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The role of life cycle assessment in supporting sustainable agri-food systems: A review of the challenges

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ABSTRACT

Life cycle thinking is increasingly seen as a key concept for ensuring a transition towards more sustainable production and consumption patterns. As food production systems and consumption patterns are among the leading drivers of impacts on the environment, it is important to assess and improve food-related supply chains as much as possible. Over the years, life cycle assessment has been used extensively to assess agricultural systems and food processing and manufacturing activities, and compare alternatives “from field to fork” and through to food waste management. Notwithstanding the efforts, several methodological aspects of life cycle assessment still need further improvement in order to ensure adequate and robust support for decision making in both business and policy development contexts. This paper discusses the challenges for life cycle assessment arising from the complexity of food systems, and recommends research priorities for both scientific development and improvements in practical implementation. In summary, the intrinsic variability of food production systems requires dedicated modelling approaches, including addressing issues related to: the distinction between technosphere and ecosphere; the most appropriate functional unit; the multi-functionality of biological systems; and the modelling of the emissions and how this links with life cycle impact assessment. Also, data availability and interpretation of the results are two issues requiring further attention, including how to account for consumer behaviour.

1. Introduction

Food production addresses one of the most important and basic human needs and has developed in parallel with the evolution of humanity to ensure steady provision (Diamond, 2002), safety and variety of food as well as improved nutritional composition. Currently, food production responds to a basic need and also to a plethora of social, cultural and even aesthetic needs and wants. However, with the requirement to feed seven billion people, this food production comes with a huge environmental cost (Tilman et al., 2001; Garnett, 2011). Farming approaches have been depleting the Earth's resources and contributing significantly to greenhouse gas emissions, to soil fertility and biodiversity loss, to water scarcity, and to the release of large amounts of nutrients and other pollutants that affect ecosystem quality (McMichael et al., 2007). If nothing changes in the way we produce and consume food, and in light of the need to increase food production by more than 60% by 2050 (FAO, 2006; FAO et al, 2015), the environmental impacts associated with food production systems will become even more severe and will increasingly surpass the planetary boundaries.

Improving food production and consumption systems is at the heart of every discourse on sustainable development from both environmental and socio-economic perspectives. Recent studies have suggested a research agenda for food sustainability. For example, Soussana (2014), who addressed the European context specifically, prioritised for the production side: i) the sustainable intensification of European agriculture, ii) the operationalisation of

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agriculture within limits for greenhouse gases, energy, biodiversity and contaminants, and iii) the improvement of resilience to climate change in agricultural and food systems. For the consumption side, Soussana prioritised: i) the identification of the determinants of a healthy diet including physical activity, ii) the development of healthy, high-quality, safe and sustainable foods, and iii) the fight against diet-related chronic diseases.

Focusing on livestock, Scollan et al. (2011) identified priorities related to balancing the need for increased production of animal products coupled with a lower environmental footprint and addressing societal needs in terms of product quality as perceived by the consumer. Indeed, there are synergies between research needs and strategies dedicated to improving food quality and safety, and those dedicated to reducing the environmental impact of ruminant livestock production. The main research priorities identified were related to the capitalization of knowledge to ensure optimization of production, application of life cycle based assessment, and use of an adaptation strategy based on selection of profitable animals under different production systems. Putting this into practice, the Livestock Environmental Assessment and Performance (LEAP) Partnership of FAO (2015) has developed a multi-stakeholder partnership for benchmarking and monitoring the life cycle-based environmental performance of the livestock sector. The initiative aims to promote adoption of life cycle thinking as a way to understand and improve the environmental profile of livestock production systems.

Several authors have proposed that the coupled assessment of environmental- and human health-related concerns together should be the building block of future research activities (e.g Tukker et al., 2011; Adams and Demmig-Adams, 2013). Bridging the conservation of natural capital, on the one hand, and human health issues on the other hand, using a life cycle perspective may lead to breakthroughs in the sustainability assessment of food systems (Soussana, 2014).

In recent research, dietary shifts have been identified as one of the most powerful ways to increase the sustainability of our food systems (as in the recent review of Hallström et al., 2015; and in studies focusing on combining environmental and nutritional aspects such as Röös et al., 2015; as well as in studies addressing the complementary role of technological innovation and demand-side changes as in Bryngelsson et al., 2016), and are usually associated with reducing meat consumption. However, any dietary shift may imply burden shifting (from one stage to another in the life cycle of different foods, or from one impact category to another). Therefore, there is a strong need for use of life cycle-based methods to ensure that dietary shifts are coupled with improved sustainability of food systems.

In view of the above-mentioned research priorities, this paper describes some of the main research challenges that need to be addressed in order for Life Cycle Assessment (LCA) to more fully support decision-making and the transition towards sustainable food systems. Given the importance of the environmental consequences of food production and consumption, the added value of this study is related to the identification of the needs and the means for better accounting those, adopting life cycle assessment. Specifically, section 2 further discusses the additional advances needed in food-related LCA. Section 3 focuses on the intrinsic high variability of food systems and the related challenges for food-related LCA. Section 4 deals with the main modelling issues specific to LCAs of food systems. The importance of further developing reliable LCA databases is discussed in Section 5. The role of consumers and of the industry in driving the development of food-related LCA is discussed in Section 6, while the role of LCA in assessing food waste is addressed in Section 7. The final Section 8 deals with the interpretation of food-related LCA studies, including the necessity of understanding how to best use LCA study results to avoid misleading conclusions.

2. State of the art and challenges for agri-food LCA in answering sustainability and food security issues

Increasingly, Life Cycle Thinking (LCT) is recognised as fundamental for addressing current challenges and research needs related to the sustainability of food production and consumption systems. However, progress towards environmentally sustainable systems requires improving the methods for quantitative, integrated assessment and promoting the use of these methods in different domains. Indeed, LCT and the different life cycle-based methodologies, such as Life Cycle Assessment (LCA) (ISO, 2006a,b), Life Cycle Costing (LCC), Social Life Cycle Assessment (sLCA) and the overall Life Cycle Sustainability Assessment (LCSA) may support a transition toward increasing the sustainability of current patterns of production and consumption. Given the importance of adopting a life cycle approach, literature on application of LCA to food system has been thriving (Notarnicola et al., 2012a; van den Werf et al., 2014; Notarnicola et al., 2015; Nemecek et al., 2016).

Notwithstanding the positive and peculiar features of LCA-based methodologies compared to other environmental assessment methodologies (Sala et al., 2013a,b), a number of food-related challenges need to be addressed in order to further advance the currently available approaches and methods. Consider for example, the comparisons between more or less intensive agriculture, and organic versus non-organic agriculture; many LCA studies find that agricultural intensification leads to less overall environmental impacts per functional unit (Sonesson et al., 2016a; Kulak et al., 2013) (but see Chobtang et al., 2016a (*this volume*) for an exception to this generalisation). The rationale is that higher yields per hectare of land or per animal are beneficial if the level of used resources does not increase to the same extent. In the face of increasing pressure on agricultural land for other purposes such as bioenergy, and pressure from urbanisation and desertification, increased efficiency of land use seems a logical way forward. However, the current LCA method is incomplete and does not comprehensively assess some aspects that are critical for long-term sustainable food production e.g. decreased soil quality and fertility, increased erosion, reduced ecosystem services due to intensification, biodiversity loss. The challenge here is that the missing aspects are dependent on synergies between many factors that are not all “captured” in current LCA methods. Also, the information needed to describe these aspects is often at the landscape level - and landscape attributes are only partly dependent upon production management at the field level. LCA studies usually focus on the field level and therefore, by not acknowledging such emergent aspects, the conclusions from an LCA study might support less preferred policies and actions from a sustainability perspective. LCA-based methods that combine both the field and landscape perspectives are needed in order to capture environmental impacts at these different scales. For example, several agricultural measures for mitigating impacts (such as having field margins acting as buffer zones) are not captured by current land use inventories, whereas reporting their presence may be regarded as a “credit” in terms of reduction of environmental impacts. Besides, the landscape pattern (e.g. the patchiness index, Weissteiner et al., 2016) is fundamental to understand the level of threat to biodiversity posed by an agricultural system. This is information at landscape level which is completely missing in LCA. Besides, two of the most incomplete modelling challenges in LCAs of food systems are the significant inconsistencies between emission inventory modelling and impact assessment of pesticides (Rosenbaum et al., 2015), and assessment of land use change associated with off-farm inputs to agricultural production systems.

Compared with other economic sectors, food systems are inherently more variable in the inventory data (e.g. in the same area, on the same crop, two different active ingredients could be applied for the same purpose) and in the reliability of impact assessment (e.g. the impact on biodiversity due to a particular land use may change dramatically from one ecoregion to another). Yet, current data available in food LCA databases and life cycle impact assessment (LCIA) models, are mostly non-spatially and temporally resolved (Hauschild et al., 2012). This results in severe limitations when agricultural systems are being evaluated. This is further exacerbated by the fact that, in a globalised world, consuming a food product in one particular location may be associated with environmental impacts occurring in many other countries (Lenzen et al., 2012).

In addition, other sustainability aspects considered highly relevant by many consumers, such as working conditions and animal welfare, are largely neglected in LCA. In moving from just the environmental to inclusion of more socio-economic aspects, so far, several Social LCAs studies have been conducted, especially in developing countries (e.g. Feschet et al., 2013 on bananas, Lemeilleur and Vagneron, 2010 on coffee, and Kruse et al., 2009 on salmon). A review of social and economic tools combined with LCA of food products can be respectively found in Settanni et al. (2010) and Kruse (2010).

Ultimately, cultural-related aspects are a fundamental component of food supply chains and heavily affect patterns of consumption – hence generating another source of variability. Inclusion of cultural aspects in LCSA has been discussed recently by Pizzirani et al. (2014), and addressed in a case study of forestry products (Pizzirani et al., 2016). Cultural values influence the way we produce and consume food, and also influence assessment of the environmental impacts associated with food systems.

3. Variability in food LCA

The importance of distinguishing between variability and uncertainty in LCA studies has been highlighted by a number of researchers (e.g. Hauck et al., 2014; Huijbregts, 1998, 2001; Steinmann et al., 2014). Uncertainty may be reduced by additional research but variability describes actual differences amongst alternative processes and/or products (and thus cannot be reduced unless there are changes in the systems under analysis) (Huijbregts, 1998; Steinmann et al., 2014).

For agricultural systems in particular, there is potential for considerable variability in inventory data between individual agricultural enterprises. Some of the aspects that underlie this variability include different management practices, soil types and climates, seasonality, the life cycle of perennial crops, and distances (and related transportation modes) between locations of activities in the life cycle of product systems.

Regarding management practices, farmers may utilise different practices based on their own preferences and expertise. Sets of practices may be grouped together and labelled as “organic”, “biodynamic”, “integrated”, “heated greenhouse” production. However, in reality there is usually a continuous spectrum of practices both within and between these categories. A number of LCA studies compare these categories of activities for specified products or production systems e.g. dairy systems (O'Brien et al., 2012; Thomassen et al., 2008; van der Werf et al., 2009); pig production systems in France (Basset-Mens et al., 2006), greenhouse production in Europe (Torrellas et al., 2012), bean cultivation in Greece (Abeliotis et al., 2013), tomato production in France (Boulard et al., 2011), wheat production in the USA (Meisterling et al., 2009). However, relatively few LCA studies have actually focused on the variability within these categories. Notable studies include Fenollos et al. (2014) on tigernuts in Spain, Mouron et al. (2006) on Swiss apple production systems, da Silva et al. (2010) on Brazilian soybean production, and a number of studies on dairy systems (e.g. Chobtang et al., 2016b; Thomassen et al., 2009).

Beyond human-controlled variability, differences in management practices and in yields may be related to soil types and climates e.g. some soils require regular application of lime to raise the soil's pH, and dryer climates require use of irrigation systems. However, in their review of LCA studies of vegetable products, Perrin et al. (2014) note that many of these studies fail to specify the representativeness of data with respect to soil and climate conditions.

Some sources of variability are related to the timescale adopted for the study. Within a single year, seasonality may contribute to differences in LCA results for food products. For example, Hospido et al. (2009) found differences in environmental impacts for Spanish and English lettuce consumed in the UK – but Foster et al. (2014) found relatively small differences in environmental impacts for Spanish and English raspberries consumed in the UK - at different times of the year. Between years the environmental impacts of a single crop may vary due to differences in yields related to variable weather conditions. Furthermore, over a period of several years, perennial crops exhibit a cycle of increasing and then decreasing yields and this is often not accounted for in LCA studies (Bessou et al., 2013).

Finally, variability may be related to the different transport distances (and modes) between the locations of agricultural production in relation to production of inputs used in agricultural production (such as fertilisers and compost), and subsequent processing, retailing and consumption activities. LCA studies on this aspect that discuss a range of food products include those by Michalský and Hooda (2015), Rothwell et al. (2016), Webb et al. (2013), and Wiedemann et al. (2015).

There is also potential for significant variability to arise at other stages in the life cycle of food products. In particular, this variability may be related to storage time, packaging and food preparation (including related wastage). For example, Meneses et al. (2012) found that the climate change and acidification potential indicators were higher for plastic bottles than for aseptic cartons (of various sizes) for Spanish milk packaging; Keyes et al. (2015) found that storage activities contributed the majority of the result for four out of eleven environmental indicators studied in an LCA of Nova Scotian apple production and delivery to a retailer in Halifax, Canada. And Schmidt Rivera et al. (2014) compared home- and ready-prepared meals in the UK, and found that home-prepared meals had lower impacts for ten out of eleven environmental indicators.

LCA studies vary hugely with respect to discussing and quantifying these sources of variability. When addressed, the variability may be represented as a range of values, a specific metric to quantify variability (e.g. Hauck et al., 2014; Steinmann et al., 2014) or using statistical analysis (e.g. Mouron et al., 2006).

So, it is often unclear as to whether or not it is important to represent these different sources of variability in LCA study results (and, in particular, those intended to support decision-making). The examples given above show that the choice of system boundaries, temporal and spatial, and choice of agricultural management practices as well as activities at other life cycle stages, can make big differences to the LCA results. Yet, for example, current Environmental Product Declaration (EPD) programmes do not generally provide detailed requirements or guidelines on representation of variability in LCA results. Instead, they simply require data to be “representative” or be calculated as averages or weighted averages, and EPDs generally present single values for different environmental indicator results. This could potentially lead to misrepresentation of products in the marketplace if comparisons are made between alternative food products on the basis of what is, effectively, biased data. It may also lead to overlooking of potential improvement options if the inferior environmental performance of individual enterprises in the supply chain of food products is “lost” in the calculation of average data for the different life cycle stages of a food product. The challenge for LCA researchers and practitioners concerns how to represent relevant variability in LCA study results without having to collect such a huge range of data that these studies become infeasible.

4. Modelling issues specific to agricultural systems

4.1. Distinction between technosphere and ecosphere in relation to modelling of environmental impacts

Traditionally, LCA has been used for the assessment of industrial systems where processes are located in the technosphere and environmental emissions are assessed in the ecosphere. Following this approach, when LCA has been applied to agricultural systems, soil has been defined as part of the technosphere (Audsley, 1997) and is regarded as merely a physical support for plants and a medium for delivery of inputs by farmers. This is actually the vision that prevailed during the early stage of the green revolution (1950–1970) where agricultural lands were managed like industrial production sites: soils were regarded as only a physical asset. As a result, impacts on soil fertility, soil structure, soil hydrology balance, and soil biodiversity are currently not included in the majority of food LCA studies despite being essential elements for ensuring the conservation of the natural capital as well as long-term security of food supply. A possible solution to overcome this flaw is to include agricultural soil in the ecosphere or to include the evaluation of these impacts under the land use impact category.

Mainly for impact categories where the use of fertilizers and pesticides is particularly relevant (i.e. eutrophication and toxicity), a further difficulty is the definition of the boundary between the technosphere and ecosphere. If some fate modelling is included at inventory analysis and it includes degradation of substances, then any environmental impacts caused by these substances prior to degradation are omitted from the analysis. Current eutrophication or toxicity LCIA models account for emissions rather than the amounts applied, and because there is no agreement on emission models to be used at inventory analysis, different modelling approaches will lead to different results (as shown by Perrin et al., 2014; Rosenbaum et al., 2015; Van Zelm et al., 2014) (see section 4.4).

4.2. Definition of an appropriate functional unit

In studies involving agricultural systems, yield in kg or area used are the most popular functional units (FU). However, even if these may be appropriate as a reference flow or unit of analysis at inventory level, this choice does not really represent the true function of agricultural products. Sticking to mass or area is actually not in line with usual LCA practices where the performance of a product must be included in the FU.

A number of researchers have proposed alternatives that include e.g. the nutritional value of the food in the FU (Heller et al., 2013). It is also important to be aware that not all the food products could be considered as nutritional per se as they may address needs that are beyond the basic ones (Notarnicola et al., 2016a,b *this volume*) and involve social dimensions like drinking wine, beers or coffee. So, a more sophisticated way of defining FU would be to include the cultural function provided by the hedonistic value of food and drink (e.g. Notarnicola et al., 2003). However, definition of a cultural function is not straightforward since it is not always defined objectively, and is thus not always feasible for the definition of FUs.

From a farmer perspective, it could be argued that economic value best represents the main function of farmer activities. Several authors (van der Werf and Salou, 2015; Notarnicola et al., 2015) support this approach as a way of including the quality of the product i.e. using a product's price as a measure of the product's quality. However, prices are usually determined by external factors that are not necessarily linked to the quality of the product (e.g. out of season products).

In addition to feeding humans and other animals, agriculture is also a provider of social services (recreational) and is regarded as a custodian of cultural and natural heritage (Koohafkan and Altieri, 2011). It could be argued that these should be reflected in the respective environmental and social assessments.

4.3. Multi-functional biological systems

Co-production is a common issue in food LCA with economic or physical allocation being the most commonly approach in food product studies due to ease of data collection. System expansion should be preferred in order to be in line with ISO; however, the method of system expansion is more complex and more demanding on data collection. Schau and Fet (2008) suggested to use biological rather than physical causality because most food production systems include biological processes e.g. reduction of CH_4 outputs by changes in the fodder composition (input). Nevertheless, the high variability of biological processes could also complicate assessment and comparisons. It is clear that different allocation methods will provide different results; in this sense, the PEF initiative and the development of Product Category Rules, as well as Environmental Product Declaration schemes, could contribute to define a consensus in establishing allocation criteria for specific products.

4.4. Modelling emissions at inventory analysis (fertilizers, pesticides and machinery)

As mentioned in section 4.1, a clear definition of the boundaries between the technosphere and ecosphere is needed at inventory analysis in order to standardise the modelling at impact assessment. However, the modelling at inventory analysis is further complicated by a number of other factors peculiar to agricultural systems. In particular, it is well known that emission flows are closely related to not only site-specific soil and climate conditions but also to the inputs of pesticides and fertilizers themselves. Several guidelines for inventory modelling are provided in different studies and reports, amongst which Nemecek et al. (2014) is an important reference.

Different models used at inventory analysis provide different environmental results for pesticides. In the ecoinvent database, for example, there are no pesticide emissions to surface water leading to no contribution of those chemicals to the aquatic toxicity impact category, whatever impact assessment model is used. An agreement on a clear definition of pesticide emission modelling is necessary and is the objective of a current international effort for finding a consensus (Rosenbaum et al., 2015).

In relation to emissions from use of fertilizers, it is possible to distinguish between synthetic and organic fertilizers, and between emissions to different compartments (air, soil and water). Air emissions are better defined thanks to the IPCC (IPCC, 2006), which provides data for greenhouse gas emissions, and EEA guidelines (EEA 2013) which provide data for a number of other air emissions.

But a consensus is still missing on a globally applicable model for calculating soil and water emissions (i.e leaching, erosion and run-off) which are more dependent on soil conditions (e.g. pH, clay content, slope, etc.).

Similar issues arise for emissions from the use of machinery where fuel consumption is dependent not only on hours of work but also on aspects such as tractor power, type of operation and soil conditions (Hansson and Mattsson, 1999; ASAE, 2003).

4.5. Impact categories such as land use, water use, biodiversity, toxicity, particulate matter

Traditionally, LCIA methods have mostly relied on generic, non-spatial, and steady state multimedia environmental models that focus predominately on energy-related impacts. However, in the agricultural sector, site dependent and closely related environmental aspects, such as natural resources (i.e., water and land) and ecosystems quality, acquire special relevance (Antón et al., 2014). Unlike the so-called global impact categories, such as climate change and ozone depletion, regional impact categories (e.g. acidification, eutrophication, toxicity) need to have spatially differentiated models because evidence shows that differences in fate and effect factors such as exposure mechanisms and sensitivity can vary significantly in different geographical contexts (Sala et al., 2011; Ciuffo and Sala, 2013).

Although water and land use in agriculture could have major environmental consequences, most LCA studies represent these impacts as mere flows expressed in m^2 or m^3 and do not assess the potential environmental damage arising from these uses. The International Life Cycle Data System (ILCD) (EC-JRC, 2011) recommended two models to be applied with caution, respectively the Swiss Ecoscarcity model for water and Mila i Canals et al. (2007a,b) for land use. Those recommendations are currently under revision to improve the robustness of the models.

At global level, much research has been undertaken in order to provide operational, site specific and globally-applicable methods. Under the efforts of the UNEP/SETAC Life Cycle Initiative, a flagship project is being conducted aiming to provide global guidance and building consensus on environmental LCIA indicators. In a first stage, work has been focused on the impacts of climate change, particulate matter, water use and biodiversity damage due to land use (Jolliet et al., 2014).

Regarding toxicity impacts, the new version of USEtox (Rosenbaum et al., 2008 in the current version, Usetox 2.0, 2013) provides new subcontinental characterization factors which allow a more site-specific assessment for human and aquatic toxicity. For some substances, e.g. metals, more detailed geographical factors would be required. However, terrestrial ecotoxicity characterization factors are still missing.

5. Databases in food LCA

In order to effectively implement LCA of any product system, the inventory data need to be reliable and up-to-date. In the case of background data used in an LCA study, the specific information is typically extracted from databases. With the increasing interest in the sustainability of food product systems, databases have evolved from ones that focussed mostly on industrial processes to ones that also focus on agri-food systems. Examples of commercial databases that deal with the food sector are ecoinvent (that also deals with other non-food systems) and more specific ones such as Agrifootprint, Food LCA-DK and Agribalyse (Blonk Consultants, 2014; Frischknecht et al., 2007; Nielsen et al., 2003; Koch and Salou, 2015).

As is often the case in food-related LCA, the datasets in these databases are usually created using data representing specific sites at specific times. This means that different databases are not interchangeable with each other and need to be used with caution by LCA practitioners. In many cases, the data are presented in a non-transparent manner that will not allow LCA practitioners to accurately adapt such data to their specific case studies. This can obviously lead to studies that have ambiguous interpretations and conclusions that are not comparable to those of other studies. For non-food inventories, this lack of site-specificity is usually not such a big issue. Different manufacturing sites producing PET are most likely using the same processes, no matter where the PET is produced (at least in Europe), and have small variability in the input-output inventories. In contrast, production of a food item in different locations of the same region of the same country can make a huge difference to the inventory data. Therefore, to allow a fair and meaningful comparison of food production systems, a high level of geographical specificity is needed for agri-food systems. There is thus a need for specific and regionalised databases that are also well-documented and implemented with flexible data structures that will allow the user to tailor the data to specific case studies.

6. Role of consumers, governments and of the industry

6.1. Role of consumer behaviour and governments towards more sustainable food

As with any market sector, the consumer can potentially play a direct role in determining the success of sustainable food products and of government policies targeted at reducing the environmental impacts of food production and consumption systems.

In some countries, LCA is becoming a mainstream tool for supporting policy development, such as the case of the EU where LCA is a fundamental instrument of its Integrated Product Policy (EC, 2001). The implementation of such policies partially addresses consumers by covering aspects such as the use of Environmental Product Declarations, Eco-labels, and the development of Green Public Procurement. However, in general consumers at present are still not playing an operative role that effectively influences the use of LCA in the food sector. This is due to the average consumer's lack of knowledge about environmental sustainability issues, and due to the fact that there are too many labelling systems that in many cases do not communicate information in a clear and direct manner. In this context, consumers' associations can have a fundamental role in promoting the transfer of knowledge to the consumers and influencing their behaviour and food habits, thus indirectly transferring their feedback back to the supply chain (Notarnicola et al., 2015). It is also essential that the initiatives such as that of the EU concerning a harmonised and unique LCA based product footprint (Product Environmental Footprint – PEF; EC, 2013) become active in order to effectively and concisely communicate environmental information about food products to consumers.

6.2. Role of changes in diet

Many LCA studies (e.g. Muñoz et al., 2010; Meier and Christen, 2012; Heller et al., 2013) have shown that the dietary choices of the consumer significantly affect the environmental sustainability of food consumption. Perhaps the most prevalent insight that emerges is that vegetarian diets seem to generate less environmental burdens compared to animal based ones, and also that the domestic/use phase is not necessarily negligible in terms of environmental impacts (Foster et al., 2006; Hallström et al., 2015). However, in these assessments the positive aspects of animal production are often not included. These positive aspects, such as ecosystem services, soil fertility, use of resources otherwise not available as food, and low levels of pesticide use, are elements where the LCA approach today is generally weak (see Sections 4 and 7). Furthermore, replacing the essential nutrients of animal-based foods poses nutritional challenges (Millward and Garnett, 2010). As indicated by Heller et al. (2013) and by Smedman et al. (2010), environmental assessment of a diet cannot consider only the daily intake food or its fat, energy or protein content, but must also comprise other more qualitative aspects of a diet. LCA of dietary aspects and health issues must consider more particularised and inclusive nutrition-based functional units (Stylianou et al., 2016).

Furthermore, dietary choices and the related consumption styles of individuals vary greatly from region to region. People in developed countries, when compared to developing countries, tend to have diets that are characterised by high consumption of animal-based unsaturated fats and proteins (Carlsson-Kanyama et al., 2003). Also, in general, in countries with colder climates, the diets tend to involve high caloric consumption of dairy products and meat. This highlights the fact that assessment of dietary choices must take into account that many different factors arise when food choices are made including social and cultural ones (see section 4.3).

6.3. Role of the food industry

The first use of LCA in the food industry dates back to 1969, when the Coca Cola Company used it as a means of evaluating aspects concerning the packaging of its products (Hunt and Franklin, 1996). Since then LCA has been widely applied to food packaging, since packaging has been a subject of public debate and it is an area where producers can both make and communicate improvements, such as the Packaging Impact Quick Evaluation Tool (PIQET), introduced by the Sustainable Packaging Alliance (SPA), and the Instant LCA Packaging tool (Intertek, 2015). The Tetrapak company has carried out many LCA studies on their food container products in order to investigate and improve their environmental sustainability (Tetrapak, 2016). The Nestlè company has included in its web-based product life cycle management (PLM) software DevEx, a module called Eco-Design Tool to help the company employees to assess and develop food products not only during the packaging phase but during all life cycle stages (Notarnicola et al., 2012b). Other food producers have implementing LCA to guide environmental improvement of the whole life cycle of their food products as part of their environmental policy e.g. Unilever (Unilever, 2016) and Arla Foods (Flysjö and Modin-Edman, 2014).

In 2013, the ENVIFOOD Protocol was developed as a food and drink-specific guidance document created by the European Food Sustainable Consumption and Production Roundtable, a multi stakeholder initiative co-chaired by the European Commission and business associations from the food and beverage supply chains. Specifically, the Protocol is intended as complementary guidance for the PEF pilot testing launched by the European Commission (Saouter et al., 2014).

The European Feed Manufacturer's Federation (FEFAC) and the American Feed Industry Association (AFIA) set up a consortium in 2011 with a view to collaborate on environmental footprinting. The FEFAC and AFIA consortium together with the International Feed Industry Federation (IFIF), has joined the UN FAO-led Partnership on benchmarking and monitoring the environmental performance of livestock supply chains (FEFAC, 2014). Also, the International Dairy Federation and the UN Food and Agriculture Organization (FAO, 2010) commissioned a study on greenhouse gas emissions from the dairy sector. A review of other similar initiatives can be found in Notarnicola et al. (2015). Also, the beverage industry has developed, via its Beverage Industry Environmental Roundtable (BIER), protocols for carbon and water footprinting beverage products (e.g., BIER, 2013).

A number of initiatives have also been developed by food retailers regarding the environmental footprint of products on their shelves. This is the case, for example, for supermarket retailers such as Casino and Leclerc (France), Migros (Switzerland), and Tesco (UK) that have evaluated the carbon footprint of their products (Notarnicola et al., 2012a,b).

Environmental Product Declarations (e.g. Environdec 2015) and other types of labels entailing the use of LCA, have been extensively used in industry as a means of communicating transparent and comparable information about the life-cycle environmental impact of food products. As already mentioned, such labelling schemes have not always been successful in communicating environmental sustainability information in an immediate and transparent fashion. As we have highlighted before, the development of an EU LCA based environmental footprint (PEF) is intended as a means to address these issues and should set an example for the future development of similar labelling schemes in other regions of the world.

7. Modelling food waste with LCA

Food waste is a globally critical aspect for sustainable development, both from an environmental- and food security perspective but is also a social issue. About 1.3 billion tons of edible food are globally wasted along food supply chains, corresponding to one-third of the food produced for human consumption (FAO, 2011). This food loss represents a huge 'avoidable' environmental burden and of course a huge concern from a social point of view. However, the implementation of food waste reduction measures is complicated (Mourad, 2016; Priefer et al., 2016). Each year, nearly 10 million die of hunger and hunger-related diseases (Nellemann et al., 2009). FAO (2013) has estimated that the environmental impacts associated with food wastage are about i) 3.3 $GtCO_{2eq}$, of greenhouse gases (GHG), which makes food wastage the 4th GHG producer after China, USA and EU, ii) 240 000 m^3 of irrigation water wasted, and iii) 1.4 billion hectares cultivated in vain.

Modelling food waste in LCA is common practice because the reference flow is defined as a part of the functional unit, and thus the inventory will include waste generated along the chain (relative to the reference flow). However, to specifically assess the impact of food waste a dedicated effort is needed to highlight this and agreed modelling guidelines are still missing (Bernstad Saraiva Schotta and Cánovas, 2015; Corrado et al., 2016.this issue). Following the standard procedure as described above will mean that usually the impacts associated with food waste are "hidden" in the impact assessment results for different life cycle stages in a supply chain; for food systems, this is most often the primary production (agriculture, fishery). This is nothing that demands new methods but require a dedicated interpretation of the results, which in turn put demands on how the LCA model is structured so that the specific impacts of food waste can be extracted. Food waste can be seen as a symptom of dysfunctional food supply chains, dysfunctional in both technologically and managerial ways. The latter implies that there are important non-technological aspects of the solutions to be developed. When addressing food waste, it is critical that the measures taken really contribute to real improvements and not just shift problems around. Thus LCA is an appropriate tool for the assessment of both the technological and managerial solutions. However, there is a need to apply LCA in a conscious way. Quantifying the environmental impacts of initiatives to reduce food waste demands a thorough understanding of the importance of methodological aspects such as system boundaries, systems expansion, and the time dimension. It also leads into a discussion of how to connect LCA to the challenge of food security. Another strength of LCA is that it can provide a platform for discussion and mutual learning among supply chain stakeholders since LCA provides an overarching framework for evaluating initiatives.

Additionally, and in the frame of circular economy, LCA can play an important role in the evaluation of waste management, including logistics, by environmentally assessing aspects such as:

- Nutrients recovery used as fertilizers and therefore avoided fertilizer consumption;
- Water-efficiency measures and reuse of treated wastewater; and
- Improved management along the whole production chain in order to reduce food losses during production and distribution, in shops, restaurants, catering facilities, and at home.

8. The risk of misusing LCA in the food sectors

8.1. Use of LCA results to support decision-making

The “cradle-to-grave” perspective and the multi-criteria approach of LCA makes it a suitable method for supporting decision making and assessing if alleged eco-innovations are effectively preferred when considering all life cycle stages and assessing across different impact categories. However, these multi-phase and multi-criteria attributes make LCA results very complicated to analyse and interpret without having a deep understanding of the modelling of the system studied and the meaning of the impact categories.

There is a clear tendency today to try to include all possible environmental questions, worries and concerns into LCA with the aim that this tool will be able to address all environmental issues at once. In theory, with unlimited time, unlimited resources, with access to all data instantaneously this could indeed be possible, but in practice it is not. Today there are huge issues in data availability, database inter-comparability, modelling approaches, validity and relevance of some impact assessment methods, in normalisation and in weighting. There is a real risk that rushing into additional definition of new impact categories, more exposure compartments, more complexity, more of everything, will undermine the value and credibility of LCA. Probably a more efficient approach is to apply appropriate tools in combination with LCA. Examples of such tools are Environmental impact assessment, Environmental management schemes, Human risk assessment, Environmental risk assessment, Material flow analysis, and Resource efficiency assessment of products. Use of impact categories that most decision-makers can easily understand and communicate is also required. Ideally, all impact categories should have the simplicity and accuracy of the GHG impact category.

The current PEF experience (EC, 2013) has been (and still is) extremely beneficial for highlighting, from 27 different sectors who applied the same method at the same time, what still needs to be fixed to make LCA a robust decision making tool. There is still a lot of work ahead, especially in the food sector where LCA was introduced more recently and where the method must be adapted to fit this particular sector.

In conclusion, to promote LCA as a decision-making tool, both for industry and for government policy, more robustness, reliability and representativeness are needed.

8.2. Interpretation of results– how to avoid misinterpretation

Preventing or minimizing misinterpretation of LCA studies requires, firstly, clarifying the questions the LCA study is addressing. This involves consideration of the system boundaries, the uncertainties and completeness of the data sources, the allocation approach, the way impacts are assessed, and the robustness and completeness of the modelling approaches. However, often LCA practitioners jump into overly strong conclusions. This can seriously jeopardize the uptake of the, in principle, very sound approach used in LCA. An area where serious misinterpretation often occurs is related to the comparison of intensive versus extensive production, where the impacts associated with each unit of product are reduced in relative terms (less impacts per unit of products) while in absolute terms the impacts may be higher (e.g. more pressure on soil quality, more pressure on aspects not modelled/included in the evaluation e.g. biodiversity etc.). Regarding the interpretation of the impact assessment results, this is another area where often results are misinterpreted. Here we discuss the case of the toxicity-related impact categories (aquatic toxicity and human toxicity), discussing the complexities and related uncertainty within this impact category as an example of the issue of misinterpretation in LCA.

LCA cannot be used to address questions related to the risk associated with a product. The current naming of some of the LCA impact assessment categories suggests exactly the opposite to a non-expert. ‘Freshwater Ecotoxicity’ and ‘Human toxicity – cancer – non cancer effect’ reflect the potential impact on ecosystem and human health, but should not be used to suggest that product A is actually safer that product B. Toxicity of a chemical must be assessed by considering concentration in the specific receiving compartment, not just via a quantity as reported in life cycle inventory. No matter how sophisticated the modelling of the impact assessment is, LCA does not capture essential information such as the volume of the receiving compartment and time of exposure to assess the true toxicity of a chemical when released into the environment. To accommodate the data requirements of LCA (only calculated as mass), the toxicity impact assessment category is a ‘time-integrated effect per unit mass of chemical release into the environment’ (Hauschild and Pennigton, 2002). This is a pragmatic and operational approach to deal with toxicity in LCA but it should never be interpreted as an assessment of the real toxicity impact of a product. There is just no causality between a mass of a substance and its toxic effect (Owens, 1997).

Applied to food product LCAs, the aquatic toxicity impact categories as currently modelled can lead to wrong conclusions when, for example, comparing conventional and organic farming (see section 4.4). As discussed earlier, the impact of pesticides on agricultural soil biota, and on plants, butterflies, birds and pollinators are not included yet, although they are essential for food production. Recently, a proposal for integrating new targets species in LCA, e.g. pollinators, have been made (Crenna et al. 2016).

Overall, to be more meaningful, impact categories in food-related LCA studies need to reflect the known impacts of agriculture production on the environment (loss of soil fertility, loss of soil structure, loss of pollinators, etc.). The usual list of impact categories developed for industrial systems cannot just be reapplied to food systems without considering the specificity of these systems. A recent review on soil modelling (Vidal Legaz et al., 2016 *this issue*) illustrates that several soil models have been developed focusing specifically on agriculture sectors. To the contrary, the challenge for many models is related to how they can be applied in the LCA context. In fact, when very comprehensive modelling is proposed (e.g. the SALCA approach, Oberholzer et al., 2012), this is often associated with a requirement for a huge amount of inventory data at a field scale, leading to difficulties in applying the model to large systems. Balancing data demand, comprehensiveness and applicability is the way forward.

The LCA concept is easy to understand and very appealing for academic, industry and policy makers because of its holistic nature, but being able to correctly interpret the results require a high level of expertise. However, the user-friendliness aspect of LCA tools puts LCA at the fingertips of almost everyone without the need to understand all underlying assumptions. As for other disciplines, running and interpreting an LCA should require a validated diploma with several years of experience. This is certainly not the case today although steps have been taken in the right direction through the development of LCA practitioner certification schemes such as that of the American Life Cycle Association (ALCA, 2016).

9. Conclusions

Ensuring a transition towards more sustainable production and consumption patterns requires a holistic approach and life cycle thinking is increasingly seen as a key concept for supporting this aim (Sala et al., 2015). As food production systems and consumption patterns are among the leading drivers of impacts on the environment, over time the applications of life cycle thinking and assessment to food-related supply chains have flourished. Life cycle assessment has been applied extensively to assessment of agricultural systems, and processing and manufacturing activities, and for comparing alternatives “from field to fork” and up to food waste management. However, despite the increasing number of LCA food studies and a flourishing literature on both methodological aspects and case studies, several challenges still need to be addressed in order to ensure that LCA is delivering robust results.

The main challenges highlighted in our analysis are related to different methodological aspects. Firstly, there is a need to move beyond the simple rationale that more output per hectare is sufficient to ensure increasing eco-efficiency. In fact, notwithstanding that increased efficiency of land use seems a logical way forward, in face of the increasing pressure on agricultural land for other purposes such as bioenergy, and pressure from urbanisation and desertification, the current LCA method is incomplete and does not comprehensively assess some aspects that are critical for long-term sustainable food production.

The inherent variability of the agricultural system is one element affecting the assessment at the inventory, impact assessment, and interpretation phases. Compared with other sectors, food systems are inherently more variable both in the inventory data and in the possible impacts (e.g. the impact on biodiversity due to a particular land use may differ dramatically from one ecoregion to another). How to represent relevant variability in LCA study results without having to collect such a huge range of data that these studies become infeasible requires the development of specific methodologies. This also suggests a need for specific guidelines for agricultural inventories, including improving the quality of the data available in LCA databases. The above mentioned variability and the geographical specificity of food systems calls into existence the need for clearly structured regionalised datasets that will allow the LCA practitioner to tailor the data to specific case studies.

Current LCA modelling approaches should be complemented by other approaches in order to improve the understanding of what is happening in-field (and potentially subject to specific comparisons, e.g. organic versus non-organic agriculture), and what is off-field (aka background systems) and which is affected by the reliability of secondary datasets. For example, for ecotoxicity-related impacts, it is often observed in LCA studies that the relative share of impacts associated with the substances applied on field (e.g. pesticides) is limited compared to the substances used in background systems that are off-field.

There is a clear need for consensus on more meaningful FUs for food products, and there is initial work in this field aiming at developing functional units covering the nutritional function of food (Sonesson et al., 2016b *this issue*). Regarding the comprehensiveness of the impacts, there is a need to address the threats to the environment that are still not properly addressed and modelled in LCA studies. This implies on the one hand, a need to enlarge and improve life cycle impact assessment, and on the other hand to find a way of integrating knowledge coming from other scientific domains when modelling within LCA is unfeasible (e.g., integrate qualitative considerations or warnings related to missing potential drivers of impacts, for example GMOs). This suggests a need to find a balance between quantities and qualities as well as exploring possibilities for implementation of semi-quantitative models in LCA. The goal should be to have comprehensive and scientifically sound measures. This requires simplifications in order to be applicable, which in turn puts more pressure on finding ways to collaborate between disciplines. Even more, this calls for the provision of clear guidelines for interpretation of results, including additional guidance by life cycle impact method developers in clarifying what their models is actually assessing and which are possible limits and uncertainties in the assessment.

The structure of food systems is very much influenced by consumers' choices and behaviours. Understanding this will lead to better modelling (e.g. the use phase). It will also lead to consideration of the main different aspects that influence the choice of a product, the potential for dietary shifts towards less impacting diets, changes in the perceived environmental quality associated with different products, the way in which products are consumed and, even, the amount of wastage associated with food systems. Both scientific and grey literature represent immense sources of knowledge. However, capitalizing on this knowledge through its correct use and interpretation are still open challenges within and beyond LCA. Further research on these challenges will contribute to making LCA more robust in its role of supporting decision-making that varies from individual farm decisions up to national and international policymaking for more sustainable future food systems.

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