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Assessing the environmental impact of agriculture

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Introduction

In spite of its decreasing share of global economic output, agriculture is still responsible for a large part of overall physical output, both to the rest of the economy and to the environment. For example, it has been estimated that agriculture and land use are responsible globally for around 24% of the greenhouse gas emissions fuelling climate change. A prerequisite to improving the sustainability of agriculture are reliable methods to identify and quantify types of environmental impact. These can then be used to identify priorities, set targets for improvement and monitor progress. This book provides a review of current research on the use of life cycle assessment (LCA) and other modelling techniques to measure and model the environmental impact and improve the sustainability of agriculture. The book is divided in three parts. Chapters in Part 1 review key issues in using LCA and modelling applied to farming systems. Part 2 provides more detail on the modelling of three particular impacts, namely freshwater, pesticides and social impacts. Part 3 looks in more detail on options for reducing the environmental impact and improving the performance of both crop and livestock farming.

Part 1 Life cycle assessment

Life cycle assessment (LCA) is increasingly applied to agricultural systems. However, in spite of the standardisation of this assessment technique through the ISO 14040 series, there are still many different interpretations of the requirements laid down by the standards, resulting in a plethora of ways in which LCA is applied in practice. In turn, this results in the generation of sometimes conflicting results. Chapter 1 provides a review of the most important issues in agricultural LCA and demonstrates how to deal with challenges such as system boundary delimitations, defining the functional unit, handling co-production and choice of impact assessment methods.

Chapter 2 provides a broader view on modelling of agricultural activities and their environmental impacts. The focus of the chapter is on model quality as dependent on the quality of data and the accuracy of modelling decisions to support mitigation strategies or management changes towards environmental friendlier agricultural products. Applying uncertainty and sensitivity analysis can reduce uncertainty in model outputs and direct data collection efforts towards the most important variables, thereby contributing to overall model quality. The chapter includes two detailed case studies that show the benefits of including uncertainty propagation and sensitivity analysis in model assessment. The chapter ends by describing future trends in modelling, especially in facilitating the reuse of models.

Chapter 3 describes a range of farm-level greenhouse gas modelling tools, and critically assesses their ability to function as a tool for farm-level benchmarking and mitigation assessment. The authors conclude that the majority of farm-level tools do not effectively account for their methodological choices and that the disparity in interpretation and adaptation leads to considerable differences in output. The chapter considers the challenges, limitations and opportunities offered by further development of different modelling approaches.

Chapter 4 provides a review of recent LCAs on ruminant production systems and considers the particular challenges in modelling, especially in pasture-based ruminant production systems with their high degree of variation in physical, chemical and biological processes. A case study demonstrates how uncertainty surrounding climate impacts of ruminant systems can potentially be reduced through on-farm measurements of greenhouse gas fluxes, but that not all measurements carry the same degree of value.

Chapter 5 examines the use of LCA for comparisons of intensive and extensive agricultural systems. Most often LCAs of agricultural systems compare conventional agriculture as an intensive farming system with certified organic agriculture as an extensive farming system, even though production intensity in terms of yields and inputs may vary over a wide range in both production systems. Nevertheless, comparative LCA studies on organic and conventional farming are well suited to discuss advantages and limitations of using LCA to assess the environmental sustainability of farming systems of different production intensities. The chapter assesses the validity of the modelling of reactive nitrogen and heavy metal emissions in different farming systems.

Part 2 Modelling particular impacts

Chapter 6 looks at modelling the impacts of agriculture on freshwater consumption, water quality and salinization. 80-90% of human freshwater consumption is accounted for by agriculture, mainly due to the high volumes of irrigation water for crop production. Additionally, agricultural activities affect hydrological cycles through land use changes and soil modifications, and through pollution of water bodies with fertilizers and pesticides. The chapter addresses data availability and databases, the issues of high geographical and temporal variability, and current gaps in modelling. Notable gaps are found in hydrological data and models for groundwater as well as in the assessment of impacts on ecosystems.

Chapter 7 reviews how pesticides are currently addressed in impact assessment. A real-life scenario study of four selected pesticides applied to potatoes illustrates and provides guidance on the quantification of emissions, exposure and toxicity. The chapter discusses the relevance of spatiotemporal variability in modelling emissions and the toxicity and ecotoxicity impacts of pesticides, and how substitution scenarios can be used to identify more sustainable pesticides.

Chapter 8 looks at the wider economic and social impacts of agriculture. Economic assessments remain the dominating form of socio-economic assessment in agriculture, but its practical implementation often suffer from severe limitations. The chapter describes how much of the early criticism has slowly been adopted and integrated in the form of significant improvements in the consistency and completeness of economic assessment techniques. Nevertheless, a better integration with the qualitative understandings developed in the social impact assessment community is still largely missing, notably with respect to power, politics and institutions, and viewing human and social capabilities as direct contributors to wellbeing, i.e. beyond being just economic factors of production. The chapter suggests that these limitations can be overcome by combining existing models. The unique importance of agriculture in sustainable development is highlighted through a discussion of how the benefits of increases in agricultural productivity are distributed, as illustrated by the role of inequality, land tenure, and food pricing. Most of these issues can only be effectively addressed by well-functioning public institutions. The final section of the chapter discusses to what extent certification and fair-trade schemes might address these issues in the absence of public governance.

Part 3 Improvement options

Part 3 looks assesses the available options for mitigating environmental impacts within agricultural crop production (Chapter 9), horticulture (Chapter 10), and pork and poultry production (Chapter 11). A final chapter looks at the options for valorising by-products. In general, the four chapters point out the large variation between local contexts, crops, and types of impacts. Nevertheless, some general conclusions can be drawn. The authors of Chapter 9 highlight the promising results from research on agro-ecology and other quantitative evidence on designing highly efficient production systems. Chapter 10 examines key findings from LCA of horticultural crops and production systems and finds a relatively low environmental impact compared to other food types. For fruit crops, the largest impacts occur at the farm level, most often related to machinery use, pesticides and fertilizers.

Currently, pork and chicken account for nearly three quarters of global meat consumption. With increasing concern for the environmental impacts associated with human activity, and due to the magnitude of the pork and poultry sectors, there is mounting pressure to meet the growing demand for these animal-based proteins with fewer resources and lower emissions. Chapter 11 points out that the emission intensity of pork and poultry operations is largely determined by the feed conversion ratio and manure handling. The authors outline improvement options and provide case studies for making pig and poultry production more sustainable. In their outlook for the future, they highlight the role of data science, data-driven decision-making and automation in order to continue improving efficiency in agriculture.

Chapter 12 provides an overview of opportunities to reduce the environmental impact of valorising agricultural by-products, i.e. crop residues, industrial by-products, and animal manure. The authors estimate the available by-products and the maximum and realistic greenhouse gas reduction potentials in North Western Europe, assessing the relative importance of availability and collectability, as well as broader issues such as land use changes, soil carbon sequestration and pollution swapping.

To the reader

The chapters of this book can be read individually. As editor, I have sought to ensure consistency in the use of terms and methodological recommendations. It is my hope that this book can serve as a standard reference for the agricultural science and LCA research community and that it will support further research to improve measuring and modelling the environmental impact of agriculture.

Part 1

Life cycle assessment

Chapter 1

Life cycle assessment methodology for agriculture: some considerations for best practices

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1 Introduction

Along with the increased demand by a growing and wealthier human population for healthy and nutritious food, growing awareness and concerns in today's society over the environmental impacts associated with agricultural systems have shifted focus to sustainable consumption and production (Mazur-Wierzbicka, 2015). As a result, agriculture has been the subject of considerable public debate and scrutiny (Jansen and Vellema, 2004).

Life cycle assessment (LCA) is an approach formalized in the 14040 and 14044 standards of the International Organisation for Standardisation (ISO) (ISO, 2006a,b). These standards provide a methodological framework for quantitatively evaluating the environmental impacts and as a further step, the trade-offs among the corresponding LCA results, economic and social impacts can be considered for sustainability assessment (Lee and Inaba, 2004). While focus on socio-economic objectives, such as employment effects and contribution to gross domestic product (GDP), historically took predominance over environmental concerns (FAO, 2006), the latter are swiftly catching

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up by drawing the interest of policy makers and expressed preferences of consumers. One of the limitations of typical LCA studies is the exclusion of social and economic aspects of product systems, which may favour a decision that otherwise would not be favoured. A sustainability assessment study that excludes these impacts is incomplete. However, incorporating life cycle costing (LCC) and social impacts has proven challenging, as pointed out by Dreyer et al. (2006), who noted that 'recommendations based on LCA fail to address possible trade-offs between environmental protection and both social and economic concerns in the product life cycle'. In an attempt to overcome this issue, a wide range of studies have been conducted on life cycle social assessment (LCSA) and LCC (Klöpffer and Renner, 2008; Finkbeiner et al., 2010). Although including a social assessment into agricultural LCA studies is still in its infancy, literature exists on some case studies and applications. In this regard, agriculture and agro-food systems are the first sectors in which social LCA has been applied – for example, tomatoes and citrus fruits from Italy, milk and eggs from Canada, bananas from Cameroon, roses from Ecuador and the Netherlands, and oysters from Denmark (Petti et al., 2018; Pelletier, 2018; Revéret et al., 2015; De Luca et al., 2015; Franze and Ciroth, 2011; Wangel, 2014; Feschet et al., 2013).

In order to manage the social performance of a production system, it is required to ensure that a wide variety of sustainability concerns are met, including labour issues, human health, resource availability, animal welfare, biodiversity, and food security, in addition to the quantification of the impacts of emissions to air, soil, and water. In this context, LCA can be an important tool for measuring, benchmarking and, in turn, improving the social performance of the agri-food sector. This is relevant, in particular, when seeking to understand and make comparisons between traditional subsistence and industry and farming, organic and conventional production systems, or animal-based and animal-free diets.

The concept of social responsibility has been defined as 'the commitment of business to contribute to sustainable economic development, working with employees, their families, the local community and society at large to improve their quality of life' (Holme and Watts, 2000). In ISO 26000 (ISO, 2010), this concept has been defined more comprehensively as: 'responsibility of an organization for the impacts of its decisions and activities on society and the environment, through transparent and ethical behaviour that (1) contributes to sustainable development, including health and the welfare of society; (2) takes into account the expectations of stakeholders; (3) is in compliance with applicable law and consistent with international norms of behaviour; and (4) is integrated throughout the organisation and practised in its relationships'. Responsibility allocation is an issue on which there is no general agreement and the literature shows different viewpoints (Mankiw and Taylor, 2006; Ferng, 2003; Lenzen et al., 2007; Rodrigues et al., 2006). Based on the literature, allocation of responsibility can be done according to criteria representing either *producer responsibility*, *consumer responsibility*, or *income responsibility* (Weidema et al., 2018). Although these three types of responsibility are discussed in the context of global regulation of greenhouse gas (GHG) emissions, they can be generally applied to all environmental impacts of the product life cycle. According to the first alternative, the producer bears the main responsibility for the environmental impact of a goods item or a service (we use the term *product* to denote both goods and services) throughout the entire life cycle. This allocation method ignores the important role of the final consumers as the main drivers for environmental impacts and that outsources the environmental impacts to countries with fewer commitments (Pedersen and de Haan, 2006; Weidema et al., 2018). The second alternative ascribes all environmental impacts related to production and consumption of a product to the consumers. This encourages the consumers to purchase products with the least environmental impacts. The last alternative – income responsibility – allocates all of the environmental impacts in the value chain to the activities that receive income from the value chain (Weidema et al., 2018). Since LCA studies are performed in the context of life cycle management and social responsibility, the relationship between LCA models and social responsibility needs to be investigated. Accordingly, Weidema et al. (2018) identified three social responsibility paradigms, namely *value chain responsibility*, *supply chain responsibility*, and *consequential responsibility* for social assessments.

In summary, the current commitments to continuous improvements over time in social responsibility and environmental performance are major challenges for policy makers, producers, and consumers, who rely on scientifically robust tools with which to quantify sustainability impacts and meet self-imposed targets. The LCA can play a key role in supporting the transition to socially and environmentally responsible systems to obtain the systematic and scientifically robust sustainability results that it can produce.

2 Identifying a functional unit

The functional unit (FU) is a key element in LCA and must be defined clearly. ISO (2006a) states that 'LCA is a relative approach, which is structured around a FU. This FU defines what is being studied. All subsequent analyses are then relative to that FU, as all inputs and outputs in the life cycle inventory (LCI) and consequently the life cycle impact assessment (LCIA) profile are related to the FU'. In other words, an FU for the product of an agricultural system ought to represent a quantifiable value associated with the function of the agricultural production system under study (Caffrey and Veal, 2013a). Defining the appropriate FU is dependent on the system being analysed, the goal of the study, and the system boundary of the LCA and is generally chosen to reflect the way the product is traded (Harris and Narayanaswamy, 2009); in case of comparing different alternatives, an equivalent basis of the alternatives can be considered (e.g. energy content in biofuels and gasoline or diesel as a transport fuel). Weidema et al. (2004) provided some useful recommendations on key issues in LCA in the form of a guideline. Based on this guideline, the FU needs to be, as far as possible, related to the function of the product rather than to the quantity of the product. For example, for agricultural tillage machinery 'the tillage area (with specific depth) per hour' can be selected rather than choosing 'the mass of tillage machine'. Failing to define an appropriate FU makes it impossible to compare different LCA results (Caffrey and Veal, 2013b) (e.g. comparing organic and conventional systems). Depending on the type of study and product, different product attributes can form the basis for defining the proper FU in the modelling of agricultural systems.

2.1 Mass

The majority of agricultural studies use a mass-based FU. In this category, the environmental impacts of a product are related to 1 kg or tonne of a product, regardless of the quality and commercial value of the product (Cerutti et al., 2013; Pishgar-Komleh et al., 2017a). For simplicity, the mass-based FU is widely used in LCA of agricultural products. Studies on animal production tend to use mass-based FUs for milk, butter, yogurt, cheese, ice cream and whey products (e.g. 1 kg of raw milk) (Basset-Mens et al., 2009). Examples of agricultural LCAs that adopt mass-based FU are (Basset-Mens et al., 2009; Haas et al., 2001; Mollenhorst et al., 2006; Fantin et al., 2017; Anton et al., 2005; Hayashi, 2005; Nienhuis and de Vreede, 1996; Pishgar-Komleh et al., 2013).

2.2 Energy

In studies that deal with bioenergy and biofuel production, an energy-based (or a proportional function, such as distance driven) FU can be used, for example, 1 MJ, since the function of the system is to deliver energy. Studies that have employed an energy-based FU in the LCA of agricultural crops include Hanegraaf and Biewinga (1998), Heller et al. (2003), Monti et al. (2009) and Fazio and Monti (2011).

2.3 Other functional units

In some studies, alternative FUs are relevant (Cerutti et al., 2013). Since land use *per se* is not seen directly as a production function, land-based FUs are not commonly used in LCA. However, this FU can be correctly applied when the goal of the study is, for example, related to supporting the minimization of environmental impacts associated with cultivating a certain area (Mouron et al., 2006a,b). When comparing different land-management options (e.g. food, fuel, feed, fibre timber, or carbon sequestration), given that the functions of the different land uses are different, it is necessary to balance each alternative so that they can be compared. The FU (e.g. 1 ha) would thus reflect no net production (Brandão, 2012). The land-based FU has been used to compare low input–low output systems with high input–high output systems (Cerutti et al., 2013), although the outputs across alternatives must be equal to enable comparability. In addition, this FU has been used to compare crop-rotation systems including fallow land (Karlsson and Sund, 2016). Defining the appropriate FU for comparing and contrasting farming systems, such as organic and conventional systems, is very important. Applying different FUs may reverse the ranking of the options being compared (e.g. organic and conventional product systems seen from a mass- or land-area-based FU). Intuitively, it is widely believed that organic farming is an environmentally superior alternative to conventional farming; however, this viewpoint is often challenged by the insights gained from LCA studies, where results are normalised by the quantity of product. Tuomisto et al. (2012) found that organic systems may decrease nitrate leaching, while increasing land requirements and soil erosion. Nemecek et al. (2011) showed organic farming has better environmental outcomes at the farm scale (per ha), but not per unit of output (per kg product). Several LCAs have used land-based FUs in the modelling of agricultural and animal products (Haas et al., 2001; Nemecek et al., 2001; Huguenin-Elie and Nemecek, 2004; Basset-Mens and van der Werf, 2005; Hayashi, 2005; Mouron et al., 2006a; Xiao et al., 2019; Pirlo and Lolli, 2019); however, results must be interpreted with caution because the comparability between alternative systems, which requires that systems to produce an equivalent amount of all products, is not always ensured.

In other studies, value-based or economic-based FUs have been adopted, which can be useful because they integrate product quantity and quality in a single index (Mouron et al., 2006b; Cerutti et al., 2013). The disadvantage of this FU is that it does not enable comparison across countries and years. It is strongly influenced by the economic context in which the farm is located and can change significantly from one year to another (Cerutti et al., 2013).

Despite the need to adapt FUs to the product properties and purpose of the study, the majority of LCA studies on agricultural products use the above FUs. For example, in studies on animal products, FUs have been defined as:

- 1 kg or tonne of energy corrected milk (ECM) leaving the farm-gate over a specific period of time (Cederberg and Stadig, 2003; Cederberg and Mattsson, 2000; Cederberg and Flysjö, 2004; Casey and Holden, 2005);

- 1 kg or tonne of fat and protein corrected milk (FPCM) (Pirlo and Lolli, 2019; Gerber et al., 2011; Thomassen et al., 2008b, 2009; Basset-Mens et al., 2009).
- 1 litre or tonne of milk produced and sold per year per herd size (Hospido et al., 2003; Williams et al., 2006; Eide, 2002);
- number of products (such as 1,000 eggs produced) (Williams et al., 2006);
- 1 kg of bone-free meat leaving the farm-gate (Basset-Mens et al., 2009; Cederberg and Darelius, 2002; Cederberg and Stadig, 2003).
- 1 kg or tonne of live or dead animal carcass, such as pig, chicken, and beef leaving the farm-gate (Basset-Mens et al., 2009; Haas et al., 2000; Eriksson et al., 2005; Basset-Mens and van der Werf, 2005; Williams et al., 2006; Casey and Holden, 2006; Katajajuuri, 2007; Pishgar-Komleh et al., 2017b).
- 1 kg or tonne of animal protein, such as pork, chicken, or beef protein (Zhu and van Ierland, 2004).

3 Unit processes as the building blocks of LCI

3.1 Collecting data on inputs and outputs: completeness vs. precision

A 'unit process' is one or more grouped operations in a production system that can be defined and separated from others. According to ISO (2006a,b), unit processes are the smallest elements in the LCI of product systems for which input and output data are quantified. With high data resolution (i.e. geographical, technological, and temporal), it can refer to an activity such as transportation, planting, harvesting, tillage, cleaning, and separation; if less resolved data are available, it can refer to a collection of activities that aggregate into a higher-level process, like crop production, post-harvest and so on. The outputs of a unit process are main products, by-products, emissions, and wastes, while inputs refer to resources, such as raw materials and energy, intermediate products, or other products (e.g. fertilizers or pesticides).

Like any assessment tool, data are a driving force behind LCA (Curran, 2015). Data acquisition is one of the most complex issues facing LCA practitioners (Klöpffer and Grahl, 2014). Large numbers of interlinked processes need to be considered to complete the LCI. Raw material inputs, energy use, ratio of main product to co-products, production rates, and environmental releases must all be quantified for each process in the production system (Curran, 2015). LCI data can be classified into two categories: foreground and background. Foreground data are typically those unique to the product system being modelled and are collected directly from sources including on-site measurements, interviews, questionnaires or surveys, and online or offline data collection tools. Background data represent common industrial processes that may be shared across multiple product systems and typically come from databases, open literature, and statistics. Once raw data are collected, data modelling approaches are applied to transform raw data into unit process datasets. Unit process datasets are the basis of every LCI database.

Depending on the LCI data collection method, LCA can be classified into three types: process-based, input–output (IO) based, and hybrid LCA (Heijungs and Suh, 2002). To select the most appropriate method for a given study, important factors such as boundary completeness and precision should be considered carefully with respect to the goals of the study (Lenzen and Crawford, 2009).

In process-based LCA, all the resources and emissions over the entire life cycle (or specific life cycle stages) of a product are considered. This allows calculating the complete environmental impact of a product. However, finding physical data for all inputs and outputs of a background process can be challenging, which can lead to the underestimation of impacts (Curran, 2015; Mattila, 2018; Crawford et al., 2018). It is challenging, if not impossible, to have a complete LCI with process data.

To overcome this problem, IO-based LCA can be applied. The IO-based LCA relies on sectoral-exchange information in a given year for a country's entire economy. This approach uses an IO table containing the annual transactions between all the sectors of the economy, along with data for sector-specific resource inputs and emissions (Curran, 2015; Mattila, 2018). Some of the limitations of this approach include price variability, sector aggregation, and the lack of input-specific data (Teh et al., 2018). The IO-based LCA requires detailed sector-level data regarding resource inputs and emissions. These are often available for certain kinds of emissions that have particular policy relevance – for example, GHG emissions – but will often be unavailable for many standard impact categories in LCA. Since the IO-based models use aggregated sectoral data, product-level LCIs will be less accurate (Huijbregts et al., 2001a). For instance, different agricultural systems (notorious for high variability among products and regions), such as organic and conventional farming systems, belong to the same sector, and thus the results of environmental impact assessments for products from these sectors will be similar, which is not valid (Curran, 2015). This concern applies to all agricultural commodities, which are considered equal in terms of resource use and emissions, when IO LCA results are calculated. Numerous LCA studies of agricultural products used process-based models (Cellura et al., 2012; Pelletier et al., 2010; Stoessel et al., 2012; Lardon et al., 2009) while few studies used IO-based models (Yan et al., 2013; Joshi, 1999).

In order to have both the completeness of the IO-based LCA and the precision of process-based LCA, a hybrid LCA approach can be applied (Suh and Huppes, 2005; Curran, 2015; Yang et al., 2017; Pomponi and Lenzen,

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