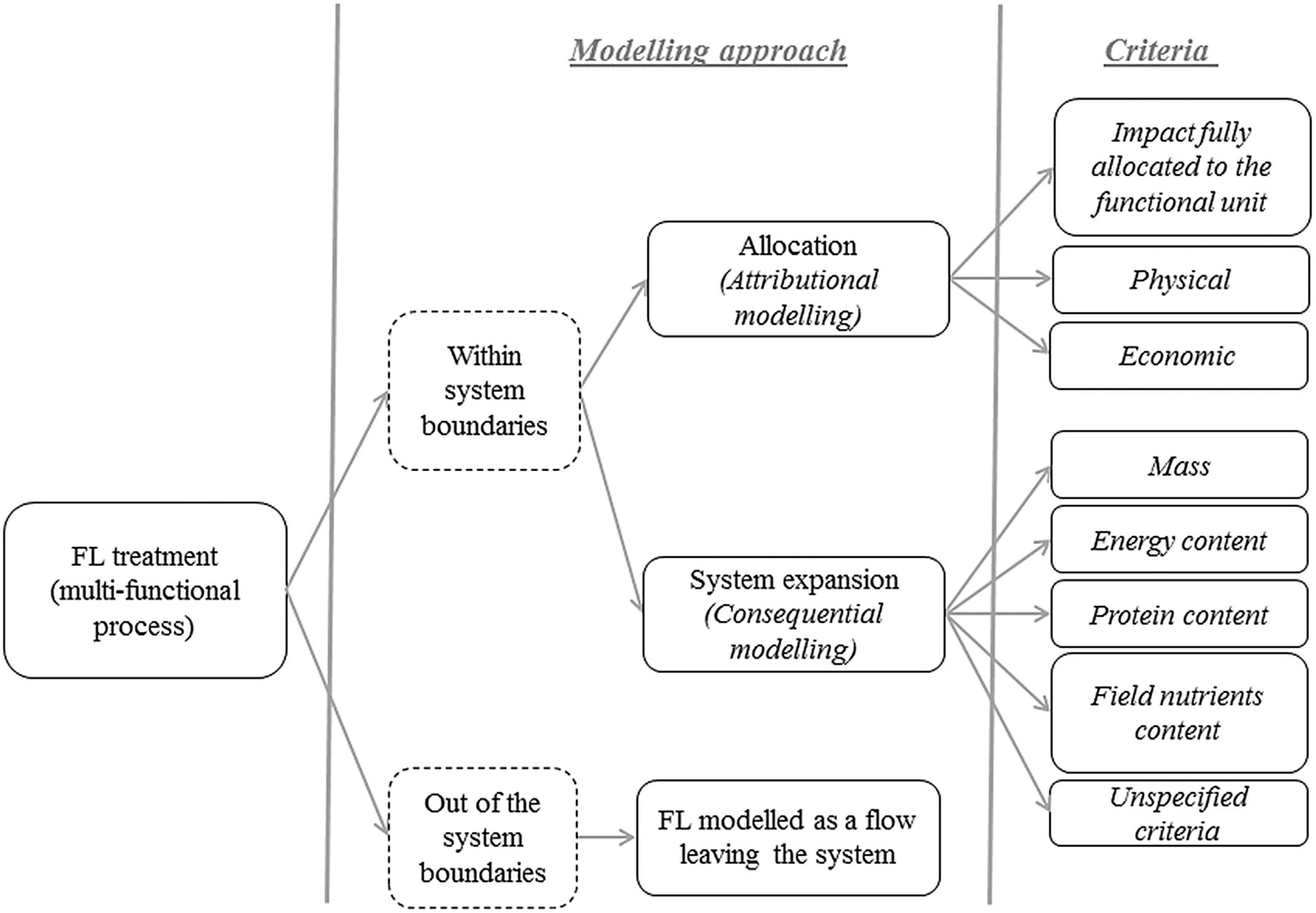

**Fig. 3.** Approaches adopted in the analysed studies to model the treatments of the FL. The graph refers to both industrial ecology applications and waste treatments.

González-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. Milà 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 by-products 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. González-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; González-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 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 multi-functionalities 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) $\text{Avoidable}_{\text{FL}} = \sum_i^n \textit{Avoidable food loss}_i$ (with ‘i’ lifecycle stage)
2) $\text{Unavoidable}_{\text{FL}} = \sum_i^n \textit{Unavoidable food loss}_i$ (with ‘i’ lifecycle stage)
3) $\text{Total}_{\text{FL}} = \text{Avoidable}_{\text{FL}} + \text{Unavoidable}_{\text{FL}}$

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

**Table 3**
Summary of the critical aspects observed in the present study and of potential recommendations for LCA practitioners to handle them.

| LCA stage[a] | Critical aspects | Rationale for criticality | Recommendation(s) | Strengths of recommendation(s) |
|---|---|---|---|---|
| GS, R | Systematic exclusion of food loss (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<br>- 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<br>- 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<br>- Wider knowledge of the FL generation dynamics |
| GS, R | Exclusion of waste treatments from the system boundaries | 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 the performance of the waste treatment | Check the representativeness of the data used to model the waste treatment | Avoidance of having misleading results related to improper waste modelling |
| M, R | Unclear description of the allocation procedure adopted to model FL and outputs of waste treatments | - Limited reproducibility of the study (ISO requirement)<br>- Allocation procedures can have a strong influence on LCA results | - Report clearly allocation procedures. Particularly, in case of allocation:<br>• Allocation criteria<br>• Allocations factors<br>In case of system expansion:<br>• Substitution criteria<br>• Amount of product substituted<br>• Accurate description of the product system substituted<br>- Assess the representativeness of the substitution criteria<br>- Perform a sensitivity analysis (including also the ”pessimistic scenario” without any credits from the waste treatments) | - Improved transparency of the study and reproducibility of the results<br>- Better understanding of the influence of modelling choices on the results of the study |

[a] GS = goal and scope definition; I = inventory or data collection; M = modelling approach; R = reporting.

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. 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 co-product.

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 summarised 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.

## 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 recommendation 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.

## References

Almeida, C., Vaz, S., Cabral, H., Ziegler, F., 2014. Environmental assessment of sardine (Sardina pilchardus) purse seine fishery in Portugal with LCA methodology including biological impact categories. Int. J. Life Cycle Assess. 19, 297–306. http://dx.doi.org/10.1007/s11367-013-0646-5.

Andersson, K., Ohlsson, T., Olsson, P., 1998. Screening life cycle assessment (LCA) of tomato ketchup: a case study. J. Clean. Prod. 6, 277–288. http://dx.doi.org/10.1016/S0959-6526(98)00027-4.

Arafat, H.A., Jijakli, K., Ahsan, A., 2015. Environmental performance and energy recovery potential of five processes for municipal solid waste treatment. J. Clean. Prod. 105, 233–240. http://dx.doi.org/10.1016/j.jclepro.2013.11.071.

Ardente, F., Beccali, G., Cellura, M., Marvuglia, A., 2006. POEMS: a case study of an Italian wine-producing firm. Environ. Manage 38, 350–364. http://dx.doi.org/10.1007/s00267-005-0103-8.

Ardente, F., Cellura, M., 2012. Economic allocation in life cycle assessment: the state of the art and discussion of examples. J. Ind. Ecol. 16, 387–398. http://dx.doi.org/10.1111/j.1530-9290.2011.00434.x.

Ardente, F., Cellura, M., Brano, L., Mistretta, M., 2009. Life cycle assessment-driven selection of industrial ecology strategies. Integr. Environ. Assess. Manag. 6, 52–60. http://dx.doi.org/10.1897/IEAM.

Arvanitoyannis, I.S., Kotsanopoulos, K.V., Veikou, A., 2014. Life cycle assessment (ISO 14040) implementation in foods of animal and plant origin: review. Crit. Rev. Food Sci. Nutr. 54, 1253–1282. http://dx.doi.org/10.1080/10408398.2011.631170.

Ayer, N.W., Tyedmers, P.H., Pelletier, N.L., Sonesson, U., Scholz, A., 2007. Co-product allocation in life cycle assessments of seafood production systems: review of problems and strategies. Int. J. Life Cycle Assess. 12, 480–487. http://dx.doi.org/10.1007/s11367-006-0284-2.

Battistel, P., 2014. Produzione integrata Evoluzione inarrestabile. Terra Vita 24, 38–43.

Berlin, J., 2002. Environmental life cycle assessment (LCA) of Swedish semi-hard cheese. Int. Dairy J. 12, 939–953. http://dx.doi.org/10.1016/S0958-6946(02)00112-7.

Bernstad, A., Cànovas, A., 2015. Current practice, challenges and potential methodological improvements in environmental evaluations of food waste prevention – a discussion paper. Resour. Conserv. Recycl. Recycl 101, 132–142. http://dx.doi.org/10.1016/j.resconrec.2015.05.004.

Bernstad, A., la Cour Jansen, J., 2012. Review of comparative LCAs of food waste management systems–current status and potential improvements. Waste Manag. 32, 2439–2455. http://dx.doi.org/10.1016/j.wasman.2012.07.023.

Bernstad, A., la Cour Jansen, J., 2011. A life cycle approach to the management of household food waste - a Swedish full-scale case study. Waste Manag. 31, 1879–1896. http://dx.doi.org/10.1016/j.wasman.2011.02.026.

Blengini, G.A., Busto, M., 2009. The life cycle of rice: LCA of alternative agri-food chain management systems in Vercelli (Italy). J. Environ. Manage 90, 1512–1522. http://dx.doi.org/10.1016/j.jenvman.2008.10.006.

Blonk Agri-footprint BV, 2014. Agri-footprint Description of Data - Version 1.0.

Cederberg, C., Stadig, M., 2003. System expansion and allocation in life cycle assessment of milk and beef production. Int. J. Life Cycle Assess. 8, 350–356. http://dx.doi.org/10.1007/BF02978508.

Cellura, M., Ardente, F., Longo, S., 2012. From the LCA of food products to the environmental assessment of protected crops districts: a case-study in the south of Italy. J. Environ. Manage 93, 194–208. http://dx.doi.org/10.1016/

j.jenvman.2011.08.019.

Cerutti, A.K., Beccaro, G.L., Bruun, S., Bosco, S., Donno, D., Notarnicola, B., Bounous, G., 2014. Life cycle assessment application in the fruit sector: state of the art and recommendations for environmental declarations of fruit products. J. Clean. Prod. 73, 125–135. http://dx.doi.org/10.1016/j.jclepro.2013.09.017.

Chen, H., Jiang, W., Yang, Y., Yang, Y., Man, X., 2016. State of the art on food waste research: a bibliometrics study from 1997 to 2014. J. Clean. Prod. http://dx.doi.org/10.1016/j.jclepro.2015.11.085.

Chertow, M.R., 2000. Industrial symbiosis: literature and taxonomy. Annu. Rev. Energy Environ. 25, 313–337.

Chiusano, L., Cerutti, A.K., Cravero, M.C., Bruun, S., Gerbi, V., 2015. An Industrial Ecology approach to solve wine surpluses problem: the case study of an Italian winery. J. Clean. Prod. 91, 56–63. http://dx.doi.org/10.1016/j.jclepro.2014.12.002.

Cleary, J., 2010. The incorporation of waste prevention activities into life cycle assessments of municipal solid waste management systems: methodological issues. Int. J. Life Cycle Assess. 15, 579–589. http://dx.doi.org/10.1007/s11367-010-0186-1.

Coltro, L., Mourad, A., Oliveira, P., Baddini, J., Kletecke, R., 2006. Environmental profile of brazilian green coffee. Int. J. Life Cycle Assess. 11, 16–21. http://dx.doi.org/10.1065/lca2006.01.230.

Contreras, A.M., Rosa, E., Perez, M., Van Langenhove, H., Dewulf, J., 2009. Comparative Life Cycle Assessment of four alternatives for using by-products of cane sugar production, 17, 772–779. http://dx.doi.org/10.1016/j.jclepro.2008.12.001.

Cordella, M., Tugnoli, A., Spadoni, G., Santarelli, F., Zangrando, T., 2008. Lca of an Italian lager beer. Int. J. Life Cycle Assess. 13, 133–139. http://dx.doi.org/10.1065/lca2007.02.306.

Davies, R.W.D., Cripps, S.J., Nickson, a., Porter, G., 2009. Defining and estimating global marine fisheries bycatch. Mar. Policy 33, 661–672. http://dx.doi.org/10.1016/j.marpol.2009.01.003.

Davis, J., Sonesson, U., 2008. Life cycle assessment of integrated food chains - a Swedish case study of two chicken meals. Int. J. Life Cycle Assess. 13, 574–584. http://dx.doi.org/10.1007/s11367-008-0031-y.

De Menna, F., Vittuari, M., Molari, G., 2014. ScienceDirect Impact evaluation of integrated food-bioenergy systems: a comparative LCA of peach nectar. Biomass Bioenergy 73, 48–61. http://dx.doi.org/10.1016/j.biombioe.2014.12.004.

Duchin, F., 2005. Sustainable consumption of food. A framework for analyzing scenarios about changes in diets. J. Ind. Ecol. 9, 99–114.

Eberle, U., Fels, J., 2015. Environmental impacts of German food consumption and food losses. Int. J. Life Cycle Assess. 1–14. http://dx.doi.org/10.1007/s11367-015-0983-7.

Eckelman, M., Chertow, M.R., 2009. Quantifying life cycle environmental benefits from the reuse of industrial materials in Pennsylvania. Environ. Sci. Technol. 43, 2550–2556. http://dx.doi.org/10.1021/es802345a.

Emanuelsson, A., Ziegler, F., Hornborg, S., Sonesson, U., 2009. D4.7 LCA Case Study Fisheries: Bechmarking Swedish Cod and Herring Products by Spatial-temporal Life Cycle Assesement. LC-IMPACT project.

Emanuelsson, A., Ziegler, F., Pihl, L., Sköld, M., Sonesson, U., 2014. Accounting for overfishing in life cycle assessment: new impact categories for biotic resource use. Int. J. Life Cycle Assess. 19, 1156–1168. http://dx.doi.org/10.1007/s11367-013-0684-z.

European Commission - Joint Research Centre - Institute for Environment and Sustainability (EC-JRC), 2011. Supporting environmentally sound decisions for bio-waste management. In: A Practical Guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA). http://dx.doi.org/10.2788/53942.

European Commission - Joint Research Centre - Institute for Environment and Sustainability (EC-JRC), 2010. International Reference Life Cycle Data System (ILCD) Handbook - General Guide for Life Cycle Assessment – Detailed Guidance. http://dx.doi.org/10.2788/38479.

European Commission (EC), 2011. COM (2011) 571, Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions - Roadmap to a Resource Efficient Europe.

European Commission (EC), 2013. REC 2013/179/EU, Commssion Reccomendation of 9 April 2013 on the use of common method to measure and communicate the life cycle environmental performance of products and organisations. Off. J. Eur. Union. http://dx.doi.org/10.3000/19770677.L_2013.124.eng.

European Commission (EC), 2014. COM (2014) 398, Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Towards a cricular economy: A zero waste programme for Europe.

European Commission (EC), 2015. COM (2015) 0614. Communication from the Commission to the European Parliament, the Council, the European economic and social committee and the committee of the regions. Closing the loop – An EU action plan for the Circular Economy.

European Union (EU), 2008. European Parliament and Council, Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on Waste and Repealing Certain Directives.

Eyjólfsdóttir, H.R., Yngvadóttir, E., Jónsdóttir, H., Skúladóttir, B., 2003. Environmental Effects of Fish on the Consumers Dish – Life Cycle Assessment of Icelandic Frozen Cod Products. Icelandic Fisheries Laboratory. Report 06–03.

Fantozzi, F., Bartocci, P., D'Alessandro, B., Testarmata, F., Fantozzi, P., 2015. Carbon footprint of truffle sauce in central Italy by direct measurement of energy consumption of different olive harvesting techniques. J. Clean. Prod. 87, 188–196. http://dx.doi.org/10.1016/j.jclepro.2014.09.055.

FAO, 2016. Environmental Performance of Large Ruminant Supply Chains - Guidelines for Assessment - Version 1.

FAO, 2014a. The State of Food Insecurity in the World.

FAO, 2014b. Working Paper - Definitional Framework of Food Loss.

FAO, 2013. Food Wastage Footprint. Impacts on Natural Resources. Summary Report. doi:ISBN 978-92-5-107752-8.

FAO, 2011a. Global Food Losses and Food Waste - Extent, Causes and Prevention.

FAO, 2011b. Global Initiative on Food Loss and Food Waste Reduction [WWW Document]. URL. www.fao.org/save-food/en/.

FAO, 1981. Food Loss Prevention in Perishable Crops. Rome.

Foley, J. a., Ramankutty, N., Brauman, K. a., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337–342. http://dx.doi.org/10.1038/nature10452.

FUSIONS EU Project, 2015. FUSIONS Food Waste Data Set for EU-28.

Gentil, E.C., Gallo, D., Christensen, T.H., 2011. Environmental evaluation of municipal waste prevention. Waste Manag. 31, 2371–2379. http://dx.doi.org/10.1016/j.wasman.2011.07.030.

González-García, S., Gomez-Fernández, Z., Dias, A.C., Feijoo, G., Moreira, M.T., Arroja, L., 2014. Life Cycle Assessment of broiler chicken production: a Portuguese case study. J. Clean. Prod. 74, 125–134. http://dx.doi.org/10.1016/j.jclepro.2014.03.067.

González-García, S., Hospido, A., Moreira, M.T., Feijoo, G., Arroja, L., 2013. Environmental life cycle assessment of a galician cheese: San Simon da Costa. J. Clean. Prod. 52, 253–262. http://dx.doi.org/10.1016/j.jclepro.2013.03.006.

Grönroos, J., Seppälä, J., Silvenius, F., Mäkinen, T., 2006. Life cycle assessment of Finnish cultivated rainbow trout. Boreal Environ. Res. 11, 401–414.

Gruber, L.M., Brandstetter, C.P., Bos, U., Lindner, J.P., 2014. LCA study of unconsumed food and the influence of consumer behavior. In: Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-food Sector.

Hallström, E., Carlsson-Kanyama, A., Börjesson, P., 2015. Environmental impact of dietary change: a systematic review. J. Clean. Prod. 91, 1–11. http://dx.doi.org/10.1016/j.jclepro.2014.12.008.

Heller, M.C., Keoleian, G.A., 2014. Greenhouse gas emission estimates of U.S. Dietary choices and food loss. J. Ind. Ecol. http://dx.doi.org/10.1111/jiec.12174 n/a–n/a.

Hornborg, S., Belgrano, A., Bartolino, V., Valentinsson, D., Ziegler, F., 2013. Trophic indicators in fisheries: a call for re-evaluation. Biol. Lett. 9, 20121050. http://dx.doi.org/10.1098/rsbl.2012.1050.

Hornborg, S., Nilsson, P., Valentinsson, D., Ziegler, F., 2012. Integrated environmental assessment of fisheries management: swedish Nephrops trawl fisheries evaluated using a life cycle approach. Mar. Policy 36, 1193–1201. http://dx.doi.org/10.1016/j.marpol.2012.02.017.

Hospido, A., Moreira, M.T., Feijoo, G., 2003. Simplified life cycle assessment of galician milk production. Int. Dairy J. 13, 783–796. http://dx.doi.org/10.1016/S0958-6946(03)00100-6.

Humbert, S., Loerincik, Y., Rossi, V., Margni, M., Jolliet, O., 2009. Life cycle assessment of spray dried soluble coffee and comparison with alternatives (drip filter and capsule espresso). J. Clean. Prod. 17, 1351–1358. http://dx.doi.org/10.1016/j.jclepro.2009.04.011.

Iribarren, D., Moreira, M.T., Feijoo, G., 2010. Implementing by-product management into the life cycle assessment of the mussel sector. Resour. Conserv. Recycl 54, 1219–1230. http://dx.doi.org/10.1016/j.resconrec.2010.03.017.

ISO, 2006a. ISO 14040 Environmental Management - Life Cycle Assessment - Principles and Framework.

ISO, 2006b. ISO 14044 Environmental Management - Life Cycle Assessment - Requirements and Guidelines. International standard. http://dx.doi.org/10.1007/s11367-011-0297-3.

Jensen, J.K., Arlbjørn, J.S., 2014. Product carbon footprint of rye bread. J. Clean. Prod. 82, 45–57. http://dx.doi.org/10.1016/j.jclepro.2014.06.061.

Jungbluth, N., Nathani, C., Stucki, M., Leuenberger, M., 2011. Environmental impact of swiss consumption and production. In: A Combination of Input-output Analysis with Life Cycle Assessment.

Kim, D., Thoma, G., Nutter, D., Milani, F., Ulrich, R., Norris, G., 2013. Life cycle assessment of cheese and whey production in the USA. Int. J. Life Cycle Assess. 18, 1019–1035. http://dx.doi.org/10.1007/s11367-013-0553-9.

Koroneos, C., Roumbas, G., Gabari, Z., Papagiannidou, E., Moussiopoulos, N., 2005. Life cycle assessment of beer production in Greece. J. Clean. Prod. 13, 433–439. http://dx.doi.org/10.1016/j.jclepro.2003.09.010.

Lal, R., 2008. Soils and sustainable agriculture: a review. Sustain. Agric. 28, 57–64. http://dx.doi.org/10.1007/978-90-481-2666-8_3.

Laurent, A., Bakas, I., Clavreul, J., Bernstad, A., Niero, M., Gentil, E., Hauschild, M.Z., Christensen, T.H., 2014a. Review of LCA studies of solid waste management systems—part I: lessons learned and perspectives. Waste Manag. 34, 573–588. http://dx.doi.org/10.1016/j.wasman.2013.10.045.

Laurent, A., Clavreul, J., Bernstad, A., Bakas, I., Niero, M., Gentil, E., Christensen, T.H., Hauschild, M.Z., 2014b. Review of LCA studies of solid waste management systems—part II: methodological guidance for a better practice. Waste Manag. 34, 589–606. http://dx.doi.org/10.1016/j.wasman.2013.12.004.

Leigh, M., Li, X., 2015. Industrial ecology, industrial symbiosis and supply chain environmental sustainability: a case study of a large UK distributor. J. Clean. Prod. 106, 632–643. http://dx.doi.org/10.1016/j.jclepro.2014.09.022.

Lowenthal, M.D., Kastenberg, W.E., 1998. Industrial ecology and energy systems: a first step. Resour. Conserv. Recycl 24, 51–63.

Manfredi, M., Vignali, G., 2014. Life cycle assessment of a packaged tomato puree: a comparison of environmental impacts produced by different life cycle phases. J. Clean. Prod. 73, 275–284. http://dx.doi.org/10.1016/j.jclepro.2013.10.010.

Manfredi, S., Cristobal, J., Torres de Matos, C., Vasta, A., Sala, S., Saouter, E., Tuomisto, H., 2015. Improving Sustainability and Circularity of European Food Waste Management with a Life Cycle Approach. Publication office of EU Commission, Luxembourg. http://dx.doi.org/10.2788/182997.

Mattila, T., Lehtoranta, S., Sokka, L., Melanen, M., Nissinen, A., 2012. Methodological aspects of applying life cycle assessment to industrial symbioses. J. Ind. Ecol. 16, 51–60. http://dx.doi.org/10.1111/j.1530-9290.2011.00443.x.

Mattila, T., Pakarinen, S., Sokka, L., 2010. Quantifying the total environmental impacts of an industrial symbiosis-a comparison of process-, hybrid and input-output life cycle assessment. Environ. Sci. Technol. 44, 4309–4314. http://dx.doi.org/10.1021/es902673m.

Meier, T., Christen, O., 2013. Environmental impacts of dietary recommendations and dietary styles: Germany as an example. Environ. Sci. Technol. 47, 877–888. http://dx.doi.org/10.1021/es302152v.

Meier, T., Christen, O., Semler, E., Jahreis, G., Voget-Kleschin, L., Schrode, A., Artmann, M., 2014. Balancing virtual land imports by a shift in the diet. Using a land balance approach to assess the sustainability of food consumption. Germany as an example. Appetite 74, 20–34. http://dx.doi.org/10.1016/j.appet.2013.11.006.

Milà i Canals, L., Burnip, G.M., Cowell, S.J., 2006. Evaluation of the environmental impacts of apple production using Life Cycle Assessment (LCA): case study in New Zealand. Agric. Ecosyst. Environ. 114, 226–238. http://dx.doi.org/10.1016/j.agee.2005.10.023.

Mirabella, N., Castellani, V., Sala, S., 2014. Current options for the valorization of food manufacturing waste: a review. J. Clean. Prod. 65, 28–41. http://dx.doi.org/10.1016/j.jclepro.2013.10.051.

Mogensen, L., Kristensen, T., Nielsen, N.I., Spleth, P., Henriksson, M., Swensson, C., Hessle, a, Vestergaard, M., 2015. Greenhouse gas emissions from beef production systems in Denmark and Sweden. Livest. Sci. 174, 126–143. http://dx.doi.org/10.1016/j.livsci.2015.01.021.

Nemecek, T., Schnetzer, J., 2011a. Data Collection of Inputs and Yields in LCIs of Agricultural Production Systems in Switzerland and Other European Countries - Data V. 3.0.

Nemecek, T., Schnetzer, J., 2011b. Methods of Assessment of Direct Field Emissions for LCIs of Agricultural Production Systems - Data v3.0.

Nessi, S., Rigamonti, L., Grosso, M., 2012. LCA of waste prevention activities: a case study for drinking water in Italy. J. Environ. Manage 108, 73–83. http://dx.doi.org/10.1016/j.jenvman.2012.04.025.

Niutanen, V., Korhonen, J., 2003. Industrial ecology flows of agriculture and food industry in Finland: utilizing by-products and wastes. Int. J. Sustain. Dev. World Ecol. 10, 133–147. http://dx.doi.org/10.1080/13504500309469792.

Notarnicola, B., Puig, R., Raggi, A., Fullana, P., Tassielli, G., De Camillis, C., Ruis, A., 2011. Life cycle assessment of Italian and Spanish bovine leather production systems. Afinidad 68.

Nucci, B., Puccini, M., Pelagagge, L., Vitolo, S., Nicolella, C., 2014. Improving the environmental performance of vegetable oil processing through LCA. J. Clean. Prod. 64, 310–322. http://dx.doi.org/10.1016/j.jclepro.2013.07.049.

Ostergren, K., Gustavsson, J., 2014. Biotechnology, S.-T.S.I. For F. and, Hilke, B.-B., Timmermans, T., UR, W., Hansen, O.-J.,. FUSIONS Definitional Framework for Food Waste.

Papargyropoulou, E., Lozano, R., Steinberger, K., Wright, J., Ujang, N., Bin, Z., 2014. The food waste hierarchy as a framework for the management of food surplus and food waste. J. Clean. Prod. 76, 106–115. http://dx.doi.org/10.1016/j.jclepro.2014.04.020.

Parfitt, J., Barthel, M., Macnaughton, S., 2010. Food waste within food supply chains: quantification and potential for change to 2050. Philos. Trans. R. Soc. Lond. B. Biol. Sci. 365, 3065–3081. http://dx.doi.org/10.1098/rstb.2010.0126.

Peereboom, E.C., Kleijn, R., Lemkowitz, S., Lundie, S., 1998. The influence of inventory data sets on life cycle assessment results: a case study on PVC. J. Ind. Ecol. 2, 109–130.

Pelletier, N., Ardente, F., Brandao, M., De Camillis, C., Pennington, D., 2015. Rationales for and limitations of preferred solutions for multi-functionality problems in LCA: is increased consistency possible? Int. J. Life Cycle Assess. 20, 74–86. http://dx.doi.org/10.1007/s11367-014-0812-4.

Rajaeifar, M.A., Akram, A., Ghobadian, B., Rafiee, S., Heidari, M.D., 2014. Energy-economic life cycle assessment (LCA) and greenhouse gas emissions analysis of olive oil production in Iran. Energy 66, 139–149. http://dx.doi.org/10.1016/j.energy.2013.12.059.

Romero-Gámez, M., Audsley, E., Suárez-Rey, E.M., 2014. Life cycle assessment of cultivating lettuce and escarole in Spain. J. Clean. Prod. 73, 193–203. http://dx.doi.org/10.1016/j.jclepro.2013.10.053.

Röös, E., Sundberg, C., Hansson, P.A., 2011. Uncertainties in the carbon footprint of refined wheat products: a case study on Swedish pasta. Int. J. Life Cycle Assess. 16, 338–350. http://dx.doi.org/10.1007/s11367-011-0270-1.

Saarinen, M., Kurppa, S., Virtanen, Y., Usva, K., Mäkelä, J., Nissinen, A., 2012. Life cycle assessment approach to the impact of home-made, ready-to-eat and school lunches on climate and eutrophication. J. Clean. Prod. 28, 177–186. http://dx.doi.org/10.1016/j.jclepro.2011.11.038.

Salomone, R., Ioppolo, G., 2012. Environmental impacts of olive oil production: a Life Cycle Assessment case study in the province of Messina (Sicily). J. Clean. Prod. 28, 88–100. http://dx.doi.org/10.1016/j.jclepro.2011.10.004.

San Martin, D., Ramos, S., Zufía, J., 2016. Valorisation of food waste to produce new raw materials for animal feed. Food Chem. 198, 68–74. http://dx.doi.org/10.1016/j.foodchem.2015.11.035.

Schau, E.M., Fet, A.M., 2008. Lca studies of food products as background for environmental product declarations. Int. J. Life Cycle Assess. 13, 255–264.

Schmidt, J., Weidema, B., Brandao, M., 2015. A framework for modelling indirect land use changes in Life Cycle Assessment. J. Clean. Prod. 99, 230–238. http://dx.doi.org/10.1016/j.jclepro.2015.03.013.

Schmidt Rivera, X.C., Espinoza Orias, N., Azapagic, A., 2014. Life cycle environmental impacts of convenience food: comparison of ready and home-made meals. J. Clean. Prod. 73, 294–309. http://dx.doi.org/10.1016/j.jclepro.2014.01.008.

Secchi, M., Castellani, V., Collina, E., Mirabella, N., Sala, S., 2016. Assessing eco-innovations in green chemistry: life Cycle Assessment (LCA) of a cosmetic product with a bio-based ingredient. J. Clean. Prod. 129, 269–281. http://dx.doi.org/10.1016/j.jclepro.2016.04.073.

Simboli, A., Taddeo, R., Morgante, A., 2015. The potential of Industrial Ecology in agri-food clusters (AFCs): a case study based on valourisation of auxiliary materials. Ecol. Econ. 111, 65–75. http://dx.doi.org/10.1016/j.ecolecon.2015.01.005.

Smil, V., 2004. Improving efficiency and reducing waste in our food system. Environ. Sci. 1, 17–26. http://dx.doi.org/10.1076/evms.1.1.17.23766.

Sokka, L., Lehtoranta, S., Nissinen, A., Melanen, M., 2011. Analyzing the environmental benefits of industrial symbiosis: life cycle assessment applied to a finnish forest industry complex. J. Ind. Ecol. 15, 137–155. http://dx.doi.org/10.1111/j.1530-9290.2010.00276.x.

Sonesson, U., Flysjö, A., Modin-Edman, A.-K., Gustavsson, J., Hessle, A., Patel, M., Woodhouse, A., Witthöft, C., 2015. Protein quality as functional unit – a methodological framework for inclusion in Life Cycle Analysis (LCA). In: LCA Fod “Feeding the Planet and Energy for Life.”.

Strid, I., Eriksson, M., 2014. Losses in the supply chain of Swedish lettuce – wasted amounts and their carbon footprint at primary production, whole sale and retail. In: 9th International Conference on Life Cycle Assessment in the Agri-food Sector.

Stuart, T., 2009. Waste: Uncovering the Global Food Scandal. W.W. Norton & Company.

Svanes, E., Aronsson, A.K.S., 2013. Carbon footprint of a Cavendish banana supply chain. Int. J. Life Cycle Assess. 18, 1450–1464. http://dx.doi.org/10.1007/s11367-013-0602-4.

The Daily Meal, 2015. 5 Ways Animal Skin Is Eaten Around the World [WWW Document]. URL. http://www.thedailymeal.com/eat/5-ways-animal-skin-eaten-around-world (accessed 2.3.16).

UNEP, 2006. Circular Economy. An Alternative Model for Economic Development.

United Nation - Department of Economic and Social Affairs, 2015. World Population Prospects, the 2015 Revision [WWW Document]. URL. http://esa.un.org/unpd/wpp/unpp/p2k0data.asp (accessed 10.20.15).

Vanham, D., Bouraoui, F., Leip, A., Grizzetti, B., Bidoglio, G., 2015. Lost water and nitrogen resources due to EU consumer food waste. Environ. Res. Lett. 10, 084008. http://dx.doi.org/10.1088/1748-9326/10/8/084008.

Vázquez-Rowe, I., Hospido, A., Moreira, M.T., Feijoo, G., 2012. Best practices in life cycle assessment implementation in fisheries. Improving and broadening environmental assessment for seafood production systems. Trends Food Sci. Technol. 28, 116–131. http://dx.doi.org/10.1016/j.tifs.2012.07.003.

Weinzettel, J., Pelletier, N., Tyedmers, P., 2012. Understanding who is responsible for pollution: what only the market can tell us-comment on “an ecological economic critique of the use of market information in life cycle assessment research.” J. Ind. Ecol. 16, 455–456. http://dx.doi.org/10.1111/j.1530-9290.2012.00460.x.

Williams, I.D., Schneider, F., Syversen, F., 2015. The “ food waste challenge ” can be solved. Waste Manag. 41, 1–2. http://dx.doi.org/10.1016/j.wasman.2015.03.034.

World Organisation for Animal Health, 2015. Animal Health. A Multifaceted Challnge.

WRAP, 2013. Household Food and Drink Waste in the United Kingdom 2012.

WRAP, 2009. Household Food and Drink Waste in the UK.

WRAP, 2008. The Food We Waste.

Ziegler, F., Nilsson, P., Walther, B.M.Y., 2003. Life Cycle assessment of frozen cod fillets including fishery-specific environmental impacts. Int. J. Life Cycle Assess. 8, 39–47. http://dx.doi.org/10.1007/BF02978747.