Chapter 4
Life Cycle Assessment in the Cereal and Derived Products Sector

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Abstract This chapter discusses the application of life cycle assessment methodologies to rice, wheat, corn and some of their derived products. Cereal product systems are vital for the production of commodities of worldwide importance that entail particular environmental hot spots originating from their widespread use and from their particular nature. It is thus important for tools such as life cycle assessment (LCA) to be tailored to such cereal systems in order to be used as a means of identifying the negative environmental effects of cereal products and highlighting possible pathways to overall environmental improvement in such systems. Following a brief introduction to the cereal sector and supply chain, this chapter reviews some of the current cereal-based life cycle thinking literature, with a particular emphasis on LCA. Next, an analysis of the LCA methodological issues emerging from the literature review is carried out. The following section of the chapter discusses some practices and approaches that should be considered when performing cereal-based LCAs in order to achieve the best possible results. Conclusions are drawn in the final part of the chapter and some indications are given of the main hot spots in the cereal supply chain.

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4.1 Introduction

Cereal grains, the fruit of plants belonging to the grass family (Gramineae), represent the most important group of food crops produced throughout the world.

The agricultural revolution some 10,000 years ago made grains the major food raw material for humans (Diamond 2002). The global importance of cereal crops to the human diet and moreover their role in the recorded history of mankind and in agriculture cannot be overstated. Cereal crops are energy dense, containing 10,000–15,000 kJ/kg, about 10–20 times more energy than most succulent fruits and vegetables. Nutritionally, they are important sources of dietary protein, carbohydrates, the B complex of vitamins, vitamin E, iron, trace minerals and fibres (Cordain 1999).

Cereals occupy an important role as global commodity products, being bought almost immediately after harvest and sold on bulk markets. The growing and exporting of grains are vital for many countries of the world and account for significant contributions to their agricultural outputs.

The types of grain cultivated around the world depend on an array of environmental, cultural and economic factors, and the most critical environmental factors are temperature and water availability, which determine the crops grown in a given region. For instance, in regions where there is water availability, rice, and to some extent corn, tend to dominate. Neither corn nor rice can withstand frost and they

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must be grown in warm environments. Wheat, on the other hand, is grown in a wider variety of environments with a broad range of water availability and ambient temperatures; hence, it is widely produced in the temperate regions both in the winter and in the spring (Kirk-Othmer 1984). Consequently, climate change-related hazards and overall global warming as well as land use, water management, fertiliser and pesticide use, and food waste and losses (wastage) are critical factors affecting the productivity of cereal product systems.

The main challenge for such systems is not only to fulfil the need for more productive agricultural and food systems but also to make them more sustainable: in other words, producing 'more with less'. This task is made even more daunting by the combined effects of land degradation, over-extraction of groundwater, climate change, energy scarcity, the increase in the world population and the overall risk of species extinction. Specifically, cereal crops play a crucial role in agriculture intensification, characterized by increasing harvests, growing use of water resources and synthetic fertiliser and pesticide use beyond sustainable levels, all of which erode the sustainability of the platform upon which food production is based.

Cereal grain availability, environmental impacts and social issues are directly related not only to agronomic practices and water consumption in the cultivation phase, but also to the respective entire supply chain (Sect. 4.1.1). In this context, supply chain environmental analysis, with systemic use of life cycle assessment (LCA), is the central element in evaluating its 'goodness' and in proposing alternative configurations (Venkat 2007). Besides, a sustainable supply chain implies the management of material, information and capital flows aiming to achieve simultaneous balancing of economic, environmental and social goals, through cooperation between the actors involved and meeting customer needs (Seuring and Müller 2008).

Because of their central role in the world's agricultural production and in the human diet, both on the Italian and on the international level, rice, wheat, corn and some of their derived products will be the chief focus of this chapter, with the aim of analysing the approaches (Sect. 4.2) and methodological issues (Sect. 4.3) that need to be considered to complete a useful LCA of the cereal sector. Furthermore, since qualitative and quantitative methods and tools that are able to address environmental performance metrics will be fundamental in supporting policy makers, management strategies and operative decisions for the development of cereal food systems, in both industrialized and developing country contexts, some lessons learned for the optimised application of LCA to the cereal sector will also be highlighted (Sect. 4.4).

4.1.1 Introductory Scenario of the Cereal Supply Chain

The number of plant species nourishing humanity is extraordinarily limited. In fact, fewer than about 20 plant species provide 90 % of mankind's food supply (Cordain 1999), of which some cereals, such as rice, wheat and corn, represent a significant percentage in terms of both value and volume (Table 4.1).

Table 4.1 Top ten world crop productions (2011). (Source: FAO 2011a)

Commodity	Production	
	Int $ 1000	Gt
Rice, paddy	186,667,648	722,559.6
Wheat	84,281,536	701,395.3
Soybeans	65,903,601	262,037.6
Tomatoes	58,223,483	159,347.0
Sugar cane	56,903,836	1,800,377.6
Corn	55,478,433	885,289.9
Potatoes	49,681,577	373,158.3
Vegetables, fresh	45,936,531	268,833.8
Grapes	39,494,901	69,093.3
Apples	31,706,244	75,484.7
Total	*674,277,790*	*5,317,577.1*

Some cereals have been primary sources of nourishment for humans for thousands of years and today roughly half of the world's cropland is devoted to growing cereals. If we combine their direct intake (e.g. as cooked rice, bread, etc.) with their indirect consumption, in the form of non-vegetable foods like meat and milk (about 40 % of all grain is currently fed to livestock), cereals account for approximately two-thirds of all human calorie intake (Dyson 1999).

Developing countries depend more on cereal grains for their nutritional needs than the developed world (from 60 to 80 % of calories are derived directly from cereals in developing countries and approximately 30 % of calories in the developed world) (Awika 2011). In particular, in Europe, the average annual consumption of cereal grains is 131 kg per capita, wheat making up the majority (108 kg/capita/year), whereas in Asia, about half of the annual cereal consumption is rice. Wheat and rice are the most important cereals globally with respect to human nutrition, whereas corn is important especially in Central and South America and sorghum and millet are important in Africa (FAO 2011b). The world's population is predicted to exceed 9 billion people by 2050 and recent FAO estimates indicate that to meet the projected demand, global agricultural production will have to increase by 60 % from its 2005 to 2007 levels (FAO et al. 2013). This equates roughly to the additional production of around 1000 Mt of cereals and around 200 Mt of meat and fish per year by 2050 (FAO 2011c). These production gains are largely expected to come from increases in the productivity of crops, livestock and fisheries. However, unlike the 1960s' and 1970s' green revolution, our ability to reach these targets may be limited in the future by a scarcity of raw materials and energy resources.

The food supply chain varies greatly in relation to its location, the productive capacity of the producers and obviously the goods themselves (e.g. fresh or processed foods, etc.). In the business context, it may be considered as a complex network of chain actors, with many interdependencies and steps, encompassing different flows of materials, services and information. In Fig. 4.1, a simplified cereal supply chain is schematised. Moreover, the way in which cereals are handled, processed and transported throughout the entire chain influences not only the characteristics and prices of the products, but also several other issues.

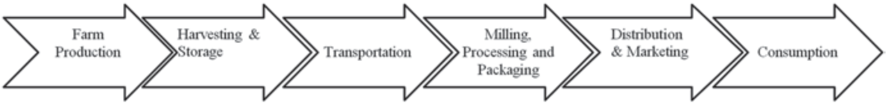

Fig. 4.1 The supply chain scheme for cereals and cereal-based products

Fig. 4.2 The main elements of sustainable cereals and cereal-based product supply chains

These issues are mainly related to the environmental and social dimensions, according to the triple bottom line approach (Elkington 1997), whereby performance is to be achieved in the economic, environmental and social dimensions (Fig. 4.2). The inclusion of environmental and social aspects in the analysis of cereal supply chains stems from two principal considerations. The first is related to market globalisation and to the growing role of multinational organisations, which has led to the lengthening of supply chains. In fact, the value provided to the customers derives from the complex aggregate of all the 'value added' along the entire supply chain. Secondly, the increasing pressure from developed countries has made necessary the close observance and monitoring of the sustainability approach of all the links in the supply chain (Editorial of Journal of Cleaner Production 2008). This is because consumers have become increasingly aware not only of the end-product consequences, but also of supply chain sustainability.

The need for better environmental performance will increase in forthcoming years, in terms of both the rising concern with national and international regulations and the ever-growing attention of end consumers to sustainability issues. Almost all products reach consumers through supply chain management and in the food sector each link of the supply chain affects the availability, affordability, diversity and overall nutritional quality of foods as well as their safety.

The result of the continuing sequence of food scandals and incidents, in almost every area of the world, has determined that food safety is currently considered the most important issue for all stakeholders. In fact, consumers' perceptions show a consolidated interest in the properties of the food they consume. The increasing need for transparent information has involved the entire cereal supply chain and is supported by several tools, most of which are based on the concept of traceability. In fact, with the globalisation of markets, consumers have become increasingly concerned about the origins of their food, the way in which agricultural land is used, working conditions and human rights, and whether the production, transportation and storage methods can guarantee product safety and environmental and social sustainability. This is reflected in the demand for improved traceability from 'farm to table'. Food traceability can be defined as the 'history' of a food crop and its subsequent transformations on its journey during its life cycle. Nowadays, traceability is becoming mandatory in many countries (the European Union, the United States and Japan), starting with some food products (Kraisintu and Zhang 2011). Legislation, protocols and quality assurance schemes perform different functions, but have in common that they all require compliance information to be recorded. The ability to collect this information and to use it to ensure product quality in 'real time' provides tangible benefits to the cereal supply chain. The latter can be especially complex since many different processing steps are taken in order to turn the coarse grain into a large spectrum of value-added products, ranging from meals to baby foods and pet foods, but also as a constituent of a large variety of other goods, such as beverages, drugs, hydrocolloids, biofuels, etc. Examples of the widespread food products deriving from cereal processing include: bread and pasta generally made, respectively, from wheat and durum wheat, couscous (a very fine grain cereal made from wheat), flour rice, crispy rice for breakfast, parboiled rice, corn meal, porridge, biscuits, snacks and many other derived products. For instance, the germ part of corn can also be refined to make a valuable vegetable oil or as a key ingredient in some margarines (Proto 1988). It is important to highlight that from the cereal cultivation stage, besides grains, representing the main product, there are many crop residues (such as cereal straw, corn stovers and rice hulls) that have the potential to be used as animal feed, bioenergy feed stocks and raw materials for a large number of both traditional and innovative industrial sectors.

4.1.2 Key Sustainability Aspects Associated with the Cereal Sector: Rice, Wheat, Corn and Derived Products

Generally, the first step in the cereal supply chain—after cultivation practices, harvesting, storage and transportation of cereal grains—is milling, a primary process that turns grain into flour. The cereal grain is composed of an embryo (the germ of the new plant), an endosperm (the starchy fraction), which accounts for about 80 % of the bulk of grain, and a protective layer of the seed coat (the bran fraction). Milling is one of the most important steps for the cereal food sector, from which many kinds of white flour and several by-products derive. The flour obtained from cereals such as wheat, corn and rice is the main raw material for the production of a wide range of products, the processes of which are obviously very different.

The increased industrialisation of the food system has been accompanied by rapid integration of the various links in the cereal supply chain (Reardon and Timmer 2012) and among these transport and logistics have a relevant role.

Cereal supply chains generally span long distances and consequently require extensive use of fossil fuels to deliver goods to customers (wholesalers, retailers, final consumers). The global energy use and related atmospheric emissions of a cereal supply chain depend principally on the characteristics of the places of agricultural production, on the storage locations and on the markets to reach. In the traditional cereal systems, and overall in developing countries, consumers buy certain types of food (bulk grains, etc.) from small independent retailers (open markets, small shops, etc.), and others (processed and packaged foods) prevalently in stores and supermarkets. Conversely, a modern supply chain for cereals and derived products is a managed process, based on a combination of knowledge and skills, spanning biology and the social sciences of economics and laws, engineering and human behaviour, and more. All are set in an integrated framework that in the past was often based upon a vertical model whereby upstream and downstream activities were managed by one organization (FAO 2013a). However, the commodity nature of cereal grains promotes relationships that are transactional, in which parties are not even interested in establishing a close, long-term supply chain relationship.

As regards the supply chain of cereals and cereal-based products, the developed countries landscape is characterised by enormous complexity and it is better outlined as 'supply networks', in which some issues related to sustainability (for instance, climate change, resource depletion, food wastage) have a significant impact on the projected food supply and security.

Among the multiple paths to improving the production, food security and overall social and environmental performance of food, and particularly of the cereal sector, the minimisation of food loss (wastage) and waste appears to be a pivotal issue.

As mentioned in the first chapter, food is wasted throughout the food supply chain, from the initial agricultural production down to the final household consumption. The food losses in industrialised countries are as high as those in developing countries, but in the latter more than 40 % of the food losses occur at the post-harvest and processing levels, while in industrialised countries, more than 40 % of the food losses occur at the retail and consumer levels (Lipinski et al. 2013).

The FAO (2013a) estimates that each year roughly one-third in weight of all food produced for human consumption in the world is lost or wasted. In total, 54 % of the world's food wastage occurs 'upstream' during production, post-harvest handling and storage, while 46 % occurs 'downstream', at the processing, distribution and consumption stages. In terms of the measured calories of the various wasted foods, cereals are the largest source of wastage, representing more than half of the total (Fig. 4.3).

In particular, for cereals, wheat is the dominant crop supply in medium-and high-income countries, and the consumer phase involves the largest losses, which range from 40 to 50 % of the total cereal food waste. In low-income regions, rice is the dominant crop, especially in the highly populated region of South and Southeast Asia, where agricultural production and post-harvest handling and storage are stages in the food supply chain with relatively high food losses, as opposed to the distribution and consumption levels.

The worldwide cereal wastage is a twofold dimension: the first part is undoubtedly linked to ethical issues, because of the pervasive poverty of many people on the planet, and the second aspect is related to the need to avoid numerous environmental impacts (water pollution, atmospheric emissions, waste, etc.) deriving from the wastage in the multiple steps of supply chains. Indeed, cereal wastage represents

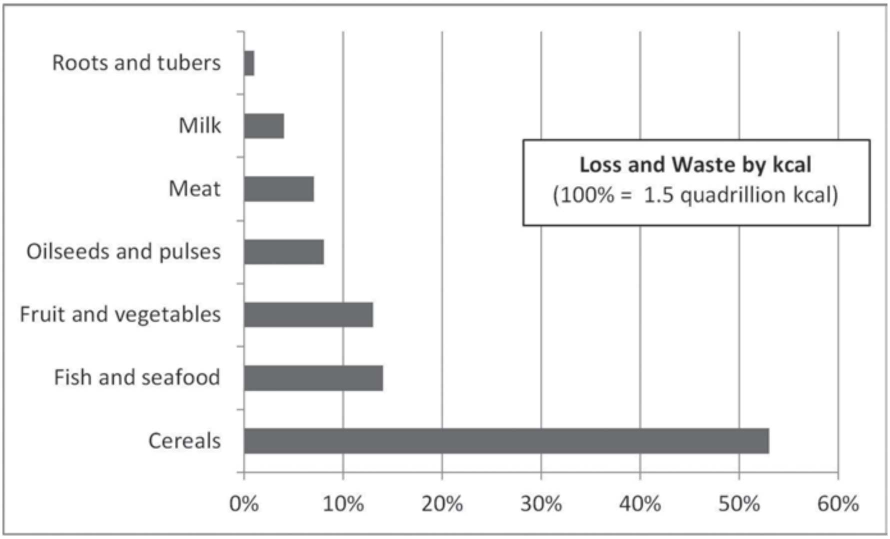

Fig. 4.3 The world's food wastage (2009). (Source: Adapted from FAO (2011b))

the main missed opportunity to improve global food security, on one hand, and to mitigate the environmental impacts and to implement more efficient resource use, on the other.

For the most important cereals (wheat, rice, corn), due the nature of the cereal-based product supply chains, it is difficult to assess the multitude of sustainability aspects in a systematic and coherent framework. Over the last decades, many studies have focused their analysis on specific practices in different phases of the supply chain tailored to a particular geographical context and often related to a single impact category. Many of these studies refer to the agricultural practices, which, as detailed in the following sections of the chapter, are often the most environmentally burdening phases of cereal-related supply chains. The use of fertilisers and pesticides is often responsible for such burdens, together with issues related to the use of the ever-decreasing available agricultural land (especially in Europe) and fossil fuel consumption. The wheat, rice and corn cultivation in the world accounts at present for some 60 % of global fertiliser use, and is expected still to account for just over half of fertiliser consumption by 2050 (Place and Meybeck 2013). Furthermore, agricultural practices are strongly site-specific, hence the impact deriving from the choice of a particular crop and the consumption of water and other resources are largely dependent on the characteristics of the production area.

It is clear that, in order to meet the increasing world future food demand, the cereal sector will increasingly face greater uncertainty and risks, both natural and economic. The first are linked to environmental damage resulting from the agricultural production system (externalities, such as biodiversity, soil loss, land degradation, GHG emissions, water pollution and solid waste production).

The economic challenges encompass principally the price volatility of both inputs and outputs.

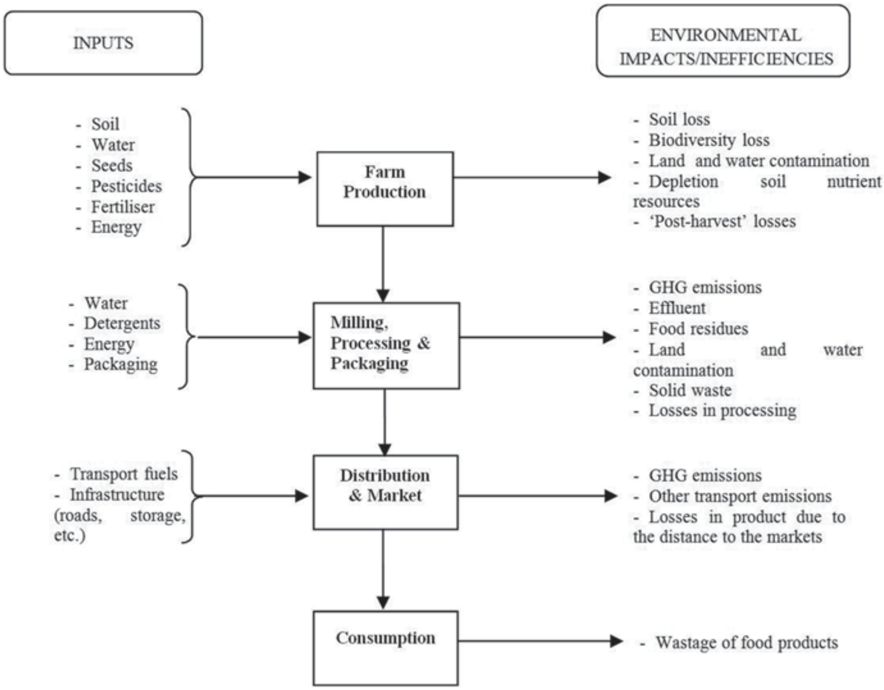

Fig. 4.4 The cereal and cereal-based product supply chain: an environmental perspective

An environmental perspective of the supply chain of cereals and cereal-based products is schematised in Fig. 4.4.

Progressing towards a more sustainable cereal supply chain requires an innovative management approach to enable measuring, assessment and monitoring capable of creating more efficient use of resources at every life cycle stage, from the farm to the consumer level. This necessary systemic vision, able to maximise the supply chain global performance, is the core concept of the life cycle thinking (LCT) framework. Taking an LCT perspective requires the development of new knowledge to look beyond the traditional vision in order to prioritise and set sustainability targets, improving the top-down and bottom-up cooperation along the supply chain.

4.2 Life Cycle Thinking Approaches Applied to the Production of Cereals and Derived Products: The State of the Art

As mentioned in the first chapter of the book, the sustainability of food products, including cereal-based ones, has become a main concern since a large part of the environmental burdens deriving from private consumption is attributable to such products. This has brought about the application of life cycle thinking approaches to the cereal sector, which has generated numerous life cycle studies of cereals and derived products.

This section encompasses a review, containing an indication of the methodologies, main findings and hot spots, of some of the main work concerning life cycle approaches applied to rice, wheat, corn and some of their principal derived food products.

To accomplish this reviewing process, a bibliographic search was performed via the consultation of scientific databases, such as the CASPUR Virtual Library (an Italian inter-university database), Science Direct, Scopus and Google Scholar, together with specific LCA conference proceedings. Combinations of key words such as 'life cycle, LCA, life cycle assessment, LCI, life cycle costing, footprint, sustainability' combined with logical expressions and other key words such as 'cereal, rice, wheat, maize, corn, pasta and bread' were used to identify the desired literature. Grey literature was excluded from the selection process. Seventy-nine publications were identified in total, together with documentation regarding cereal-related Product Category Rules (PCR) (see Table 4.2 and Sect. 4.2.6).

Of all the publications listed in Table 4.2, ten papers were excluded from the review process since they concern the use of cereal for energy purposes, thermoplastic production or other non-food production.

4.2.1 Classification of the Reviewed Life Cycle Thinking Approaches Applied to the Cereal Sector

The first studies regarding the application of life cycle approaches to the cereal sector date back to the beginning of the twenty-first century (e.g. Braschkat et al. 2003; Notarnicola and Nicoletti 2001; Petti et al. 2000). Since then, cereals have been studied intensively, in terms of their sustainability, via life cycle methodologies, confirming the topic as one of the major subjects of the last two international conferences on LCA in the agri-food sector held in Bari, Italy, in 2010 (Notarnicola et al. 2012a) and Saint Maló, France, in 2012 (van der Werf et al. 2013).

Specifically, such life cycle approaches have been extensively applied to the three most important cereals produced worldwide (rice, corn and wheat). As mentioned in the previous section, 69 studies, including LCAs regarding these three cereals, were reviewed: 65 % of them were published in scientific journals between 2000 and December 2013; 29 % of them are proceedings from conferences and workshops; and 5 % of them are research and/or project reports. The relevance of each paper to a specific cereal or derived product is illustrated in Fig. 4.5: rice (33 %), corn (11 %), wheat (38 %) and its derivatives (bread, 9 % and pasta, 9 %). Even if this review is certainly not exhaustive, it is undoubtedly a sound representation of life cycle studies in the cereal sector.

The scope of the reviewed studies can be broadly classified into four groups, which are summarized below.

Table 4.2 List of all the identified life cycle publications (prior to final screening and selection for review) concerning cereals or derived products

Reference	Main purpose	Methodology	Functional unit
Petti et al. (2000)	Applicative case study on pasta production	LCA	1 kg of pasta in primary and secondary packaging delivered to the company's direct customers
Notarnicola and Nicoletti (2001)	Comparative case study on pasta and couscous production	LCA	Quantity of product needed to produce a meal for 4 people
Kim and Dale (2002)	Applicative study of ethanol production from corn	LCA	1 kg of ethanol
Braschkat et al. (2003)	Comparative case study on bread with various cultivation methods	LCA	1 kg of bread ready for consumption
Brentrup et al. (2004)	Applicative case study on wheat	LCA	1 t of wheat grain
Narayanaswamy et al. (2004)	Applicative case study on bread, beer and canola cooking oil	LCA	Loaf of bread, 1 hL beer, 1 L cooking oil
Notarnicola et al. (2004)	Hybrid LCA approach applied to pasta production	LCA-IO	1 kg of pasta
Salomone and Ciraolo (2004)	Applicative case study on pasta production	LCA	1 t of dry packaged pasta
Breiling et al. (2005)	Rice-related GHG	Top-down approach CF	n.a.
Roy et al. (2005)	LCI rice processing	LCI	1 t of rice
Spatari et al. (2005)	Corn-derived ethanol automobiles	LCA	n.a.
Charles et al. (2006)	Applicative case study on wheat production system for bread making performed to optimise fertilisation	LCA	ha; 1 t of grain; 1 t of grain with constant quality (13 % of protein in dry grain)
Bevilacqua et al. (2007)	Applicative case study on pasta production	LCA	0.5 kg of packaged durum wheat pasta sold in the Italian market
Harada et al. (2007)	GHG of conventional puddling, no-puddling and no-tilling rice cultivation	CF	60 m^2 of land
Roy et al. (2007)	LCA+cost assessment of rice in Bangladesh	LCA	1 t of rice
Kim and Dale (2008a)	Effects of N fertiliser GHG from corn cultivation		1 kg of dry corn grain
Kim and Dale (2008b)	Ethanol derived from corn grain via dry milling	LCA	1 kg of ethanol

Table 4.2 (continued)

Reference	Main purpose	Methodology	Functional unit
Nemececk et al. (2008)	Comparison of cereal crop rotation with and without grain legume	LCA	Land, economic and energy-specific FUs
Pelletier et al. (2008)	Assessment of generic life cycle models of contemporary conventional and organic cultivation of cereals and other crops	LCA	1 t of each crop
Renouf et al. (2008)	Comparison of sugar production from corn sugarcane and beet	LCA	1 kg of monosaccharide
Schmidt (2008)	Illustrative framework for defining system boundaries in consequential agricultural LCA	LCA	1 kg of wheat demanded in Denmark
Yossapol and Nadsataporn (2008)	Applicative case study on rice production in Thailand	LCA	1 t of unmilled rice grain
Aldaya and Hoekstra (2010)	WF due to Italian consumption of pizza and pasta	WF	1 kg pasta, 725 g pizza
Blengini and Busto (2009)	Applicative case study of alternative agri-food chain management systems in northern Italy	LCA	1 kg of refined and packed rice
Kasmaprapruet et al. (2009)	Applicative case study on milled rice production in Thailand	LCA	1 kg of milled rice
Meisterling et al. (2009)	Case study of organic and conventional wheat	CF	0.67 kg of wheat flour
Roy et al. (2009a)	Review of LCA on some foods	LCA	n.a.
Roy et al. (2009b)	LCI of different forms of rice consumed in households in Japan	LCI	1 kg of final product or 1 MJ energy supplied by the final product
Kim et al. (2009)	Case study of corn grain and corn stover for ethanol in the US		1 kg of dry biomass
Vidal et al. (2009)	Case study of thermoplastic production from rice husk and other materials	LCA	1 kg of thermoplastics
Aldaya et al. (2010)	Case study of water footprint of wheat and rice (and cotton) in Asia	WF	m^3/ha
Biswas et al. (2010)	Comparative study on greenhouse gas emission due to the production of Australian wheat, meat and wool	LCA	1 kg of wheat
Hayashi et al. (2010)	LCI inventory DB study for crop production in Japan	LCI/IO	n.a.

Table 4.2 (continued)

Reference	Main purpose	Methodology	Functional unit
Kägi et al. (2010)	GHG rice—study on the use of confidence limits for communicating LCA results	CF	1 kg of rice
Khoo et al. (2010)	Calculation of GWP of beef, chicken, tofu, rice and tomatoes delivered to Singapore	LCA	1 kg of protein per food
Kissinger and Gottliebb (2010)	Profile of the environmental burden of production of grain in Israel	EF	1 t of grain
Kløverpris et al. (2010)	Methodological case study on wheat, with a focus on land use	LCI	1 t of wheat
Ruini and Marino (2010)	Applicative case study on durum wheat production	CF, EF, WF	kg of wheat considering the specific yield
Ruini et al. (2010)	Applicative case study on pasta production	EF	1 t of product
Seda et al. (2010)	Environmental assessment of different crops using different FUs	LCA	land area, yield end economic FUs
Settanni et al. (2010b)	Applicative study of a costing model to pasta	LCC	0.5 kg of pasta
Yoshikawa et al. (2010)	Case study of the CF of ecologically cultivated rice using sampling survey theory	CF	4 kg of polished rice
Wang et al. (2010)	Applicative study of rice production in China	LCA	1 t of rice
Williams et al. (2010)	Applicative and comparative study of wheat bread and other products	LCA	1 t of product
Berthoud et al. (2011)	Case study of freshwater eco-toxicity assessment due to wheat production	LCA (partial)	1 kg of wheat; 1 ha of land
Chapagain and Hoekstra (2011)	Applicative study regarding the WF of rice	WF	1 t of paddy rice
Espinoza-Orias et al. (2011)	CF of bread consumed in UK	CF	800 g loaf of bread
Ferng (2011)	Case study footprints on consumption of wheat and rice in Taiwan	EF	Hectare of land
Gan et al. (2011a)	Case study on field crop cultivation in semi-arid regions with crop rotation	CF	1 kg of wheat

Table 4.2 (continued)

Reference	Main purpose	Methodology	Functional unit
Gan et al. (2011b)	Case study on wheat cultivation with crop rotation in Canada	CF	1 kg of wheat
Lo Giudice et al. (2011)	Case study on Sicilian pasta production	LCI	1 kg of pasta
Nalley et al. (2011)	Comparative study on different agricultural techniques relative to corn, rice, wheat and other products	CF	1 acre of land
Prasara and Grant (2011)	Comparative study for energy production from rice husk	LCA	1000 t of rice husk
Röös et al. (2011)	Applicative case study on pasta production	CF	1 kg of pasta in paper packaging
Ruini (2011)	Environmental assessment and labelling of a multinational company	LCA, CF, WF	1 kg of product
Li Borrion et al. (2012)	Bioethanol production from wheat straw	LCA	1 kg of ethanol
Brankatschk and Finkbeiner (2012)	Comparison of different allocation methods for different agricultural products	LCA	n.a.
Van Stappen et al. (2012)	Scenario evaluation for food and non-food uses of cereal	LCA	Any useful output per hectare in an average year
Drocourt et al. (2012)	Applicative case study on rice production in a French region	LCA	1 t of rice
Fallahpour et al. (2012)	Comparative study of different farming systems for barley and wheat	LCA	1 t of cereal
Hokazono and Hayashi (2012)	Comparison of conventional and organic rice farming in Japan	LCA-IO	n.a.
Huang et al. (2012)	Applicative case study on wheat and maize	WF	1 kg of grain
Delivand et al. (2012)	Case study on the use of rice straw for ethanol production in Thailand	CF	1 t of dry rice straw
Kulak et al. (2012)	Case study of environmental improvements for bread production	LCA	1 kg of bread
Laurent et al. (2012)	Availability of data in existing LCI databases regarding cereals	LCA	n.a.

Table 4.2 (continued)

Reference	Main purpose	Methodology	Functional unit
Malça and Freire (2012)	Land use issues regarding wheat-based bioethanol	CF	1 MJ of fuel energy content
McConkey et al. (2012)	Applicative case study on wheat and maize	WF	1 t of grain
Muñoz et al. (2012)	Comparative study of conventional and organic wheat in Chile	LCA	1 t of wheat
Roer et al. (2012)	Applicative case study on wheat, barley and oats	LCA	1 kg of grain
Ruini et al. (2012)	Environmental assessment and labelling of a multinational company	LCA, CF, WF	1 kg of product
Shafie et al. (2012)	Case study of electricity generation in Malaysia from rice husk	LCA	1.5 MWh of electricity
Yoshikawa et al. (2012)	Comparison of different fertilisation techniques for rice production in Japan	CF	1 kg of rice
Eshun et al. (2013)	Case study of GHG emission in Ghana related to rice	CF	Land surface
Murphy and Kendall (2013)	Study of allocation issues for corn and stover production	LCI	1 ha of land
Ruini et al. (2013)	Comparison of different crop rotation systems	CF, WF, EF and others	1 t of durum wheat
Xu et al. (2013)	GHG study related to rice production in Chinese districts	CF	1 t of rice
Yan and Boies (2013)	Case study on uncertainty of GHG for wheat ethanol	CF	n.a.
Yoo et al. (2013)	Case study of water use issues for rice production in Korea	WF	n.a.
Zhang et al. (2013)	Effects of tillage on the soil organic carbon sequestration rate, C footprints and C sequestration in wheat–maize systems	CF	n.a.

LCA life cycle assessment, *LCIA* life cycle impact assessment, *LCI* life cycle inventory, *LCC* life cycle costing, *IO* input output, *CF* carbon footprint, *EF* environmental footprint, *WF* water footprint, n.a. not available

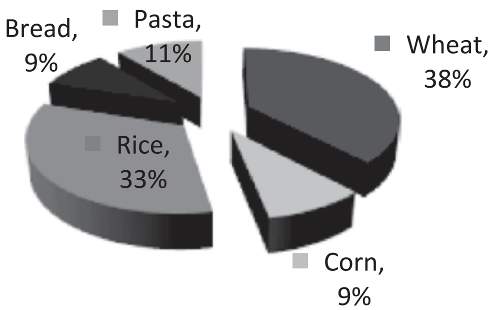

Fig. 4.5 Percentages of reviewed studies regarding crops/derived products

The *first group* of studies comprises papers (approximately 60 % of the total) that profile the environmental burden of a cereal/cereal product or compare different farming practices: Notarnicola and Nicoletti (2001) compared the life cycle of two foods (pasta and couscous); Roy et al. (2005) assessed the LCI of fresh parboiled and fresh rice produced by different production processes; Roy et al. (2009b) evaluated the life cycles of different forms of rice; Lo Giudice et al. (2011) focused on the LCI of pasta, taking into account all the different phases of the productive cycle; Aldaya and Hoekstra (2010) applied water footprint (WF) and LCA to pasta and pizza; Biswas et al. (2010) used LCA with the aim of calculating the GHG emissions for wheat, meat and wool; Röös et al. (2011) applied LCA to wheat and pasta; similarly, Salomone and Ciraolo (2004), Bevilacqua et al. (2007) and Röös et al. (2011) carried out an applicative case study on pasta; Berthoud et al. (2011) used the USEtox model to assess the share of the total freshwater ecotoxicity impact due to pesticide use and to identify active ingredients to replace these high-impact pesticides and estimate the effect of such substitution; Espinoza-Orias et al. (2011) adopted a simplified LCA vs. the carbon footprint (CF) calculation for bread consumed in the UK; Kulak et al. (2012) also studied bread with a focus on possible environmental improvements to its production; Fallahpour et al.'s (2012) study aimed to analyze the impact assessment of wheat and barley; Kasmaprapruet et al. (2009) applied LCA to milled rice; whilst Xu et al. (2013) and Yoo et al. (2013) calculated the carbon and water footprints of rice, respectively, from China and Korea.

This group of studies, which includes the analysis of the environmental impact of different agricultural practices, is very interesting since it provides useful information on the available choices for sustainable farming practices and deserves to be investigated further, as already pointed out by Benedetto et al. (2013). In particular, Braschkat et al. (2003) compared different industrial practices in the supply chain, whilst Brentrup et al. (2004) evaluated different N rates in wheat production. Kim et al. (2009) estimated the county-level environmental performance for continuous cultivation of corn grain and corn stover in various corn-growing locations in the Corn Belt states; two cropping systems were under investigation: corn produced for grain without collecting stover and corn produced for corn grain and corn stover harvesting. Nemececk et al. (2008) compared crop rotation with and without grain legumes; Kim and Dale (2008a) evaluated the global warming effects of N fertiliser application rates in the US using data at the county level; in their study, Meisterling et al. (2009) compared organic and conventional wheat practices; Ruini and Marino (2010), Ruini et al. (2013) compared durum wheat cultivation in two regions with different cropping characteristics and different kinds of rotation; Nalley et al. (2011) compared 57 different farming practices for cotton, rice, sorghum, soybeans and wheat; Hokazono and Hayashi (2012) used a multi-year comparative LCA of agricultural production systems with the aim of identifying the variability in environmental impacts during the conversion from conventional to organic farming; Muñoz et al. (2012) compared conventional and organic wheat crop systems; Kägi et al. (2010) carried out a comparison between conventional, organic and upland rice production; Charles et al. (2006) used an LCA approach for the optimisation of fertiliser use for wheat destined for bread production; Harada et al. (2007) studied GHG emission deriving from conventional puddling, non-puddling and no-tillage rice cultivation; and Gan et al. (2011a, b) studied the possibilities of reducing global warming effects due to wheat cultivation from the diversification of crop rotation. Yoshikawa et al. (2010) calculated the carbon footprint of ecologically cultivated rice in Japan using data from multiple producers in order to ensure the representativeness of the inventory results. In their other carbon footprinting study, Yoshikawa et al. (2012) studied the effects on GHG emissions of different fertilisation techniques for rice production. Finally, Zhang et al. (2013) identified carbon-friendly tillage systems for the North China Plain by evaluating the effects of different types of tillage on the sequestration rate of soil organic carbon for double-cropping cultivation systems based on wheat and maize.

The *second group* of studies focuses on comparing the environmental burden of different food products, e.g. Narayanaswamy et al. (2004) carried out an LCA case study for wheat-to-bread, barley-to-beer and canola-to-cooking oil with the objective of identifying the key environmental impacts along the food chain and assessing the relative contributions of pre-farm and farming to the total life cycle environmental impacts of products produced and consumed in Western Australia; Pelletier et al. (2008) generated a generic LC model of contemporary conventional and organic production systems in Canada in order to predict the 'cradle-to-farm-gate' cumulative energy demand for canola, corn, soy and wheat; Seda et al. (2010) analysed and compared the LCAs of wheat and corn, as well as horticultural crops, using different functional units and suggested the best alternative crop; Khoo et al. (2010) compared beef, chicken, soy-tofu, rice and tomato production; McConkey et al. (2012) applied the WF approach to compare the maize and wheat production processes; the LCA carried out by Renouf et al. (2008) compares different types of sugar production based on corn, sugarcane and beet; and Williams et al. (2010) evaluated the different burdens of bread wheat, oilseed rape and potatoes produced in various parts of the UK.

The *third group* includes studies that have adopted differing approaches to LCA or methodologies used in combination with LCA, or even different methodologies for the assessment of environmental impacts. It includes: Notarnicola et al. (2004), who applied LCA and IO-LCA to pasta production; Breiling et al. (2005), who used IO-LCA referred to GHG emissions for rice; Roy et al. (2007), who used LCA and cost assessment to determine the environmental load and production cost of rice in Bangladesh; Aldaya et al. (2010), who calculated a green and a blue WF for wheat, rice and cotton; Settanni et al. (2010b), who applied a novel costing model to pasta and LCC based on IOA to evaluate its consistency with LCA; Chapagain and Hoekstra (2011), who carried out a WF for rice production; Ferng (2011), who evaluated an environmental footprint (EF) in terms of crop land and energy land; Laurent et al. (2012), who analysed the available data in existing LCI databases regarding cereals and cereal products; Van Stappen et al. (2012), who carried out an environmental cereal LCA (attributional and consequential) together with a social LCA; Brankatschk and Finkbeiner (2012), who demonstrated the benefit deriving from the use of the cereal unit as a functional unit for a better allocation procedure in LCA studies of agricultural systems; and Murphy and Kendall (2013), who evaluated three different allocation methods for solving the problem of multi-functionality in the case of the production of corn grain.

The *fourth group* of studies concentrates its efforts on profiling the environmental burden of cereal production in a given area or on identifying the environmental hot spots in production systems' performance: e.g. Harada et al. (2007) estimated GHG emissions in northern Japan; Kissinger and Gottliebb (2010) assessed the ecological footprint for grain-based consumption in Israel (wheat, barley, maize); Huang et al. (2012) used WF and LCA for wheat and maize in China's main breadbasket basins; Drocourt et al. (2012) evaluated the environmental assessment of rice production in Camargue; Eshun et al. (2013) estimated the GHG emission and energy impact in rice production systems in Ghana; Schmidt (2008) focused his development of a framework for the definition of system boundaries in consequential LCA on Danish wheat production; and Yossapol and Nadsataporn (2008) carried out an LCA on rice produced in Thailand. Blengini and Busto (2009) applied the LCA methodology to rice production in northern Italy; Ruini and Marino (2010) calculated footprints for wheat productions in the south-west US and southern Italy; in their work, Hayashi et al. (2010) developed LCI data for Japanese crop production; similarly, Kløverpris et al. (2010) developed inventory data for land use deriving from wheat production in Brazil, China, Denmark and the USA; Muñoz et al. (2012) evaluated conventional and organic wheat crop systems in Chile; and Roer et al. (2012) assessed the life cycle environmental impact of grain production in central south-east Norway on a typical grain farm with a mix of barley, oat and spring wheat. The work of Ruini and Marino (2010) is an example of the second investigated issue of this group of studies, and is an application of the EF as a key performance indicator of a large-scale pasta producer.

In general, the LCA case studies have played a key role in supporting decision making in the cereal sector, but some authors highlight hot spots and methodological issues (some of the latter are discussed in detail in Sect. 4.3). For example, Schmidt (2008) presents a framework for defining system boundaries in consequential agricultural LCA. The framework is applied to an illustrative case study, i.e. the LCA of increased demand for wheat in Denmark. Different scenarios for meeting the increased demand for wheat show significant differences in emission levels as well as land use. The comparison of scenarios shows significant differences in the contribution to the included impact categories (climate change, eutrophication and land use). Therefore, the modelling of how increased demand can be met in an LCA appears to be crucial for the outcome of any study involving cereals.

Furthermore, most life cycle inventory data for crops do not include the ultimate (marginal) land use induced by crop consumption. Land use and land use change are usually considered at the inventory level in terms of land occupation and land transformation. Kløverpris et al. (2010) present, document and discuss a method that addresses this problem via its application to wheat consumption in Brazil, China, Denmark and the USA. The analysis shows that a combination of economic modelling, geographical data and agricultural statistics can resolve some of the obstacles to identifying ultimate or marginal land use changes when applying consequential LCA to crop production such as wheat.

Of all the LCA studies reviewed, only a few include pesticide diffusion models (Berthoud et al. 2011), nutrient balance models (Brentrup et al. 2004; Charles et al. 2006; Seda et al. 2010; Williams et al. 2010; Yoshikawa et al. 2012) and carbon storage accounting (Roer et al. 2012; Yoshikawa et al. 2012). On the other hand, the majority of the studies consider the emissions associated with fertiliser use (Berthoud et al. 2011; Blengini and Busto 2009; Brentrup et al. 2004; Charles et al. 2006; Espinoza-Orias et al. 2011; Fallahpour et al. 2012; Hokazono and Hayashi 2012; Kim et al. 2009; Meisterling et al. 2009; Muñoz et al. 2012; Murphy and Kendall 2013; Narayanaswamy et al. 2004; Nemececk et al. 2008; Pelletier et al. 2008; Renouf et al. 2008; Roer et al. 2012; Schmidt 2008; Seda et al. 2010; Yoshikawa et al. 2012). The most frequently applied method for calculating fertiliser emissions is the IPCC model, even though in some cases other methodologies were used, e.g. the DNDC model (Yoshikawa et al. 2012).

The most common hot spots identified when assessing the agricultural activities are fertiliser and pesticide production and use and fuel-related emissions (e.g. Braschkat et al. 2003; Roer et al. 2012; Williams et al. 2010). According to some authors (Braschkat et al. 2003; Pelletier et al. 2008), the adoption of organic cropping systems could improve the environmental profile of agricultural activities by lowering the overall impact, even though, as Blengini and Busto (2009) state, the lower grain yields obtained with organic systems could cancel this benefit.

4.2.2 LCA of Cereal Product Systems

Since over 78 % of the identified studies involve the implementation of classical environmental LCAs of cereal systems, involving multiple impact categories, the remaining part of this section reviews such LCA work regarding corn (Sect. 4.2.2.1), rice (Sect 4.2.2.2), wheat (Sect 4.2.2.3) and wheat-derived products (Sect 4.2.2.4). Following life cycle costing studies, simplified and hybrid LCA studies and footprints are discussed in detail in Sects. 4.2.4 and 4.2.5. Finally, Sect. 4.2.6 discusses cereal-related EPD labels and the relative PCRs.

4.2.2.1 Corn

Corn is an annual herbaceous plant that is widely cultivated throughout the world: over 170 million hectares are dedicated to corn cultivation (FAO 2013b). The United States produces 40 % of the world's harvest (273,832,130 t in 2012, according to the FAO), followed by China (208,130,000 t). This cereal is used both as human food and livestock feed and as a feedstock for the production of ethanol fuel; according to the RFA (2010), most ethanol produced in the United States is derived from corn grain.

Most of the studies regarding corn from a life cycle perspective consider this cereal as a feedstock for biodiesel production (e.g. Kim and Dale 2002, 2008b; Spatari et al. 2005). Only a few studies with the objective of evaluating the environmental performance of corn cultivation were identified and are discussed in this section.

Specifically, the major aims of the articles under study differed: while the study by Murphy and Kendall (2013) focused on the life cycle inventory for corn production, the goal of the study carried out by Kim et al. (2009) was to estimate the environmental performance of corn cultivation in various corn-growing locations in the Corn Belt states. Nalley et al. (2011) performed an environmental assessment of six of the largest row crops (among which is corn) produced in Arkansas, taking into consideration only the GHG emissions. Pelletier et al. (2008) compared different conventional and organic crops, including corn, in Canada. The inventory data used referred to the average agricultural practice specific to the area under study. All the studies under analysis used a cradle-to-farm-gate perspective.

As for the functional unit, both area units (i.e. 1 ha of corn and stover production (Murphy and Kendall 2013)) and mass-based units, such as 1 kg of dry biomass (Kim et al. 2009) or 1 kg of corn (Pelletier et al. 2008), were selected.

An important issue associated with LCA-oriented studies for multifunctional processes consists of the most appropriate choice of the allocation approach. In fact, as Murphy and Kendall (2013) demonstrate, the allocation method selected can heavily affect the results of the analysis. Along with corn grain, stover is also produced; as a by-product, it can be left on the field to maintain the soil condition, collected to be used as cattle fodder or harvested for biofuel production (Murphy and Kendall 2013). Different allocation approaches were performed in the studies analysed: no allocation, when corn stover is not collected (Kim et al. 2009), system expansion, when both corn grain and stover are harvested (Kim et al. 2009), and energy-based, economic allocation and subdivision, for the three different scenarios assessed by Murphy and Kendall (2013). The aim of this last study was in fact to explore these three allocation strategies for corn and stover, pointing out the advantages and disadvantages of each of them. The authors stated that 'value-based allocation methods, like energy and economic allocation, may be most appropriate when they reflect the goals of the production system. In addition, value-based methods are typically simple to apply thus may be more transparent'.

Another important issue when dealing with the environmental assessment of agricultural activities is represented by field emissions. Different models are available for estimating these emissions. Murphy and Kendall (2013) as well Pelletier et al. (2008) estimated N emissions from fertilisers according to the IPCC methodology

(IPCC 2006), while Kim et al. (2009) applied the DAYCENT model (Natural Resource Ecology Laboratory 2005), which is the daily time step version of the CENTURY model, a multi-compartmental ecosystem model. This model was also used to predict the carbon sequestration by soil.

The following impact categories were taken into account when assessing the environmental performance of corn cultivation: climate change, acidification, eutrophication, fossil energy (Kim et al. 2009; Murphy and Kendall 2013; Pelletier et al. 2008) and ozone layer depletion (Pelletier et al. 2008).

Regarding the most impacting materials identified, the studies under analysis show consistent results: the production and use of fertilisers generally dominate the total GHG and fossil energy impacts in conventional cropping systems. Pelletier et al. (2008) show that, for organic crop systems, the second major contributor after fertiliser emissions is fuel use.

According to Pelletier et al. (2008), the choice of the cultivation system (conventional or organic) affects the environmental performance of corn production. In their study, they show in fact that the organic crop production models generated consistently lower contributions to all the impact categories: this reduction was mainly due to the substitution of conventional nitrogen fertilisers with green manure.

4.2.2.2 Rice

Rice production is the second-largest cereal production worldwide, but in terms of dietary intake, rice is first in the world ranking, as the bulk of the world rice production is destined for food use, although some is used in domestic animal feeding. Rice is the primary staple for more than half the world's population, with Asia representing the largest producing and consuming region. In recent years, rice has also become an important staple food throughout Africa (FAO 2013b).

From an LCA perspective, most LCA studies on rice have Asia as the geographical boundary, i.e. Japan (Harada et al. 2007; Hokazono and Hayashi 2012; Roy et al. 2009b; Yoshikawa et al. 2010, 2009), Bangladesh (Roy et al. 2005, 2007), Thailand (Kasmaprapruet et al. 2009; Yossapol and Nadsataporn 2008) and China (Wang et al. 2010), followed by Europe, namely France (Drocourt et al. 2012) and Italy (Blengini and Busto 2009).

Some of these LCA studies are limited to the life cycle inventory level (Roy et al. 2005, 2009b); meanwhile, others report only the GHG emissions (Harada et al. 2007; Roy et al. 2009b).

The cultivation phase of rice emerged as the hot spot in the life cycle of rice (Kasmaprapruet et al. 2009). Rice is present in many varieties, e.g. brown, partially milled, well-milled, germinated brown and parboiled (i.e. rice that has been boiled in the husk), which differ in their production process and therefore also in their environmental impacts, as already discussed by Roy et al. (2009a) in their review paper of LCA of food products. When parboiled rice was compared with non-parboiled rice, the latter showed lower environmental loads (Roy et al. 2005), whilst the partially milled rice (milling 2 %) was found to be the most environmentally friendly rice by Roy et al. (2009b). Apart from the reduction in the environmental impacts, the choice of different types of rice has implications for their nutritional context, e.g. the partially milled rice leads to the retention of some of the nutrients that are beneficial to human health (Roy et al. 2009b), and parboiling improves the milling yield, storability and nutritional content (Roy et al. 2007). However, this kind of comparison can be misleading as the different varieties have different tastes and require different amounts of water during cooking. As mentioned by Roy et al. (2009b), to gain a certain amount of energy from cooked rice, greater amounts of parboiled rice need to be consumed compared with the well-milled rice because of the higher volume expansion ratio. The environmental load is dependent not only on the form of the rice but also on the packaging used; as shown by Roy et al. (2009b), the paper bag packaging option seems to be preferable to the polyethylene bag option. Further differences are connected with the cultivation techniques, e.g. organic farming or upland farming (upland rice is rice cultivated without submersion and grown under a reduced water regime). The study performed by Blengini and Busto (2009) shows that organic and upland farming have the potential to decrease the impact per unit of cultivated area. However, due to the lower grain yields, the environmental benefits per kg of the final products are greatly reduced in the case of upland rice production and almost cancelled for organic rice. The comparison between conventional and organic farming is a delicate issue, the results of which can be biased by the assumption that the year-to-year variations in agricultural production are negligible. However, as demonstrated by Hokazono and Hayashi (2012), it is necessary to investigate the variability in environmental impacts during the conversion period, because the performance of organic farming in the conversion process from conventional farming is unstable. Although the environmental impacts of organic rice production are higher than those of conventional rice production on average, they decrease to the same level as conventional farming in the last phase of conversion. Other options to decrease the environmental impacts of rice production refer to the use of alternative types of fertiliser. Yoshikawa et al. (2012) compared two types of cultivation: reduced chemical fertiliser use and green manure use. The results show that the utilisation of green manure reduces the production costs and the impact due to energy consumption and eutrophication, though it increases the farmer's labour time and GHG emissions. Furthermore, in order to reduce the total impact of rice production, improved water management would provide a significant benefit for green manure use.

One peculiar aspect of the rice life cycle is connected with water management practices, mainly due to long submersion times, which lead to the anaerobic decomposition of organic matter and the consequent methane production, which determine the GHG emissions (Blengini and Busto 2009; Drocourt et al. 2012; Harada et al. 2007).

4.2.2.3 Wheat

The LCA studies performed on wheat can be distinguished into two main categories: studies addressing wheat as cereal, without indication of its final use

(Berthoud et al. 2011; Brentrup et al. 2004; Fallahpour et al. 2012; Roer et al. 2012; Schmidt 2008), and studies of wheat used for bread production (Charles et al. 2006; Meisterling et al. 2009; Williams et al. 2010). One key issue in the LCA of wheat is the assessment of the impact of the fertilization rate on the final results, particularly nitrogen (N) fertiliser (Brentrup et al. 2004; Charles et al. 2006). However, as concluded by Berthoud et al. (2011), pesticides and their effects on the ecosystems should not be neglected, as they strongly contribute to freshwater ecotoxicity impacts.

A further main topic for cereal LCAs—and wheat LCA in particular—is the comparison between different farming techniques, i.e. conventional vs. organic farming (Meisterling et al. 2009) or irrigated vs. rain-fed farming (Fallahpour et al. 2012). In the case of climate change, Meisterling et al. (2009) show that when conventional and organic wheat are transported the same distance to market, the organic wheat system produces less CO_2-eq per functional unit than the conventional wheat system. The shipping distance of the wheat, as well as the transport mode of the finished product, could cancel out or enhance the advantage of the organic wheat. With regard to the irrigation issue, Fallahpour et al. (2012) show that under low consumption of N fertiliser, the environmental impacts of rain-fed wheat are lower than those of irrigated wheat.

When other impact categories are included in the assessment, similar trade-offs can be expected between impact categories, as well as different results according to the different FUs chosen for the assessment. This is the reason why some studies (Charles et al. 2006; Roer et al. 2012) include a sensitivity analysis with different functional units, mainly mass (1 kg of dry matter) or area (1 ha of cultivated land). Defining the functional unit in terms of mass is not always a good measure of the quality of the food produced; the energy (MJ) and protein content (kg) can be of greater interest (Roer et al. 2012). To compare different systems of production managed with different fertilisation intensities, it is necessary to consider both the yield and the quality of the product. Assessment of the wheat production system shows that increased fertilisation needs a sufficient increase in yield to justify the additional emissions (Charles et al. 2006). Finally, in order to reveal the importance of system boundaries, attention should be paid to the inclusion/exclusion of factors in LCA studies, such as machinery manufacturing, buildings, pesticide production and use, humus mineralisation and nitrous oxide loss from the use of mineral fertiliser, as shown by Roer et al. (2012).

4.2.2.4 Wheat Products

Pasta and bread have been the object of various LCA analyses. One key element when assessing these processed products is the system boundary selection: the majority of the studies adopted a cradle-to-grave approach, including all the life cycle phases up to disposal in the analysis (Bevilacqua et al. 2007; Espinoza-Orias et al. 2011; Notarnicola and Nicoletti 2001; Notarnicola et al. 2004; Salomone and Ciraolo 2004). Taking into account the whole life cycle of a product appears to be an important element since, in some cases, the last stages of the product life cycle were found to be not negligible. In fact, besides the cultivation phase, which resulted as being determinant in all the studies carried out on pasta and bread, other stages of the life cycle, such as distribution and use, indicated 'environmental importance'. While for bread the impact of the consumption phase resulted as significant depending on the consumer's behaviour (if bread is refrigerated or toasted) (Espinoza-Orias et al. 2011), the use phase associated with pasta appeared to be relevant in terms of energy consumption and related impacts (Bevilacqua et al. 2007; Ruini et al. 2013). In some studies (i.e. Kulak et al. 2012; Salomone and Ciraolo 2004), the production (pasta production and bread baking) and distribution phases (Kulak et al. 2012) were also found to be critical.

A comparative approach was used in different studies: Notarnicola and Nicoletti (2001) assessed two different wheat-derived products (pasta and couscous), Bevilacqua et al. (2007) compared the results obtained with alternative production systems designed to reduce the environmental impact of pasta (conventional vs. organic crop systems, plastic vs. cardboard packaging), while Braschkat et al. (2003) analysed eight different scenarios for bread production. When considering the use of organic wheat for pasta or bread production, lower impacts were obtained, but more land area was required (Braschkat et al. 2003). Different milling and baking technologies were also assessed by Braschkat et al. (2003), revealing that home-made bread has a greater overall impact when compared with industrial bread.

All the functional units selected are based on mass, i.e. 1 kg of bread (Braschkat et al. 2003; Kulak et al. 2012), 1 kg of packaged pasta or the amount of pasta needed to prepare four portions (Notarnicola and Nicoletti 2001).

The typical impact categories taken into account in the studies regarding wheat products are global warming, acidification, eutrophication, ozone layer depletion, eco-toxicity and abiotic depletion. In some of the studies related to pasta, the normalisation phase of the results was carried out. The normalised results reveal the most affected impact categories, i.e. land use and fossil fuel, followed by respiratory inorganics and climate change, according to Bevilacqua et al. (2007).

4.2.3 Life Cycle Costing (LCC)

Evaluating the costs of a product system from a life cycle perspective is a task performed with the general intention of evaluating possible new investments in a supply chain (SC) or with the aim of optimising the existing resources and reducing the costs along the whole SC, including the consumption and end-of-life phases. Since the underlying framework of LCC is similar to that of LCA, such costing methods are also implemented as a means of evaluating the environmental costs of a product system or the most cost-effective method of making environmental improvements to it. In this respect, typically, LCC approaches combine some discounted cash flow analysis with LCA. However, in order to apply LCC effectively in the same holistic manner as LCA to the entire life cycle of non-durable products such as cereal products, input–output analysis (IOA) based approaches (Settanni et al. 2010a) need to be implemented from a microeconomic point of view. The application of such approaches allows, from a supply chain management (SCM) perspective, both the evaluation of the economic performance of an SC and the inclusion of considerations for environmental concerns. This input–output cost accounting methodology, fundamentally different from typical costing activities, has not yet become as mainstream as pure environmental LCA, and is thus still an object of academic research.

In Settanni et al. (2010b), a novel IOA costing model is applied to the fresh pasta supply chain. Here the authors address the problem of representing a southern Italian pasta factory as a series of interacting processes, including those with the suppliers and customers, and then transforming such relationships into financial transactions via matrix operations. This kind of modelling involving reciprocal interdependences, in terms of interconnected material flows, among the processes of an SC allows the management of inter-organisational cost issues. Specifically, such an approach permits the assessment of the activity levels along the pasta SC, together with the expected required resources, the related environmental burdens, and, subsequently, the internal production costs along such an SC. Even though the purpose of the paper is that of demonstrating the effectiveness of the approach, which in this case, for simplification purposes, excludes the agricultural stage, direct process and unit product costs were obtained, for the various types of pasta produced, and were simultaneously combined with the relative environmental burdens calculated with an LCA based on the same inventory data structure. Furthermore, the model is able to indicate the costs of inefficiencies along the different stages of the production process. To increase the robustness of the method, a non-deterministic analysis was performed via the use of uncertainties related to the main technical–economical parameters used for the study; this allowed the unit costs to be turned into calculated probability distributions.

The reviewed cereal based literature also encompasses work concerning costing activities, such as for example Roy et al. (2007), carried out in parallel with an LCA study, without a common integrated life cycle framework, using standalone calculations. In this case, the authors used the LCA methodology to determine the environmental load of different rice production processes in Bangladesh together with estimating the production cost of the rice in order to aid the decision-making processes employed for the identification of potential improvements of a product, a process or an activity.

Specifically, the production costs of milled and head rice were calculated, both per unit mass and energy. The results indicate that the production cost of untreated rice (per unit mass or energy) is higher than that of parboiled rice for the head rice option. However, the production cost of parboiled rice is found to be higher than the untreated rice for the milled rice option (probably due to the difference in rice yield and energy consumption in the production processes). The analysis of the production costs per tonne of rice (ranging from US$ 135.9/t to US$ 145.5/t) indicates that milled rice would be acceptable for the local consumers, in economic terms, whilst the untreated rice would be the best choice for sustainable consumption.

4.2.4 Simplified Life Cycle Assessment (S-LCA) and Hybrid Methods

Guinée et al. (2002) define simplified LCA as: '…a simplified variety of detailed LCA conducted according to guidelines not in full compliance with the ISO 14,040 standards and representative of studies typically requiring from 1 to 20 person-days of work'. Such a type of LCA is usually implemented due to time and/or cost constraints and typically leads to an indication of the main environmental criticalities of the product system analysed as opposed to a reliable quantification of the various burdens occurring during its life cycle. The simplification can occur at the inventory level and/or during the impact assessment phase of the study.

Simplified LCI, when based on process analysis, can be achieved via modelling simplification and data collection simplification strategies. Modelling simplification involves the cut-off of life cycle sub-phases or the removal of smaller elementary flows of the product system, or the modelling of a series of processes as a unique process. Data simplification processes usually involve the use of generalised non-detailed or non-specific databases in order to overcome the difficulties (time requirements, costs, confidential nature) of data collection.

Another approach to the simplification of the inventory phase is IO LCI, which uses IO tables (Suh and Huppes 2005). Such an approach is successful if sufficiently detailed applicable sectorial environmental data exist for the desired country.

LCIA simplification generally involves the exclusion of certain indicators or the aggregation of some of these into a unique new one, for example the cumulative energy demand indicator (Huijbregts et al. 2010).

Hybrid LCA methods can involve the integration of information from IO accounts coupled with process-specific data not as a means of simplification, but rather as a means of avoiding cut-off or truncation errors and hence making the study more complete.

In such a context, Notarnicola et al. (2004) evaluated how the conjunct adoption of a typical LCA approach and an IOA-based one can help improve the inventory set-up of the pasta sector. Furthermore, the study tried to quantify the hybrid approaches in order to improve the overall results. The IO-LCA methods are based on the utilisation of environmental matrices and input–output tables developed in America and in Europe. A comparison of the ISO 14,040 LCA results with those of the IO-LCAs highlights the capability of the IO approach to avoid truncation errors from cradle to gate and the capability of avoiding closed loops. At the same time, the results highlight the problem of gate-to-grave truncation of the IO approaches due to the nature of the input–output tables, of which the European ones seem to be less detailed than the US ones. Hybridising the approach via conjunct adoption of the above-mentioned approaches tended to obtain more detailed results, in particular for the impacts due to fertilisers and pesticides, for which the traditional LCA approach lacks specific data. The authors conclude that, for the pasta case study, IO-LCA approaches should not be used as standalone simplified methods but instead should be used in a hybrid approach with traditional LCAs, keeping in mind that the quantity and quality of the available data will determine the level and the type of adopted combination of approaches.

Similar conclusions are explicated by Hayashi et al. (2010). In this study, when comparing their LCI database regarding crops (including rice—conventional and non-conventional) with those derived from Japanese IO tables, the authors express the need to be cautious when validating results derived from IOA due to the coarseness of such a method for assessing agricultural technologies.

Other IO approaches have also been identified in the literature regarding the life cycle of rice. Specifically, Hokazono and Hayashi (2012), in their study concerning the change in environmental impact during the conversion from conventional to organic rice farming, due to the lack of data regarding duck rice farming, implemented IO analysis, via Japanese input tables, to build the inventories describing such a type of farming. Breiling et al. (2005), when evaluating rice-related greenhouse gases in Japan, used data from IO tables to track primary and secondary CO_2 emissions and detailed the secondary emissions using the IO tables of different Japanese prefectures. The main findings of this work are presented in Sect 4.2.5.

4.2.5 The Carbon, Water and Ecological Footprints

4.2.5.1 The Ecological Footprint (EF)

The ecological footprint (EF) is nowadays one of the most widespread indicators used for assessing the sustainability of humanity's demands on nature. Over the years, the assessment of the EF has been increasingly applied to food products. Regarding cereal production and cereal-derived foods, most of the studies concern wheat and pasta production.

The EF has been used as one environmental assessment method (together with the carbon footprint and water footprint) for the Environmental Product Declaration (EPD) of several cereal-based products from the Barilla company (see also Sect 4.2.6). Barilla, the largest world producer of pasta, was the first private company to develop a system based on the International EPD PCR (Product Category Rules) to certify the results of its LCAs, not only in terms of its carbon footprint, but also in terms of its water and ecological footprints. Ruini and Marino (2010) assessed the EF for wheat cultivation in southern Italy; considering a grain yield of 3.2 t/ha, they assessed the EF of wheat as being equal to 9.5 global m^2/kg: this value is considerably higher than the value assessed for the south-western USA (4.7 global m^2/kg). Ruini and Marino (2010) evaluated the EF of semolina pasta made by Barilla considering a cradle-to-factory-gate boundary. The objectives of this work were to quantify the environmental appropriation of each phase of pasta production, including the phase of final consumption. The total footprint for dry durum semolina pasta is 1.63 global ha/t of the final product (16.2 global m^2/kg) at the platform (this result regards the part of the productive chain from cradle to gate). The largest contribution, 77.6%, is due to durum wheat cultivation, followed by packaging (14.4 %), while other industrial processes, such as milling, pasta production and transport, usually associated with heavier pressure on ecosystems, are far less land-intensive, covering on the whole only 8 %. In another study, Ruini (2011) calculated for semolina pasta an EF equal to 10.9 global m^2/kg of pasta considering durum wheat cultivation (84.4 %), milling (1.0 %), packaging (5.5 %), pasta production (7.3 %) and distribution (1.8 %). In this study, cooking was excluded from the system boundary, but the authors estimated for this process an EF ranging from 2 global m^2/kg of pasta (when carried out using natural gas) to 6 global m^2/kg of pasta (when carried out using electricity). In the same study, the authors also report an EF for rice equal to 14 global m^2/kg of rice.

According to Cerutti et al. (2012), the agricultural phase accounts for almost 49 % of the whole product EF, the industrial phase (which includes elaboration, packaging and distribution) accounts for 9 % and the consumer phase (which includes the impact of cooking) accounts for 42 % of the whole product's EF.

Ferng (2011) measured Taiwan's rice and wheat consumption footprints in terms of cropland and energy land from 1989 to 2008 and identified the cropland location by source country. The results of this study indicate that Taiwan has continuously enlarged and dispersed the cropland for its rice and wheat consumption footprints in foreign countries, but it has decreased its footprint in domestic territories.

Kissinger and Gottliebb (2010) analysed the ecological footprint of grain-based consumption in the state of Israel during the last two decades. They found that most of Israel's grain footprint falls on North America, followed by the Black Sea region. The study also shows that while the overall consumption of grain products has increased throughout the research period, the size of the footprint has been dropping in recent years as a consequence of changing sources of supply and grain composition.

4.2.5.2 The Carbon Footprint (CF)

Food systems include agricultural phases as well as transport, processing and disposal and are among the main contributors of anthropogenic GHG emissions. Over the years, considering the importance of GHG emissions for the climate change impact category, the need to account for the emissions associated with the agricultural sector has become increasingly relevant. Consequently, the carbon footprint (CF) has become one of the key indicators of environmental sustainability aiming to identify the hot spots and stimulate emission reduction.

With regard to the agricultural sector, during the last 10 years, several studies focusing on the evaluation of the CF of different cereals have been carried out. Cereals represent one of the most important agricultural commodities and their cultivation is widespread worldwide in developed as well as in developing countries. Although the final destination of cereals can differ, their cultivation practices are quite standardised in the different geographical areas and involve significant GHG emissions.

Kim and Dale (2008a) evaluated the impact of nitrogen fertilisation on the CF of maize production in the USA (Corn Belt states). Depending on N application, the

CF ranges from 227 to 518 kg CO_2eq/t and N_2O is responsible for between 31 and 59% of the overall GHG emissions, whilst 63–97 kg CO_2eq per tonne of dry corn grain are associated with nitrogen fertilisation. Biswas et al. (2010) studied the CF of the main products of Australian agriculture. The CF for wheat is 0.40 kg CO_2eq/kg, but soil tillage as well as soil carbon sequestration were not taken into account in this study. Meisterling et al. (2009) assessed the CF of organic and conventional wheat in the USA; lower values are reported for organic agriculture (160 kg CO_2eq/t) than for conventional cultivation (190 kg CO_2eq/t) and N_2O emissions are the main GHG sources. Seed transport for 420 km involves an emission of GHG equal to 30 kg CO_2eq/t.

Ruini and Marino (2010) evaluated the CF for wheat cultivation in southern Italy. The CF is equal to 610 kg CO_2eq/t; the bigger contributions are caused by the production and use of fertilisers causing principally nitrous oxide releases. Diesel use is also an important contributor to the total. The CF of wheat cultivated in different cropping systems was evaluated by Gan et al. (2011b) for semi-arid climatic conditions in Canada. When wheat cultivation is followed by the cultivation of another cereal (on the same land area), the CF is higher (460 kg CO_2eq/t) than crop systems with legumes (200 kg CO_2eq/t) or canola (301 kg CO_2eq/t). The main hot spots identified are the production and application of N fertilisers, which account for about 57–65% of the CF, and crop residue decomposition (16–30%). Besides the choice of different cropping systems, in this study the other strategies and practices evaluated for lowering the CF include an improvement of N use efficiency, the increment of the Harvest Index and the improvement of crop residue management in farming systems. With the correct combination of these strategies, a CF reduction varying from 25 to 34%, depending on the soil conditions, can be achieved. A second study, aimed at evaluating the impact of different cropping systems on the CF of durum wheat, was conducted by Gan et al. (2011a) under the same climatic conditions. The total GHG emissions from the decomposition of crop residues along with various production inputs were used for estimating the CF. On average, the emissions from the decomposition of crop straw and roots accounted for 25% of the CF, those from the production, transportation, storage and delivery of fertilisers and pesticides to farm gates and their applications accounted for 43% of the CF and emissions from farming operations accounted for 32% of the total. Regarding the impact on the CF of different cropping systems, the authors report that durum wheat: (1) preceded in the previous year by an oilseed crop had a CF of 0.33 kg CO_2eq/kg of grain (7% lower than durum in a cereal–cereal–durum system); (2) preceded by a biological N-fixing crop the previous year lowered its CF by 17% compared with durum preceded by a cereal crop; (3) produced in a pulse–pulse–durum system had a CF of 0.27 kg CO_2eq/kg (34% lower than durum grown in cereal–cereal–durum systems). In addition, Ruini et al. (2013) assessed the CF of durum wheat cultivated after different crops. The trend is similar to the one highlighted by Gan et al. (2011a): the CF is higher when the wheat follows another cereal (580 kg CO_2eq/t), while the cultivation of durum wheat after vegetable (405 kg CO_2eq/t) or leguminous cultivation (380 kg CO_2eq/t) contributes to reducing the GHG significantly.

A CF study of bread was carried out by Espinoza-Orias et al. (2011) in the United Kingdom. In particular, the authors assessed the CF for a standard 800 g loaf of sliced bread, made of wheat flour on an industrial scale and consumed at home. Specifically, the CF of bread depends on the thickness of the slices, packaging and types of flour. For example, the CF ranges from 1.11 kg CO_2eq/loaf for wholemeal bread cut into thick slices to 1.24 kg CO_2eq/loaf for white bread cut into medium slices. For bread packaged in plastic bags, the results range from 0.98 kg CO_2eq/loaf for thick-sliced wholemeal bread to 1.10 kg CO_2eq/loaf for medium-sliced white bread.

A CF evaluation for durum wheat semolina dried pasta produced in Italy and packaged in paperboard boxes was carried out by Ruini and Marino (2010); a CF value of 1.284 kg CO_2eq for 500 g of pasta is reported. The main contributions to the CF are cooking (39 %), wheat cultivation (36 %), pasta production (13 %), grain milling (5 %), packaging (4 %) and transport (3 %). Röös et al. (2011) assessed the CF for Swedish pasta; they report, for wheat at the farm gate, CF values varying between 0.25 and 0.47 kg CO_2eq/kg wheat. The mean CF of 1 kg of Swedish pasta is 0.50 kg CO_2eq/kg (0.31 kg CO_2eq/kg wheat before the milling process). The main contributing processes are N_2O from soil (74 %) and fertiliser production and application (21 %).

When compared with wheat, only a few rice studies on CF evaluation have been carried out. Xu et al. (2013) assessed the CF for the five main rice-growing regions in China. In this study, the material and energy consumptions were estimated for these five regions using governmental statistical data, industrial standards and relevant technical data. The CF of rice production ranges from 2504 to 1344.92 kg CO_2eq/t. As possible mitigation strategies, the reduction of urea applications and intermittent irrigation are proposed in the paper. The CF of milled rice produced in Thailand and imported into Singapore was evaluated by Khoo et al. (2010). In this study, the authors compared different food products (beef, tofu, tomatoes and rice) in terms of protein content. The CF for milled rice is 219 kg CO_2eq/kg of protein; the methane emissions from paddy fields represent the main emission source of GHG.

Nalley et al. (2011) estimated the GHG emissions of the six largest row crops (corn, cotton, rice, sorghum, soybeans and wheat) produced in Arkansas using 57 different production practices. The CF estimation was carried out using a cradle-to-farm-gate LCA on a county-by-county basis. For rice, the CF value ranges from 2250 kg CO_2/ha (with conventional seeding and cultivation carried out in clay soils) to 2082 kg CO_2/ha (with no tillage and water seeded). For corn, the CF ranges from 640 kg CO_2/ha in furrow clay soil to 533 kg CO_2/ha in loamy soil, while for wheat, the CF shows a higher value (318 kg CO_2/ha) when the cultivation takes place after rice and a lower value (269 kg CO_2/ha) in sandy or silt soil after the cultivation of other cereals. Eshun et al. (2013) assessed a rice CF equal to 477 kg CO_2/ha in Ghana with 'cradle-to-national-retailer' system boundaries.

Furthermore, Yoshikawa et al. (2010, 2012) evaluated the CF of rice in Japan. In the first study, the carbon footprint of ecologically cultivated rice was evaluated. The functional unit in this study is a 4 kg package of polished rice. The system boundary includes raw material production, rice polishing, distribution and retailing, rice cooking and waste treatment. Environmental loads related to durables (agricultural equipment, facilities, cooking equipment, etc.) are not included because of uncertainty regarding their durability. The results show that the carbon footprint of rice is 7.7 kg CO_2eq/package (for 4 kg of polished rice, which amounts to 1.925 kg CO_2eq/kg). About 65 % of the emissions are related to the raw material production stage; almost all the emissions derive from agricultural production. CH_4 emission from paddy fields is caused by anaerobic fermentation and accounts for 50 % of the CF from agricultural production. Besides CH_4 emission, the emission of GHGs from fertiliser, energy and the transportation of input materials each accounted for more than 5 % of the CF in the agricultural phase. In the second study, the assessment was carried out considering two types of cultivation: one with reduced use of chemical fertiliser and another in which green manure is utilised. The assessment is carried out considering 'cradle-to-factory-gate' boundaries and the FU is the mass of white rice. The CF is 2.25 kg CO_2eq/kg for polished rice with cultivation that involves chemical fertiliser reduction (73, 12 and 8 % of the life cycle GHG emissions are due to field emissions—CH_4, N_2O, fertiliser production, and fuel and electricity, respectively), while the CF is equal to 4.89 kg CO_2eq/kg for polished rice when cultivation is carried out using green manure (the CH_4 emissions—2.8 times higher than rice cultivation with mineral fertilisers—represent about 80 % of the global GHG emissions). The impact of different forms of tillage management on the rice CF in Japan was studied by Harada et al. (2007). In particular, scenarios for conventional puddling and no-tilling rice cultivation were compared. The CF from the no-tilling field is 1741 kg CO_2/ha, lower than that from the conventional puddling field. In conclusion, considering that the fuel consumption is also lower, the authors state that no-tilling rice cultivation has the potential to save 1783 kg CO_2/ha from paddy fields. Breiling et al. (2005), for Japan, estimated the CF of rice considering not only direct rice-related GHG emissions but also GHGs hidden in the other categories, primarily energy, industry and waste. The study highlighted that since 1990 the GHG emissions in rice production have been reduced, but this reduction has been offset by the increase in other sectors.

4.2.5.3 The Water Footprint (WF)

Cereal cultivation, in particular maize and rice, involves the consumption of high water volumes. Irrigation is essential to reach good production levels and it can help to stabilise the yields. Moreover, in the case of flooded paddies, water plays an important role in thermoregulation, allowing cultivation in temperate areas characterised by a cold spring as well. For spring wheat cultivation, irrigation is not usually carried out, but water is needed for the processing operations and, in particular, for pasta production. In recent decades, the water availability, for agricultural use as well as for industrial processes, has decreased due to climate change. Consequently, the importance of water footprint (WF) assessment has greatly increased and several studies have been carried out for the evaluation of water consumption during the life cycle of food products, as well as for the identification of the hot spot processes. Generally, for WF assessments for which only the cultivation system is considered ('from-cradle-to-farm-gate' boundary), more attention has been paid to rice than to maize and wheat.

Aldaya et al. (2010) carried out a WF evaluation for rice cultivation in Asia and they obtained a WF equal to 2600 m^3/t in Kazakhstan, 3500 m^3/t in Uzbekistan and 4000 m^3/ha in Tajikistan; lower WF values refer to clay soils and arid conditions. Chapagain and Hoekstra (2011) assessed the WF for the 13 major rice-producing countries. Although an average value of 1325 m^3/t was attained, the study highlights pronounced differences between different countries: rice production in Pakistan shows the highest WF (2874 m^3/t), followed by India (2020 m^3/t) and Thailand (1617 m^3/t), while the lowest values are reported for Vietnam (638 m^3/t), Japan (802 m^3/t) and the USA (829 m^3/t). Besides big differences in WF values, the share of green, blue and grey water also varies greatly; in India, Indonesia, Vietnam, Thailand, Myanmar and the Philippines, the green water fraction is substantially larger than the blue one, whereas in the USA and Pakistan, the blue water footprint is four times higher than the green component. The WF of rice cultivated in Asia has also been evaluated by Yoo et al. (2013). Specifically, this study refers to rice cultivation in South Korea. The WF of rice is 844.5 m^3/t, and green, blue and grey water accounts for 294.5, 501.6 and 48.4 m^3/t, respectively.

McConkey et al. (2012) evaluated the WF for maize and wheat cultivation in Canada. In semi-arid conditions, maize cultivation with irrigation has a WF equal to 3310 m^3/t. In these areas, the spring wheat shows a WF ranging from 4110 m^3/t, with irrigation, to 19,200 m^3/t when cultivation takes place on summer fallow without irrigation. In sub-humid and humid areas, the maize is not irrigated and it shows a WF equal to 5540 m^3/t (sub-humid conditions) and 7290 m^3/t (humid conditions), while the WF values for spring wheat range from 10,500 to 19,600 m^3/t. Generally, the grey water, computed following the Canadian environmental objectives in terms of P concentration, represents about 80 % of the total WF. Huang et al. (2012) compared the WF of wheat and maize in China's main breadbasket basins. The authors report remarkable differences for wheat cultivated in the different regions (from 1262 to 31 m^3/t). The water footprints of maize range from 35 to 515 m^3/t. Ruini and Marino (2010) compared the WF of durum wheat cultivation in Italy with that of other countries. For Italy, the WF values are 450, 920 and 1100 m^3/t, respectively, for northern, central and southern regions. These values are higher than the WF for durum wheat in France (450 m^3/t), but they are lower than the ones obtained for other European countries (Spain 1400 m^3/t, Turkey 1520 m^3/t and Greece 1220 m^3/t) as well as for the northern USA (2230 m^3/t) and Australia (2750 m^3/t). For pasta, the authors report a WF equal to 0.7 m^3 per 500 g of product.

Compared with rice, few wheat WF studies, relative to the cultivation phase, have been carried out, but, unlike the case of rice WFs, some papers analyse in particular detail the processing steps needed for derived products, such as pasta and bread. Aldaya and Hoekstra (2010) analysed the WF related to pasta and pizza margherita. They report a WF equal to 1574 m^3/t for durum wheat (748, 525 and 301 m^3/t, respectively, for green, blue and grey water). For pasta, the durum wheat grains need to be processed into flour. Considering a semolina yield of 72 % (the rest consists of the wheat bran and germ) and that the semolina constitutes 88 % of the total value of the separate products, the authors calculated the WF of semolina to be 1924 m^3/t and that of pasta (assuming that it is made from semolina (1 kg), water (0.5 dm^3) and salt) to be 1924 dm^3/kg. The authors also report a WF of bread wheat of 786 m^3/t and a WF of bread wheat flour of 961 m^3/t. Finally, the reported WF of pizza margherita is 1216 dm^3/kg.

4.2.6 Product Category Rules (PCRs) and Environmental Product Declarations (EPDs)

The European Union (EU) promotes environmental strategies and policies oriented towards the development of a European market characterised by an exchange of greener products. As such, as illustrated in the first chapter, one of the most important actions is the development of life cycle assessment (LCA) based environmental labels, on one hand stimulating producers to improve the environmental performance of their products and on the other allowing consumers, with their choices, to privilege the market for more ecological products.

One of the most interesting types of LCA-based environmental labels is the Environmental Product Declarations (EPDs) and their relative Product Category Rules (PCRs). They are considered complementary to the general requirements of the EPD programmes and they form the basis for third-party verification of LCA studies on the products and the related statements.

The current EDP systems in the agri-food sector, which use the type III programme according to the requirements of the ISO standard 14025:2006, are the International EPD® System (EU), EPD Norge (EU), Earthsure® (USA), the Sustainability Measurement and Reporting System (SMRS) (the USA) and the Ecoleaf environmental label (Japan).

Focusing on the cereal sector, due to its wide variety, the field of application of this chapter is restricted to three main cereals: wheat, rice and corn. Among these, wheat is characterised by a higher number of PCRs and EPDs because, even though it has quite a limited area of production interest in the world, there is a growing market demand in new geographical areas, especially for wheat-derived products like pasta and bread. The following paragraphs illustrate the PCRs published in the section on food products, basic module grain mill products, starches and starch products and other food products, with their related EPDs, with regard to the International EPD® System (Environdec 2014), which is the most widespread scheme among the ones mentioned above:

1. PCR 2010:01 (CPC 2371): Uncooked pasta, not stuffed or otherwise prepared. The EPDs based on this PCR are:
   - developed by the company Lantmännen—Kungsörnen spaghetti, Kungsörnen Macaronis ‘Gammaldags Idealmakaroner’, Kungsörnen Ideal Macaroni in bulk packs, Kungsörnen pasta in bulk packs, Kungsörnen wholegrain pasta in bulk packs, Kungsörnen white fibre in bulk packs;

- developed by Barilla—dry semolina pasta from durum wheat EPD, which covers the classic semolina pasta cuts (spaghetti, penne, fusilli, etc.), piccolini (miniatures of classic semolina cuts) and specialità (reginette, orecchiette, ruote, etc.);
- developed by De Cecco—pasta di semola De Cecco EPD, which includes a traditional range manufactured in cello bags (spaghetti, penne, fusilli, etc.) and the specialities (farfalle, zita, etc.);
- developed by Sgamboro—Pasta Sgamboro Etichetta Gialla.

2. PCR 2011:07 (CPC 2372): Pasta, cooked, stuffed or otherwise prepared; couscous—for this PCR no EPDs have been actually published;
3. PCR 2012:06 (CPC 234): Bakery products—this PCR incorporates the Product Category Rules 2010:05 (CPC 2349): Bread and other bakers' wares valid until 9 March 2013 and the Product Category Rules 2010:06 (CPC 2343): Pastry goods and cakes valid until 9 March 2013. Several EPDs, based on the abovementioned deregistered PCRs, are valid until the end of the year 2014, and they are not presented because they are not relevant to the research topics. All the products in the CPC Group 234 'Bakery products' are included in this PCR, especially the following classes:
   - CPC 2341: Crispbread; rusks, toasted bread and similar toasted products
   - CPC 2342: Gingerbread and the like; sweet biscuits; waffles and wafers
   - CPC 2343: Pastry goods and cakes, fresh or preserved
   - CPC 2349: Other bread and other bakers' wares

   The EPDs based on this PCR are:
   - developed by Barilla—Mulino Bianco Pan Bauletto Bianco, Mulino Bianco Fette Biscottate dorate, integrali, malto d'orzo e cereali, Muli-no Bianco Tarallucci, Mulino Bianco Girotondi, Mulino Bianco Bat-ticuori, Cracker Gran Pavesi, Ringo Pavesi, Mulino Bianco Flauti, Mulino Bianco Plumcake, Petit Pavesi, Mulino Bianco Pagnotta di Gran Duro, Mulino Bianco PanCarrè, Grancereale classico alla frutta, Harrys American Sandwich Complet, Harrys American Sandwich Nature, Harrys Brioche Tranchée, Harrys Extra Moelleux, Mulino Bianco Granetti, Mulino Bianco Saccottini, Mulino Bianco Michetti, Mulino Bianco Pan Goccioli, Pan di Stelle, Mulino Bianco Cracker salati e non salati;
   - developed by Lantmännen—Kungsörnen pancake with diced pork, Kungsörnen potato pancakes (raggmunk), Lantmännen batter pudding (ugnspannkaka).

4. PCR 2013:04 (CPC 231): Grain mill products—this PCR covers products belonging to the UN CPC Group 231 'Grain mill products' and replaces PCR 2010:03 (CPC 2313): Groats, meal and pellets of wheat and other cereals, which expired on 29 April 2013.

   This group includes the following CPC classes:
   - 2311—Wheat and meslin flour
   - 2312—Other cereal flours
   - 2313—Groats, meal and pellets of wheat and other cereals

- 2314—Other cereal grain products (including cornflakes)
- 2316—Rice, semi-or wholly milled, or husked
- 2317—Other vegetable flours and meals
- 2318—Mixes and doughs for the preparation of bakers' wares

The EPDs based on this PCR (CPC 2313) are:

- developed by the company Lantmännen—Axa oatmeal 'Havre Gryn', Kungsörnen plain flour, Kungsörnen oat berry, Kungsörnen pearled barley, Kungsörnen wheat berry, Kungsörnen pearled barley in bulk packs, Kungsörnen wheat flour with whole grain, Kungsörnen graham flour, Kungsörnen plain flour in bulk packages, Axa oatmeal in bulk packs.

The above-mentioned EPD list highlights the dominance of the Lantmännen and Barilla companies as the main EPD developers. Lantmännen, part of the Lantmännen Cerealia team, develops, manufactures and markets mainly cereal-based products under strong brands such as AXA, Golden Eagle, Home, Gyllenhammar, Gooh, GoGreen, Soups, Amo, Kornkammeret and Regal. The range consists mainly of breakfast foods, flour, flour mixes, pasta, pancakes, beans/lentils and dishes that are sold in grocery stores in northern Europe. In addition, Barilla occupies a representative position in terms of experience in EPD development: it is the first private company to have developed an EPD Process System. This company is one of the top Italian food groups, producing more than 100 products in about 50 plants around the world. The company has been using the LCA methodology for more than a decade. Since 2008, life cycle thinking has made its way into the company strategy, as an instrument to study the production chain thoroughly and localise the most substantial environmental impacts. Moreover, Barilla, at this moment, developing 56 % of the above-mentioned EPDs, could be considered as a guide in this field.

Indeed, Barilla defines a common system process according to its experience, the 'funnel process', which, in three main steps, represents an internal standard to develop PCR–EPDs, gathering, aggregating, analysing and processing the data, to reduce them to more manageable results. Specifically, such steps are:

- data collection and management—the identification and gathering of product-specific information regarding the product recipe (the amount of food raw materials per unit of product), the bill of materials packaging list, the production plants in which the product is manufactured and the related production volume, logistic distribution data for the finished product and other relevant environmental aspects;
- data processing—elaboration of the product system using the LCA database distinguished in data module groups (raw materials, packaging raw materials, energy, plants and transport). This step occupies a central role in PCR–EPD elaboration because PCRs describing the product category include requirements for the LCA that provides the basis for an EPD: the functional unit, system boundaries, cut-off rules, allocation rules, data quality requirements and indicators. All the data modules are internally verified and are ready to be used for EPD purposes;
- result management—the product group model calculation tool carries out a collection of the results in a specific LCA data sheet, which is developed for each product group in a specific fashion following the PCR and is internally vetted.

The reliability of the EPDs is ensured by several verification levels, carried out internally by the Data Assessor and Process Assessor and externally by the Verification Body. Internal verification is applied in all three steps in a continuative way, in order to verify the LCA calculation and maintain conformity. Indeed, internal assessments, at planned intervals, are conducted to determine the reliability, relevance and independence of the EPD.

The Verification Body, an external auditing body, represents an accredited body certified for auditing management systems that verifies the entire EPD process system.

The use of the Barilla EPD Process System has shortened the EPD publication time, which now lasts about 8–10 weeks.

The EPD represents an environmental success action. From the EPD process results (i.e. by LCA calculation of semolina pasta), Barilla has achieved a mix of environmental objectives; the company has obtained a reduction in GHG emissions by acting on the phases of cultivation of durum wheat and pasta production. The result is due to a combination of actions:

- the rotation of cultivation and the careful use of fertilisers, changing the production rules, avoiding −55 % of the GHG emissions (390 kg of GHG per tonne of produced durum wheat);
- the increase in the proportion of recyclable packaging from 92 to 95 %;
- the rationalisation of logistics with the optimisation of the transport saturation avoiding—8 % of GHG emissions;
- the reduction of water recommended for cooking, from 1 to 0.8 L of water per 100 g of pasta, avoiding—5 % GHG emissions;
- the improvement in the efficiency of the energy management systems in factories, introducing a CHP (combined heat and power) plant in pasta production, using renewable energy, avoiding −13 % of GHG emissions.

In conclusion, the analysed PCRs do not present relevant differences; given the amount of information, the following list summarises the main differences:

- in the functional unit section, PCR 2012:06 (CPC 234) defines 1 kg of product (as do the others), but the packaging weight is not included in the kg of product;
- in the core process section, only PCR 2010:01 (CPC 2371) is limited to the product production (pasta in this case);
- in the downstream processes section, not only the PCRs include the distribution (e.g. the PCR 2010:01 (CPC 2371));
- in the allocation section, all the PCRs allow partitioning with the allocation by mass, but PCR 2010:01 (CPC 2371) underlines that the products that are not compliant with the quality requirements and are destined for other chains (such as animal food) must be considered waste.

Instead, all the above-mentioned PCRs, in order to communicate and compare the environmental performances of different products, in the additional environmental information section (Sect. 10.4) and annex 1–2, include in the LCA report some additional optional indicators, widening the EPD scope (e.g. in the case of Barilla) in terms of the ecological footprint, carbon footprint and virtual water content (e.g. PCR 2010:01 (CPC 2371) introduces the ecological footprint, the water footprint, land use, land use change and forestry, marine water eutrophication and aquatic ecotoxicity indicators). However, comparability remains a critical factor for the EPDs (Schau and Fet 2008); hence, it is best to compare only products that are similar to each other and that are within the same class included in each PCR. Finally, on the basis of the foregoing analysis, the LCA–PCR link seems to be a critical aspect that still needs a wide and deep action for harmonisation from a global perspective.

4.3 Review of the Methodological Issues in the Cereal Sector

4.3.1 Definition of the Functional Unit (FU)

The main applications of LCA in the cereal sector (maize, wheat and rice) have been devoted to different goals: identifying the environmental hot spots in production systems' performance, profiling the environmental burden of production in a given area, comparing the environmental burden of different food products and different farming practices, as well as evaluating the environmental properties of a supply chain.

As a consequence, different LCA studies of the same product systems can have different functional unit (FU) definitions, making the choice of the FU very controversial (Reap et al. 2008). In fact, as mentioned in the first chapter, this can lead to different or even contrasting LCA results. Ideally, an LCA with multiple FUs can provide a better picture of the sustainability of the product systems under assessment and at the same time make the study more comparable with others.

In most of the studies reviewed in the current chapter, the FU is based on mass: about 31 % of the reviewed papers use 1 kg of the investigated product at different stages of the value chain. For example, for the agricultural stage, 1 kg of dry corn grain or milled rice at the mill gate is considered. If the LCA considers the final consumption stage, the FU can be that of 1 kg of pasta, in primary and secondary packaging, delivered to customers, 1 kg of bread ready for consumption, 1 kg of refined rice packed and delivered to the supermarket, 1 kg of short pasta, 1 kg of pasta or a 725 g pizza margherita.

In some cases, the FU includes quality aspects, such as the energy content of the final product. Roy et al. (2009b) express the FU in MJ of energy supplied by different forms of cooked rice to enable the comparison among them. In such a context, other FUs are the protein content per food (Khoo et al. 2010), the content of glucose and fructose (Renouf et al. 2008) or dry biomass production (Kim and Dale 2002).

Over 14 % of the reviewed studies use mass as the FU but refer to metric tons (Brentrup et al. 2004; Drocourt et al. 2012; Fallahpour et al. 2012; Kissinger and Gottliebb 2010; Kløverpris et al. 2010; McConkey et al. 2012; Muñoz et al. 2012;

Roy et al. 2005, 2007; Ruini and Marino 2010; Salomone and Ciraolo 2004; Williams et al. 2010). Some studies, mainly dealing with derived cereal products, use different FUs: 1 loaf for bread; 1 hl for beer; 1 L for canola cooking oil (Narayanaswamy et al. 2004); 1 t of grain; or 1 t of grain with constant quality (13 % protein in dry grain (Charles et al. 2006)). Nemececk et al. (2008) define three FUs, one for a land management function (i.e. hectares/years), one for a financial function (gross margin 1 in €) and finally one for a productive function (MJ gross energy of the product).

Another commonly used FU is based on land surface area: Van Stappen et al. (2012) adopt any useful output per hectare in an average year and illustrate the competition for land between food and non-food products; Murphy and Kendall (2013) choose 1 ha of corn and stover production; Eshun et al. (2013) and Ferng (2011) use 1 ha; Seda et al. (2010) use land surface area together with yield and economic benefit; Harada et al. (2007) use 60 m^2; and Nalley et al. (2011) use acres.

Some studies perform the LCA according to different FUs. Seda et al. (2010) choose the land area (ha), yield (tonnes) and economic benefit (€) as a basis for comparing the environmental impacts of cereals and horticultural crops. The land area functional unit provides explicit information on the intensity of use of agricultural inputs. Yield as an FU is a reflection of agricultural activity as a producer of market goods, and it can be used to evaluate the effect of cultivation techniques on yield (e.g. different rates of fertilisation). The study also includes a cost–benefit analysis in order to define the eco-efficiency concept better, which is the management philosophy encouraging business to search for environmental improvements that obtain parallel economic benefits. In general, when using land surface area, the impacts of horticultural products are higher. The cost–benefit analysis reveals that the economic benefit of the horticultural crop alternative is seven times higher than that of cereals. From this case study, it can be concluded that horticultural crops would be a suitable choice based on productivity and economic terms (the weight of the product or the economic benefit were used as functional units). The differences could be attributed to the higher yield and retail prices of horticultural crops in comparison with cereals. On the contrary, when land area is used as an FU, the cereal crops tend to be more sustainable.

Roer et al. (2012) illustrate the importance of carefully selecting the functional unit, choosing 1 kg of cereal dry matter as the FU and then performing a sensitivity analysis in terms of energy content (MJ), protein content (kg) and area occupied (ha). Even though the choice of the FU does not change the overall rating of the included cereals, the relative differences change. Furthermore, despite the widespread use of kg dry matter, this unit is not always a good measure of the quality of the food produced; the energy and protein content can indeed, as reviewed above, be more appropriate. The situation changes completely if area is used as the functional unit, as in this case more intense management per ha can overshadow the higher productivity. This factor underlines the need to be very specific regarding the motivation and scope of the study and thus the selection of the functional unit, which should be goal-driven. For example, in the case of the selection of alternative agricultural production systems, Hayashi (2013) recommends the definition of decision criteria rather than trying to make decisions on the basis of multiple functional units. The author shows that a comparison based on the functional units is not fair because the product information (yield) is not contained in impacts per area unit. When decision criteria are introduced, two aspects need to be considered: impacts per area unit (which have to be minimised) and yield per area unit (to be maximised). The ratio of the former to the latter leads to the definition of impacts per product unit, which can be recognised as an integrated upper-level criterion (Hayashi 2013).

4.3.2 System Boundaries and Cut-Off Criteria and End-of-Life Aspects

Among the reviewed studies, more than 61 % explicitly specify the system boundaries. To outline better the definition of the system boundaries of cereal-related life cycle studies, the reviewed studies have been subdivided into five categories, according to the cereal considered: wheat, rice, maize, derived products and a combination of these.

The authors who have dealt with wheat as a case study all start at the wheat cultivation phase, apart from Röös et al. (2011), who also consider the seed planting stage. Charles et al. (2006), Muñoz et al. (2012), Ruini and Marino (2010), Ruini et al. (2013) and Schmidt (2008) stop at the stage of cultivation, while Bevilacqua et al. (2007) and Salomone and Ciraolo (2004) consider all the stages of the production cycle (processing, packaging, transportation, distribution, consumption and disposal), and Röös et al. (2011) stop at the distributive stage. Brentrup et al. (2004) and Meisterling et al. (2009) consider the extraction of raw materials, production and transportation of input (fertiliser, pesticides, machinery, fuel).

Only for 56 % of the studies on rice it is possible to analyse the process coverage, and the definition of the system boundaries, when specified, is very different: for example, Blengini and Busto (2009) consider agricultural processes, drying and storing, and refining and packaging; Roy et al. (2009b) include the cultivation, processing and distribution of rice produced and consumed; Khoo et al. (2010) take into account land use, cultivation, harvesting, milling, drying, refining and storage, and transportation to a national retailer; and Hokazono and Hayashi (2012) define the boundaries in a generic manner: ‘all farm-level and upstream processes of rice production in the paddy fields’.

Studies that analyse wheat-derived products (mainly pasta and bread) have a greater process coverage and include all the stages of production, packaging, transportation, distribution and consumption of the processed product, including planting and cultivation, and in some cases they also consider the stage of waste disposal (Bevilacqua et al. 2007; Espinoza-Orias et al. 2011; Salomone and Ciraolo 2004).

The studies on corn mainly focus on the agricultural phase and consider farming operations (such as soil tillage, seedbed preparation, sowing, fertiliser and pesticide application, harvesting, collection and collection of stover). Regarding the studies focusing on the analysis of a combination of products, out of a total of 19 studies, 5 do not indicate the system boundaries; of the 14 remaining, some consider only the cultivation phase (Brankatschk and Finkbeiner 2012; Fallahpour et al. 2012; Gan et al. 2011b; Pelletier et al. 2008; Zhang et al. 2013). Nalley et al. (2011), in addition to farming, also consider the production of inputs; others (Huang et al. 2012; McConkey et al. 2012; Roer et al. 2012; Seda et al. 2010) consider only plantation and cultivation; and Biswas et al. (2010) and Narayanaswamy et al. (2004) also consider storage, processing, retail, consumption and transport.

In terms of the overall main boundaries and the cut-off of some stages of the life cycle under analysis, 40 % of the studies reviewed in this chapter implement 'cradle to farm gate' as the main boundaries; about 9 % consider 'cradle to consumption'; over 6 % 'cradle to grave including end of life'; 5 % 'cradle to factory gate (end of transformation)'; 4 % 'cradle to national retailer'; and two studies chose 'cradle to international retailer'. Breiling et al. (2005) delimit the analysis 'from land preparation to harvesting', whilst Settanni et al. (2010b) delimit the analysis by 'entry gate to output gate of factory'. Geographical boundaries are specified in over 65 % of the reviewed studies, and time boundaries in only 28 %. The choice of geographical boundaries should be consistent with the system boundaries of unit process data sets, because of the critical issues that may arise due to the fact that some phases are carried out within the geographic boundaries of the country indicated, while other phases (e.g. sales, use, treatment and waste) can take place in other areas.

The inclusion or exclusion of process units in the system boundaries is a subjective choice, which can be relevant to the outcomes of an LCA, even in the case of cradle-to-farm-gate analysis. The lack of data, which is one of the constraints of LCA, often contributes to spreading the tendency towards simplification (see Sect. 4.2.4), e.g. excluding the contribution of some inputs, such as capital goods (machinery and buildings), which can, in certain circumstances, contribute significantly to the total impact of the production systems (Blengini and Busto 2009; Roer et al. 2012). Furthermore, the contribution of the production and use of pesticides, mineralisation in humus and nitrogen oxides from mineral fertilisers is usually neglected, with consequent underestimation of the actual total environmental impact, but as demonstrated by Roer et al. (2012), all of this has a significant environmental impact in the cereal sector.

The end of life is a relevant step in the life cycle of agricultural products, but in the specific case of cereals and their derivatives, it assumes a marginal role, as cereals are usually used for human consumption or as raw materials for the manufacturing of other products. The end of life is considered in the case of packaging materials, e.g. rice (Kägi et al. 2010), bread (Espinoza-Orias et al. 2011) or, in the case of the Product Category Rules, e.g. bakery products, cooked and uncooked pasta (see Sect. 4.2.6).

Most of the studies investigated in this chapter adopt an attributional LCA: in this case, the life cycle of the system is modelled as it is and the principal system boundaries and included life cycle stages can be derived from the goal and scope of the study. When consequential modelling is considered, processes of other systems (other than those specifically assessed) are to be included in the system boundary of the analysed system. Schmidt (2008) presents a framework for defining system boundaries in consequential agricultural LCA using wheat production as a case study. He also argues that the proposed methodology contributes to increased completeness of the identification of the processes actually affected.

4.3.3 Criteria for the Allocation of Multifunctional Processes

Allocation is a crucial issue in LCA studies, because the uncertainty of LCA results is largely dependent on the methodological choices related to allocation criteria (Brankatschk and Finkbeiner 2012; Curran 2008; Gnansounou et al. 2009; Kim and Dale 2002). In the cereal production sector, the allocation problem is particularly relevant because this type of production almost always implies cultivation systems that produce multiple products (co-products or by-products) by rotating crops or processing diverse parts of a plant for different uses (e.g. for food or energy purposes).

By analysing the scientific papers included in the state-of-the-art analysis for the cereal sector presented in Sect. 4.2 (see Table 4.2), some elements regarding allocation methodologies and criteria can be highlighted.

First of all, it should be highlighted that three papers specifically investigate the allocation issue, presenting different criteria to face the problem with applicative examples in the cereal sector:

- In Kim and Dale (2002), the authors focus on the ethanol production system from corn grain and present a study in which allocation is avoided through system expansion. In order to avoid the allocation procedures completely, five mutually interdependent product systems were required (ethanol production from corn dry milling and corn wet milling, corn grain production, soybean products from soybean milling, urea production). The system expansion approach is equivalent to assuming that the environmental burdens associated with ethanol from dry milling are equal to those associated with ethanol from wet milling. This approach is interesting because it can be used to compare the environmental burdens associated with ethanol with those associated with petroleum-based fuel as well. However, the proposed approach would not work for an LCA study aiming to compare the environmental burdens between different ethanol production technologies;
- In Brankatschk and Finkbeiner (2012), the authors demonstrate the benefits deriving from the use of the 'cereal unit' (CU) as an allocation procedure in LCA studies of agricultural systems, presenting a comparison of different allocation methods (cereal unit, mass allocation, energy allocation and economic allocation) for different agricultural products (barley, soybeans, sugar beet plant, wheat plant, sunflower plant, rapeseed plant and rape seeds). The CU is a common denominator that could be used for evaluating agricultural products and by-products based on the feeding value of agricultural products. The results highlight that the application of CU allocation could reduce the variability and potential bias in the LCA results of agricultural systems;

- Murphy and Kendall (2013) explored three different approaches to allocation for corn and stover: economic allocation, energy-based allocation and a subdivision approach, which assigns to stover only those additional activities caused by its harvest. For most indicators, subdivision produces impacts approximately equal to those of economic allocation. Both economic allocation and subdivision assign lower impacts to stover than energy allocation. No definitive conclusions on which allocation criteria to be preferred are defined in the paper, but the authors argue that in the long term, once commercial production systems and associated markets are established, economic allocation may be preferable, while for current LCAs of stover production, the most reasonable approach could be that of using a range of values based on multiple allocation methods.

Of the remaining scientific articles (taking into account only the case studies of cereals used for food purposes), it should be noted that only eight of them clearly report whether allocation was applied, also specifying the methods used (for which the economic allocation is the most frequent); how allocation was treated in these studies is briefly presented in the following paragraphs.

In Blengini and Busto (2009), the LCA methodology is applied to the rice production system, from the paddy field to the supermarket. Rice production generates different marketable products and by-products (refined rice, broken grains, rice flour, husk straw, etc.) for which the allocation of burdens to the co-products was based on relative economic value, as suggested by Williams et al. (2005) (which is the same method as that used by Williams et al. (2010), as reported below and in footnote 1).

Kasmaprapruet et al. (2009) present an LCA analysis of milled rice production, from rice cultivation to the mill. The allocation step was performed based on economic allocation from which resulted the following allocation of environmental burdens: 51 % to milled rice, 27 % to broken rice, 20 % to rice bran and 2 % to rice husk.

Biswas et al. (2010), using LCA methodology, compared the life cycle global warming potential of three important Australian agricultural productions (wheat, meat and wool), including two major life cycle stages: pre-farm and on-farm. In order to calculate the inputs and outputs of the co-products, the authors chose to apply an economic allocation method in which the allocation factors to partition the greenhouse emissions to the various products (wool, sheep meat and wheat) are derived using the ratio of market value for those products; the method used was based on Guinée et al. (2004).

Hokazono and Hayashi (2012) present an LCA of three rice production systems in Japan: organic, environmentally friendly and conventional. The allocation procedure was applied to brown rice and rice ducks (a by-product of paddy fields sold as poultry in small markets at relatively high prices) following economic allocation criteria (approximately 10 % of the impact was allocated to rice ducks, which varied from 8.1 to 10.4 % depending on the rice yields). Allocation was also conducted between white rice and rice bran (both obtained by polishing brown rice), again using economic criteria (99.6 and 0.4 % of the impact were allocated, respectively, to white rice and rice bran since the economic value of rice bran is much lower than that of white rice).

Espinoza-Orias et al. (2011) estimated the carbon footprint of bread produced and consumed in the UK. In this case, allocation problems arise in the wheat milling stage, which co-produces flour, wheat germ and bran. The authors decided, in the absence of data to perform system expansion, to face allocation using an economic value approach (suggested both by PAS 2050 and by ISO 14,044). The result is that GHG emissions deriving from the wheat milling stage were allocated 88 % to white flour, 92.5 % to wholemeal flour and 90 % to brown flour.

Williams et al. (2010) describe the production burdens of three organic and non-organic arable crops (bread wheat, oilseed rape and potatoes). For the specific case of wheat (which is grown for bread making), the burdens were allocated between the bread and the feed fractions according to their economic value.[1]

In Notarnicola and Nicoletti (2001), a comparative LCA between pasta and couscous is presented. Two different allocations were performed: one referring to the stage of agricultural production, from which grain and straw are obtained, and the other related to semolina production, from which flour, bran and fodder grain are obtained. The allocations were made with a combination of economic and mass criteria by applying the following formula:

$$A = \frac{q_i \text{ x } p_{ui}}{\sum q_i \text{ x } p_{ui}}$$

where

A = economic factor of allocation;
q_i = mass allocation factor;
p_{ui} = relative price.

Also in Notarnicola et al. (2004), the allocation problems related to the co-production of durum wheat and straw or semolina, pollard, millfeed, screenings and germ were solved on the basis of the relative quantities and marker prices.

It should be pointed out that the Product Category Rules (PCRs) published for this sector also suggest different allocation rules depending on the specific type of product, in particular:

- the PCRs on 'grain mill products', 'bakery products', 'pasta, cooked, stuffed or otherwise prepared and couscous' and 'uncooked pasta, not stuffed or otherwise prepared' suggest the use of mass allocation when the inputs and outputs of the system should be partitioned between the different products or functions;

[1] 'The total burdens of producing grain and straw are: $T = H + (1 - p_s) I + p_s B + D$. Then the burden allocated to grain is: $G^* = (H + I)\ (Y_g / (Y_g + v_s p_s Y_s)) + D$, and the burden allocated to straw is: $S^* = \frac{(H + I)(v_s p_s Y_s)}{(Y_g + v_s p_s Y_s)} + p_s (B - I)$ where H is the vector of burdens of producing grain up to the end of combine harvesting per hectare, I is the vector of burdens of chopping for incorporation for all straw produced, D is the vector of burdens of drying and storage of grain, B is the vector of straw baling burdens for all straw produced, p_s is the proportion of straw baled and harvested, Y_g is the net yield of grain per hectare at standard DM content, Y_s is the yield of straw per hectare (whether harvested or not) at standard DM content, and v_s is the relative value of the straw prior to baling versus the grain, typically 0.05' (Williams et al. 2010).

- the PCRs on 'bread and other bakers' wares', 'groats, meal and pellets of wheat and other cereals' and 'pastry goods and cakes' report that 'allocation between different products and co-products shall be based on economical allocation'.

To avoid allocation, as recommended by the ISO requirements, system subdivision or system expansion should be implemented when possible. System subdivision means dividing the unit process to be allocated into two or more sub-processes and then collecting the input and output data related to these sub-processes (ISO 2006), while system expansion means that the system boundary is expanded in order to include the displacement of substitute products (the co-products) in the market, which will generate environmental credits due to the avoided production of displaced products (Ekval and Weidema 2004).

Of the papers reviewed in this chapter, system subdivision was implemented by Murphy and Kendall (2013) (as reported above), while system expansion was specifically mentioned only by Renouf et al. (2008), in which an LCA of sugar-cane production and processing in Australia is presented and this system is then compared with other sugar-producing crops (US corn and UK sugar beet). Among the various conclusive remarks stressed by the authors of this case study, it should be noted that they state that a crop's agronomic characteristics can influence its environmental performance and one of the main characteristics is the nature and quantities of co-products deriving from crops, which can displace other products in the markets, giving environmental credits.

All the other case studies included in the state-of-the-art analysis do not mention allocation criteria in any way, except for two cases (Petti et al. 2000; Schmidt 2008), in which the authors specify that, for simplification reasons, all the burdens were allocated to the main crop/product, and one paper (Nemececk et al. 2008), in which it is specified that only allocation for shared infrastructure (machinery and buildings) was performed (in particular following the procedures described by Nemecek and Erzinger (2005) and Nemecek and Baumgartner (2006).

4.3.4 Data Availability and Quality

Data availability and data quality are one of the main problems that LCA practitioners face when developing an LCA study; the significance of the problem is also demonstrated by the fact that the Society of Environmental Toxicology and Chemistry (SETAC) has set up a working group on this specific topic (SETAC LCA Working Group on Data Availability and Data Quality—Bretz 1998) and many other initiatives, at the national and international levels, have been initiated to deal with this issue.

LCI data availability is particularly significant in some specific industrial and productive sectors, such as agri-food, in which there is still a lack of complete and reliable data for many processes and kinds of food.

Concerning the cereal sector, it can be highlighted that, starting from the state-of-the-art analysis for the cereal sector presented in Sect. 4.2 (see Table 4.2), only three papers specifically investigate the problem of data availability:

- In Notarnicola et al. (2004), LCA and IO-LCA are applied to the pasta life cycle in order to verify whether the adoption of these two tools could improve the quality of the inventory set-up. In particular, in the study, two IO-LCA approaches are considered: the Economic Input–Output Life Cycle Assessment (EIOLCA), developed in the US, and the Missing Inventory Estimation Tool (MIET), developed in the Netherlands; the input–output tables of EIO-LCA are substantially different in nature from LCA, while MIETs are more similar in structure to LCA. The results show that hybrid approaches (involving the integration of IO with LCA and vice versa) may resolve the truncation error problems of LCA, together with the closed-loop incompleteness issues. The study also highlighted that, in general, IO-LCA approaches should not be used to carry out streamlining LCA, but to make the LCA set-up and the LCA results more comprehensive as well as to make them less site-independent, keeping in mind data quality and quantity;
- In Hayashi et al. (2010), a life cycle inventory (LCI) database for crop production in Japan (the NARO LCI database) is presented. The database was developed using modularisation techniques; SimaPro 7.2 was utilised for database construction and management, and Ecoinvent 2.1 was used as the basis of the development. The database includes inventories for paddy and upland field crops, agricultural work, fertilisers, pesticides and agricultural machinery;
- In Laurent et al. (2012), a summary of the results of an analysis aimed at assessing the available data in existing LCI databases regarding cereals (wheat, barley, maize, sorghum, rice and rye) and cereal-containing products is presented. The analysis was conducted on ten French and international databases, eight of which include cereal-related data (Ecoinvent, DiaTerre, LCA Food, Bilan Carbone®, AUSLCI, CPM Database, USLCI and Agri-Footprint), while two (Probas and BUWAL 250) do not include cereal-related data. The analysis highlights that the Ecoinvent Database is by far the most complete database, with Swiss and European data for agricultural raw materials, inputs and processes. Data about some cereal-based finished products can be found in the LCA Food Database (wheat, bread, pastries, oat flakes) and in the French Bilan Carbone® database. However, very few data can be found in the databases about agricultural processes, food industry processes, storage or mass-market retailing. Only the Ecoinvent Database and the LCA Food Database provide specific geographic data: Swiss data in Ecoinvent and Danish data in the LCA Food Database. The study also raises the issue of methodological comparability: all the databases set their own hypotheses and methodological rules (allocation, cut-off rules, etc.) and major differences can be found between data from different databases. This variability makes it difficult to implement environmental labelling of cereal-based products with sufficient accuracy and comparability.

Concerning data quality, it is interesting to stress that different papers applied some form of data quality check: sensitivity analysis was applied in seven cases (Charles et al. 2006; Kägi et al. 2010; Kim et al. 2009; Kløverpris et al. 2010; Narayanaswamy et al. 2004; Nemececk et al. 2008; Pelletier et al. 2008); completeness and consistency checks in one case (Narayanaswamy et al. 2004); uncertainty checks in one case (Röös et al. 2011); and comparisons with other studies and/or LCI databases in five papers (Aldaya et al. 2010; Espinoza-Orias et al. 2011; Harada et al. 2007; Hayashi et al. 2010; Williams et al. 2010). This indicates that data quality checks are increasingly gaining importance in LCA practice since they can be used to verify the reliability of uncertainty data and can assess more carefully the kind of influence such data can have on the final results.

4.3.5 Life Cycle Impact Assessment (LCIA)

The LCIA is a crucial phase of an LCA in which large quantities of data regarding natural resource use and emissions are transformed into useful information for the evaluation of the product system under analysis in terms of impacts on human health and on the environment. Unlike traditional risk assessment analysis, LCIA is not site-or emission-specific nor time-dependent (Margni and Curran 2012). However, the nature of food products, including cereals and derived products, and at times the type of food LCA study, is such that site-specific data must be considered (Notarnicola et al. 2012) in order to assess the potential impacts properly. In fact, as pointed out in the first chapter of the present book, especially for the agricultural phase of a food LCA, site specificity can greatly influence the results of the impact assessment. For example, the pedoclimatic conditions can heavily influence the impact deriving from the use of fertiliser and pesticides or the water use impact category.

The evolution of LCIA methods over the last decades has brought about numerous models, mainly involving a combination of midpoint and endpoint modelling, with numerous characterisation models that can potentially generate different results. The most widely used LCIA methods for cereal LCA are CML (Muñoz et al. 2012; Narayanaswamy et al. 2004; Nemececk et al. 2008; Notarnicola et al. 2004; Pelletier et al. 2008; Salomone and Ciraolo 2004; Williams et al. 2010), Ecoindicator (Bevilacqua et al. 2007; Petti et al. 2000; Renouf et al. 2008), EDIP (Nemececk et al. 2008; Schmidt 2008), ReCiPe (Roer et al. 2012), Impact (Drocourt et al. 2012) and LIME (Yoshikawa et al. 2012). Furthermore, as pointed out by Margni and Curran (2012), the rapid and fervent development of methodologies indicates that LCIA has not yet reached a stable and generally accepted standard; hence, methodologies that are older than 10 years may not reflect the state of the art and may entail methodological weaknesses that have been resolved with more recent methodologies. Thus, when consulting cereal-related LCA results dating back at least 10 years or when using one of the older methodologies, the results should be carefully analysed and if possible compared with similar results obtained with a more recent LCIA methodology implementation. However, despite this fervid development and the ongoing discussion, there are still some LCA studies that do not report the LCIA method used, therefore preventing any sort of comparison or relative assessment.

As far as specific impact categories are concerned, the IPCC (Intergovernmental Panel on Climate Change) and the WGO (World Meteorological Organization) models are the only internationally accepted ones commonly used for the GWP (global warming potential) and ODP (ozone depletion potential) assessment in all the methodologies. This is reflected in the explicit use of such methodologies in LCA regarding cereals, with particular regard to the GWP and the use of the carbon footprint methodology (e.g. Drocourt et al. 2012; Eshun et al. 2013; Espinoza-Orias et al. 2011; Gan et al. 2011a, b; Murphy and Kendall 2013; Nalley et al. 2011; Ruini et al. 2013). For other indicators, there are multiple characterisation models, not all unanimously accepted and each with limitations that inevitably will produce variability among LCA results regarding similar products. A list of LCIA methods, identified as the best among the existing characterisation models, was provided for each impact category in the context of the ILCD Handbook (EC-JRC 2011). If the identified model was judged of sufficient quality, it was recommended and the list of the recommended models for each impact category was provided by Hauschild et al. (2013). Some examples of applications of these recommendations are already present in the cereal sector, e.g. with regard to Usetox for toxicity-related impacts (Berthoud et al. 2011). Furthermore, all LCIA models assume that the emissions (with the exception of those relative to global warming and ozone depletion) occur in the country where the methodology was developed, which is not necessarily true and may need to be accounted for in the interpretation phase of the LCA study.

In order to deal with the above-mentioned site specificity, characterisation methodologies (e.g. IMPACT World+ 2014) are being developed in order to address the regionalisation of impact categories. Furthermore, software producers, of products such as Ecoinvent (2014), are moving towards more regionalised data sets (when applicable), but such effort is limited to data regarding different macro geographical regions (countries or areas of continents). In reality, data sets of a specific region of a country can produce LCIA results that differ considerably from the average national impact values; see for example Laurent et al. (2012) in the cereal sector.

Water use and land use issues related to cereal crops are undoubtedly affecting the ecosystem worldwide. This is particularly true for cereal crops such as wheat or rice that are used for the production of staple foods in many countries. The impact assessment for such impact categories is by no means standardised (Notarnicola et al. 2012) but should nonetheless, if possible, be included in order to improve the overall quality of the LCA of cereal or derived products (e.g. Kløverpris et al. (2010) for land use changes in wheat production). However, most of the cereal-related LCA studies include land use and land use change in terms of land occupation and land transformation, therefore neglecting a proper impact assessment (e.g. Bevilacqua et al. 2007; Braschkat et al. 2003; Brentrup et al. 2004; Charles et al. 2006; Drocourt et al. 2012; Kulak et al. 2012; Ruini et al. 2013; Schmidt 2008; Williams et al. 2010).

Of the studies reviewed in the previous sections of this chapter, many include pesticide and fertiliser production, but very few actually include the modelling of their diffusion in the environment. In most cases, it is assumed that all the pesticide or fertiliser is absorbed by the cereal plant. In reality, the pedoclimatic conditions and farming practices can strongly influence how much of these chemicals are transferred to the environment. Some studies adopt the PestLCI model by Birkved and Hauschild (2006) and Dijkman et al. (2012), which estimates pesticide emissions to air, surface water and groundwater for use in life cycle inventory (LCI) modelling of field applications, e.g. Berthoud et al. (2011); however, such a method is rather complicated and requires large quantities of data for its correct implementation. If the data are not available, there is a risk of basing the evaluation of the pesticide diffusion on too many assumptions, hence making the modelling ineffective. Furthermore, the results obtained with the PestLCI model need to be applied in combination with characterisation factors obtained from emission route-specific impact assessment models, such as USEtox.

Finally, LCIA includes options for the normalisation, grouping and weighting of impacts. The implementation of such approaches is subjective (e.g. weighting factors may be based on economic, political or environmental considerations) and can make the results of the LCA inapplicable to product systems of the same nature originating from different geographical areas. Therefore, in the cereal sector, weighting is only rarely implemented (Bevilacqua et al. 2007; Brentrup et al. 2004; Fallahpour et al. 2012; Notarnicola and Nicoletti 2001, 2004 Petti et al. 2000).

4.3.6 Interpretation and Comparison of the Results

There are many tools to assess the robustness of an LCA: one of these is the completeness check for both process coverage and I/O coverage (e.g. all the included material or energy input and emission associated with the system under analysis). Such coverage is seldom complete and complicated by the high variability of the system boundaries of cereal-related LCAs, even within the same LCA approach (cradle to gate or cradle to grave), which makes the comparison between different studies analysing the same product a harsh task. In this regard, Schau and Fet (2008) stress the need for a set of rules to determine the system boundaries for different product categories so that a comparison of the environmental impacts of different batches of products can be possible.

Most cereal-related studies have an interpretation phase that entails a description of significant related aspects and a discussion on the limitations and recommendations. However, not all of the analysed studies carried out a sensitivity or an uncertainty analysis to test the extent to which the results are affected by specific methodological choices.

In accordance with ISO 14,044, the sensitivity and uncertainty analysis should be based on those model choices known to have a major influence on the results of the study, such as (Guinée et al. 2002):

- allocation rules: examples for the cereal sector could be economic vs. energetic vs. mass allocation
- boundary setting: examples could be the inclusion or exclusion of the transport of agricultural inputs, the production of agricultural machinery, etc.
- process data: examples could be the variations of the type of fertilisers used, pesticides, fertiliser emissions, etc.

- cut-off criteria: changing the cut-off rules (the boundary between processes that are relevant and irrelevant to the product system)
- characterisation method: alternative characterisation methods, which could be adopted instead of the baseline method
- normalisation data and weighting method (if carried out)

Once one or more variables from the above list have been selected, the changes produced by their variation in the LCA results should be analysed.

Within the cereal and derived products context, several authors have implemented a sensitivity analysis in their studies (Blengini and Busto 2009; Charles et al. 2006; Drocourt et al. 2012; Espinoza-Orias et al. 2011; Kägi et al. 2010; Kløverpris et al. 2010; Meisterling et al. 2009; Roer et al. 2012; Yoshikawa et al. 2012).

Specifically, for such analyses, the following parameters were taken into account: field emissions (Blengini and Busto 2009; Charles et al. 2006; Kägi et al. 2010; Roer et al. 2012), allocation criteria (Blengini and Busto 2009), transportation distance and yields (Drocourt et al. 2012; Meisterling et al. 2009) and water requirements (Blengini and Busto 2009). Regarding field emissions from fertiliser use, two alternatives are available: either varying the emissions factor within the range provided in the model (i.e. the IPCC method), as performed by Roer et al. (2012), or using different models to assess the emissions (Charles et al. 2006). Espinoza-Orias et al. (2011), who determined the carbon footprint of bread, opted for a non-agricultural variable, carrying out a sensitivity analysis with different percentages of bread waste (from 10 to 30 %).

As for the results of the sensitivity analysis (expressed in terms of percentage variations with respect to the results obtained), the following considerations can be drawn: the range of variation in field emissions (N_2O and, in the case of rice cultivation, also CH_4) affected climate change from 11 (Roer et al. 2012) up to 36 % (Blengini and Busto 2009) and photochemical oxidant, terrestrial acidification and particular matter formation from 32 up to 53 % (NH_3 and NO_x) (Roer et al. 2012). In addition, Drocourt et al. (2012) and Kägi et al. (2010) underline the importance of the field emission parameters on LCA results. In fact, when it comes to direct field emissions, Kägi et al. (2010) found that the results varied from 15 (upland rice) up to 31 % (organic rice).

According to Blengini and Busto (2009), in their study on rice, the maximum variation caused by allocating burdens to straw was −10 % (for eutrophication potential, photochemical ozone creation potential and water use), while the change in water for irrigation affected the total water requirement by ±27 %. According to Murphy and Kendall (2013), the allocation approach could have a great influence on LCA results. In fact, in their study on corn, they state that, for most indicators, the subdivision approach produces impacts approximately equal to those of economic allocation and both economic allocation and subdivision assign significantly less impact to stover than energy allocation. Finally, in their study on bread, Espinoza-Orias et al. (2011), by varying the percentage of waste, calculated a variation of 10–12 % in GHG emissions.

In conclusion, in order to evaluate the robustness of an LCA model, a sensitivity analysis is most effective. Normally, the most uncertain parameters have to be taken into consideration to run a sensitivity test. These variables are often associated with field emissions when agricultural activities, such as cereal cultivation, are evaluated. These emissions, in fact, can strongly affect the results of an LCA study. Allocation criteria are also likely to influence the results of a study and should therefore also be carefully considered in this step of the analysis.

4.4 Some Lessons Learned from the Application of Life Cycle Assessment in the Cereal Sector

In this section, some indications are given of the best possible application of the LCA methodology based on the issues discussed in the previous sections. Such indications are by no means exhaustive or absolute but should be considered, whenever possible, when performing an LCA of a cereal product system, in order to fulfil the scope of the study and achieve the best possible results.

4.4.1 Goal and Scope Definition

The first phase of an LCA study consists of defining the goal and the scope, which aim to provide a description of the product system.

According to the ISO standard (ISO 14,040 2006), the goal of the study should define the application and the reason for carrying out the study, the intended audience and whether the results are intended to be disclosed to the public. The scope should clearly describe the system of the studied product or process and its boundaries, the system functions, the functional unit and reference flow, the environmental impact assessment methodology applied, the data requirements and finally the assumptions and limitations.

The goals of the cereal LCA studies reviewed in this chapter were different. The majority of them aimed to profile the environmental burden of a cereal, in order to identify the environmental hot spots of the system investigated. Other studies privileged a comparative approach, aiming to evaluate different farming and industrial practices or different cereal products. Investigating different agricultural practices (which includes evaluating different N rates used in the fertilisation phase or comparing organic and conventional systems), as well as industrial alternatives, could be very useful since it provides support in choosing the most productive approach in terms of environmental sustainability. From this perspective, effort should be made to identify the best agricultural techniques to put in place for cereal cultivation, consistent with the geographical specificities, in order to achieve a lower environmental burden. The same conclusion could be drawn for the industrial practices to be pursued along the supply chain of cereal-derived products.

4.4.2 The Functional Unit

The identification of an FU is the core of any LCA, providing a reference unit to which the inventory data are normalised. As already mentioned, the results of an LCA are strongly dependent on the FU chosen and this introduces a kind of uncertainty. Particularly in the cereal studies field, a 'one-size-fits-all' solution cannot be envisaged.

In the wide array of LCA studies found in the literature regarding cereals and cereal-based products, the selected FUs vary to a great extent, reflecting the significant differences in the characteristics of goods. Such differences concern not only the type of cereal (wheat, corn, rice) but also the kind of product deriving from a specific step in its supply chain (e.g. grains, flours, pasta, derived products, etc.). At times, the choice of FU is based on the mass or volume. This is often inadequate because the qualitative characteristics of grain can differ widely, as can those of cereal-based products. Most differences derive from complex production processes, which entail different technologies. Thus, cereal-derived products are not always comparable, even when belonging to the same cereal. Therefore, it may be useful to choose a set of multiple FUs based on mass, volume, cultivated area, economic value and qualitative characteristics, such as nutritional content. Charles et al. (2006), for example, in order to compare properly different wheat cultivation systems for bread making, included a quality parameter in the selected FU (13 % of protein in dry grain). The inclusion of quality parameters could in fact be an important aspect when it comes to comparing different systems within the same study or different studies assessing the same product.

Furthermore, since the main goal of an LCA is that of supporting market operators in their decisions, it is crucial to identify an appropriate FU—or, better, several FUs (Notarnicola et al. 2012)—in order to increase the relevance and impact of LCA sustainability information. A suggested tailored range of FUs for the cereal supply chain is schematised in Fig. 4.6.

4.4.3 System Boundaries

The definition of the system boundaries is a crucial step in the scope definition of every LCA study, but in the specific case of cereals, some aspects need to be considered. The recommendations are different according to the life cycle inventory modelling techniques used, i.e. whether it is an attributional or consequential LCA.

As demonstrated by Roer et al. (2012), in the case of attributional LCA, the system boundaries should be set in such a way that important processes for the environmental impact caused by food production are not excluded. In most cereal LCA studies, the manufacturing of machinery, buildings, humus mineralisation, production and the use of pesticides and/or nitrogen oxide loss due to the use of mineral fertiliser are excluded, but Roer et al. (2012) show that all these factors have a significant environmental impact, but with different contributions according to the

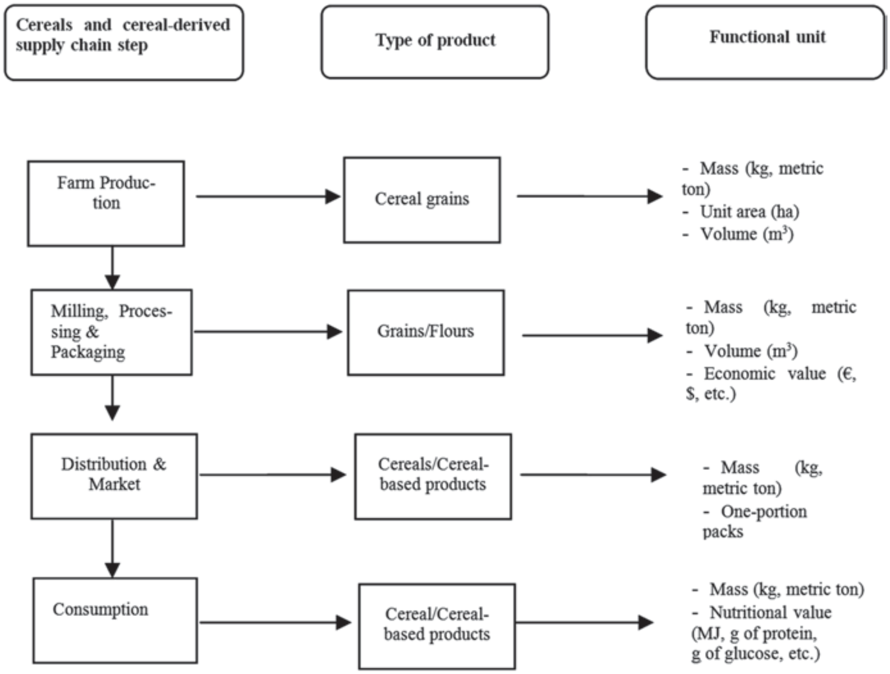

Fig. 4.6 Possible FUs related to cereal and cereal-derived product systems

impact categories considered. Therefore, the inclusion of these factors should be considered according to the impact categories included in the LCA study:

- the influence of the production of machinery was found to be relevant to all the impact categories, mostly the toxicity ones;
- the production of buildings had only a minor influence on the total environmental impact, although excluding buildings decreased the impact of the climate change impact category by 6–7 %;
- humus mineralisation only had an impact on climate change, and the exclusion of this factor decreased the carbon content by 2–30 %;
- the exclusion of the production and use of pesticides determined the highest reductions for the ecotoxicity impact categories;
- the exclusion of NO_X loss from the use of mineral fertiliser gave high reductions for particulate matter formation (65–71 %), photochemical ozone formation (83–82 %) and terrestrial acidification (68–75 %).

If all the above-mentioned factors are excluded simultaneously, the climate change impact is reduced by more than 40 %, with a consequent underestimation of the actual total environmental impact and increased difficulties in comparing different studies.

In the case of consequential LCA, the processes of further systems in addition to the one analysed need to be included in the system boundary of the analysed system in order to take into account the effects of an increased demand for cereals in one region. Schmidt (2008) presents a framework for defining system boundaries in consequential agricultural LCA, through the definition of different scenarios describing how the increased demand for wheat can be met. The comparison of scenarios shows significant differences in the contribution to the included impact categories (climate change, eutrophication and land use).

4.4.4 Availability and Quality of Data

The availability and quality of data issues of the cereal sector overlap with the problems of the more general agri-food sector. In particular, the former are connected to the lack of availability of data in the agricultural phase, such as the production of some fertilisers, herbicides and pesticides, the dispersion of chemical compounds into the environment (air, water and soil), the balance of CO_2 emissions, etc.

According to ISO 14,044, the quality of data should be carefully evaluated considering its time-related, geographical and technological coverage, and any information should be precise, complete, representative, consistent and reproducible, paying a high degree of attention to the source of the data and the uncertainty of the information.

The quality of data is strictly related to the availability of primary data, so it can be generally suggested to use literature data for the background system and primary specific data for the foreground. Estimations are very frequently not accurate, so, in principle, they should be avoided. Nonetheless, this is not always possible and provided that secondary data are derived from careful source selection and estimations, they can be used to obtain reasonably accurate LCA results. For example, in their study, Espinoza-Orias et al. (2011) performed a CF assessment of bread by following the PAS 2050 methodology using primary data and then performed the same study using secondary data, attaining similar results. Of course, in the case of the use of secondary data, all assumptions and estimations should be clearly declared and fully explained in order to avoid incomparability among different case studies. Furthermore, even if primary or specific data are available, a statistical approach to data collection and its evaluation should be used whenever possible (e.g. the use of confidence intervals or variance analysis). This implies taking, whenever possible, multiple measurements and readings of primary data in order to check the representativeness of the sampled data and also to verify that these data sets originate from the same stochastic distributions. Such approaches are implemented in very few LCA studies (e.g. Harada et al. 2007; Williams et al. 2010; Yoshikawa et al. 2010) and their importance is illustrated in Kägi et al.'s (2010) study on rice CF. Finally, considering the variability of data affecting the sector caused by the lack of specific and/or primary data, a consistency check of the data quality should be carried out, whenever possible, in order to evaluate how the different choices affected the results.

4.4.5 Allocation Methods

As highlighted in Sect. 4.3.3, in the cereal production sector, the allocation problem is particularly important because this type of production almost always implies cultivation systems that produce multiple products or imply the processing of diverse parts of a plant for different uses.

As suggested by ISO 14,044, allocation should be avoided through system expansion to include the additional functions related to the co-products or dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes. The choice of system expansion or of subdivision of the processes in which the allocation problem occurs should always be preferred to avoid data distortion, but when allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the physical relationships between them.

In the cereal sector, the literature analysis highlighted that, when allocation was carried out, the most common solution was to use economic allocation methods, although other methods were also experienced, such as mass allocation, the cereal unit allocation (Brankatschk and Finkbeiner 2012), energy-based allocation or a combination of economic and mass criteria. In addition, the PCRs published for this sector suggest different allocation rules depending on the specific type of product; in particular, of the four PCRs actually published for this sector, three suggest using mass allocation while the other suggests economic allocation. Moreover, as highlighted in Sect. 4.2.6, the products that are not compliant with the quality requirements and/or are destined for other chains (such as animal food) must be considered as waste (PCR 2010:01 (CPC 2371)).

In the opinion of the authors of this chapter, and also in line with the suggestions carried out for other agri-food sectors (see for example the chapter on the olive oil sector), when allocation cannot be avoided, the allocation procedures in the cereal sector (which mainly involve by-products), should take into account both the mass and the economic value of the by-products in order to balance the huge quantities of by-products obtained with their low economic value.

4.4.6 Life Cycle Impact Assessment (LCIA)

The scope of an LCA study in the cereal sector can be varied: evaluating/identifying environmental hot spots, comparing different options, obtaining some kind of sustainability label, for marketing purposes, or other aims. Hence, such a scope will determine whether the LCIA phase of the study should consider optional elements, such as normalisation, grouping and weighting of the results. For example, if the scope of the study is to compare LCAs of the same cereal product system, based on the approaches used by the other studies with which one wishes to compare the results, it may or may not be appropriate to exclude these optional steps of the impact assessment phase. As mentioned in Sect. 4.3.5, a cereal-based LCIA will have to deal with particular site-and time-dependent issues when addressing the impacts deriving from fertilisers, pesticide application and land and water use. Specifically, many of the cereal-based LCAs reviewed in this chapter do not specify (or simply completely ignore) how to deal with the modelling of the impacts deriving from the diffusion of pesticides in the environment and the balance of nutrients.

Fertiliser applications have been identified as one of the main emission sources, in particular for impact categories such as acidification and eutrophication. However, the importance of these emission sources can vary greatly in different cultivation areas because of the different climatic conditions and cultivation practices. In particular, the emissions of nitrous oxide (N_2O), ammonia (NH_3) and methane (CH_4) from the soil depend firstly on:

- The amount of nitrogen applied and the type of fertiliser. The application of organic fertilisers involves higher emissions of ammonia and methane compared with the application of the same amount of nitrogen with mineral fertilisers;
- The temperature, wind, soil and water content at the moment of fertiliser application (and during the days following the application);
- The method of application. For example, for organic fertilisers, the injection into the soil strongly reduces the ammonia emissions and a similar effect can be achieved with fast soil incorporation after the spreading. In such a context, Carozzi et al. (2013) report strong reductions of NH_3 emissions if spreading is carried out with specific techniques (higher than 80 % when the organic fertiliser is injected into the soil or is quickly incorporated into the soil).

In the absence of primary data regarding these emissions, some methods have been developed for their assessment. The method proposed by the IPCC is one of the most utilised due to its simplicity. Nevertheless, others methods have also been developed. For example, Brentrup et al. (2000) assessed the emission of NH_3, NO_3 and N_2O not only considering the amount of N applied but also taking into account the timing of application, the climatic conditions (temperature, wind and rain) and the soil conditions (water content, structure, texture and field capacity). When this information is available, this method should be utilised instead of the IPCC's method. In particular, when the analysis is focused on the agricultural step of the production system, simplification should be avoided because it will not be possible to evaluate the differences linked to the use of different fertilisers or to spreading with different spreading machines.

Furthermore, when the LCA regards cultivation carried out in specific conditions (for example in greenhouse or soilless cultivation or with high surface irrigation), detailed methods must be utilized for the assessment of fertiliser-related emissions (see the chapter on the fruit sector).

Various models are available for the quantification of the emissions from pesticide use (e.g. Audsley et al. 2003; Birkved and Hauschild 2006; Dijkman et al. 2012) and should be carefully chosen based on the complexity and age of the model and the specific data available for its implementation. Furthermore, due to the non-univocal definition of technosphere (anthropogenic system) and ecosphere (environment) in the case of agricultural soil, different approaches can be used to account for the impact from pesticide use with a subsequent effect on the impact assessment step. An operative framework for toxicological assessments of pesticides is proposed by van Zelm et al. (2013), who defined a procedure to help LCA practitioners to gather the right data and use the proper models to include all the relevant emission and exposure routes where possible. Furthermore, very few cereal-based LCAs deal with the impacts deriving from land and water use. Even though no standardised methodologies have been adopted for such impact categories, if site-specific data are available, the implementation of one of the many methods that are described in the literature (Notarnicola et al. 2012), provided that the relative assumptions made are clearly stated, can give an indication of the effects of land and water use that are undoubtedly responsible for a large part of the overall impacts attributable to cereal product systems. For example, a method for identifying ultimate or marginal land use changes when studying crop consumption via LCA is proposed by Kløverpris et al. (2010). Furthermore, in such a respect, LCIA methods that deal with site specificity are being developed (e.g. IMPACT World+ 2014) and should be used whenever possible to improve the overall results of the LCA.

Finally, as highlighted in the literature review (Sect. 4.2—e.g. Braschkat et al. 2003; Pelletier et al. 2008), organic cultivation (e.g. duck rice) and certain agricultural practices (e.g. reduced tillage) can have a beneficial effect on the sustainability of cereal product systems. In such a context, with regard to the climate change impact category, the LCIA phase should carefully account for any sequestration or specific emissions of biogenic greenhouse gases, which in many studies are erroneously assumed to generate an overall GHG balance of zero.

4.4.7 Interpretation

Generally the main recommendation for the interpretation of the LCA results is to perform combined sensitivity and uncertainty analysis in order to test the influence of the variability of the input data and the uncertainty connected with subjective choices on the final outcomes of the LCIA results.

In the reviewed cereal-related LCA studies, the most important issues with regard to the sensitivity check emerged as being linked to the cultivation phase with reference to the emissions related to the use of input (mainly N emissions from fertiliser use); the yield levels; the comparative analyses; and the transport and waste phases. Some authors also stress the need to perform a sensitivity check for methodological issues, such as the choice of the functional unit or system boundaries; the methodological exploration of LCI modelling of land use; and to investigate the effect of uncertainty due to the level of precision of the collected input data, the variations in climate and farm practice schedules.

Concerning uncertainty analysis, a practical solution for testing the model outputs according to the variability of inventory data is suggested by Niero et al. (2012). The approach is based on a combined qualitative and quantitative analysis, implemented through a qualitative assessment by data quality indicators and a quantitative analysis through the Monte Carlo sampling technique. If empirical data are available for calculating the uncertainty distribution, this should be the preferred option, instead of using expert judgement to make qualified estimates.

With these combined tools, the conclusions of the LCA study can be strengthened and the robustness and transparency of the study can be improved.

Conclusions

Cereals and their derived products represent agricultural commodities of worldwide importance, with particular environmental hot spots originating from their widespread use and from their particular nature. The review illustrated in this chapter of the life cycle approaches related to cereals has highlighted that the agricultural phase is in most cases the one responsible for a larger share of the impacts of such product systems. Specifically, fertiliser and pesticide production and use and fuel-related emissions seem to be a common source of impact. Fuel use is responsible for a large contribution to the energy demand and acidification. Fertilisation and pesticide usage are also responsible for a large quota of the overall energy use during the life cycle of cereal-based products and hence are also responsible for the production of GHG. Such energy demand, together with ozone-depleting and acidifying emissions typical of intensive agricultural systems, are the reason for the lower impact of alternative organic types of cultivation that generally do not involve the use of pesticides and avoid the production of fertilisers, including nitrogen-based ones. However, even though organic agricultural approaches can potentially lower the overall impact of cropping systems, the lower fertiliser use and relative energy use in such systems can at times be counterbalanced by larger energy use for fieldwork and lower yields, which in turn lead to overall greater land occupation needed for the cereal production.

The growing number of CF studies highlights an emphasis on the study of the effects of cereal systems on climate change. In this context, rice differentiates itself from corn and wheat since there are many types of rice and production processes, all of which are responsible not only for the above-mentioned major contributions deriving from the production and use of fertilisers but also for the contribution of methanogenesis occurring in the waterlogged soil. Accordingly, there still seems to be no unique methodology for the reduction of the impacts of crop production, and the results from the studies vary substantially; however, the literature indicates that careful use of organic approaches and controlled water use and puddling methods need to be considered in order to reduce both WFs and CFs. What can certainly improve the overall sustainability of cereal production is the use of correct crop rotation by following the cereal cultivation, whenever possible, with legume or vegetable cultivation.

The literature review has also highlighted that the user behaviour when dealing with cereal-based products, for example in terms of transport distance and typology for the purchase of such products or the disposal of waste, can heavily influence the overall environmental sustainability of such product systems. There is thus a need to inform customers better and enhance their awareness of the possibilities of contributing to more sustainable cereal product systems with particular reference to the end of life of the product, which is often one of the least-studied LCA phases.

Overall, the implementation of LCA approaches, at an institutional level (both in developed and in developing countries), at the large corporate firm and SME levels, has increased the environmental consciousness of the people involved in the cereal sector, including users and customers, with an overall reduction of the burdens deriving from such product systems. There is in fact growing use of LCA in the cereal sector for the obtaining of environmental labels (e.g. EPDs). However, there is still a need for a better understanding of the difficulties that can be encountered when performing an LCA of a cereal product system in order to gain the best possible results, which can be used to improve the sustainability of the system. Some of these methodological aspects have been discussed in this chapter, such as the site and time dependency of pesticide diffusion modelling, the need for a deeper analysis and a standardised methodology for calculating the effects of land use on the quality of soil and biodiversity and the need for better quantification and qualification of water use. Among the reviewed studies, the system boundary and functional unit definition appear to be critical stages of the LCAs, in which it may be necessary to use one or more FUs, inevitably using allocation for environmental burden partitioning, and in which certain assumptions have to be made in order to progress with the overall assessment and overcome the lack of data and time or cost issues. The sources of information at the base of these assumptions are not always accurate; hence, it is important to explicate them carefully and evaluate the representativeness of the results and their variability in order to produce useful cereal LCA work that can help understand and improve the sustainability of such product systems.

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