Chapter 29
LCA of Food and Agriculture

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Abstract This chapter deals with the application of Life Cycle Assessment to evaluate the environmental sustainability of agriculture and food processing. The life cycle of a food product is split into six stages: production and transportation of inputs to the farm, cultivation, processing, distribution, consumption and waste management. A large number of LCA studies focus on the two first stages in cradle-to-farm gate studies, as they are the stages where most impacts typically occur, due to animal husbandry and manure handling, production and use of fertilisers and the consumption of fuel to operate farm machinery. In the processing step, the raw agricultural product leaving the farm gate is converted to a food item that can be consumed by the user. Distribution includes transportation of the food product before and after processing. In the consumption stage, environmental impacts arise due to storage, preparation and waste of the food. In the waste management stage, food waste can be handled using a number of technologies, such as landfilling, incineration, composting or digestion. A number of case studies are looked at here where the life cycles of typical food products (meat, cheese, bread, tomatoes, etc.), and an entire diet are discussed. Other case studies deal with what LCA can conclude on the differences between conventional and organic farming, and the perceived advantages of local food items. Finally, methodological issues in agricultural LCA are discussed: the choice of functional unit, setting the boundary

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between technosphere and ecosphere, modelling flows of nutrients and pesticides, and the generally limited number of impact categories included in LCA studies.

29.1   Introduction

Agricultural production systems have changed dramatically over the course of the past century. The introduction of the combustion engine around 1900 started the replacement of human labour with fossil energy. In 1910, the Haber–Bosch process to bind nitrogen from air was commercialised. Synthetic fertilisers could now be produced on an industrial scale. In the decades after the Second World War synthetic pesticides became widely used, which together with improved plant breeding led to the so-called Green Revolution: a spectacular increase in yields worldwide, but especially in Asia and Latin America. More recently, techniques to alter the genetic material of crops have been applied to develop new plant varieties, such as herbicide-resistant maize. Especially in Europe this development has led to controversy.

   These developments do not mean that there are no challenges remaining for agriculture: the world population keeps growing. The global population increased from 1.65 billion in 1900 to 6 billion in 2000. In 50 more years, an additional 2 to 4.5 billion people will be added to the population (FAO 2012). Part of this growing population is becoming increasingly affluent. More affluence results in a higher demand for food in general—a year-round supply of fruits and vegetables either imported off-season from distant countries or produced in artificialised production systems such as heated greenhouses—for meat in particular. Part of this population, especially in poor countries, is also increasingly living in cities (50% in 2007 at the world scale, Kulikowski 2007), implying the import of food from rural areas and the development of urban farming. Urban farming is showing a great potential for securing food supply, creating jobs and alleviating poverty in the Global South, but it can be accompanied by environmental and human health risks due to intensified and not always well-controlled practices.

   Today, food production is associated with major environmental problems. Rachel Carson's famous 1962 book Silent Spring addressed an important environmental problem: the effects of pesticide use on non-target animals, including humans. Later, focus shifted to issues such as eutrophication due to nutrient runoff. Greenhouse gas (GHG) emissions caused by agriculture are high on the agenda today. For example, the United Nations Food and Agricultural Organization (FAO) estimates that livestock alone contributes 18% to the global GHG emissions and that 50% of the methane emitted into the atmosphere by human activity is due to crop and livestock production (FAO 2013). Another environmental problem is the disruption of the nitrogen cycle. Human extraction of nitrogen from air, mainly for fertiliser production, is larger than all natural processes extracting nitrogen from air. Subsequent application of the produced fertiliser results in extensive nitrogen emissions to surface water. Last, but not least, agriculture is the economic sector

with the largest requirements of water and land and the main driver of land use change, e.g. through the conversion of forests into agricultural land (FAO 2013).

In order to gain insight into the environmental performance of agriculture and agricultural products, a considerable number of Life Cycle Assessments (LCA) of these products have been carried out in recent years. Also, several initiatives encourage the environmental life cycle-based assessment of food products, such as the Envi Food protocol (Food SCP RT 2013), the Product Environmental Footprint (PEF) pilots (European Commission 2016), and the well-established international LCA Food conference.

This chapter focuses on the LCA of food products. Foods are here defined as products of plant or animal origin that provide macro and micro nutrients and energy to the human body. They can be either produced by a form of land-based agriculture and aquaculture or collected in the environment, such as seafood or fungi. The structure of the chapter is as follows. First, the life cycle of food products is presented and discussed. After that, a selection of LCA studies is reviewed, illustrating typical LCAs of food products and describing some relevant cases. Finally, the main results and methodological issues found in the case studies are summarised.

29.2   The Life Cycle of Agricultural Food Products

Six stages can be distinguished in the life cycle of agricultural products, which are somewhat different from the usual stages in LCA (see Chap. 6). A large part of the agricultural LCAs carried out are cradle-to-farm gate studies (see Fig. 29.1), because of the importance of agricultural production, and because the agricultural stage often bears the largest environmental impacts.

29.2.1   Production and Transportation of Inputs

In most modern food systems, the first stage of the life cycle is the production and transportation of inputs, such as agrichemicals, machines, building elements, seeds and energy carriers such as fuel and electricity to the farm. The production of these inputs has a vast geographic scope and requires substantial transportation.
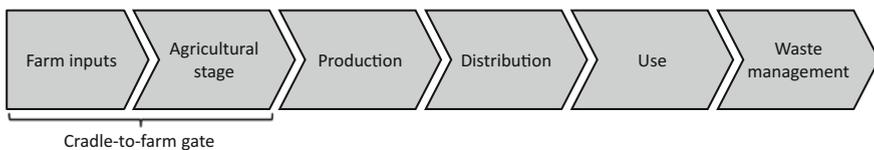
Fig. 29.1 The six stages in LCA of agricultural products

Agrichemicals include pesticides and fertilisers. Pesticide is a general term covering all chemicals (insecticides, fungicides, herbicides or others) used to protect the farm product from different pests, diseases or undesired plant growth. Both the production of pesticides and of fertilisers contribute to environmental impact potentials on the site of their production. The manufacturing of both nitrogenous and phosphorous fertilisers requires substantial energy inputs: according to a 2004 estimate (Swaminathan and Sukalac 2004 as cited in IPCC 2007), the fertiliser industry consumes 1.2% of the total annual energy use, and contributes similarly to GHGs emissions, mainly carbon dioxide and nitrous oxides. Moreover, phosphorus is derived from phosphate rock, which is a non-renewable, overexploited resource whose reserves may be depleted in 50–100 years (Cordell et al. 2009). In animal husbandry, the production and transport of animal feed is associated with emissions of N_2O and CO_2, mainly caused by fertiliser production and fuel use.

The production of capital goods, such as machinery, buildings for animals, greenhouses or glasshouses can have contrasted contributions to overall food impacts. Usually, farm equipment is used for a longer period of time, whilst an LCA study typically considers the production of a given mass of product, or the production from 1 ha of land, so the impacts of the production and disposal of the farm equipment would have to be allocated over different product systems. Consequently, the impacts of the equipment are often relatively small. In some cases though, for example growing tomatoes in tunnel greenhouses, the production of the greenhouse may be an important source of environmental impacts as reported by Torrellas et al. (2012a). Previous studies have shown the importance of including agricultural capital goods in environmental assessments. In particular, for protected crops, structural components of unheated greenhouses may account for nearly 30% of the total impacts in environmental impact categories such as resource depletion and global warming (Antón et al. 2014). In accordance with ISO standards (ISO 2006a, b), in order to be accurate when assessing the environmental impact of products, infrastructure must be taken into account as capital goods are explicitly part of the production system. Most guides recommend including capital goods in the assessment when they contribute more than 5% of the total impacts per impact category (EU-JRC 2010).

29.2.2   Agricultural Stage

The second stage in the life cycle of food products is the agricultural stage. This stage starts at the origin of the food product, for example seeds, fertilisers, pesticides, water and energy in case of crops, and breeding of animals in case of meat or dairy products. In the agricultural stage, all these inputs are used to produce the food product. In this section, the most important processes are described. However, some LCA studies may include other processes than the ones listed here.

29.2.2.1 Application of Agrichemicals

Aside from their beneficial effects on the crop growth and yield, application of agrichemicals results in emissions from the field and from the buildings in which animals are kept. Pesticides are mainly considered because of their toxic impacts on non-target species during or after application and bioaccumulation in harvested parts of the crops, potentially contributing to ecotoxicity and human toxicity impacts. Because pesticides are distributed over the agricultural field when used, the disposal stage is usually not included in LCAs. However, pesticide residues are present in packaging materials, in the sprayer and in the water used to clean the sprayer after application (van Zelm et al. 2014). Ideally, the disposal and subsequent fate of these residues should be included in LCA studies.

Fertilisers are applied to supply nutrients to crops, mainly nitrogen (N), phosphorous (P) and potassium (K). Fertiliser consumption typically contributes to potential impacts due to field emissions into all environmental compartments: air, water and soil. More specifically, on-farm use of fertilisers results in emissions of ammonia (NH_3), nitrous oxide (N_2O) and nitrogen oxides (NO_x) to air, contributing to impact categories such as acidification, climate change and eutrophication. In addition, emissions of N in the form of nitrate (NO_3^-), and of P in the form of phosphate (PO_4^{3-}) and particulate P through erosion, result in eutrophication of nearby water bodies and ultimately of the ocean and the sea. In LCIA practice, N emissions are considered to result in marine eutrophication, and P emissions result in eutrophication of freshwater bodies (see Chap. 10).

Farming practice influences the emissions of pesticides and fertilisers to the environment. When performing an LCA study, the choice of agricultural data should reflect the goal and scope of the LCA study. The study might focus on either average practices, i.e. the agricultural practice that is most common among farmers, even if these are non-optimal practices, or innovative and alternative agricultural practices. Worst, best or alternative practices should only be modelled when primary data is present to document this behaviour, or when the LCA study aims to compare different practices (van Zelm et al 2014).

29.2.2.2 Animal Husbandry and Manure Management

Apart from application of agrichemicals for forage production specified in the section above, enteric fermentation and manure handling are important contributors to impacts such as climate change, eutrophication and acidification. Enteric fermentation in the digestive tract of ruminants (cattle, sheep, goat) produces methane. This process is one of the major contributors to GHG emissions in LCAs of products from these animals. In a typical cradle-to-farm gate study, it will contribute approximately one-third of total climate change impacts. Excretion and manure handling from both ruminants and monogastric animals (pigs, chickens and other birds) result in direct emissions of CH_4 and N_2O. Manure can be applied as fertiliser, thus replacing synthetic fertilisers. Prior to application on the field,

manure can be treated mechanically, biologically or chemically. Ten Hoeve (2015) showed that all manure treatment technologies have inherent environmental advantages and disadvantages, and hence, the choice of technology depends on local policy preferences, costs and practicality.

29.2.2.3 Farming Operations

Fuel is consumed during a number of on-farm operations. Ploughing is an especially energy-intensive activity because of the large volume of soil that needs to be moved. Other fuel-consuming activities are the application of fertiliser and pesticides, roughage production, harvesting, heating of greenhouses, transport of the product, etc. Because the fuel used on farms is normally from a fossil origin, these actions induce non-renewable resource use and climate change impacts.

The use of machinery in farm operations negatively affects soil quality through compaction and erosion. Degraded soils are less productive and require extra inputs of fertilisers to maintain food production steady on the short term. In addition, soil ecological functions, such as buffering and filtering of toxic chemicals, water retention and soil-biota, are also affected. Hence, at the long term, unsustainable farming practices may lead to irreversible soil degradation. To maintain food production, land cover is transformed to accommodate new cropland areas, many times at the expense of natural vegetation. Today, croplands and pastures have become one of the largest biomes on the planet, occupying around 40% of the land surface (FAO 2011) and still expanding to feed the growing population.

29.2.2.4 Irrigation

Irrigated crops sustain 40% of the global food production (Abdullah 2006) and are responsible for around 70% of global water withdrawal taken from surface and ground water bodies (FAO 2014). Especially in arid countries (e.g. Australia, India, Spain) and for water-intensive-crops (e.g. almonds, rice) impacts derived from water consumption might be one of the main contributors to overall food production environmental impacts. Water leaving the farm is a vector of salts, toxic and nutrient-rich pollutants, potentially affecting aquifers and surface water bodies downstream. In many situations, water withdrawal is also responsible for a large use of non-renewable energy resources, with important associated environmental impacts, in order to transport the water to the field.

29.2.3 Processing

Processing is any step to convert the raw farm product into a (packaged) food item that will be preceded and followed by logistic phases further described below.

There is a wide variety of processing steps that can be performed, some of which have been included in LCA studies. Both the types and importance of potential environmental impacts from food processing vary a lot in function of the food item considered, as showed in the selected case studies commented in Sect. 29.3.

29.2.4   Distribution

All food life cycles include distribution stages dealing with (often refrigerated) transportation to the warehouse and to the retailer, sorting fruits, conditioning, packaging and cool storage for good maintenance of food properties. Together these processes can contribute a large share to the potential impacts of food products. These impacts are mostly related to the use of non-renewable energy, such as fuel use in transportation and electricity in cool storage. These distribution phases are particularly important in fresh products' life cycles such as fruits and vegetables. In animal products' life cycle, where the contribution of the agricultural stage is large, the impacts from distribution will appear minor. In contrast, in fruit and vegetable products, the stage of refrigerated transport and storage can have a major contribution to the total impacts. This is particularly true for fruits and vegetables transported over long distances, especially when these are air-freighted (see Table 29.1). For instance, Sim et al. (2007) revealed that the transportation of French beans by airplane from Kenya to England constituted 95% of their overall impacts. The mode of transportation more than the distance itself will play a role.

29.2.5   Consumption

The use stage of food mostly includes food transport from the retailer to the point of consumption as well as energy use for cooking and storing. This stage might include, depending on the LCA case study, private households or restaurants and institutional kitchens (Sonesson et al. 2003). Most studies that have included the use stage conclude that its environmental impacts are related to food storage or preparation. In the complete life cycle of a food product; however, these steps usually are minor contributors to overall impacts in most categories (Schau and Fet 2008).

Table 29.1 CO_2-eq emissions per tkm (1 tonne transported over 1 km) for different modes of transport (Cristea et al. 2013)

Mode of transport	Emissions per tkm (g CO_2-eq)
Road	120
Rail	23
Ship	5–12
Air	475–1000

An increasing part of meals are consumed in restaurants, institutional kitchens, caterers, etc. According to some studies, the variations in energy use for cooking between restaurants and institutional kitchens are large as are the variations between different dishes. Dishwashing, which is a direct effect of home cooking and eating, uses large and varying amounts of either cold or hot water. Hot water is an important contributor to a household's total energy use.

An aspect that is currently finding its way into LCA practice is exposure of humans to pesticide residues in food during consumption, leading to human toxicity impacts. Even though most countries have regulations in place to limit human exposure to pesticides to levels considered safe, LCA practice aims to quantify any effects on humans, no matter how small these may be. With the release of the dynamiCROP model (Fantke et al. 2011) and USEtox 2.0 in 2015 (USEtox 2015), this pathway is covered in LCIA practice.

29.2.6 Waste Management

The disposal stage of a food product consists of both handling of food waste generated along the entire life cycle, and the treatment of human excretion resulting from food intake. An alternative not considered here is to allocate the impacts of waste handling to the life cycle stage where the waste arises.

The food sector is wasteful. About one-third of all food produced in Europe for human consumption is lost or wasted before people consume it. For fruits and vegetables, this number may reach $\sim$45%. In general, 20% of food produced is wasted along the supply chain, from agricultural production (9%) to post-harvest handling and storage (4%), processing and packaging (5%) and distribution (3%). The consumer discards between 15 and 33% (Williams and Wikström 2011). Reduction of food waste in the use stage has been shown to be an effective way to reduce food environmental impacts. Avoidance or at least reduction of waste must be the priority. Once waste is produced it is also a challenge to close loops of nutrients and other materials. The potential to extract valuable bio-chemicals or recover energy and nutrients from various waste streams is significant, namely the recovery of energy and nutrients through digestion and composting is one of the most common methods in the food sector (Ellen MacArthur Foundation 2015). A number of studies have dealt with food waste handling, considering options such as landfilling, incineration, centralised and decentralised composting, digestion to produce biogas and conversion to animal feed. The results of the studies do not uniformly point in one direction and there also appear to be trade-offs between different impact categories.

For some impact categories, food waste and human excretion can result in substantial contributions to the impacts found. For example, various GHGs are emitted from wastewater treatment and subsequent sludge disposal. Depending on the wastewater treatment facility, emissions of N and P to surface water may

contribute to the eutrophication potential (see more on LCA of wastewater treatment in Chap. 34).

29.3   Selected LCA Studies on the Food Sector

This section will start by describing a number of contrasted examples of LCAs of food products, followed by a number of case studies about various developments intended to lower the environmental cost of supplying food. This selection of LCA case studies does not aim at being exhaustive, but rather at illustrating the diversity of case studies.

29.3.1   Examples of Food Product LCAs

Below, we describe a number of LCAs of different types of food products: meat, cheese, bread and tomato. The studies have been chosen based on representativeness of their outcomes among LCA studies for similar products, and inclusion of processes beyond the farm gate. This chapter closes with an LCA study of a full diet to give an overview of the relative magnitudes of impacts of different food items.

29.3.1.1   Meat

Dalgaard et al. (2007) conducted an LCA of Danish pork exported to the United Kingdom to determine the environmental hotspots. The functional unit of the study was 1 kg of Danish pork (carcass weight) delivered at the port of Harwich.

The system boundaries in this study included the pig farm, the slaughterhouse and the use and maintenance of transport infrastructure required to transport the pork to the UK. Because of the chosen consequential-LCA approach (see Chaps. 8 and 9), the feed products considered were limited to grain and soybean meal, which is the most competitive feedstock. Grain is mixed with soybean meal to achieve the optimal protein content in the feed.

Manure and other by-products are considered as co-products of pork meat. The manure is used as natural fertiliser or is anaerobically digested into biogas for district heating and electricity production. The animal by-products are used as feed or for energy purposes. For these by-products, a system expansion was done in which environmental impacts of synthetic fertiliser, fossil fuels, grain and soy for feed production were subtracted as avoided impacts.

The study was limited to three environmental impact potentials: global warming, eutrophication and acidification. The impact assessment method used was EDIP97 with updated global warming characterisation factors for methane and nitrous

oxide. Normalisation and weighting steps, which are optional in LCA, were not done in this study.

The results, summarised in Table 29.2, showed that the environmental hotspots were in the farm's input stage (i.e. production of grain and soybean meal used as forage). Approximately two-thirds of the global warming impact, which was 3.6 kg CO_2-eq over the entire life cycle, was attributed to the production of farm inputs out of which 2 kg CO_2-eq came from the production of grain. The authors did not specify where in the production of grain these impacts arise. The agricultural stage (mainly pig housing) contributes a further 0.9 kg CO_2-eq due to methane emissions from manure. Meat processing in the slaughterhouse contributed about 5% of the global warming impacts, while transport to the UK contributed less than that. The use of manure as natural fertiliser resulted in a negative impact: although methane was emitted from manure, the avoided impacts from not producing synthetic fertiliser were greater.

More than 99% of the eutrophication impacts were associated with grain production, manure application and ammonia emissions from pig housing: about 122 and 47 g NO_3^--eq, respectively, out of 232 g NO_3^--eq in total. The acidification impacts are also highest in the agricultural stage: ammonia emissions from pig housing contributed 53% and grain production contributed 38% to the total impact of 45 g SO_2-eq.

Based on their findings, the authors proposed to further reduce the protein consumption in pig feed by shifting from soy meal to grain. Reducing the protein content in feed will reduce nitrogen excretion and emissions from pig manure in the pig housing or when spread out on the field as manure. Moreover, as soy meal has a

Table 29.2 Contribution of the life cycle stages to the overall impacts of 1 kg of pork, at UK port

Process	Global warming potential	Eutrophication potential	Acidification potential	Photochem. ozone formation	Ozone depletion potential
Soybean meal	8	1	5		
Grain	61	53	39		
Pig housing	26	20	56		
Energy use in pig housing	4	1	1		
Manure application to field	-6	27	1		
Slaughterhouse	5	-1	1		
Transport after slaughterhouse	1	1	1		
Total impact	3.6 kg CO_2-eq	232 g NO_3^--eq	45 g SO_2-eq	1.3 g C_2H_4-eq	0.7 mg CFC11-eq

The contributions are expressed as percentages of total impact (%) (Dalgaard et al. 2007). The total percentage does not sum up to 100% because contributions between 0 and 1 or −1% are all referred to as, respectively, 1 or −1%

higher global warming impact per kg than barley, the global warming impact will decrease as well when shifting to grain.

Now, one can ask whether pork production is a good representative of meat production. Or, how would the picture look like when looking at LCAs of poultry and beef? De Vries and De Boer (2010) reviewed 16 cradle-to-farm gate LCA studies on livestock production in nonorganic farming systems in OECD-countries. They recalculated the results found in the reviewed papers to fit three functional units: kg of meat, kg of protein and kg daily intake. Here we will focus on their findings for the first functional unit. All impacts found were fully allocated to the edible part of the products. The impact potentials considered were fossil energy use, global warming, acidification, eutrophication and land use.

Among all meat products, beef showed the greatest fossil energy use. Energy use results for pork and chicken production were in the same range (see Table 29.3). For global warming, the reviewers found the highest impacts for beef, followed by pork, then chicken. Global warming, energy use and land use impacts from beef were considerably higher than impacts from pork and chicken. Ruminants (beef) emit high amounts of methane that contribute to global warming impacts. Energy consumption is also greater for beef production than for pork and chicken production. In terms of land use, beef has greater feed requirements (i.e. feed conversion ratio, kg feed per kg meat), which means that a larger extension of land is needed, both as direct land use (pasture), and as indirect land use for forage production. Moreover, cows live longer and because most cows only have one calf per year a larger breeding stock is needed. Regarding acidification and eutrophication, the variation within each type of meat was larger than the variation between the three types of meat. These variations were mainly attributed to differences in emissions of NH_3 caused by different agricultural practices and climatic conditions.

Concluding, chicken and pork meat at the farm gate show comparable impacts, while beef meat generally causes higher impacts. The internal variability of the impacts within each meat product was high, reflecting the variability of practices and environmental conditions.

Table 29.3 Comparison of minimum and maximum environmental impact potentials per kg of chicken, pork and beef (De Vries and De Boer 2010)

Livestock animal	Energy use (MJ)	Global warming (kg CO_2-eq)	Acidification (g SO_2-eq)	Eutrophication (g PO_4^{3-}-eq)	Land use (m^2)
Chicken	15–29	3.7–6.9	0.062–0.29	0.004–0.079	8.1–9.9
Pork	18–34	3.9–10	0.043–0.74	0.032–0.17	8.9–12.1
Beef	34–52	14–32	0.11–0.90	0.063–0.33	27–49

29.3.1.2 Cheese

A full LCA of a Swedish cheese was carried out by Berlin (2002). Though now relatively old, this study yielded conclusions that were confirmed by later studies.

As a functional unit "1 kg of Ängsgården semi-hard cheese wrapped in plastic" was chosen. The product system included milk production at farm level, cheese production at dairy factory, retailing, household consumption and waste management. Data from 1 farm and 1 dairy, both in Southwest Sweden were used. In order to produce cheese, apart from milk, several other ingredients were used: rennet (enzymes from calves' stomach), calcium chloride, saltpetre, salt and water. Packaging material was made of plastic and cardboard. A number of cleaning agents in the dairy was also included. Capital goods such as buildings and equipment were excluded from the study.

The impacts from the farm were allocated over the milk and meat produced. The dairy factory produced four different kinds of cheese as well as a range of other products. An economic criterion was used to allocate impact over the different co-products of the factory. The following impact potentials and flow indicators were considered: global warming potential, acidification, eutrophication, photochemical ozone formation, material and energy use. Ozone depletion and ecotoxicity were dealt with in a qualitative way. For this reason, these 2 impact categories will not be extensively discussed here. Impact assessment was stopped at the characterisation step. Hence no normalisation or weighting was done. Characterisation factors from different methods and guidelines were used to calculate impacts. The results are summarised in Table 29.4.

Almost 95% of global warming impacts were attributed to farming, with methane emissions from fermentation in the cow rumen being the most important impact source. N_2O emissions from soil processes and fertiliser production also played a key role. The remaining 5% of the impacts were attributed to cheese

Table 29.4 Results of the LCA of 1000 kg of Swedish semi-hard cheese (Berlin 2002)

Life cycle stage	Global warming (kg CO_2-eq)	Acidification (kg SO_2-eq)	Eutrophication (kg O_2-eq)	Photochemical ozone formation (kg C_2H_4-eq)
Farm inputs and agricultural stage	8300	135	2120	2.4
Other inputs	67	<1	3.3	<0.1
Processing	369	<1	10	<0.1
Retail	48	<0.1	<1	<0.1
Use	12	<0.1	<1	<0.1
Waste management	−2	<−0.1	<−1	<−0.1
Total	8794	136	2134	2.5

production and retailing. Here, CO_2 emissions from natural gas and fuel use caused the largest share of the global warming impact. Impacts from electricity use were low, because the Swedish electricity grid mix was used for modelling the product system. Since Sweden's electricity mix is largely made up of hydropower and nuclear power, GHG emissions from electricity production are low.

Of the photochemical ozone formation impacts, in which only volatile organic compounds (VOC) were taken into account, 93% of the impact was attributed to farm emissions. As was the case for global warming impacts, the cow-produced methane was the main source of VOCs.

The farming step in the life cycle of cheese was the source of more than 99% of the eutrophication and acidification impacts. These impacts were attributed to ammonia volatilisation from manure for both impact categories. Nitrate leaching from the soil was another important contributor to eutrophication.

We have seen that, in this study, the major contribution to the impact categories considered arose from the farm inputs and agricultural stage (i.e. the process of milk production). The cheese making process accounted for most of the remaining impacts. The study was published in 2002, but most of the data are from the mid to late 1990s. Despite the age of the study, the conclusion that most impacts of dairy products arise at the farm input stage and agricultural stage was also found in more recent LCAs. However, compared with newer studies, the share of these stages (93 to >99%) as found by Berlin (2002) is at the high end. For example, in their LCA study of the production of cheddar and mozzarella in the USA, Kim et al. (2013) found that the farm inputs and agricultural stages (feed production and on-farm emissions) contributed to more than 60% in seven out of nine impact categories. For cumulative energy demand, marine and freshwater eutrophication and human toxicity, other steps such as manufacturing, retail and consumption were identified as hotspots. In other studies dealing with cheese production in the Netherlands (Van Middelaar et al. 2011), New Zealand (Basset-Mens et al. 2007), Spain (González-García et al. 2013) and Serbia (Djekic et al. 2014), the agricultural stage was confirmed to be the most important environmental hotspot. In the distributive step the mode of transport more than the distance itself plays a role. In their study of New Zealand cheese exported to England, Basset-Mens et al. (2007) found that the contribution of ship transport over more than 20,000 km from New Zealand port to England was lower than the contribution of truck and consumers' car transport together in all impact categories except acidification (Basset-Mens et al. 2007). These authors also found that the average sewage treatment in England for human excretion was the second main contributor for eutrophication (31%) after farm production.

Comparing the Berlin (2002) study with newer studies raises two relevant points. Firstly, the limited number of impact potentials that were quantified shows that the field of LCA has seen a considerable methodological development in the last 15 years. Secondly, the lack of data that led to the ozone depletion potential being discussed qualitatively in the study illustrates the data availability issue that LCA practitioners traditionally have to deal with. For example, the authors of the study described the locations where cooling equipment was used, as well as the

refrigerants used, but could not quantify the resulting environmental impacts because data about refrigerant leaking was not available at that time.

29.3.1.3 Bread

One of the first studies of a whole food product was an LCA of bread production carried out by Andersson and Ohlsson (1999). The main aim of this study was to compare different scales of baking. A secondary goal was to identify environmental hotspots. This study was selected for this chapter to illustrate some typical pitfalls of LCA application, which often arise due to the complexity of the modelled systems in real life, data unavailability and lack of transparency of the modelling choices made.

In cooperation with industrial partners from the bakery industry that provided data, the authors compared the environmental impacts of 1 kg of white bread ready for consumption at home produced in four different Swedish scenarios. Two of these were industrial bakeries. The first produced bread distributed through the entire country, the second operated at a regional scale, the third a small local bakery, whilst the fourth corresponded to the baking of bread at home.

Included in the system boundaries were the farm inputs, wheat cultivation, wheat milling and baking, production of packaging material, the household stage (freezing the bread), waste handling and transport. Capital goods were excluded and so were all other ingredients of the bread other than wheat. These ingredients were excluded because little variation was observed between the recipes. Unless the impacts of these ingredients were minor, which has not been tested, excluding them makes it hard to conclude on the environmental hotspots: an unknown part of the impacts are not quantified. The wholesale and retail steps were also excluded because they were expected to contribute little to the overall impacts.

Impacts from wheat cultivation and milling were allocated to wheat and flour, respectively. Allocation of the impacts of the bakeries was done differently. For the two industrial bakeries, allocation was done on basis of the mass of the products produced there. In contrast, economic allocation was used in the local bakery scenario. This option was chosen because data for the masses produced were not available.

The impact categories and indicators included in the study were energy use, land use, global warming potential, eutrophication potential, acidification potential and photo-oxidant formation potential. No normalisation or weighting step was done. The results are presented in Table 29.5. We will not discuss the details of the results here, but summarise the main findings of the hotspot analysis. A hotspot was defined as a sub-system that contributes more than 20% to the impact in a given impact category or flow indicator. With the exception of primary energy use and photo-oxidant formation, the farm inputs and agriculture stages were hotspots in all of the impact categories for all scenarios. Eutrophication impacts were dominated by wheat cultivation. Transport was a large contributor to global warming and acidification potentials in the industrial bakery scenarios. In the bakeries using

natural gas or oil in their ovens, the baking process also was a hotspot for the global warming impacts. Food processing, i.e. ethanol released when baking bread, was the main hotspot for the photo-oxidant formation impacts, even though transport was also a hotspot in this impact category.

Summarising, we see that in contrast to what was observed in the previous chapters, the agricultural stage is not the single dominating stage in this case study.

The main aim of the authors was to compare different scales of bread production. It is, however, doubtful how useful the results were to study differences of scale, because of a number of methodological inconsistencies that a practitioner should try to avoid.

First, the system boundaries were set in a way to include the consumers' transport to buy the bread, or the raw materials to bake the bread at home, except in the local bakery scenario. The authors claim that the local bakery is visited on foot or by car on the way home. As the transport by car hence does not add extra kilometres, the impacts were not accounted for, while the impacts due to the transportation could also have been allocated to both the travel back from work and the bread production. It is not clear from the study why this would not be the case when purchasing industrially produced bread at supermarket. Lack of transparency in modelling choices makes it difficult to identify if the systems compared are equal.

Second, the allocation procedure is not done consistently: the impacts from the industrial bakery were allocated by mass, those from the local bakery economically. The authors justify the approach by mentioning that the fractions of impacts allocated to the bread were found to be similar for the industrial bakeries on one hand and the local bakery on the other hand. However, this does not validate the approach: the similarity in outcomes may be accidental.

A factor that further complicates the interpretation of the comparison relates to the differences between bakeries. One of the industrial bakeries used a natural gas oven while the local bakery had an oil-fuelled oven. In the second industrial bakery and the home baking scenarios, electric ovens were used. It is not clear from the

Table 29.5 Summary of the results of the LCA of bread production

Studied systems	Primary energy (MJ)	Global warming (kg CO_2-eq)	Acidification (mol H^+)	Eutrophication (g O_2)	Photochem. oxidant formation (g C_2H_4-eq)
Industrial bakery 1 (national)	22	940	0.15	160	5.4
Industrial bakery 2 (regional)	14	630	0.1	99	3.2
Local bakery	12	660	0.1	120	2.6
Home baking (heat from electricity)	18	620	0.078	88	2.4
Home baking (heat from oil)	17	630	0.09	89	2.6

Functional unit 1 kg of white bread ready for consumption at home. Electricity production from waste incineration is assumed to replace electricity produced from oil combustion (Andersson and Ohlsson 1999)

study whether or not the use of different fuels for the oven was related to the different scales of the bakeries. If not, this would mean that the systems compared were not equivalent regarding the goal set, i.e. compare different scales of baking. However, if the aim of the study had been to compare most common practices of baking at a given scale, the same definition of systems would have been fair to answer the question aimed. Summarising, when conducting an LCA it is important to act consistently and to design a model of a product system that suits the aim of the study.

29.3.1.4   Tomato

As an example of an LCA of a vegetable product, we will discuss here the study on tomatoes produced in a greenhouse in Spain, carried out by Torrellas et al. (2012a). The study was carried out to investigate how the environmental performance of greenhouse tomatoes could be improved.

The functional unit of the study was 1 tonne of loose tomatoes at the farm gate, indicating that this study is a cradle-to-farm gate study. Included in the system boundaries were the manufacturing of greenhouse components, auxiliary equipment such as the irrigation and water collection systems and the substrates in which tomato plants are grown, products needed for greenhouse management such as fertilisers, pesticides, water and electricity, waste handling including transport to waste management from the site of the greenhouse.

The environmental impacts were calculated for six categories and flow indicators: cumulative energy demand, abiotic depletion, acidification, eutrophication, global warming and photochemical oxidation. If we use the same definition of a hotspot here as in the last paragraph, i.e. a process or group of processes to which more than 20% of an impact can be attributed, then the results (see Fig. 29.2) showed that the structure of the greenhouse was a hotspot in all of the categories. The steel in the frame and the plastic of the cover were the main contributors. Auxiliary equipment represented a hotspot in all impact categories apart from eutrophication. For acidification, eutrophication and global warming, fertiliser production and application constituted a hotspot as well. A large part of the environmental impacts was associated with the greenhouse itself: for the impact categories abiotic depletion, acidification, photochemical oxidation and cumulative energy demand the sum of greenhouse structure and auxiliary equipment accounted for more than 75% of the total impacts. For eutrophication and global warming the percentages were 46 and 66%, respectively. Therefore, the impacts of the greenhouse were relatively high. Because the studied greenhouse was located in Spain, no heating was needed. Another study that assessed different geographical greenhouse tomato scenarios showed that the most important contributor was the greenhouse heating, highlighting the need to reduce energy consumption and use renewable energy sources where greenhouse heating is necessitated (Torrellas et al. 2012b).

Since the study by Torrellas et al. (2012a) was a cradle-to-farm gate LCA, the role of transport in the life cycle of vegetables and fruit is not highlighted. Other work, for example the aforementioned study by Sim et al. (2007) on French beans flown from Kenya to England, showed that transportation may be determining for the environmental impacts of vegetables and fruit. The same was concluded by Mithraratne et al. (2010), who analysed the carbon footprint of a tray of kiwifruit produced in New Zealand and consumed in Europe. Here, shipping contributed to 44% of the footprint.

29.3.1.5   A Full Diet

In the previous chapters various food items were discussed in isolation, with functional units of 1 kg (or 1 tonne) of the food item. Although this gives a good overview of the impacts of single food products, it provides little information about where environmental impacts arise in a diet. After all, humans usually do not consume identical amounts of different foods. Therefore, we finish this chapter by reviewing a study by Muñoz et al. (2010), who did a full LCA of an average Spanish diet. The functional unit of their study was given as 'the supply of food for a Spanish citizen in the year 2005'. The authors included all processes needed to provide food to the consumer in the study: agriculture stage, processing of food, distribution, retailing, storage at home and preparation. Furthermore, the end of life of food was considered via food waste management and wastewater treatment of human excretion.

Impact assessment was limited to global warming, acidification, eutrophication and primary energy use. The results, presented in Fig. 29.3, showed that food production, including agricultural and processing stages, represented the largest source of impacts in all categories. Meat and dairy alone made up 54% of the global
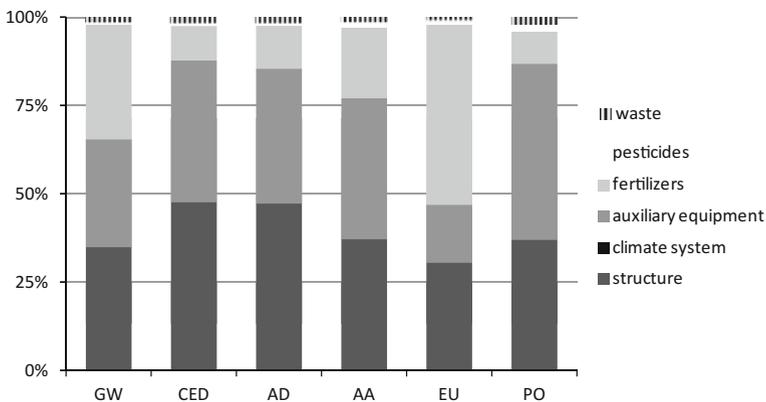
Fig. 29.2 Results of the LCA of the production of 1 tonne cold greenhouse tomatoes. Adapted from Torrellas et al. (2012a)

warming potential. Wastewater treatment contributed 17% of the impact potential when including emission of biogenic CO_2 while it was only 3% when excluding these emissions. These CO_2 emissions can be excluded from the impact assessment, as they are formed from carbon that was taken up by the plant during its growth.

Meat, dairy and beverages represented 60% of the eutrophication potential. Here, wastewater treatment was the second most important stage, contributing 17% to the total impact. Home storage and cooking, which was considered as one process, was the second contributor to the acidification potential, with a contribution of 12%. This process was also the second largest user of primary energy (22%).

With regard to the percentages above, the authors mentioned that there was a fair level of uncertainty involved in the study. As an example, for some food products data were missing. For other products the data used were collected for Danish rather than Spanish circumstances, following a consequential approach, instead of the attributional approach used by the authors.

From the results it is clear that for a full diet, the agricultural stage and the production its inputs are the stages contributing most to the impacts. Among the different products, meat production was associated with the largest impacts in all of the four impact categories studied. Dairy and beverages were other considerable contributors to environmental impacts.

The authors of this study did not discuss options to reduce the environmental impacts of diets. Other authors recommend exploring a reduction of food waste and
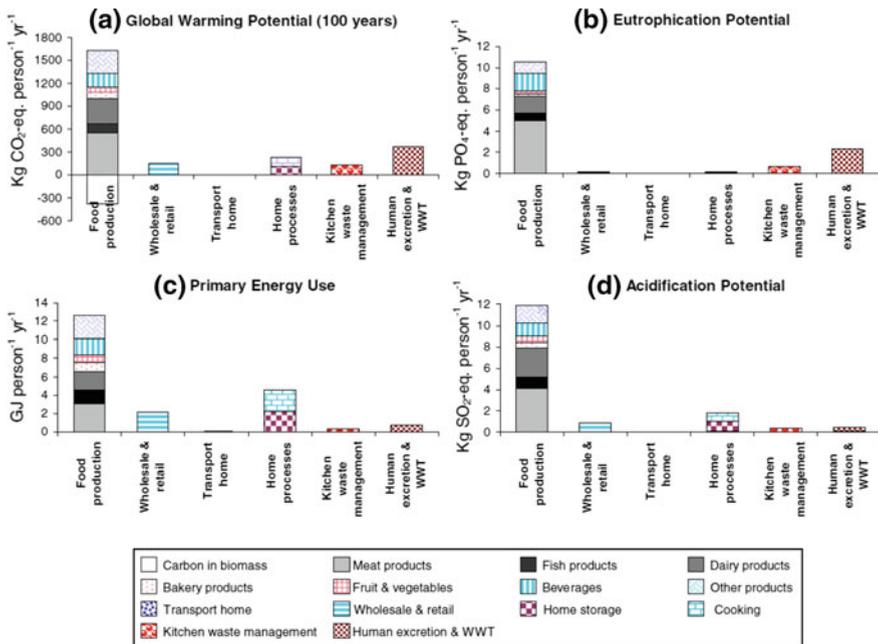

Fig. 29.3 results for the LCA study of the average Spanish diet. Taken from Muñoz et al. (2010)

switching to a (partly) vegetarian diet. In the study of the Spanish diet, the authors found that 23% of food purchased was discarded. This waste consisted of a smaller fraction of inedible parts of the food products, but the largest waste fraction were edible parts not consumed for one reason or another.

Regarding switching to a diet containing less meat, Saxe and Jensen (2014) compared the environmental consequences of switching from an average Danish diet (ADD) to a New Nordic Diet (NND). In the NND, focus is on local ingredients, produced organically. In addition, the diet contains less meat, but more fish, wholegrain products, nuts, fruits, berries and vegetables than the average Danish diet. The NND was shown to reduce environmental impacts, but at the expense of increased cost for the consumer.

29.3.2 Case Study: Conventional, Integrated and Organic Farming

Most agricultural practices in Western countries are chemically intensive, aiming at maximising the production by using external inputs. This production mode is associated with environmental problems mostly due to nutrients and pesticides emissions to environmental compartments and decreased near farms.

In order to reduce the environmental burdens of agriculture, new production methods have been introduced. Organic agriculture aims at producing while sustaining the health of soils and people and preserving biodiversity. In practice, this form of agriculture differs from conventional agriculture in the sense that it avoids the use of synthetic-agrochemical pesticides and mainly uses manure and compost as fertiliser. In organic milk farming, as discussed below, cows spend most of their time outside in order to stimulate their natural behaviour. Feed should consist of 60% of roughage, produced organically, preferentially on-farm. Another alternative is integrated farming. Based on the principles of integrated pest management, it aims to achieve optimal long-term results from both an environmental and economic point of view. Furthermore, pesticide application has to be targeted and limited, the soil has to be protected in winter and the crop rotation needs to be diversified.

In this chapter, two comparative cases are discussed: organic and integrated farming in Switzerland, and organic and conventional milk production in the Netherlands.

29.3.2.1 Crop Production

Nemecek et al. (2011) compared the environmental impacts of conventional/integrated and organic farming in Switzerland. Environmental impacts were reported as yearly averages calculated from 7-year crop rotations. The authors

identified three functions of agriculture, and defined functional units accordingly: land management, providing income to the farmer and production of food. The focus in this chapter will be on the first and third functions. In the land management function, the impacts are expressed per hectare per year. In the food production function, they are reported per kg dry matter or per MJ net energy content, depending on the food or feed product.

The environmental impacts were assessed using the Swiss Agriculture Life Cycle Assessment (SALCA) framework. In this framework, the usual impact categories and flow indicators energy resources, global warming potential, ozone formation, eutrophication, acidification, terrestrial and aquatic ecotoxicity and human toxicity are considered along with biodiversity and soil quality.

The results found for normal fertilisation levels are shown in Fig. 29.4.

The results for biodiversity, not given in Fig. 29.4, showed that biodiversity was higher in an organic farming system, mainly as a consequence of banning synthetic pesticides. The soil quality indicators did not vary much between the three systems.

The results presented in Fig. 29.4 show that, when looking at the land management function, the potential impacts of organic farming were considerably less than those caused by conventional/integrated farming. However, the land required to produce the same amount of product was 25–30% higher for organic farming. For that reason, the differences between organic and conventional/integrated farming were smaller when looking at the results from the production function. Still, the authors showed a significantly lower impact for organic farming for most of the impact categories when looking at the impacts per kg dry matter, especially for toxicity impacts. The exceptions here were land use, which was higher for organic farming, and ozone formation, acidification and eutrophication, where the differences were not statistically significant. The large difference in toxicity-related impact categories was explained by the reduced use of pesticides in the organic farming systems. Not taken into account in the method applied was the use of copper as a fungicide in organic farming. Therefore, the toxicity impacts of organic farming have been underestimated. The reduction in energy demand and global warming potential in the organic farming systems was due to reduced use of mineral fertiliser in this type of farming.
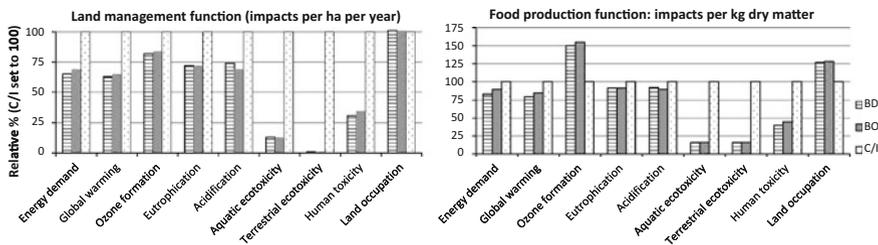
Fig. 29.4 Relative environmental impacts for crop rotations of three types of farming in Switzerland. Impacts for conventional/integrated farming set to 100% (reference system). *Abbreviations BD* bio-dynamic, *BO* bio-organic, *C/I* conventional/integrated

In addition to calculating the impacts associated with crop rotations, the authors also looked at the impacts of individual crops in organic and integrated crop production. Here it was found that some organic products had higher environmental impacts in some categories than their conventional/integrated counterparts. This was partially explained by lower yields in organic farming. Based on this the authors stressed the importance of looking at product systems instead of looking at products in isolation.

Summarising, depending on the selected function, the outcome is either that organic farming is the environmentally favourable option, or that the results are not conclusive to decide on the most environmentally friendly option for agriculture.

29.3.2.2 Milk Production

Based on Dutch data from 2003, Thomassen et al. (2008) performed a comparative LCA of conventional and organic milk production. This cradle-to-farm gate study used 1 kg of fat and protein corrected milk leaving the farm gate as the functional unit.

The studied system included the on-farm processes and the inputs required on the farm: breeding of animals, production of feed concentrates, roughage and bedding material, transport of animal manure used as fertiliser. In addition, the conventional milk production also required pesticides and artificial fertilisers for feed and roughage production.

The studied systems had a number of multifunctional processes, most notably the cow which produced not only milk, but also meat, calves and hides. In order to deal with these co-products, economic allocation was applied.

The study considered five impact categories and flow indicators: land use, energy use, climate change, acidification and eutrophication. Stratospheric ozone depletion was excluded because previous studies had shown that milk production does not produce significant ozone depletion impacts. Furthermore, human toxicity, terrestrial and aquatic ecotoxicity were excluded. The reason for this was the absence of data on pesticides and heavy metals used. The impacts obtained are summarised in Table 29.6.

The difference in land use observed in Table 29.6 was mainly explained by the lower yields for feed production and a lower intensity of animals per hectare in the organic system. The difference in energy use is related to the absence of fertiliser and pesticide production in the organic system and the smaller use of concentrate in the feed mix. Likewise, the production and use of concentrates and artificial fertiliser explained the higher eutrophication impacts in the conventional farming system. The small difference in acidification impacts may seem counterintuitive: the conventional system emitted more ammonia from manure storage and application, and fertiliser application per hectare. However, the animal intensity was much higher in the conventional system, leading to lower overall impacts when looking at mass of milk produced. The climate change impacts were similar for both systems. The organic system had higher on-farm impacts because of the higher number of

Table 29.6 Comparison of environmental impacts of conventional and organic milk production per kg of fat and protein corrected milk leaving the farm gate (Thomassen et al. 2008)

Impact category	Conventional system	Organic system
Land use (m^2)	1.3	1.8
Energy use (MJ)	5.0	3.1
Eutrophication (kg NO_3^--eq)	0.11	0.07
Acidification (g SO_2-eq)	9.5	10.8
Climate change (kg CO_2-eq)	1.4	1.5

cows needed to produce the same volume of milk. In contrast, the off-farm emissions were higher in the conventional system, mainly due to the purchase of feed concentrates. Concentrates are feed products that contain a high density of digestible nutrients and are usually low in fibre content (FAO 1995).

This case shows the importance of the perspective of the study, as reflected in the functional unit, on the outcome of the study. For example, taking the perspective of a farmer with a fixed land area at his disposal who wants to produce milk in a more sustainable way, switching from conventional to organic farming would probably result in lower environmental impacts. In this case, the functional unit will be area-related (e.g. hectare per year). However, the main motivation of a farmer usually is to maximise income for the agricultural production. Looking at how to minimise environmental impacts per currency unit might be more realistic from the farmer viewpoint. In that case, the functional unit will be income-related (e.g. €). As can be seen from Table 29.6, using the perspective of producing food, the functional unit chosen was kg of milk. The picture of which option is most favourable to the environment is not clear.

In both the crop and the milk examples, results showed that neither conventional farming nor organic farming was a more environmentally favourable practice in all respects. LCA was powerful at identifying environmental hotspots and margins of improvement of the studied systems. Agriculture is multifunctional and LCA outcomes depend a lot on the agricultural function studied, especially when comparing systems with contrasted intensification levels. This conclusion is also found in studies conducted by other authors, such as Cederberg and Mattsson (2000), De Backer et al. (2009) and the review by Foster et al. (2006). In addition, based on a review of 34 agricultural LCAs, Meier et al. (2015) argue that many LCA studies do not sufficiently capture the differences between two production systems, for a number of reasons. The goal and scope definition does not differentiate the characteristics of the farming systems. The inventory data used for N and C flows from the field are based on models, often developed for modelling conventional farming, and do not represent actual circumstances. Finally, LCA studies apply LCIA indicators of all readily available impact categories. Other important categories for agriculture, such as land use impacts on biodiversity and soil quality, water use and (terrestrial) toxicity are currently the object of fast scientific development and will be normal practice in a few years.

29.3.3 Case Study: 'Local' Food

Consumers are increasingly aware of environmental problems connected to food production. Many food products are transported over long distances: air-freighted French beans from Kenya, apples from Chile or peas from Egypt can be found in European and North American supermarkets. The "food miles" concept (defined in 1994 by Paxton as the distance food travels from producer to consumer) has become popular in the UK and USA and has led to increased interest in local food, which has been produced in close proximity of where it is consumed. Apart from questionable environmental benefits, local food is also associated with other values: taste, naturalness, local economy, to mention a few examples given by Edwards-Jones et al. (2008). These authors have analysed the interest in local food and food miles from an environmental and ethical point of view. Here we will focus on the environmental aspects that have been analysed using LCA.

Transporting food products over large distances may appear as a waste of resources and an unnecessary cause of GHG emissions. Especially air transport causes high emissions per tonne kilometre. In order to have a measure of the distance over which food is transported the concept of 'food miles' is used. However, when looking from an LCA perspective, transport is only one of the stages in the life cycle of a food product where GHG and acidifying emissions occur. Apart from the distance, the transportation mode is of the utmost importance for the environmental impacts of the transportation phase. Moreover, for many field-grown crops the production of fertiliser has a large global warming impact potential, whilst in crops grown in greenhouses the use of electricity for heating and lighting can cause considerable impacts. Therefore, only looking at the transported distance is not sufficient to conclude on the environmental benefit of local food.

To underline this, the authors give an example of an assessment of global warming potential for apples consumed in the UK, taking a life cycle perspective. Researchers from the UK had found that local production results in the lowest GHG emissions, whilst researchers from New Zealand found the opposite: apple production in New Zealand followed by transport to the UK results in the lowest global warming impacts. Therefore, there are two contrasting conclusions. Which one is right? The answer is, surprisingly, both. The two studies used different system boundaries. When looking at a full calendar year, and including the cold storage that is required to store the apples between harvest and consumption, it can be shown that apples from the UK are favourable in most parts of the year. However, when these apples have to be stored for a long time, importing freshly harvested apples from New Zealand is environmentally favourable. Similarly, it was found that, in terms of energy, it is more efficient to import off-season tomatoes from Spain to the UK, rather than growing the tomatoes locally in heated greenhouses (Smith et al. 2005). Unfortunately, in terms of water use impacts, the imported tomatoes have much greater impacts than their local counterpart (Payen et al. 2015).

In addition, in the study about Danish pork meat exported to the UK, which was described in Sect. 29.3.1 the 'food miles' concept was criticised. The authors

pointed out that the different transport steps contribute less than 1% to the overall global warming impacts. Hence looking at reducing these impacts, transport is not the place where significant reductions can be obtained. The authors of the pork study call the concept of food miles misleading.

Food-miles can be concluded to be a simple social representation of a complex system that help people engage but it is not reliable as an indicator of the environmental impacts of a product system.

29.4 Methodological Issues

In the previous sections, a number of Life Cycle Assessments of food and food products has been discussed. These studies showed that in the full life cycle of a food product, it is often the farm inputs and agricultural stages where most environmental impacts arise. Within these agricultural stages, a few trends can be observed. Firstly, global warming impacts can be attributed to animal husbandry or fertiliser production. Secondly, acidification and eutrophication impacts are associated with the production and use of fertiliser or animal manure. Finally, toxic impacts are related to pesticide use and are still seldom considered.

The two papers described in the section about organic, integrated and conventional farming not only gave an indication of the differences in environmental impacts between these farming systems, they also illustrated the importance of considering land use in LCAs. The case studies on Swiss farming practices and local foods stressed the need to consider systems as a whole. Studying crops in isolation or only at a certain moment in time might lead to deceptive conclusions.

Besides those, a number of other methodological issues relating to the environmental assessment of food and food production remain.

First of all, the choice of the functional unit of a study has to reflect a product's function. As discussed in Sect. 29.3.2, some authors ascribe three functions to agriculture: land management, providing an income for farmers and production of food. The choice of the functional unit will depend on the goal and scope of the LCA study (see Chaps. 8 and 9). It will often be relevant to express the results by different functional units within the same study to give a fair picture of the compared systems, especially in the case of highly contrasted intensification levels. From a consumer's point of view, the primary function of food products is to provide sufficient energy and nutrients to the human body. Beyond the mass of food produced, research is therefore looking for more qualitative functional units, taking into account the food's nutritional value in a harmonised way. As an example, increasing yields of wheat may negatively affect the nutritional quality of the grains. It is therefore recommendable to clearly define the nutritional quality of a food product in the functional unit, especially in comparative LCA studies (Schau and Fet 2008).

A second issue is the definition of the system boundaries (see Chap. 9). This discussion has two important aspects. The first aspect is setting the border between

technosphere (the product system) and ecosphere (the natural environment). This border is essential in LCA, as only material and energy flows crossing it are considered inputs or outputs. Especially in LCAs of agricultural and food products this boundary might be hard to identify clearly, because in food production the technosphere is closely linked to the ecosphere. An example can be found in the emissions of pesticides to agricultural soil. It can be argued that these should be marked as emissions to the ecosphere because the pesticides might affect various forms of life in the soil: worms, beetles, which are not necessarily the target organisms for the pesticide. On the other hand, one can reason that the soil of a field is part of the technosphere, because it is manipulated by humans to an extent where it is incomparable to natural soils. Setting the system boundaries is dependent on the goal and scope of the LCA study: it is not possible to objectively define one correct boundary setting that works for all agricultural LCAs (Dijkman 2014; Rosenbaum et al. 2015). For this reason, it is important to explicitly define the system boundaries in the goal and scope definition. Ideally, the boundaries between technosphere and ecosphere, and thus of LCI and LCIA for modelling the inflows and outflows, should be defined uniformly in order to produce a consistent LCA study. The second aspect of setting the system boundaries relates to the processes that are included in the study. Often, a cradle-to-farm gate study is done because it is assumed that most impacts arise at the agricultural stage or because the post-farm gate processes are identical. This might result in overlooking product losses during processing or consumption, while the reduction of food waste can contribute to lowering environmental impacts elsewhere in the life cycle.

A third issue in the LCA of food products is the inclusion of the diversity of production systems. Most LCA studies in the past have relied on a very small number of farms, while agricultural production systems are generally very diverse due to the interaction between the farmers' skills and practices and their environment. In studies where the variability of systems has been explored, the variability of LCA results is larger within one production group than between the studied alternatives. This also leads to the question of uncertainty of LCA results, which is generally not evaluated (see Chap. 11).

As a fourth issue, the modelling of flows from the agricultural field can be improved. Meier et al. (2015) argued that the modelling of nutrients needs to be improved in LCA practice, because especially nitrogen flows are responsible for many environmental impacts from on-field processes. Many studies reviewed by Meier et al. (2015) did not calculate the N balance from the field. When the N balance was calculated, differences in the N surplus (defined by Meier et al. (2015) as the nitrogen potentially emitted to the environment via different pathways) between conventional and organic farming systems were not always reflected in differences in the eutrophication potentials calculated for both farming systems. However, because the N surplus is a measure of the amount of N that is available for leaching to surface water or other environmental compartments, differences in the eutrophication potential should be related to differences in the N surplus. Moreover, Meier et al. (2015) found that the N emissions exceeded the N surplus in four processes representing Swiss agriculture in the EcoInvent database.

In addition, N emissions from manure, which are dependent on the excretion of N by livestock animals, are seldom adjusted for differences in the animals' dietary composition. Because the N balance of a field is dependent on many factors, such as the chemical or organic fertilisers used, the uptake into the crop and local soil and climate conditions, a simple model of the N balance of a field for the use in LCA is not available. The authors also stated the need for better modelling of flows of carbon from the field and manure. Apart from nutrient flows, modelling of pesticide flows can be improved. Databases with inventory data, such as ecoinvent (Ecoinvent Centre 2007) and the US LCI database (NREL 2003), often assume fixed emissions to one or two environmental compartments, independent of the pesticide applied, the application technology used, the climatic circumstances and the crop or soil onto which the pesticide is applied. Here, models such as PestLCI 2.0 (Dijkman et al. 2012) and the forthcoming results of ongoing pesticide consensus work can be used to better represent the influence of chemical properties and local circumstances on pesticide emissions (Rosenbaum et al. 2015).

A final issue in agricultural LCAs is the limited number of impact categories and flow indicators that are usually included: global warming, acidification, eutrophication, ozone depletion and energy use. Land use-related impacts are increasingly considered. This is an important impact category, because land is a scarce resource. Most land that is suitable for agriculture, is currently already in use as such. Moreover, considering land use may also help to illustrate the trade-offs of, e.g. organic farming: more land is required per unit of product, resulting in an expanded use of land for that product when switching to organic farming (under the assumption that the demand for the product remains unchanged). This direct land use change in turn results in indirect land use change (iLUC) because the organic product displaces another product, which ultimately results in the conversion of grasslands or forest into agricultural land. At the same time, chemical pesticide use is avoided, and the nutritional quality of grains may be at optimum when grown below maximum yield levels. So, even though all farmers switching to organic farming would considerably reduce certain environmental impacts of agriculture such as toxicity impacts, the amount of food available may be reduced as well on the short term.

Another impact category that is usually omitted in LCAs is toxicity (Meier et al. 2015). Because of neglecting toxicity, the effects of pesticide use are not well reflected in LCA results. Historically, toxicity was excluded because of the unavailability of emission data and impact assessment methods for these categories. On the LCI side, models such as PestLCI 2.0 can now be applied to calculate pesticide emissions to air, surface water and groundwater (Dijkman et al. 2012). Likewise, in impact assessment, models such as USEtox (Rosenbaum et al. 2008) can be used to calculate toxicity impacts. However, LCI and LCIA models do not necessarily apply the same boundaries between ecosphere and technosphere, both in terms of time and space. In order to overcome this inconsistency, and to provide guidance to LCA practitioners about modelling pesticide emissions to the environment and their impacts in LCA, a series of international workshops has been

held in an effort to establish a global consensus. Rosenbaum et al. (2015) report the objectives of the effort and the recommendations of the first workshop.

Finally, water use-related impacts are frequently omitted in LCA studies. Given the importance of water resources for economic activities in general and for food production in particular as well as for human and ecosystems health, this impact category has rapidly evolved in the last few years. Nowadays, operational methods based on regional water stress are available (e.g. Berger et al. 2014, Pfister and Bayer 2014) and the new AWARE consensus impact assessment method (WULCA 2016) to asses water deprivation impacts (see Sect. 10.15). Despite these developments, much work to improve environmental relevance of methods addressing the consequences of water use on the environment is ahead. Water use is also associated with other long lasting problems such as salinisation (Payen et al. 2016) and desertification (Núñez et al. 2010), included in a number of LCA pilot studies.

Therefore, even though quite some steps have been made in the field of LCA of food products during the last 25 years, a number of challenges and methodological issues remain to be improved.

29.5   Applying LCA to Food Products in Southern Countries

In a context of demographic increase, especially in cities, southern and generally poor countries are facing immense challenges in terms of food security, poverty reduction, food safety and environmental protection. In such contexts, global assessment tools such as LCA can help stakeholders in food supply chains focus, improve and develop the most promising technical alternatives and support the eco-design of livestock and cropping systems. However, the application of LCA to food products in southern countries is recent and associated to difficult and numerous challenges. All the previously mentioned methodological challenges are relevant, but in an even more critical way due to the extreme diversity of production systems (often including perennial crops) and practice, the data scarcity on production systems, the lack of knowledge and appropriate models for estimating the field fluxes and finally the predominance of environmental impacts for which no consensual methodology exists. This concerns all environmental impacts associated to the use of land and water, including aspects of biodiversity, water deprivation, soil quality and fertility, carbon balance and GHGs from soils, salinisation impacts, etc. This also concerns all toxicity impact categories which are complex and for which available inventory approaches are either not valid or difficult to implement due to data scarcity. Particularly important in these situations are also the social impacts associated to food supply chains for which ambitious research programmes are starting. Finally, caring for the environment in such subsistence economies where the priority is to feed the people, often appears as a luxury for wealthy people. Therefore, awareness-raising and education about eco-friendly practices,

human health risks and environmental sustainability are important priorities in association to more methodological studies to implement LCA in southern countries. Some research teams have started to implement LCA to food products in southern countries with examples on peri-urban tomato in the South of Benin (Perrin 2013), palm oil in Indonesia (Bessou et al. 2016), clementine in Morocco and mango in Brazil (Basset-Mens et al. 2016).

29.6 Conclusions

We have seen that LCAs of food products can be divided into six stages: inputs production and transportation, agricultural stage, processing, distribution, use and waste management. A large number of LCA studies are cradle-to-farm gate studies, and include only the two first stages. As a consequence of production and flows of nutrients and pesticides from the field, as well as from livestock and manure handling, the agricultural stage is often found to be the major contributor to many impact categories. This was illustrated in a number of case studies. Different impacts can arise in the food production stage. Environmental impacts from the use stage are often related to energy consumed in food storage and preparation. In the waste management stage, impacts arise due to food waste handling and treatment of human excretion. Throughout the life cycle of a food product, food waste is a major problem.

The case study about conventional and organic farming showed that, depending on the choice of functional unit and the impact categories included in the assessment, LCA can be used to conclude in favour of both conventional and organic farming practices. The case study about local food showed that local food is not by definition more sustainable. Transport is not often the decisive factor when it comes to environmental impacts of food products, so the circumstances during production weigh more heavily in determining where and how locally produced food is more sustainable. Overall, the key strength of LCA lies in the identification of the hotspots and margins of improvement of each system.

Despite methodological improvements, a number of challenges remain for agricultural LCA. Firstly, the functional unit is currently often defined on basis of mass produced, or production per unit of area, without considering the nutritional quality of the product. Secondly, setting the system boundary between technosphere and the ecosphere is difficult in those production systems showing a direct interface with nature. In the life cycle inventory, modelling of flows of nutrients, water, salt and pesticides can be improved in many studies. A challenge in the LCIA phase remains the limited number of impact categories included in most LCA studies. Moreover, some impact categories relevant for agriculture, such as land use-related impacts including soil quality aspects and biodiversity damage, remain to be further developed and operationalised.

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