Chapter 5
Life Cycle Assessment in the Livestock and Derived Edible Products Sector

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Abstract The livestock production sector represents more than 40 % of the economic value of EU primary productions. This sector consists of a huge diversity of processes and techniques depending on the animal species and the final products. Because of these differences, livestock productions are associated with several adverse effects on the environment, especially in the breeding phases and feeding composition and management; moreover, in terms of raising awareness of the environmental implications of livestock productions, LCA applications are of increasing importance for systematic assessment of the environmental burdens connected with this sector. After an overview of the structural and economic characteristics of the most significant livestock supply chain and its main environmental problems, we provide a description of the international state of the art of LCA implementations for livestock. Methodological problems connected with the application of LCA are investigated, starting with the critical analysis of international papers and the few Italian papers in the scientific literature. Finally, the best practices regarding LCA methodology implementation are proposed, in order to improve results and manage the methodological problems identified.

Keyword Livestock sector · Life cycle assessment · Life cycle costing · Environmental product declaration · Footprint labels

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5.1 Introduction

The global livestock industry accounts for almost 40 % of agricultural GDP (Steinfeld et al. 2006) and the global meat production is projected to double by 2050 following the increase in meat demand (FAO 2006). At the same time, the FAO (2006) states that the livestock sector is one of the most critical sectors in terms of environmental problems such as climate change, water and air pollution, land degradation, and biodiversity loss (Steinfeld et al. 2006). Livestock systems occupy about 30 % of free terrestrial surface area and with a value of at least $ 1.4 trillion they are a major global asset. In developed countries production and consumption of livestock products are now growing slowly, albeit at high levels of production, accounting for 53 % of their agricultural GDP (World Bank 2009). Impacts may vary significantly depending on the supply chain in question (meat and dairy, pigs, sheep, goats or chickens) and the practices and techniques employed. In recent years, this sector has received particular attention, and has been the subject of a number of studies since it was defined as one of the productive sectors with the highest environmental impacts (Steinfeld et al. 2006; Weidema et al. 2009). This is because of 75–90 % of the energy consumed by the animal in its diet is then used for body maintenance or lost in manure and by-products such as skin and bones. Livestock competes with the other productive sectors for the use of scarce resources such as land, water and energy, and, according to the Food and Agriculture Organization (FAO), it is responsible

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for 18 % of global greenhouse gas (GHG) emissions (Weidema et al. 2008; Leip et al. 2010); because of the emissions of CO_2, CH_4 and N_2O occurring during crops production for animal feed and the animal rearing (Steinfeld et al. 2006). Livestock production has significant environmental impacts including greenhouse gas (GHG) emissions (Stanford University 2010). According to the FAO (2006), the global demand for beef and milk in 2050 is expected to rise up by 72 and 82 %, respectively, compared with 2000, and thus the GHG emissions from these sectors need to be reduced considerably. Identification of the best reduction strategies requires a detailed analysis of the environmental loads produced by each food product, in particular livestock products, during the entire life cycle in order to identify the hotspots and to compare them. For this purpose, the Life Cycle Assessment (LCA), a method that analyses products along the whole production cycle, their use and waste management (Guinée et al. 2002), is well suited for this type of analysis and has been used for the determination of environmental impacts of livestock products (de Vries and de Boer 2010). The structure of this chapter differs from the others in this book because of the numerous types of supply chains in the livestock sector in terms of both species and products (meat, milk, wool, eggs). They obviously use several production technologies and different amounts of resources and so produce different environmental impacts. Consequently, the chapter is ordered according to the different supply chains starting from beef and dairy production, through sheep and goats, and ending with pigs and poultry.

5.2 Overview of Product Based Life Cycle Assessment Methods on Livestock[1]

Livestock products are often shown as amongst the most harmful for the environment. This view is, inter alia, supported by the findings of the Input-Output Environmentally-Extended Analyses conducted for the United Nations Environment Programme (UNEP 2010) and the European Commission's Joint Research Centre (Tukker et al. 2006). According to peer-reviewed environmental assessments conducted by the United Nations Food and Agriculture Organizations (FAO) at global level, the greenhouse emissions tied to livestock are also significant and effective mitigation measures should shaped and implemented (Gerber et al. 2010, 2013; MacLeod et al. 2013; Opio et al. 2013). Such studies contributed, among others, to indicate livestock as top priority for the agenda on the environment of policy makers. Also because of the numerous scientific challenges when accounting for the environmental burdens, pressures and benefits tied to livestock supply chains, several have been the efforts made to advance science on life cycle assessment in these sectors. See the following sections for more detail on the scientific literature produced so far. Building somehow on these efforts, high has been the proliferation of product-specific environmental assessment methods, mostly developed for ei-

[1] The views expressed in this chapter are those of the authors and do not necessarily represent the views of the European Commission and FAO

ther commercial purposes linked to environmental communication or in support of a suite of policy measures on eco-friendlier farming practices or low-carbon biofuel productions. This section introduces these product-specific environmental assessment methods to pave the way to the development of harmonised methods in the context of consensus building initiatives in this field such as, amongst others, the FAO-led Livestock Environmental Assessment and Performance (LEAP) Partnership (LEAP 2014a), the European Commission's Product and Organisation Environmental Footprint (EC 2014) and the European Food Sustainable Consumption and Production Round Table (Food SCP RT 2014). These initiatives involve governments, business representatives, and civil society and strive, to varying extents, for ensuring equal footing in steering the development process of the technical guides. The livestock-specific environmental assessment methods were mostly identified through a search for relevant PCRs in the repositories of the prominent programme operators of product performance-based environmental communication schemes (Subramanian 2012, GEDnet 2014) that was coupled with an internal consultation round within the European Food Sustainable Consumption Round Table in late 2013 (Food SCP RT 2013a). This search was restricted to methods that are freely available. The assessment methods found can be grouped as follows:

Group A[2]: product category rules (PCR) developed in the context of type III environmental labelling schemes established according to ISO 14025 (Boeri 2012; Brondi 2013; IERE 2006; Japanese CFP scheme 2011a, b, c; Marino et al. 2011; Palm 2010; Pernigotti 2011; Sessa 2013a, b, c).

Group B: environmental assessment methods released by and for business associations. For example, the International Dairy Federation and DairyUk have issued carbon footprinting methods (Carbon Trust 2010; IDF 2010). It must be noted that some PCRs have also been prepared, commented and endorsed by specific business associations such as, e.g., Assocarni for the PCR on meat of mammals (Boeri 2012);

Group C: product-based carbon footprinting methods developed in support of policy measures for lowering the environmental performance of farming practices both at product and at organisation level (Tuomisto et al. 2013; Bochu et al. 2013);

Group D: Sector-specific carbon footprinting methods underpinning the peer-reviewed life cycle assessments conducted by authoritative bodies such as FAO (Gerber et al. 2010; 2013; MacLeod et al. 2013; Opio et al. 2013) and the European Commission's Joint Research Centre (Leip et al. 2010);

Group E: Product-based environmental footprinting methods to calculate greenhouse gas emissions of biofuels and biogas out of livestock supply chains (EU 2009). The methods sorted out by product category under concern are presented in Table 5.7. The methods have been grouped in three major categories, namely: products from ruminant supply chains (other than dairy products), dairy products and poultry products. The names of these product categories were set for illustrative sake and are not necessarily aligned with reference international codes for prod-

[2] Further PCRs are seemingly under development in the context of The Sustainability Consortium (TSC, 2014) and the French labelling scheme laid down in the national law generally known as Grenelle de l'Environment (Cros et al. 2010, French Parliament 2010)

ucts. Given the wide range of products coming out from livestock supply chains, Table 5.7 is to be conceived as a partial overview of the methods available to date on livestock products. In addition, new methods are on the pipeline of a few labelling schemes. For instance, The Sustainability Consortium has been developing new methods on, amongst others, beef, milk, butter, cheese, yoghurt, chicken, eggs, and pork (TSC 2014). Similarly, the new PCRs in the context of the French labelling initiative are seemingly under development on dairy products, and on meat and co-products from bovine, poultry and pork supply chains (ADEME AFNOR 2014).

Despite the proliferation of environmental assessment methods in the livestock sector, it is high the methodological misalignment between technical documents applicable to the same product category. If the methods listed in Table 5.7 are analysed against the criteria set for PCR characterization purposes in the context of the PCR Guidance Development Initiative (Ingwersen and Subramanian 2013), misalignment areas can be clearly spotted. From a rapid screening, it sounds clear that, currently, there is lack of convergence on how to set functional unit, system boundaries, allocation rules and, last but not least, the scope of the methods in terms of environmental issues covered. Data quality requirements are also another major issue. Nevertheless, we decided to leave the discussion on this point out the scope of this chapter for practical reasons. The in-depth analysis of all datasets would have, in fact, required a dedicated project. Functional units look different across methods from the different groups presented in Table 5.1, across PCRs of different food categories and even across PCRs of the same product category. Of course, the application context of the methods falling into the different groups matters and seems sufficient per se to justify such misalignment. For example, the approach adopted to set up the function unit in the context of product-based B2B communication (see Group A for PCRs) is different from the sector-specific environmental reporting methods (see Group B and D). Going through PCRs, we found out that divergences on functional unit exist. For example, the functional unit for chicken meat is equal to “one pound of meat at the processing plant exit gate” according to IERE (2006), it shall be expressed as 1 kg of poultry meat and the required packaging according to the International EPD System PCR on meat of poultry (Palm 2010), and it should be expressed as “per unit weight (100 g of contents amount)” according to the PCR on chicken developed in the context of the Japanese pilot program on the carbon footprint of products (Japanese CFP scheme 2011b). The methods screened also diverge in terms of system boundary. Beyond the differences in terms of coverage of processes related to e.g. supply of capital goods (e.g. machineries, buildings, greenhouses, etc.) and labour (transport of farmers and other workers), the definition of system boundary varies across methods because of lack of harmonization on how to define co-products, by-products and waste streams. For example, manure is a co-product according to e.g. the FAO report on the global assessment of GHG emissions and mitigation opportunities (Gerber et al. 2013), and according to the European Commission’s Product Environmental Footprint (PEF) Guide (EC 2013a). In contrast, manure is not a co-product according to a number of other references such as e.g. Boeri (2012), IERE (2006), and Sessa (2013a, b, c). As no GHG emissions from the farming stage are associated to biogas and biodiesel produced

Table 5.1 Environmental assessment methods per product category and groups

Product category	PCRs (Group A)	Business association guidelines(Group B)	Carbon footprint calculator (Group C)	Reference studies (Group D)	Laws (Group E)
Products from ruminant supply chains (other than dairy products)	Meat of mammals (Boeri 2012), Meat (IERE, 2006) Finished bovine leather (Pernigotti 2011) Leather footwear (Brondi 2013)		Cattle, pigs, sheep, goats, and other small ruminants (Bochu et al. 2013)	Pig sector (MacLeod et al. 2013) Ruminant sectors (Opio et al. 2013) Large ruminants (Leip et al. 2010)	Biogas and biofuels from manure and tallow (EU 2009)
Dairy products	Raw milk (Sessa 2013a) Processed liquid milk and cream (Sessa 2013b) Youghurt, butter, and cheese (Sessa 2013c)	Dairy products (Carbon Trust 2010) Dairy (IDF 2010)		Dairy sector (Gerber et al. 2010)	Biogas and biofuels from manure and tallow (EU 2009)
Poultry products	Meat (IERE, 2006) Meat of poultry (Japanese CFP scheme 2011b; Palm 2010) Poultry eggs (Japanese CFP scheme 2011a; Marino et al. 2011) Down and feather (Japanese CFP scheme 2011c)		Poultry (Bochu et al. 2013)	Chicken sector (MacLeod et al. 2013)	

from manure and tallow in the so-called Renewable Energy Directive (EU 2009), we deduce that manure and tallow are not co-products either according to such law. How to distribute environmental burdens, pressures and benefits among livestock co-products remains one of the most debated and unresolved issues. Assessment results drastically change, depending on the allocation approach adopted. This issue is particularly evident in livestock-related LCA methods where diverging approaches for dealing with process multifunctionality exist. For example, Gerber et al. (2013) have followed the following approach for their FAO report: among edible products (e.g. meat and eggs; and beef and milk), the allocation is based on protein content; between edible and non-edible products (e.g. milk, meat and fibre), the allocation is based on economic value of outputs; no emissions are allocated to the by-products from the slaughtering stage (e.g. offal, skins, blood). In the European Commission report conducted on livestock sector contribution to GHG emissions in EU (Leip 2010), allocation of emissions between multiple products throughout the supply chain is generally performed according to the nitrogen content of the products. The only exception was the allocation of CH_4 emissions from enteric fermentation and manure management of dairy cattle, which is allocated to milk and beef on the basis of the energy requirement for lactation and pregnancy). In the context of the International EPD System PCR on finished bovine leather (Pernigotti 2011), allocation at the slaughtering stage should be conducted among raw hide, comestible goods and scraps according to physical allocation (mass). In the PCR on meat of mammals (Boeri 2012), the allocation between meat, milk and leather should be conducted according to an estimate of their economic value. According to IERE (2006), all impacts should be allocated to meat. Allocation according to energy content is recommended by the Renewable Energy Directive (EU 2009) for co-products from the production of fuels. As said, tallow and manure are considered as sources of biofuels in such context. Last but not least, methods were found diverse in terms of environmental impact categories covered. Several are the methods dealing with GHG emissions only and coming up with figures on the carbon footprint of: the livestock sector as a whole (Gerber et al. 2013; Leip 2010), dairy sector (IDF 2010; Gerber et al. 2010), dairy products (Carbon Trust 2010), pig and chicken supply chains (MacLeod et al. 2013), ruminant supply chains (Opio et al. 2013). With the exception of PCRs developed in the context of the Japanese pilot project on the carbon footprint of products, which by definition covered GHG emissions only, all other PCRs listed in Table 5.1 adopt a multi-criteria perspective in the sense that a range of impact categories are covered. Nevertheless, impact assessment models recommended are often different, especially between different schemes (cfr. EIRE 2006 with Marino et al. 2011). This is a major issue that heavily affect the interpretation of assessment results. To support decision making processes and avoid that environmental information is deliberately disclosed in a misleading way, the European Commission and the European Food Sustainable Consumption and Production Round Table have recently recommended a list of assessment models and characterisation factors to be used in the context of environmental communication (both B2B and B2C). See the PEF Guide (EC 2013b) and ENVIFOOD Protocol (Food SCP RT 2013b) for more detail. Unlike these initiatives, the LEAP Partnership has been focussing on GHG emissions and other few impact categories when developing LCA guidelines on feed, poultry, small and large ruminants (LEAP 2014). This narrower scope is justified by the seemingly more consolidated science behind GHG emission accounting. Nevertheless, LEAP will not limit itself to GHG emissions. At present, efforts are on-going to explore how best set up a common framework to assess not only negative, but also positive impacts of livestock on biodiversity. Consensus will also be sought on issues such as e.g. use of water and of nutrients (LEAP 2014).

5.3 Beef Cattle

In the following, a description of the main aspects of this sector at the international and European levels is presented. Then, 34 international LCA studies on beef cattle production published in peer-reviewed journals, scientific reports or international conference proceedings are analysed. The study selection covers all the LCA applications to beef production systems published in the last 10 years. Methodological problems connected with the application of LCA in the beef cattle production sector are analysed in detail, starting with a critical comparative analysis of the LCA case studies. Finally, hotspots for the implementation of the LCA methodology in the beef production sector are identified in order to manage the methodological problems presented above. (Table 5.2)

5.3.1 The Beef Cattle Sector: Main Aspects

Over the years, the world market for beef has suffered a decline in terms of number of farm animals, production and consumption. The largest losses are found among the developed countries, mainly because of the economic downturn. However, trends in some emerging countries, particularly Brazil, India and Argentina, have seen improvements in terms of consumption and production, as they are managing to meet their domestic demand and overcome the shortcomings of the United States and the European Union, whose countries have increased import volumes. In 2010, 56 million t of meat were produced worldwide, according to USDA data (FAO 2006). The United States appear to be the major producer with nearly 12 million t of meat produced, followed by Brazil with 9 million tons and the EU−27 with about 8 million t. In the European context, France has by far the EU's largest cattle herd, with more than 19 million animals, followed by Germany (about 12.7 million) and Britain (10.3 million.). Italy, Ireland, Spain and Poland are each home to around 6 million cattle. Cattle farming is a significant component of European livestock sector, and many LCA studies of milk and meat production have been performed in recent years (Basset-Mens 2008). In this regard, Weidema et al. (2008) estimated that the 24 % of the environmental impacts of overall European consumption were attributable to milk and meat. The breeding of cattle has always accompanied the

Table 5.2 List of references included in the literature review and their main characteristics

Reference	Fu	Method	Main boundaries	Geographical areas	Time boundaries
Basarab et al. (2010)	1 kg of live weight	LCA and economic analysis	Cradle to slaughterhouse gate LCI basis	Canada (Alberta)	*Not specified*
Beauchemin et al. (2010)	1 kg of meat	LCA and HOLOS model for LCI	Cradle to farm gate	Canada	8 years
Beauchemin et al. (2011)	1 kg of carcass weight	HOLOS model for LCI and LCA	Cradle to farm gate	Canada (Western)	8 years (2003–11)
Bonesmo et al. (2013)	kg of CO_2-eq/kg of live weight kg of CO_2-eq/FPCM[a]	HOLOS model adapted for Norwegian dairy and beef production system	Cradle to farm gate	Norway	1 year
Casey and Holden (2006)	1 kg of live weight (after the first rearing year)	LCA	Cradle to farm gate	Ireland	*Not specified*
(Cederberg and Stadig 2003)	1 kg of energy corrected milk	LCA	Cradle to farm gate	Sweden	*Not specified*
Cederberg et al. (2009)	1 kg of carcass weight equivalent 1 kg of meat exported to Europe	LCA	Cradle to gate	Brazil	1 year (2005)
Clarke et al. (2012)	1 kg of carcass and ha of land occupied	GBSM model and a partial LCA	Cradle to farm gate	Ireland	*Not specified*
De Vries and de Boer (2010)	1 kg of meat 1 kg of protein 1 kg of average daily intake	Review of LCAs	Cradle to international retailer	OECD countries	*Not specified*
Doreau et al. (2011)	1 kg of live weight gained	LCA	Cradle to farm gate	*Not specified*	*Not specified*
Edwards-Jones et al. (2009)	kg CO_2-eq/kg live weight	LCA	Cradle to farm gate	Wales	*Not specified*
Flysjö et al. (2012)	1 kg of ECM[b]	LCA	Cradle to farm gate	Sweden	*Not specified*
Foley et al. (2011)	kg CO_2-eq-per ha	GBSM model and a partial LCA	Cradle to farm gate	Ireland	1 year
Leip et al. (2010)	1 kg of carcass meat	CAPRI model and LCA	Cradle to farm gate	EU–27	*Not specified*

Table 5.2 (continued)

Reference	Fu	Method	Main boundaries	Geographical areas	Time boundaries
Nguyen et al. (2010)	1 kg of meat at the slaughterhouse	LCA	Cradle to farm gate	Europe	*Not specified*
Nguyen et al. (2012a)	1 kg of carcass weight 1 ha of land occupied	LCA	Cradle to farm gate	*Not specified*	1 year
Nguyen et al. (2012b)	1 kg of carcass weight 1 ha of land occupied	LCA	Cradle to farm gate	Francia	1 year
Nijdam et al. (2012)	kg CO_2-eq/kg product	Review of LCAs	Cradle to farm gate	*Not specified*	*Not specified*
Ogino et al. (2004)	1 live animal	LCA	Cradle to farm gate	Japan	*Not specified*
Ogino et al. (2007)	1 marketed beef calf (8 months)	LCA	Cradle to farm gate	Japan	*Not specified*
Oishi et al. (2013)	1 kg of live weight	LCA and economic analysis	Cradle to farm gate	Japan	*Not specified*
Pelletier et al. (2010)	1 kg of live weight	LCA	Cradle to farm gate	United States (Upper Midwest)	*Not specified*
Peters et al. (2010a)	*n.a* [c]	Hybrid LCA On-site and I/O LCI	Cradle to farm gate	Australia	2 years
Phetteplace et al. (2001)	*n.a.*	LCA for Carbon Footprint	Cradle to farm gate	US	*Not specified*
Ridoutt et al. (2011)	kg CO_2-eq/kg of live weight	LCA	Cradle to farm gate	Australia	*Specified*
Ridoutt et al. (2012a)	1 kg of live weight	LCA	Cradle to farm gate	Australia	*Not specified*
Ridoutt et al. (2012b)	L H_2O eq/kg live weight m^2 year/kg of live weight	LCA	Cradle to farm gate	Australia	*Not specified*
Ridoutt et al. (2013)	kg CO_2-eq/kg of live weight L H_2O-eq/kg of live weight	LCA	Cradle to farm gate	Australia	*Not specified*
Roer et al. (2013)	kg CO_2-eq/kg of carcass weight	LCA	Cradle to farm gate	Norway	*Not specified*
Roy et al. (2012)	kg of CO_2-eq/kg of meat (or gr of protein or MJ of energy)	LCA	Cradle to fork	Japan	*Not specified*

Table 5.2 (continued)

Reference	Fu	Method	Main boundaries	Geographical areas	Time boundaries
Vergè et al. (2008)	kg of CO_2-eq/kg of live weight	LCA	Cradle to farm gate	Canada	20 years
Weidema et al. (2009)	Meat and dairy products consumed in EU−27 kg $CO_{2\ eq}$/kg of carcass weight	LCA and environmental impacts, monetarisation	Whole life cycle	EU−27	*Not specified*
Weis and Leip (2012)	kg CO_2-eq/kg of raw milk kg CO_2-eq/kg of eggs	LCA with CAPRI model	Cradle to farm gate+ slaughtering	EU−27	20 years
Williams et al. (2006)	1 t of carcass weight 20,000 eggs 10,000 l milk	LCA	Cradle to farm gate	UK	*Not specified*
Zonderland-Thomassen et al. (2013)	L H_2O-eq/kg of live weight	LCA	Cradle to farm gate	New Zealand	1 year

[a] Fat and Protein Corrected Milk (FPCM)
[b] Energy Corrected Milk (ECM)
[c] *n.a.* not available

evolution of agriculture in Italy, accounting for a total of 5.592.700 heads (ISTAT 2010). As can be seen from the Italian census data, beef cattle account for 15 to 42% of the total number of cattle reared in the country, with 862.660 heads (ISTAT 2010). The differences between breeding techniques strongly influence the production and the economic results and, above all, the environmental impacts of the production system. Beef production systems are managed essentially by two techniques: the fattening of calves and the breeding of suckler cows. The first is characterised by the purchase of calves of different age and weight: new-born calves 10 days old and weighing 30–40 kg, calves aged 2–3 months weighing 70–120 kg, light weanlings between 8 and 10 months weighing 270–300 kg or heavy weanlings 14–16 months old and weighing 380–480 kg; these are fattened until they reach the ideal weight for slaughter, i.e. 550–650 kg at 15–18 months. The fattening of calves can be done by both extensive methods, that require loose housing which allows the animals to move freely and to develop their muscle mass, or intensive methods, such as tethered housing, where the animal is tied or penned within its location, and deprived of freedom of movement. The beef production achieved in lowland areas through fattening cattle of high genetic merit, often imported from abroad, can cause socioeconomic deterioration. Indeed, the high cost of imported weanlings adversely affect the economic result of the production system. Furthermore, the intensive farming is responsible for environmental degradation due to the excess nitrogen production. The suckler cow production system can be divided into two categories: the cow-calf line and the calf-heifer line. The most common is the cow-calf line, which requires the purchase of heifers or calves of high genetic merit that remain on the farm until the end of their career and are bred to provide calves to be sold and fattened in other stalls. This activity represents an economic opportunity, not only for the lowland areas, but also the hills and mountainous areas which are otherwise difficult to exploit and can be abandoned, causing their environmental degradation. The calf-heifer line is a more intensive breeding method involving the purchase of heifers on the market that are impregnated as soon as possible. The young cows are fattened for slaughter before or after weaning the calf. Both breeding systems can be conducted in the confined wild state. Breeding in confinement is typical of farms also engaged in crop production (especially corn), which have significant amounts of crop residues and use manure to maintain soil fertility. Wild and semi-wild breeding is practised only in the marginal areas for the purpose of environmental restoration. This production system mainly exploits forage resources through grazing.

5.3.2 Literature Review on LCA Application in Beef Cattle Sector

The results reported in the scientific publications collected in this review are as different as the studies analysed. The variability is essentially because of the difference in the production systems and methodological choices (functional unit, system boundaries, allocation method, etc.). The GHG emitted by specialised beef production systems vary from 22 to 40 kg CO_2 equivalents per kg of meat; whereas for meat from dairy cow systems values are lower, from 14 to 19 kg CO_2 equivalent per kg of meat (Sonneson et al. 2009). These results were confirmed by Nguyen et al. (2010) who report 27.3 kg CO_2 equivalents per kg of meat for suckler cow and calf-beef production systems and an average of 17.9 kg CO_2 equivalent per kg of meat for dairy calf and beef systems. This huge variation is largely because of the very wide variety in beef production systems, which range from very intensive to very extensive (Nijdam et al. 2012). Several studies aiming to identify ways to reduce the environmental impacts of ruminants have focussed exclusively on the analysis of GHG (Martin et al. 2010; Eckard et al. 2010). However, the most critical aspect remains the evaluation of how the implementation of these practices could produce a net reduction of environmental impacts, assessing, for instance, other impact categories (Beauchemin et al. 2011). In this regard, only five studies exclusively dedicated to GHG emission assessment are included in this review; others extend the analysis to other impact categories (e.g. energy use, acidification potential, eutrophication potential or land use). Among these, some studies focus on the analysis of the effects of different diet on the production of CH_4 from enteric fermentation (Doreau et al. 2011; Oishi et al. 2013). A significant factor in LCA analysis of beef production systems is the definition of the unit of product with respect to which the environmental impacts are defined (functional unit—FU). According to Nijdam et al. (2012), the most commonly used functional unit for meat is either kilogram of carcass weight or live weight. This uniformity is not evident in practice, as the values assigned by each author to the respective FU vary considerably depending on the production system analysed, the rearing species, and the traditional and local slaughtering activities. Different FUs are found in the studies which evaluate the effects of production process modifications on environmental impacts. Nguyen et al. (Nguyen et al. 2012a, b) when focussing on the effects produced by the different animal management strategies and different feed crop rotations, use as FU, respectively, 1 kg of carcass weight and 1 ha of land occupation; Doreau et al. (2011) evaluate the effects of different diets on the GHG emissions based on the unit increase in animals weight. A common characteristic of all the analysed studies is the heterogeneity in the system boundary (SB) definition. The variety and the complexity of beef meat transformation processes is a critical methodological point for LCA analysis. The life cycle is usually considered at the farm gate, confining the analysis to the rearing phase and disregarding the slaughtering and transformation processes. The descriptions of the productive phases that characterise the life cycle from cradle to farm gate are not always consistent in the examined studies. Basarab et al. (2010) and Cederberg et al. (2009) include the transport of animals to the slaughterhouse in the SB, whereas Ogino et al. (2004) consider the disposal of animal wastes (manure and slurry) to be part of the SB. In general, capital goods and internal and external transport are excluded from the SB (Oishi et al. 2013; Nguyen et al. 2012a, b; Ridoutt et al. 2011; Basarab et al. 2010; Beauchemin et al. 2010; Leip et al. 2010; Pelletier et al. 2010; Ogino et al. 2004, 2007; Casey and Holden 2006; Cederberg and Stadig 2003). The agricultural phase in beef cattle rearing is restricted to fodder and grassland production that consists of farm operations (fertilisation, pesticide use, etc.), and is responsible for high environmental impacts of the entire production system. The inclusion of crop production in beef rearing system impact assessment is a critical and debated question, for the analysis of which we refer the reader to Sect. 5.2.2. Only Basarab et al. (2010) use primary data because they focus on a specific area (Alberta, Canada) which specialises in beef production. Beef production systems are characterised by a high number of co-products and by-products. Thus, allocation is a key methodological issue in environmental impact assessment for this sector. The collected studies avoid co-product allocation, defining as their goal the assessment of environmental impacts generated only by beef production systems, which is totally different from assessment of milk production. Only Casey and Holden (2006) use, respectively, 1 kg of live weight and 1 kg of live weight gained as FU to avoid impact allocation between milk and meat. Manure, the main by-product, is included within the SB in all the studies analysed, because it is considered as organic fertiliser that returns directly (including the agricultural phase) or indirectly into the natural cycle. However, the polluting emissions produced by manure management operations are always included in the Life Cycle Inventory (LCI). The Life Cycle Impact Assessment (LCIA) varies among the studies, especially for the impact categories and the methods used for their assessment. All the analysed articles, excluding that by Weidema et al. (2009), stop at the classification and characterisation impact stages, which are obtained by different methods: IPCC 2007 (Climate Change 2007); EDIP (Hauschild and Potting 2003); CML (Guinée et al. 2002); CED (Frischknecht et.al. 2003); Impact 2000+ (Jolliet et al. 2003); and Ecoindicator 99 (Goedkoop and Spriensma 2001), depending on the impact indicators chosen for the assessment. Among these, the impact categories designed to measure the environmental impacts in terms of GHG, non-renewable energy use, eutrophication and acidification potential and land occupation are the most common ones. In general, other variables being constant (intensive, extensive, conventional or organic rearing systems), the cow calf-beef production system has greater impact than beef production systems. The phase with greatest impact is animal rearing, due to the emission of enteric CH_4 and NH_3 and N from animal excreta, the major source of environmental loads. Data availability remains a complex problem, as witnessed by the considerable time dedicated in all studies to system definition and inventory construction. LCA analyses built on primary data are not common, especially because of the beef production system complexity and the broad variability of climate conditions. Almost all the studies analysed in this review (n = 34) use data collected and developed by third party organisations (national statistics agencies, non-government organisations, professional associations, etc.), derived from literature, or collected from dedicated LCA databases (LCAFood or Ecoinvent). Data uncertainty and LCIA result evaluation are almost wholly absent from the analysed studies, with the exception of Casey and Holden (2006), Weidema et al. (2008), Pelletier et al. (2010), Nguyen et al. (2010), Foley et al. (2011), Bonesmo et al. (2013) and Roer et al. (2013), all of whom report the evaluation of uncertainty and sensitivity of both input data and LCIA results.

5.3.3 Strategies to Mitigate the Impacts

Given the prospects of growth in consumption of meat and milk by 2050 (FAO 2006), debate on how to produce animal products in a sustainable way is taking place among the scientific community. Environmental performance improvement options can be classified, according to their effects, in two main areas: agricultural improvements and rearing and breeding technologies enhancing environmental performance of the fostering phase. In relation to the first aspect, agronomic production techniques may be improved by substitutions of current inputs, such as chemical pesticides and fertiliser, with lower impact inputs, such as organic ones, and by replacement of fossil energy sources with renewable; it is possible to reduce nitrate leaching, N_2O and ammonia emission, by planting catch crops during winter and reducing liquid manure pH. Land use may be reduced, thus improving growing practice: cereal yields can easily be increased by increased input of fertiliser, plant protection agents, better management and intensive cereal cultivation in low yield areas. Multiple use of such cultivation techniques produces a growth of emissions per ha but, as a result of the increased fertilisation, production increases too: the emissions per ton of cereal produced will decrease and, as many authors underline, the overall effect on cereal production may be a reduction in land use and ammonia emissions with only small changes in other emissions (Weidema et al. 2009). Animal husbandry has a strong impact in all categories. The most important pollutant in the impact categories acidification, terrestrial eutrophication, and respiratory inorganics is ammonia, mostly generated by manure production and handling. Beef fattening diets are generally well balanced and have a low N content that cannot be further reduced, especially as regards grazing animals. On the other hand, ammonia emission from liquid manure represents a problem only in beef fattening units, and can be very limited (up to 60 or 70 %). Limiting ammonia emissions provides a manure richer in ammonia N for plant fertilisation and saves on chemical fertilisers. Nitrogen leaching is responsible for aquatic eutrophication impacts that can be reduced by optimised protein feeding and by the use of manure N as a substitute for fertilisers, resulting in less leaching of N and fewer N_2O emissions.

Management of feedlot and ranch can improve environmental performance of breeding farms. In particular, Capper (2011) underlined that reducing time-to-slaughter may reduce CO_2 eq emission because the growth phase requires more energy than fattening. The methane and the dinitrogen oxide from enteric rumen fermentation in cattle contribute equally to about 90 % of GHG emissions. Methane emission is correlated with fatty acid diet contents so addition of fats to cattle feed can have a positive environmental effect in relation to global warming potential (Grainger and Beauchemin 2011). GHG emission can also be reduced by the use of liquid manure for biogas production to reduce consumption of energy from fossil sources. This has a threefold effect; according to Sommer et al. (2001), the methane emission from the manure will be reduced by 40 % or 1.1 kg methane per Mg manure, the N_2O emissions will be reduced by 14 g per Mg manure, and, at the same time, the methane produced will replace energy from fossil sources and thereby reduce the overall contribution to global warming.

5.3.4 *Other Methodological Measures and Innovative Tools for Product Environmental Assessment: Carbon, Water and Land Footprints*

5.3.4.1 Carbon Footprint

The Carbon Footprint (CF) shows the amount of greenhouse gases (GHG) emitted during a product's lifecycle (Röös et al. 2013). This environmental impact indicator is an increasingly important method for reporting the climate change impacts of food production and is fast becoming one of the key indicators of environmental sustainability. With regard to the livestock sector, several studies, focussed on the evaluation of the CF of different beef products, have been carried out. Edwards-Jones et al. (2009) evaluated the impact of beef products in the UK using primary data from three farms. Within a system that considers GHG produced from cradle to farm gate, producing 1 kg of lamb releases on average almost 3 kg CO_2 eq/kg live weight and for the production of 1 kg of beef they estimated 3.15 kg CO_2 eq/kg live weight of GHG emissions. With wider system boundaries, that included production of farm crops for animal feed, the amount of GHG emitted was almost 15 times higher for both lamb and beef (Edwards-Jones et al. 2009). Likewise, differences in the amount of GHG emitted from beef production depend on the cattle farming system used (intensive fattening, extensive pastoral, etc.). Nijdam et al. (2012) reviewed 15 LCA studies on beef production in a variety of cattle farming, finding that the production of 1 kg of extensively farmed beef results in three to four times as many greenhouse gas emissions as the equivalent amount of intensively farmed beef. According to these authors, the differences in feed transformation efficiency are higher in intensive systems; but for both systems, they found that methane from enteric fermentation and emissions from manure are, by far, the most important contributors to the CF (Nijdam et al. 2012). Few studies use empirical methods; usually, the GHG emissions of livestock production systems are calculated with the standardised IPCC approach (Tier 2). Ridoutt et al. (2011) used this approach to assess the GHG emissions from beef production in Australia, extending the GHG emissions calculation to agricultural soils after inorganic nitrogen fertiliser application and to the residue of cultivated leguminous pastures. A hybrid approach was used by Peters et al. (2010a) to perform an environmental life cycle assessment of Australian red meat production. Detailed on-site process modelling and input-output analysis were used to build a Life Cycle Inventory (LCI) and to assess the CF and the total energy consumption of three different Australian supply chains. They compared the grass-fed with the lot-fed systems, finding lower total GHG emissions for the latter; the additional effort in producing and transporting feeds was effectively offset by the increased efficiency of meat production in feedlots.

5.3.4.2 Water Footprint

Freshwater consumption is another relevant impact of agriculture and related production activities, accounting for around 70% of global freshwater withdrawals (UNESCO-WWAP 2009). Consequently, several studies in recent years have focussed on the application of a single indicator LCA-based WF in agriculture, in order to find a possible solution for reducing the pressure on freshwater resources from agriculture and food production. Ridoutt et al. (2012b) applied an LCA-based WF calculation method to the Australian beef cattle production system. Taking data directly from farms, they selected six geographically defined production systems, in order to cover a broad range of production method (pasture and feedlot finishing), product (yearling through to heavy steers), environment (high-rainfall coastal country to semi-arid inland country) and local water stress (Ridoutt et al. 2012b). All the flows from surface and groundwater into the farming system were included in the LCI. Moreover, the reduction in flows from the farming land base to surface and groundwater as a result of the operation of farm dams used for livestock watering was considered, together with the direct use of water in animal rearing and the water use associated with the production of all the inputs entering the system. To calculate the water footprint in units of L H_2O eq, they multiplied each spatially differentiated instance of water use by the locally relevant WSI and divided it by the global average WSI (0.602) (Ridoutt et al. 2012b).

5.3.4.3 Land Footprint

An innovative approach to land use in LCA analysis of beef production sectors has been proposed by Ridoutt et al. (2012, 2013). They consider it from a qualitative rather than only a quantitative approach (e.g. m^2.yr), and suggest the net primary productivity of potential biomass (NPP0, g C.m^2.yr−1) as an indicator to account for land's intrinsic productivity capacity. Comparing six beef production systems (from cradle to farm gate), they report a variability of NPP0 for kg of live weight between 86 and 176 m^2.yr-e, where 1 m^2.yr-e is 1 m^2 of land occupied on global average NPP0 (Ridoutt et al. 2012). According to the authors, this indicator, called the land use footprint, is easy to calculate from existing databases and allows us to consider the different pressure exerted globally on the land resources, depending on productivity (Ridoutt et al. 2012). In a further study, Ridoutt et al. (2013) propose a normalisation step, in order to make the different life cycle impact category indicators comparable. In particular, they perform the normalisation of the carbon footprint (Ridoutt et al. 2011), the water footprint (Ridoutt et al. 2012) and the land use footprint (Ridoutt et al. 2012, 2013) in relation to beef production systems in Australia by using the global economic system for 1995 to 2000 as reference (Ridoutt et al. 2013). Although they find no correlation between these indicators, their study is a first attempt to overcome the lack of comprehensiveness when considering indicators as stand-alone environmental indicators.

5.3.5 Comparative Analysis of Different Types of Breeding and Final and Processed Products

Using data on the composition of the entire beef production system in the EU−27 (Weidema et al. 2008), Nguyen et al. (2010) studied four systems (one suckler cow-calf system and three dairy bull systems). The results of this study show that the suckler cow-calf system has the lowest environmental efficiency because the higher quantity feed for kg of meat is followed by higher manure production. The dairy bull system with calves slaughtered at 12 months emerged as the most efficient (Nguyen et al. 2010). Furthermore, in another study analysing the most common beef production system in France, Nguyen et al. (2012a) focussed on the possible scenarios for GHG emissions reduction. The analysed scenarios included changes in grazing management and in herd and diet management, as well as a combination of all such strategies. As regards the 10 alternative scenarios, the authors also found that their combination could reduce the current impact of 13–28 % per kg of live weight (Nguyen et al. 2012b). Three beef production systems in the USA were analysed by Pelletier et al. (2010): (1) directly weaned calves in the herd, (2) weaned calves on grazing which ended up in feedlots and (3) calves finished directly on pasture. They found that the last resulted in the highest impacts in terms of cumulative energy use, ecological footprint, greenhouse gas emissions and eutrophying emission impact categories. The environmental efficiency, in terms of non-renewable energy consumption and GHG emissions, of three specialised and two mixed (crop-livestock) farms, was studied by Veysset et al. (2010), in the search for management options for income maximisation. For the assessment of the economic and environmental performances of the systems they used two models: Opt'INRA to optimise the economic input, and PLANETE to assess the environmental performances. From the economic perspective, the authors found higher efficiency in mixed crop-livestock farms, because of higher management flexibility, especially in crop-based rather than grassland-based farms. However, from the environmental perspective, also crop-based farms had restricted opportunities in non-renewable energy consumption and GHG production improvement, due to the reduced number of possible solutions found by the authors, for the three studied systems (Veysset et al. 2010).

5.3.6 Hotspots

The difference in beef production systems is determined by a series of farm characteristics, including the rearing species, the number of animals, the type of production (milk or meat), the rearing system (conventional or organic; intensive or extensive, etc.), manure management, the presence or absence of agricultural activities for feed production supporting the livestock system. All the selected studies found that animal rearing was the phase of beef production system with the greatest impact (Peters et al. 2010a); this was mainly caused by the emission of enteric CH_4,

NH_3 and of N from animal excreta to be the major cause of environmental loads (Ogino et al. 2004, 2007; Casey and Holden 2006; Cederberg et al. 2009; Beauchemin et al. 2010; Nguyen et al. 2010; de Vries and de Boer 2010; Basarab et al. 2010; Veysset et al. 2010; Foley et al. 2011; Nijdam et al. 2012; Oishi et al. 2013). Many studies therefore combine LCA analysis with models for the optimisation of farm resources (e.g. Beauchemin et al. 2010; Pelletier et al. 2010; Nguyen et al. 2010) or direct their attention to the assessment of environmental impacts depending on different diets or rearing techniques (e.g. Ogino et al. 2004, 2007; Basarab et al. 2010; Doreau et al. 2011; Nguyen et al. 2012a, b; Clarke et al. 2012; Ridoutt et al. 2012). Another relevant issue in environmental impact assessment of beef production and the whole livestock sector is land occupation. Usually, land use and land use changes are considered in LCI as the amount of land occupied by processes or by raw material production (or extraction). This is also the case for beef production, whereby the land use impact category is measured in terms of the m^2 of land required to produce a certain amount of meat in a defined period of time. Moreover, the literature also suggests that LCA coupled with other approaches provides much more comprehensive information for environmentally conscious policy-makers, producers, and consumers in selecting sustainable products and production processes (Roy et al. 2009). Thus, the integration of LCA analysis with farm economic efficiency models has assumed major importance in the livestock sector and beef production in recent years (Beauchemin et al. 2011; Oishi et al. 2011; Clarke et al. 2012), although others use different methods to assess the environmental loads generated by livestock production (Veysett et al. 2010).

5.4 Dairy Cattle

5.4.1 Literature Review on LCA Application to Milk and Dairy Products and Problematic Approaches

Several research studies about the application of LCA methodology to milk and dairy products have been published in the last 10 years. In this paragraph a critical review of the literature LCA studies regarding the evaluation of the environmental performance of milk and other dairy products is reported, in an attempt to summarise the main issues, both methodological and technical, that these studies highlight. The selection of peer-reviewed LCA articles for inclusion in the comparative analysis was based mainly on the year of publication and on the main scope of the studies; the older studies (prior to 2010) were excluded because of the large number of available articles and the fact that their results are often used as primary data or for comparisons in more recent studies. In addition, the older studies are already discussed in some LCA reviews (de Vries and de Boer 2010).

According to these selection criteria, milk is the most studied dairy product (seven studies), followed by cheese (four studies) whereas only one article about yogurt has been reviewed. Among the twelve works analysed, two considered only the carbon footprint of the product (Thoma et al. 2013; Yan et al. 2011), and the remaining ten assessed a higher number of indicators. It is evident that the older studies about milk production (Castanheira et al. 2010) assessed only raw milk production, whereas more recent ones have tried to probe more deeply. First, some studies enlarged the system boundaries up to the processing plant (Gonzales-Garcia et al. 2013) or up to the end of life of the product (Thoma et al. 2013). Furthermore, some research papers compared the environmental performance of different farming approaches, focussing on the differences between intensive and extensive systems and organic and traditional ones (O' Brien et al. 2012; Yan et al. 2011; Guerci et al. 2013). One study compared the results of a traditional LCA with those of the Environmental Product Declaration (EPD) of milk in the International EPD® System (Fantin et al. 2012).

Without exception, on-farm activities have been found to be the main environmental hotspots for milk production, followed by the production of feed, particularly concentrates; seasonal-grass based systems can have a lower impact thanks to their lower resource use and their production of fewer pollutants from concentrate feed compared with forage and shorter manure storage periods. As regards other dairy products, five studies have been analysed: one about yogurt (Gonzales-Garcia et al. 2013b), and four about cheese (Gonzales Garcia et al. 2013c, d; Kim et al. 2013; Van Middelaar et al. 2011). The two articles about cheese written by the same author (Gonzales-Garcia) adopted a similar approach, although treating different kinds of cheese. All these studies agreed that the production of milk is the main hotspot for the most common impact categories considered: global warming, eutrophication, acidification and photochemical ozone formation potentials. However, in the majority of the works, the authors tried to identify the hotspots of cheese production over which the manufacturer has direct control.

An interesting approach is proposed by Van Middelaar et al. (2011), who tried to make a combined economic and environmental evaluation of cheese by using the parameter eco-efficiency, which expresses the gross value added of a unit of environmental impact (global warming, land use or energy use).

The critical analysis of the LCA studies showed that, similarly to other products derived from livestock, the handling of multifunctionality, frequently solved by applying allocation approaches, is one of the main critical issues in the environmental assessment of cheese and milk. In fact, the environmental load of milk production has to be divided between all the outputs of the rearing process: milk, meat and skin. Furthermore, the dairy factory generally produces more than one product, which implies that the whole impact should be allocated among all of them. Another problem related to the assessment of the environmental performance of dairy products transformed in medium and large dairy plants is that they often use milk supplied by different farmers with different rearing systems. Obtaining primary data from all of them is frequently a problem, and thus it is common practice to include in the inventory analysis primary average data obtained from a representative sample of farms or information from the literature. (Table 5.3)

Table 5.3 List of references included in the literature review and their main characteristics

Reference	LCA	Other tool	Product
Castaneheira et al. (2010)	X		Milk
Fantin et al. (2012)	X		Milk
Gonzales-Garcia et al. (2013a)	X		Milk
Gonzales-Garcia et al. (2013b)	X		Yogurt
Gonzales-Garcia et al. (2013c)	X		Cheese
Gonzales-Garcia et al. (2013d)	X		Cheese
Guerci et al. (2013)	X		Milk
Kim et al. (2013)	X		Cheese
O' Brien et al. (2012)	X		Milk
Thoma et al. (2013)	X	CF	Milk
Van Middelaar et al. (2011)	X	Eco-efficiency analysis	Cheese
Yan et al. (2011)		CF	Milk

5.4.2 Methodological Problems Connected with the Application of Life Cycle Assessment for Dairy and Dairy Products: Critical Analysis of International Experiences

5.4.2.1 Goal and Scope

Most of the articles analysed had a similar purpose, i.e. to evaluate the potential environmental burdens of milk and cheese production chains. Furthermore, some authors compared different production or farming systems or included economic evaluations such as economic efficiency and the evaluation of a benchmark. The main goals of the studies about milk production analysed here were both to assess the potential environmental impact of the product (Gonzalez-Garcia et al. 2013a; Castanheira et al. 2010; Thoma et al. 2013) and to compare different agricultural or breeding management systems (Yan et al. 2013; Guerci et al. 2013; O'Brien et al. 2012).Only one study compared the results of LCA and an EPD for the same product and performed a critical analysis of the existing product category rules (Fantin et al. 2012).

The goals of Gonzalez-Garcia et al. (2013a), Castanheira et al. (2010) and Thoma et al. (2013) were similar, though Thoma et al. (2013) were focussed only on the carbon footprint of milk production. Gonzalez-Garcia et al. (2013a) aimed at evaluating the environmental performance and the energy balance of the production of UHT milk in Portugal and at identifying the hotspots in the production chain. The authors chose a Portuguese dairy factory, using best available technologies (BATs) for the assessment. Castanheira et al. (2010) aimed to identify the processes with the largest environmental impact, and considered a typical Portuguese dairy farm. The goal of Thoma et al. (2013) was to determine GHG emissions associated with consumption of 1 kg of milk by US consumers. Yan et al. (2013), Guerci et al. (2013) and O'Brien et al. (2012) applied LCA methodology to different farming systems. Yan et al. (2013) performed a carbon footprint study, the purpose of which was to compare two systems for milk production in a grass-based, rotational grazing system:

one used nitrogen fertilisers for pasture production and the other used white clover, which is an alternative to nitrogen fertilisers (applies biological nitrogen fixation). Guerci et al. (2013) aimed at assessing the environmental impacts of milk production of different farming systems (organic versus conventional, confinement systems versus pasture systems and different annual production levels) and at identifying their strengths and weaknesses. Finally, O'Brien et al. (2012) compared the environmental impacts of seasonal grass-based and confinement dairy farms, following an LCA approach. The main purpose of Fantin et al. (2012) was to compare the environmental performance of milk production with the published EPD of a similar product by following the requirements of the PCR document for milk of the International EPD® System and critically analysing it. In fact, the authors discussed the main key issues affecting the comparability of different EPDs for the same product.

As regards dairy products, Gonzalez-Garcia et al. (2013b) focussed their analysis on the assessment of environmental impacts and energy balance from the production of different types of yogurt.

The studies about LCA of cheese stemmed from different needs but the goals and scope of the studies analysed show some similarities. The main difference is that Gonzales-Garcia (2013c, d) and van Middelaar et al. (2011) focussed on the environmental performance of a specific product with the aim of quantifying its environmental impact and identifying the most impacting processes. Furthermore van Middelaar et al. (2011) assessed the ecological impact in the context of the economic efficiency through the eco-efficiency parameter. Kim et al. (2013) performed a more strategic analysis, aiming at defining a benchmark for the US cheese producers and at providing stakeholders with information about the environmental impact of cheese.

5.4.2.2 Functional Unit

In milk and dairy product LCAs, two main approaches towards functional unit definition are presented: the first considers only the mass of the product regardless of its composition and its water content; the second one takes into consideration the nutritional value of the product, normalising the mass to a certain energy or fat and protein content. It should be pointed out that the choice of a "corrected functional unit", such as fat and protein or energy content, could be an efficient approach for covering the nutritional value of dairy products as well and could allow comparison of the results of different studies.

Three studies among seven on milk production referred to 1 kg of energy corrected milk (ECM), a correction factor used by the dairy industry to determine the amount of energy contained in milk and based on fat and protein content (Gonzalez-Garcia et al. 2013a; Yan et al. 2013; Guerci 2013). Three LCAs referred to a certain amount of product (1 L, 1 kg or 1 t) (Fantin et al. 2012; Thoma et al. 2013; Castanheira et al. 2010). The comparative analysis by O' Brien et al. (2012) was based on different functional units: 1 t of fat and protein corrected milk (FPCM), 1 t of milk solids (MS), the on-farm area occupied and the total area occupied.

Concerning dairy products, Gonzalez-Garcia et al. (2013b) referred to the production of 1 t of yogurt ready for consumption. In the studies about cheese production, the most commonly adopted functional unit was 1 kg of cheese (Gonzales-Garcia et al. 2013c; d; van Middelaar et al. 2011). Only one study among the four analysed referred to 1 t of cheese on a dry weight basis (Kim et al. 2013).

5.4.2.3 System Boundaries

As regards studies on milk production, the most common approaches regarding system boundaries definition are from cradle to farm gate and from cradle to gate. The former included only farms' activities until raw milk production (Castanheira et al. 2010; O'Brien et al. 2012; Yan et al. 2013; Guerci et al. 2013). On the other hand, the latter took into account the pasteurisation and packaging processes at dairy plants, excluding the distribution and use phases (Gonzalez-Garcia et al. 2013a; Fantin et al. 2012). Moreover, a third kind of study considered a cradle-to-grave approach (Gonzalez-Garcia et al. 2013b; Thoma et al. 2013).
The production of capital goods (machinery and buildings) and road infrastructures were excluded from all these studies as well as land use and soil quality changes caused by cultivation-related activities. Regarding the studies on LCA of cheese, different approaches towards system boundaries definition were adopted. In particular, the production of milk was common to all the studies analysed, whereas the other cheese production phases considered vary. Gonzales-Garcia et al. (2013a, b) analysed the process from farm to cheese manufacturing plant gate. Kim et al. (2013) considered all the processes of the life cycle, from cradle to grave, and finally van Middelaar et al. (2011) evaluated the phases between milk production and sale.

5.4.2.4 Availability and Quality of Data

As far as data quality is concerned, the studies on milk production can be divided into two main groups: studies using primary data for farms or dairies (Fantin et al. 2012; Guerci et al. 2013; Yan et al. 2013; Gonzalez-Garcia et al. 2013b); studies using secondary data for farms (Castanheira et al. 2010; O' Brien et al. 2012; Gonzalez-Garcia et al. 2013a, b; Thoma et al. 2013).

Castanheira et al. (2010) and O'Brien et al. (2012) did not use primary data, but only secondary ones from previous studies. Moreover, in Gonzalez-Garcia et al. (2013a, b) data regarding the foreground processes for the production of raw milk were obtained from Castanheira et al. (2010) The foreground data for dairy factories consisted of average annual data obtained by on-site measurements. Thoma et al. (2012) used data collected from several sources such as the USDA's National Agricultural Statistical Service and Economic Research Service, peer-reviewed literature and other technical literature and an extensive nationwide survey of dairy farm operations. On the other hand, Fantin et al. (2012) used primary data for both farm activities and dairy processing and packaging. Gonzalez-Garcia et al. (2013b) used primary data from the dairy factory regarding transportation to wholesale and retail stages, whereas they use literature data for the use phase. The authors of two studies collected primary data from a large sample of farms: Yan et al. (2013) used data obtained from experimental systems in 16 Irish farms; Guerci et al. (2013) collected data from 12 dairy farms, five from Denmark (two of which were organic), two from Germany which differed in their summer feeding systems (confinement vs. pasture), and five from Italy (all of which used confinement feeding). All studies used literature data and the Ecoinvent database for background data.

A critical methodological issue in LCAs of dairy products is often the calculation of methane and nitrogen emissions because of the management and agronomic use of chemical and organic fertilisers, such as manure and slurry. Methane emissions from enteric fermentation and manure management and emissions of nitrous oxide, nitrogen oxides and ammonia from manure management were generally calculated according to IPCC 2006 and EMEP/EEA Corinair 2009 (Castanheira et al. 2010; Gonzalez-Garcia et al. 2013a, b; Fantin et al. 2012; Guerci et al. 2013). On the other hand, O'Brien et al. (2012) excluded the emissions of manure in pastures. When considered, phosphorus emissions were often calculated in accordance with Nemeck and Kagi suggestions (Fantin et al. 2012; Guerci et al. 2013).

Nitrogen and carbon dioxide emissions from livestock respiration were not taken into account in any study. Carbon dioxide sequestration by crops was accounted for in Guerci et al. (2013) but it was not considered by O'Brien et al. (2012), Gonzales-Garcia et al. (2013c) or Fantin et al. (2012).

Data about the production of milk employed in the cheese manufacturing plants were not always available for LCA analysis. Among the four studies considered, two (Gonzales-Garcia et al. 2013c; van Middelaar et al. 2011) used primary data for the production of milk, whereas the remaining two (Gonzales-Garcia et al. 2013d; Kim et al. 2013) used secondary data from scientific literature (Castanheira et al. 2010; Thoma et al. 2012). However, in the two cases where primary data were used, milk was provided to the cheese manufacturer by different farms, but only a few, representative of the average situation, were included in the evaluation. Data related to the cheese production process derived from a primary source in all the studies included in this literature review. Similarly to milk production, background data were generally taken from the literature or databases, particularly the Ecoinvent database.

5.4.2.5 Allocation Methods

As regards milk production, different approaches were applied to solve multifunctionality problems: no authors used system expansion or substitution, whereas several studies applied allocation, mainly on an economic or biological basis. Only one study (Fantin et al. 2012) adopted a conservative approach and allocated all impacts to milk production.

Yan et al. (2013) applied economic allocation for concentrate feed production and economic allocation between milk and meat. In Gonzalez-Garcia et al. (2013a), economic allocation was applied in the case of dairy farms in order to partition the environmental burdens between meat and milk, which are based on historical market prices in Portugal. As regards the dairy factory studied by Gonzalez-Garcia et al. (2013a), different types of UHT milk were produced: simple milk and cocoa milk, as well as butter and cream. Nevertheless, the authors considered the whole system as a black box and applied mass allocation for the assessment of the co-products (milk, cream and butter). Guerci et al. (2013) and Thoma et al. (2013) used biological allocation, based on the feed energy required to produce the amount of milk and meat at farm level. Moreover, Thoma et al. (2013) used economic allocation for feed crop processing, and mass balance of milk solids (fat and protein contents) for the allocation between milk and cream.

In the production of yogurt two main co-products have been identified (Gonzalez-Garcia et al. 2013b): yogurt and animal fodder, to which impacts were allocated following a mass-based partitioning approach.

In the production of cheese two main multi-output processes have been identified: the production of milk that also implies the production of meat, manure, calves and skin, and the manufacture of cheese, generally accompanied by the production of whey and other co-products, such as cream. In three studies reviewed (Gonzales-Garcia et al. 2013c, d; van Middelaar et al. 2011) the allocation of the impacts of the farm is done on an economic basis. The allocation factor for milk is explicated in the two studies (Gonzales-Garcia et al. 2013a; van Middelaar et al. 2011) where primary data are used to model the milk production, which is respectively equal to 92 and 87%.

The allocation of impacts caused by cheese manufacturing plants is managed in different ways depending on the study. That which produces San Simon da Costa cheese (Gonzales-Garcia et al. 2013c) did not have specific equipment for whey processing, which was sent to the wastewater treatment plant. Therefore in this case cheese was the only output of the system and no allocation was needed. The authors also analysed an alternative scenario with whey valorisation in which they allocated the impact to the whey and cheese according to their fat content. Gonzales-Garcia et al. (2013b) analysed the effect of different allocation approaches and found that mass allocation improved the impact of cheese compared with the economical one, because the economic value of whey per unit of mass is lower than that of cheese. Gonzales-Garcia et al. (2013d) and van Middelaar et al. (2011) subdivided the impact of cheese production according to the economic value of co-products. Kim et al. (2013) adopted another approach: the impact of milk was subdivided between the co-products on the basis of their fat and protein content, whereas the environmental load of all the other materials, such as steam and electricity, was allocated following the economic approach.

5.4.2.6 Life Cycle Impact Assessment (LCIA)

The studies analysed applied different impact assessment methods, which included different characterisation factors and environmental indicators. However, the studies reviewed display a certain level of coherence in their choice of impact assessment method. The majority of the LCAs reviewed considered more than one impact category, whereas two studies among twelve assessed only the potential impact on climate change (Yan et al. 2011; Thoma et al. 2013). The potential impact on global warming was without exception evaluated in line with the IPCC guidelines (IPCC 2006).

The characterisation method most commonly adopted was the CML 2001 (Gonzales-Garcia et al. 2013a, b, c, d; Castanheira et al. 2010; O'Brien et al. 2012). These authors focussed their analysis on the following impact categories: abiotic depletion (ADP), acidification (AP), eutrophication (EP), global warming (GWP), ozone layer depletion (ODP), photochemical oxidant formation (POFP). Only two studies considered the impact on land use (Gonzales-Garcia et al. 2013b; Guerci et al. 2013). The toxicological impact categories and the cumulative energy demand (CED) are evaluated by Gonzalez-Garcia et al. (2013a, b). It must be noted that water depletion was not considered in any of the studies analysed. On the other hand, both Fantin et al. (2012) and Guerci et al. (2013) selected the impact assessment methods recommended by the International EPD system. Two studies evaluated biodiversity (Guerci et al. 2013; Kim et al. 2013). Finally, Kim et al. (2013) utilised the ReCiPe method (Goedkoop et al. 2009) and the Usetox (Rosenbaum et al. 2008) whereas van Middelaar et al. (2011) considered the model proposed by Thomassen et al. (2009). Two studies related the environmental impacts to eco-efficiency analysis by means of the gross value added of the product (Van Middelaar et al. 2011) or combined them into a single score expressed in monetary units (Guerci et al. 2013).

5.4.2.7 Critical Analysis

The studies analysed often performed sensitivity analyses to evaluate improving actions in the dairy production chain, discussed the effect of allocation approaches on total LCA results or compared the results with literature on LCA studies of the same product. Gonzalez-Garcia et al. (2013b) discussed, via sensitivity analysis, some improvement actions, which could contribute to the reduction of the overall environmental performance of yogurt production. As regards raw milk production, optimised farm management can lead to the reduction of the potential environmental impact. The main improvements that can be applied to farm activities are:

- increase the consumption of grass silage instead of maize silage;
- lower the use of concentrates or use concentrates with a lower environmental impact (e.g. domestic or regionally produced rapeseed meal instead of imported soybean meal);
- reduce the use of high protein concentrate meals in order to lower nitrogen losses;
- increase the length of the grazing season in order to reduce the storage of manure and store manure under aerobic conditions and target and reduce N fertiliser application (Gonzalez-Garcia et al. 2013b; O' Brien et al. 2012).

As regards the dairy factory, the minimisation of milk losses (which involves the increase of total yogurt production and the reduction of the co-product dairy fodder at the same time), the reduction in the total energy requirements in the dairy factory, the use of gas fuelled boilers instead of oil fuelled ones, reductions in travel distances and energy consumption in both retail and consumption phases would lead to environmental improvements. On the other hand, the consideration of dairy fodder as an avoided product does not allow reductions in any impact categories (Gonzalez-Garcia et al. 2013b, d; van Middelaar et al. 2011). The recovery of whey in cheese production plants increases the total environmental impacts of the processes analysed, except for the impact category EP, thanks to the reduction of phosphate emissions. However, it should be recognised that the system analysed releases two value-added outputs and the impact allocated to cheese according to the fat content is lower than in the base case scenario (Gonzales-Garcia et al. 2013c). Increasing the cheese ageing period would lead to a worsening of the results (Kim et al. 2013). Furthermore normalised results highlight that cheese production mainly affects the categories aquatic eutrophication, aquatic ecotoxicity and terrestrial acidification, and it is possible to lower these impacts through energy conservation and water conservation/treatment activities (Kim et al. 2013). Gonzalez-Garcia et al. (2013a), Fantin et al. (2012) and Castanheira et al. (2010) compared their results with other literature studies on milk production. They stated that their results fall within the range of literature values. Moreover, they found that the main flows affecting the results are the same, although their contributions to the total results are different because of the different assumptions made and models used in the studies. However, Gonzales-Garcia et al. (2013a) identified some differences from other studies, such as the allocation approach, data sources, characterisation factors, farm management practices and enteric fermentation emission factors, which do not allow a comprehensive and detailed comparison with LCA literature results. Consequently, when the results of different studies on the same product are compared, these aspects should be taken into account, and a sensitivity analysis which considers the assumptions and uncertainty of the results should be performed (Gonzalez-Garcia et al. 2013a).

Yan et al. (2013) found that the difference in carbon footprint between the systems investigated is in agreement with other similar studies (the other three studies show that white clover reduces the carbon footprint of milk) and discussed the main differences (carbon sequestration, stocking density). The authors also performed a ratio sensitivity analysis which examined the effect of emission factors on the comparison between the two systems. The analysis revealed that to reverse the ranking of white clover and nitrogen fertilisers systems, changes to emission factors and assumptions had to be much greater than the uncertainty ranges found in the literature.

As regards an allocation approach, Gonzalez-Garcia et al. (2013a) discussed the effect of different allocation methods among milk, cream and butter on the total life cycle results: in addition to a mass allocation approach, the authors performed a sensitivity analysis in which economic and protein-based allocations were applied to the system. The results showed that economic allocation improved the environmental performance of milk production by 34 %, whereas protein-based allocation worsened the results by up to 5 %. Gonzales Garcia et al. (2013b) analysed the effect of different allocation approaches and found that mass allocation improved the impact of cheese more than the economical one, because the economic value of whey per unit of mass is lower than that of cheese.

Guerci et al. (2013) performed two correlation analyses: the first on the impact categories and the second between impact categories and main parameters of dairy farms. The former showed strong and positive relations between GWP, acidification, eutrophication and energy use, whereas land use was negatively related to the four categories. The latter found that feed efficiency affected several impact categories (significant negative correlation with global warming, acidification and eutrophication). This supports the theory that better animal efficiency (in terms of feed conversion rate) is one of the ways of reducing the environmental impact in milk production. A positive relation was observed between GWP, acidification, energy use, biodiversity and the amount of grassland of the farmed area, whereby the farms with the largest amount of grassland had cows grazing during the summer season. Overall, the results of the study showed that the improvement of greenhouse gas emissions would lead to an improvement in the environmental performance of the dairy farm.

Van Middelaar et al. (2011) assessed the eco-efficiency of the processes of the supply chain, expressing the gross value added per unit of environmental impact (GWP, land use and energy use). and found that. the least eco-efficient product production process is concentrate production.

5.4.2.8 Environmental Hotspots

On-farm activities were found to be the main environmental hotspots for milk and other dairy productions regardless of the impact category considered (Gonzalez-Garcia et al. 2013a, b, c, d; Fantin et al. 2012; Guerci et al. 2013; O' Brien et al. 2012; Castanheira et al. 2010; Van Middelaar et al. 2011; Kim et al. 2013). Larger contributions from the farm subsystem are made by enteric fermentation, the production of animal feed, airborne and waterborne emissions from farm activities as well as manure management and spreading (Gonzalez-Garcia et al. 2013a, d; Fantin et al. 2012; Guerci et al. 2013). As regards the proportion of the different compounds and their environmental impacts, the global warming and the photochemical ozone formation potentials of milk production are primarily influenced by methane emissions because of enteric fermentation and manure management and secondarily by feed production (Castanheira et al. 2010; Fantin et al. 2012). Particularly, enteric methane, manure deposition, fertiliser spreading, fertiliser production, electricity production, indirect nitrous oxide emissions, slurry storage, concentrate production, and slurry spreading account for 95 % of the total GHG emissions (Yan et al. 2013). The nitrogen that volatilises in the form of ammonia from manure and fertilisers significantly affects the acidification potential and, to a lesser extent, the eutrophication potential, which is mainly influenced by nitrate emissions (Castanheira et al. 2010; Fantin et al. 2012).

Concerning the different dairy cow feeds, concentrates have the higher environmental load (Castanheira et al. 2010; O' Brien et al. 2012). Furthermore the CF of milk is 11–23 % lower for white clover systems compared with nitrogen fertiliser systems because of the fact that methane, carbon dioxide and nitrous oxide emissions are significantly higher for the latter (Yan et al. 2013). The most extensive use of land is for organic farms, supporting the theory that such farms generally need more land to produce feed because of their lower crop yields The farms that had the lowest impact on biodiversity losses were organic (Guerci et al. 2013). In addition, a simplified sensitivity analysis performed by Guerci et al. (2013) showed that, accounting for the emissions from direct land use change would increase the impact of conventional farms, whereas it would remain the same for organic farms. The same authors also found that grasslands have an important role in GWP mitigation and in reducing biodiversity losses, especially on organic and pasture-based farms. These effects are probably because of a greater capacity for carbon sequestration. In addition, farms with more grassland are more self-sufficient in feed so they avoid the heavy impact of commercial feed production and transport on total energy consumption. The influence of grassland on lowering acidification could be because of the lower fertiliser input for this type of crop. Correlation analysis showed that land occupation is significantly reduced when the farming intensity increased (stocking rate, N surplus and use of fertiliser) and when crop production on the farmland increased (Guerci et al. 2013).

As regards dairy plants, energy consumption (electricity, fossil fuels), packaging production, transport-related activities and on-site emissions are the main contributors to the environmental impacts of this phase (Gonzalez-Garcia et al. 2013a; Fantin et al. 2012).

Concerning other dairy products, the production of powdered and concentrated milk needed for yogurt production is the main hotspot for the dairy factory phase, mainly because of the high energy consumption required for their production processes. Moreover, it was found that the production of packaging materials and energy requirements contributes significantly to the yogurt environmental profile. Finally, the distribution phase, consumption at the household and final disposal showed a low contribution (Gonzalez-Garcia et al. 2013c). Although the production of milk is the main environmental concern of cheese production, Gonzales-Garcia et al. (2013c, d) and Kim et al. (2013) focussed on the environmental impact of cheese manufacturing plants. The main contributions to GWP by the manufacturing plant are related to the combustion of fossil fuel both for energy production and for transport (Gonzales-Garcia et al. 2013c, d; Kim et al. 2013). Furthermore, boilers fuelled with oil and wastewater treatment plants contribute significantly to ADP, AP, ODP, POCP and EP (Gonzalez-Garcia et al. 2013c; Kim et al. 2013). Finally, the smoking process performed with birch wood has an important influence on AP, POFP and GWP mainly because of the wood supply chain and combustion (Gonzales-Garcia et al. 2013c).

5.5 Sheep and Goat

5.5.1 Comparative Analysis of Life Cycle Thinking Approaches in the Sheep and Goat Sector

The literature review of LCT approaches in the sheep and goat sector was performed through consultation of scientific databases and search engines, including Scopus, Web of Knowledge, Google Scholar and Google, as well as LCA conferences and the websites of Ecosystem Assessment, FAO, and IPCC. Selected studies, published from 2010 onwards, refer to various countries. Following a homogeneous framework adopted for all sectors addressed in this book, the review tries to show the prospects and constraints of LCT as a tool to assess the environmental impact of sheep and goat production. There is less research, globally, on LCT involving the sheep and goat sector compared to other livestock sectors, possibly because of the great diversity of situations (including species, products, and intensity of land used), or because of its secondary economic importance and low political weight, as de Rancourta et al. (2006) argue about Mediterranean and other European areas. The literature on the environmental assessment of sheep and goat production appears to be focussed on the differences in the livestock systems, and there are few, if any, studies considering the wider systemic perspectives of both, that of sustainability and that of two-way relationships.

5.5.1.1 Life Cycle Assessment Applications: State of the Art

Very few studies have addressed the environmental impact of sheep and goat sector using LCA methodology. These studies that directly or indirectly refer to the wider term "sheep and goat", including not only meat production, but also sheep and goats in general (for milk, wool and dairy products). Generally, LCA analysis of the sheep and goat sectors presents a comparison of several species and their products, often including cow, sheep and goat, and of the breeding methods used.

In the following, firstly we briefly explain the main objectives and results of the studies reviewed, than we make a detailed and comparative analysis involving each LCA step.

As argued above, few studies were found that specifically applied LCA methodology to the sheep and goat sector. The study of Head et al. (2011) showed that sheep have a high impact on biodiversity, greenhouse gases and health. Despite significantly greater land use for goat livestock, production of goat's milk had a slightly lower impact than that of cow's milk, because dairy goats require less feed. Comparing conventional and organic lamb farming, Head et al. (2011) demonstrated that conventional lamb has almost twice the impact than that of organic lamb because the former eats a greater percentage of soy-based concentrate, with negative implications for biodiversity. Organic lamb had a slightly higher negative impact on climate change and human health, than conventional lamb, because manure was used instead of chemical fertiliser on the wheat straw.

Two studies by Kanyarushoki et al. (2008, 2010) shown that per 1000 kg of, goat milk had a greater negative impact than cow milk. Moreover, goat farms had greater impact per hectare of land occupied, except on climate change (Kanyarushoki 2010). The authors estimated the emissions, non-renewable energy and land occupation of several farms through the EDEN model, which is a Microsoft® Excel-based tool (van der Werf et al. 2009). Impacts were compared using two functional units (FU): (a) per 1 t of fat and protein corrected milk (FPCM) sold; and (b) on-farm plus estimated off-farm hectares utilised. The authors decided to avoid allocation between animal and crop products and separated the farms into two parts: production of crop products not used for animal production, and all other farm processes. In the final step, economic data was used to determine the impacts of milk and animal production.

Michael (2011) applied the standardised LCA methodology to identify and evaluate the carbon footprint, water and energy efficiency of five animal product industries. The results were significantly influenced by whether the animal species was a ruminant or non-ruminant, and whether competitive feed conversion ratios were achieved. The non-ruminant animal species were highly efficient in terms of emissions compared to the ruminants, provided that the enterprises were well managed and feed conversion rates were high. Ruminants (dairy sheep and dairy goats) had a significant burden of enteric emissions and methane.

Koch et al. (2013) applied the AGRIBALYSE® programme launched by the French Environment and Energy Management Agency (ADEME) to create a Life Cycle Inventory (LCI) database of French agricultural products. AGRIBALYSE® was built with two aims: (i) to create an LCI database to provide data for the environmental labelling of food products; and (ii) to share data to enable the agricultural and food industries to assess the production chain and reduce environmental impacts. AGRIBALYSE® provided 136 LCI data sets for arable, horticultural and livestock products. The data for the production systems and direct emission was processed using Excel®, while the indirect flows were added using SimaPro® to obtain the LCI and LCIA data sets.

After this general presentation of LCA studies involving sheep and goat production, the next section (§ 5.5.2) makes a comparison between them, following LCA steps and trying to highlight the prospects and constraints of this methodology.

5.5.1.2 Other Life Cycle-Based Methodologies and Tools: The Carbon Footprint

While the literature review does not present studies combining LCA with other methodologies, such as Social LCA and Life Cycle Costing, numerous Life Cycle Thinking (LCT) approaches of the Footprint indicator Family have been used for the sheep and goat sector. Carbon footprint (CF) is one of the most common, followed by water footprint (WF) indicator. The literature review include these two indicators because of their strong similarities with and complementarities to LCA assessment (see, for example, the EC-JRC PEF Guide, 2013; Boulay et al. 2013; Fang et al. 2014) and of the international initiatives developed worldwide (presented in § 1.2.4.2 and 1.2.4.3). The review does not intend to completely describe the state of the art of the CF literature for the sheep and goat sector. We selected studies from 2010 and strictly related to LCT approach to give some insight into the main aspects relevant to LCA methodology perspectives.

Biswas et al. (2010) compared the emissions (CO_2, N_2O and CH_4) performance of three different Australian products (sheep's wool, sheep's meat and wheat) in three adjacent plots (mixed pasture, wheat and sub-clover). The system boundaries adopted a cradle to farm gate perspective and was divided into two main stages, pre-farm and on-farm. An economic allocation method was used to calculate the input and output of co-products. The input/output data of the LCI was linked to the relevant libraries in SimaPro 7. They reported that the life cycle greenhouse gas (GHG) emissions of 1 kg of sheep's wool were approximately three times higher than the GHG emissions of the sheep's meat production. on the on-farm stage contributed the most significant portion of total emissions. CH_4 emissions from enteric methane production and from the decomposition of manure accounted for a significant portion (83–90 %) of the total emissions from sub-clover and mixed pasture production. A sensitivity analysis was carried out showing that the GHG emissions were very sensitive to the fluctuation of prices of sheep meat and wool (respectively +/−3.5 and +/−14 %).

Many authors calculated the environmental impact of goat/sheep production through the joint application of the LCA approach and specific models, including:

- the Cranfield model (Williams et al. 2006);
- the EDEN model (Kanyarushoki et al. 2008; 2010);
- the Capri model (Weiss and Leip 2012);
- the Global Livestock Environmental Accounting model (GLEAM) (Opio et al. 2013).

Willams et al. (2006) proposed first the Cranfield model, which was used as a reference point by the other studies presented below. Williams et al. (2012) developed the systems model for the stratified UK sheep industry to provide the activity data input for the life cycle assessment of the Cranfield model. This includes the biophysical performance of the lowland, upland and hill sheep flocks. The LCA analysis of the production of lamb meat took into account the different sizes of the breeds and consequent feed requirements, different types of land and consequent yields of grass (and management requirements), and different rates of lamb growth and ewe productivity. The FU was a 1000 kg edible lamb carcass at the national level; the system boundary was the farm gate. Enteric methane was calculated using the IPCC (2006) Tier 2 formula, and the results were expressed as LCIs using the characterisation factor of the IPCC for GWP and of the CML for other impacts (i.e., eutrophication potential, acidification, abiotic resource use, type of land). The baseline results were compared with alternative scenarios that considered changes in sheep management, changes in genetic potential and management quality (including animal health), and in the emission factor for enteric methane. The Cranfield model was also used in Phase One of the English Beef and Sheep Production Roadmap set by a steering group of industry organisations led by EBLEX (the organisation for sheep and beef producers in England). In the second phase of the Roadmap a different model was employed—the E-CO2 system—using real data at farm level. This model used Carbon Trust, IPCC 2006 and PAS 2050 methodology (BSI 2008) to calculate the GWP of beef and sheep production.

A GHG footprint study for exported New Zealand lamb (Ledgard et al. 2010) assessed the full life cycle CF of lamb from farms, through to cooking and eating the meat, and the disposal of waste and sewage. Emissions (CH_4, N_2O, CO_2, refrigerant) referred to a 100 g portion of raw, purchased meat as the functional unit. This study used a biophysical allocation for different animal types on farms, based on the amount of feed they consumed. An economic allocation was used for lamb meat, mutton, wool and, at the meat processing stage, meat and non-meat products. At the farm stage a private data set covering nearly 500 farms throughout New Zealand was sampled, to be statistically representative of the sheep farming sector, and stratified to cover the wide range of different farm types (from extensive high country through to more intensive rolling land). 100-year global warming potential (GWP100) conversion factors were used to convert methane and nitrous oxide emissions. The total footprint was divided into 80 % for the on-farm stage, 3 % for meat processing, 5 % for (oceanic shipping) transportation and 12 % for the consumer phase (excluding consumer transport).

Weiss and Leip (2012) carried out another interesting study to estimate GHG fluxes for all emission sources of the agricultural sector. Estimates of GHG (CH_4, N_2O and CO_2) fluxes referred to the main European livestock products (meat, milk and eggs) according to a cradle to gate attributional life-cycle assessment, including emissions from land use and land use change. Calculations were made using the CAPRI modelling system, considering on-farm and off-farm fluxes and emissions from land use changes. The quantification of methane emissions from enteric fermentation and manure management followed a Tier 1 approach for sheep and goats. Allocation of emission fluxes to multiple outputs was based on the nitrogen content in the products. As far as CH_4 emissions from dairy cattle (enteric fermentation and manure management), the energy requirement for lactation and pregnancy was used to allocate emissions from milk and young animals. For most animal products, except sheep and goat meat and milk, emissions from foregone carbon sequestration dominated enhanced carbon sequestration in managed grasslands leading to net emissions. Emission intensities differed considerably between the EU-27 countries for all products examined, due to many factors (productivity, dependency of imported feed products, and share of pasture in the animal feed diet). The comprehensive approach of this paper, and its peculiarities compared to other studies, were reviewed in Bellarby et al. (2013).

Gac et al. (2012) studied the carbon footprint (CH_4, N_2O, CO_2) of French and New Zealand lamb production from cradle to farm gate for the year 2008, comparing two contrasting systems: in-shed lamb *vs.* grass lamb. Each system was analysed using a methodology developed to fit its own country, namely GES'TIM (Gac et al. 2010) for France and the Overseer® model (Ledgard et al. 2010) for New Zealand. The impact on climate change was assessed by using the GWP100 proposed by IPCC (2006). A common mass allocation was firstly used to allocate impacts to either meat or wool. The differences in the average CF in the two countries underline the importance of country specificities of both environmental context and the socio-economic characteristics of local livestock systems. In fact, the higher carbon footprint of French lamb was due to the use of external feed input and the fact that sheep are housed in-shed for part of the year, with emissions from manure management. Conversely, in New Zealand, where productivity is often higher due to warmer climatic conditions, the animals stay outside all year eating perennial pastures and therefore there are no gaseous emissions linked to external food production and manure management. Carbon sequestration in pastoral soils can potentially have a significant effect on reducing the carbon footprint at farm level. This paper presents an interesting sensitivity analysis that showed how results were highly dependent on methodological choices. Firstly, the effects of allocation method were tested: economic *vs.* mass allocation. There was a small difference between countries when mass allocation was used, and a much larger difference using economic allocation; this was because in New Zealand wool has an economic value for carpet making, whereas in France it has little economic value. Another sensitivity analysis was performed by comparing the same methodologies across both countries. Audsley and Wilkinson (2012), using the Cranfield system model, explored options for reducing UK GHG emissions from crop and livestock production systems considering cradle to farm gate boundaries. Among livestock production, sheep systems included hill, upland and lowland, pure and crossbred flocks. Emissions were expressed as GWP100 in tonnes CO_2-eq per unit of product. For each system, emissions of nitrous oxide were calculated using the IPCC Tier 1 methodology (IPCC 2006). GHG emissions were always higher for ruminants due to the methane emitted during rumination. Differences between upland and lowland sheep were small in terms of GHG emissions/kg of product at the farm gate. The best alternative system in terms of reduced emissions compared to the combined typical systems was identified for each livestock sector using the Cranfield model. The potential reductions in GHG emissions ranged from 7 % for dairy beef and poultry meat to 21 % for sheep meat.

Chatterton et al. (2012) developed an integrated livestock-ecosystems linear programming model to assess the economic and environmental impacts of the livestock sector in the UK. For this, the Cranfield Model was combined with a grassland productivity model and a soil erosion model to assess the environmental consequences of the livestock sector. A model was also developed to calculate soil erosion. The output of the LCA model were linked within the linear programming framework. The objective function to be maximised was the sum of the various ecosystem services (Provisioning + Regulating + Cultural), which were converted to a common monetary valuation system. The results show the importance of the use of a systems-based LCA approach in identifying the trade-offs between the cultural benefits of extensive systems and the potential efficiencies of more intensive systems.

The study of Eady et al. (2012) was interesting in that it used the whole suite of approaches recommended in the ISO guidelines to model co-production at the farm level, in an attempt to best represent the mixed farming system. Studying a single case study farm in Western Australia, the authors compared the CF of products with and without quantifying the benefits of mixed farming system, and compared different methods of modelling co-products. The mixed farm being studied produced distinct products (Merino wool, sheep meat and grains) that were modelled from cradle to gate using system expansion. Co-production from the sheep activity was modelled using allocation, comparing biophysical and economic relationships. As in the other studies previously discussed, the authors concluded that when compared to biophysical allocation, economic allocation shifted the environmental burden to the higher value co-products and away from the products with high resource use.

Brock et al. (2013) determined the emission profile and carbon footprint of wool production in south Wales. GHG emissions were estimated at the pre-farm and on-farm stages of production, the second being the most relevant. This study is interesting in that it tested how the emissions profile varied according to calculation method and assumptions As in other studies, the total emissions were apportioned to wool and co-products, based on economic allocation. This study also showed that the calculated emissions for wool production changed substantially, under an economic allocation method, by changing the farm emphasis from wool to meat production (41 % decrease) and by changing wool price (29 % variability). Other sensitivity analyses referred to changes in the fibre diameter (23 % variability) and fleece weight (11 % variability). The paper excluded carbon sequestration.

The implications of land occupation for CO_2 was addressed by Schmidinger and Stehfest (2012) who calculated the missing potential carbon sink of producing or not producing a certain livestock product. The applied methodology related land occupation data from LCA studies to the potential carbon sink as calculated by the IMAGE model and its process-based spatially explicit carbon cycle model. The total GHG effect of a product was calculated as the sum of the emissions along the product chain according to conventional LCA (not including direct emissions from land-use change) plus the CO_2 emission or missed potential carbon uptake due to land-use occupation in terms of kg CO_2-eq/kg product. The authors accounted for regional differences (world region), heterogeneity in land-use, and different time horizons (30-year, 50-year and 100-year time horizon). Calculations showed that the CO_2 consequences of land occupation were in the same order of magnitude as the other process-related greenhouse gas emissions of the LCA, and depended largely on the production system. The highest CO_2 implications of land occupation were calculated for beef, sheep, and goat.

Ripoll-Bosch et al. (2013) explored whether accounting for the multifunctionality of sheep farming affected the CF of lamb meat. Three farming systems (the pasture-based system, the mixed sheep-cereal system, and the industrial system, or zero-grazing) in Spain were considered representative. The study's main data sources include the FAO and national statistics. The authors computed from cradle to farm gate because post-farm gate processes were assumed to be equal for each system, and, therefore, were excluded from the analysis. The CF assessment followed the attributional approach. They quantified emissions (CO_2, N_2O and CH_4) using a model processed in MS Excel that consisted of four main modules: (i) herd structure and performance (but no herd dynamics considered); (ii) feed production (assessed both, whether on farm or off-farm production); (iii) animal feeding; and (iv) manure management. Calculations of emission in the model were based on a Tier 2 level. The GWP values used to convert methane and nitrous oxide into CO_2-eq were taken from IPCC (2007). The highest GHGs emissions involved the pasture based livestock system. When accounting for multifunctionality, the lowest GHGs emission were for the pasture-based system and the highest for the zero-grazing system.

Liang et al. (2013) studied GHG emissions from the livestock sector (swine, cow, beef, goat and poultry) in Beijing based on average data between 2007 and

2009. They covered the structure and relative proportions of diverse livestock, and adjusted related coefficients to the local situation. In this study, the assessments of total GHG emissions (only CH_4 and N_2O) was computed, together with the relative proportion in different processes (enteric fermentation, inside barn and waste management).

The Global Livestock Environmental Assessment Model (GLEAM), used in Opio et al. (2013), is a process-based static model that simulates the functioning of livestock production systems. It consists of five main modules: herd module, manure module, feed basket module, system module and allocation module, and two additional modules for the calculation of direct and indirect on-farm energy and post-farm gate emissions. The authors presented a life cycle analysis of the GHG emissions arising from ruminant supply chains. The average emission intensity for products from ruminants were estimated in terms of kg CO_2-eq/kg fat and protein corrected milk for milk, and in terms of kg CO_2-eq/kg carcass weight for meat.

In the Italian literature, a growing interest in this topic can be seen among a group of researchers inside the Animal Science and Production Association (ASPA). From these studies, we selected two papers on the GHG emissions of the Italian sheep sector. Atzori et al. (2013a) studied the differences in primary and secondary CO_2-eq emissions among four simulated scenarios of dairy sheep production in Sardinia, all able to produce the same amount of milk per year. Using the Tier 3 approach of the IPCC (2006), an Excel® spreadsheet simulated different processes: animal categories, land use, soil management, biomass available (pasture or hay), and purchased feeds. A specific sub-model was built to estimate farm CO_2-eq emissions, including methane from enteric fermentation, methane and nitrous oxide from manure management, CO_2-eq from fertilisers and fuel and from purchased feeds. Animal requirements, dry matter intake and nitrogen excretion were estimated based on locally developed equations; enteric emissions were based on IPCC sheep coefficients, whereas emissions from manure management were based on dairy cattle IPCC tables. Simulated scenarios considered four farms with high and medium-low production levels, with or without pasture, with different percentages of on-farm or purchased feed or forage. Results suggest that a reduction of emissions takes place with high production levels and on-farm feed production.

Atzori et al. (2013b) aimed to assess the GHG emission of the Italian sheep sector by accounting for CH_4 from enteric fermentation, CH_4 and N_2O from manure management, both expressed as CO_2-eq using the Tier 3 approach. A meta-modelling approach was applied within each animal category, to estimate: diets and metabolizable energy requirements for maintenance, activity, cold stress and production; emitted methane as a percentage of metabolizable energy intake; nitrogen excretion; and emission factors for CH_4 and N_2O from manure. Their results for enteric fermentation emissions were higher than those proposed by the IPCC Tier 1 guidelines.

The study of Jones et al. (2014) estimated the cradle to farm gate CF of 64 sheep farms across England and Wales for a single year using empirical farm level production data, in terms of kg CO_2-eq/kg live weight finished lamb. Default IPCC Tier 1 emission factors and data from the literature were used for reporting direct and indirect emissions (CH_4, N_2O and CO_2). Variation in the CFs relating to both system type and management was assessed. A non-parametric test was used to make comparisons between the footprints of lowland, upland and hill farms; between farms categorised by breeding ewe flock size; and then between farms categorised by area. Multiple linear regression models and dominance analysis indicated the four farm management variables with the highest impact on the size of the carbon footprint of finished lamb (head/ewe; lamb growth rate; the percentage of ewe and replacement ewe lamb flock not mated; and concentrate use). Shared inputs, such as fertilisers, were allocated based on total grazing livestock units. Emissions were shared between categories of sheep products (finished lambs, live lambs, culls sold for meat, breeding sheep and wool) using economic allocation.

5.5.1.3 Other Life Cycle-Based Methodologies and Tools: The Water Footprint

To obtain further insight about other life cycle-base methodologies and footprint indicators we chose to study the water footprint. This choice partly reflects the same arguments as discussed for the carbon footprint review, and is partly motivated by the relevance of water impact in the sheep and goat sector. Following the same criteria as CF, we present a selection of literature and do not intend to give a complete picture of the state of art in the sector, but only to suggest some useful points for future improvement in the development of complementary LCT approaches.

Several approaches to estimating water use and its impacts have been developed, each differing in the types of water included, whether the upstream or downstream processes were considered, and the characterisation of environmental impacts. Among these approaches, we only report some of those studies below that refer to a life cycle approach. In many cases, authors calculated the water impact of sheep and goat production through the joint application of LCA and specific models, such as the MEDLI model (Peters et al. 2010b) and the OVERSEER® nutrient budget model (Zonderland-Thomassen et al. 2012). The study of Peters et al. (2010b) aimed to account for water use in southern Australian red meat production, considering three supply systems, among which was a sheep-meat supplier. The functional unit of this LCA was defined as the delivery of 1 kg of hot standard carcass weight (HSCW) meat to the meat processing works product gate for wholesale distribution. The water input and output was allocated to red meat production in accordance with the relative mass of the red meat and its by-products. The authors used a hydrological model based on MEDLI, a model for analysing effluent reuse systems, and a climate file. One critical point in the WF estimation was whether, and which, environmental consequences result from water being an input to the system. Construction of the life cycle inventory, and characteristics of the water source, such as whether (1) it is renewable, (2) extraction exceeds the renewal rate, and (3) whether the extracted water is returned to the original watercourse in full, must be understood in order to determine whether water use is sustainable. The quality of water output and the time reference are aspects relevant in the WF estimation that could be difficult to manage in an LCA framework. As suggested in Peters et al. (2010b), if the frame of reference is a particular year, then changes to foreground production systems that occur from year to year and that threaten biodiversity are overlooked. Chatterton et al. (2012), in a study for EBLEX, the organisation for beef and lamb levy payers in England, combined the Cranfield LCA model with the WaSim[3] water simulation model to establish a water footprint for English beef and sheep production. Their assessment took into account all input and output of water linked to the production of beef and sheep meat—from hill, upland and lowland ewes—to calculate water use per kilogram of meat. The total footprint was accounted for almost entirely by green water (97 %), required for feed crop and grass production. The grey water (only nitrate leaching) accounted for the remaining 3 %. Hill systems had a much higher use of blue water because grass yields were significantly lower and thus green water footprints were much greater per ton of grass required.

Mekonnen and Hoekstra (2012) estimated the WF of animal products and compared it with the WF of some crops. Different production systems (grazing, mixed and industrial) and feed composition per animal type and country (China, India, Netherland, USA) were considered. They assessed the WFs of growing feed crops using a grid-based dynamic water balance model that took into account local climate, soil conditions and data on irrigation at a high spatial resolution. They considered sheep and goat, as meat animals, together with beef cattle, chicken and pigs. Sheep and goats were the least impactful animal category. The paper is very interesting for the relevance it gave to the different production systems (grazing, miked, industrial production) in calculating the total, blue and grey WFs. The study is also interesting for the methodological problems it raised: uncertainties due to a lack of data; assumptions made and combination of different data sources. Some aspects ignored in the paper (the indirect water footprints of materials used in feed production and animal raising; the potential pollution by fertilisers other than nitrogen or by pesticides or other agro-chemicals; the grey water footprint from animal wastes) are particularly relevant for industrial production systems, resulting in an underestimation of their WF.

The WF of pastoral farming systems in New Zealand (NZ) was the topic of a study by Zonderland-Thomassen et al. (2012). Survey data from representative sheep and beef farm systems was used to deal with variation in production systems. The cradle-to-farm gate life cycle required for the production of milk, beef, and sheep meat was analysed. Economic allocation was applied when dividing the WF between milk and meat. Biophysical allocation based on feed intake was used when dividing the WF between beef cattle and sheep, while economic allocation was used when dividing the WF for sheep between meat and wool. A WF approach compliant with LCA principles was used to assess the stress-weighted WF. The eutrophication potential was also assessed. Water losses associated with evapotranspiration from irrigated pasture, as well as nitrate leaching and phosphate runoff were

[3] The Water balance-Simulation Model (WaSiM) has been developed by Schulla (Schulla, J., 1997; Schulla, J., Jasper, K., 2007) to evaluate the influence of climate change on water balance.

computed using the hydrological sub-model in the OVERSEER® nutrient budget model (Wheeler et al. 2003).

5.5.2 The Implementation of Life Cycle Assessment in the Sheep and Goat Sector: Methodological Problems

Following the same approach as used in other sectors, we analysed the LCA ISO 14044 specific requirements in the domain of sheep and goat LCA studies, suggesting some points for future reflection and improvement.

5.5.2.1 Goal and Scope

All the selected papers claimed to assess the environmental impact of different goat/sheep products, in some cases both meat and milk, and in other cases only one product. According to the different goals of the papers, some considered only a single species (sheep or goats), while many studies carried out comparative environmental assessment of several vegetable and/or livestock products. Table 5.4 lists the selected articles, specifying the methodology used for the analysis—LCA, CF, WF—and the product investigated. As shown in the table, there are studies that considered only one product, others that looked at many co-products of the same livestock, and others that compared several products among different livestock. There were case studies and sectorial analyses. Even with this variety of goals and scopes, no relevant methodological problems appeared in the definition of the first LCA step.

5.5.2.2 Functional Unit

The selected LCAs studies commonly defined the FU as the mass of the product leaving the farm gate, but with different specific criteria. Many examples of FU could be drawn from the papers (see Table 5.4) even when they referred to the same product. Different FUs do not permit comparison of results from different LCA studies because, as is well known, the FU allows the comparison of alternative systems of products with a similar function.

As argued later on, few studies test the robustness of their results against different FUs with a sensitivity analysis.

The choice of a proper FU is one aspect that deserves more attention in defining homogeneous standards, but it does not require further methodological advances. The literature review of the sheep and goat sector leads to conclusions in accordance with the guidelines proposed in the previous paragraph concerning the choice of the functional unit. This choice would require a standardisation in relation to the objectives of the study and the phases included in the system boundaries. Qualitative

Table 5.4 Articles reporting on the implementation of LCT tools in the sheep/goat sector

Reference	FU	System Boundaries	Products	Methodology
Williams et al. (2006)	1 t of sheep meat and 10 m^3 of goat milk,	Farm gate	Sheep meat and goat milk	Cranfield LCA model
Kanyarushoki et al. (2008)	1000 kg of fat-and protein corrected milk (FPCM), and hectares of land utilised	Cradle-to-farm gate and from farm to the retailer entrance gate	Goat and cow milk, dairy products	LCA
Kanyarushoki et al. (2010)	1000 kg of FPCM, and hectares of land utilised	Cradle-to-farm gate and from farm gate to the retailer entrance gate	Goat and cow milk, dairy products	LCA
Ledgard et al. (2010)	100 gm portion of lamb meat	Full life cycle of meat, from farm to consumption and consumer waste stages	Lamb meat	CF
Peters et al. (2010b)	1 kg of (HSCW) sheep meat	Cradle-to-farm gate and processing for wholesale distribution	Sheep meat	WF
Head et al. 2011	1 kg of product	From farm to the supermarket	Goat milk, lamb meat, sheep meat, and goat dairy products	LCA
Michael (2011)	1 kg of milk adjusted for fat and protein content	Farm gate	Sheep and goat milk	LCA
O'Mara (2011)	The gross energy content of the commodities	Cradle-to-farm gate	Goat/sheep meat and milk	CF
Ripoll-Bosh et al. 2011	1 kg of meat lamb	Cradle-to-farm gate	Lamb meat	CF
Audsley et al. (2012)	kg of product fresh weight, MJ edible energy, kg edible protein	Farm system	Sheep meat	Cranfield LCA model
Chatterton et al. (2012)	Ton of sheep meat	Livestock sector	Sheep meat	Cranfield LCA model
Eady et al. (2012)	Grain: 1 t, Wool: 1 kg of greasy wool, sheep: one animal	Cradle-to-farm gate	Wool, sheep meat and grains	CF
Gac et al. (2012)	1 kg of total sheep live	Cradle-to-farm gate	Lamb meat	CF

Table 5.4 (continued)

Reference	FU	System Boundaries	Products	Methodology
Mekonnen and Hoekstra (2012)	Ton of product	Cradle-to-farm gate	Sheep/goat meat	WF
Schmidinger and Stehfest (2012)	kg product, m^2/year	Product chain+carbon uptake	Beef, milk, pork, poultry, crops, sheep and goat	CF (LCA+IMAGE)
Zonderland-Thomassen et al. (2012)	kg live weight and kg fat and protein corrected milk (FPCM)	Cradle-to-farm gate	Sheep meat and milk	WF
Weiss and Leip (2012)	1 kg of carcass meat and 1 kg of raw milk	Farm gate, including slaughtering	Sheep/goat meat and milk	CF
Williams et al. (2012)	1000 kg carcass lamb	Farm gate	Lamb meat	Cranfield LCA model
Atzori et al. (2013a)	Litre of milk and no. of ewes	Cradle-to-farm gate	Sheep	CF
Atzori et al. (2013b)	1 kg of sheep meat, 1 kg of sheep milk, the number of livestock, kg of 4 % fat milk	Cradle-to-farm gate	Meat and sheep milk	CF
Brock et al. (2013)	1 kg of greasy wool	Cradle-to-farm gate	19 µ wool production	CF
Koch et al. (2013)	1 kg or 1 L of product	Cradle-to-farm gate	Sheep/goat milk and lambs	LCA
Liang et al. (2013)	The average number of major livestock	Livestock husbandry systems, waste treatment systems, and agricultural use of livestock waste/manure	Livestock	CF
Opio et al. (2013)	kg of carcass weight and kg of FPCM	Cradle-to-farm gate and farm gate to retail	Goat/sheep meat and milk	CF
Ripoll-Bosch et al. (2013)	1 kg of lamb live weight	Cradle-to-farm gate	Lamb meat	CF
Jones et al. (2014)	1 kg of life weight lamb	Cradle-to-farm gate	Sheep meat	CF

Legend: *CF* Carbon Footprint/GHG, *LCA* Life Cycle Assessment; *WF* Water Footprint indicators should be used when the assessment of environmental load is related to the final products, such as by correcting the amount of milk to an energy corrected milk (ECM) basis, or by specifying the amount of meat in terms of animal parts. As in some of the selected papers, it is suggested that selecting multiple functional units or assessing the variability of results against different FU might be more accurate in a sensitivity analysis.

5.5.2.3 System Boundaries

The reviewed studies encompass all the processes in goat/sheep production: from production and the application of fertiliser, pesticides and herbicide for forage to goat/sheep cheese or meat processing. In particular, the production of input, such as fertiliser, pesticides (Biswas et al. 2010; Koch et al. 2013; Ripoll-Bosch et al. 2011, 2013; Williams et al. 2006), herbicide (Biswas et al. 2010), and their application (Head et al. 2011; Jones et al. 2014; Koch et al. 2013; Weiss and Leip 2012; Williams et al. 2006, 2012) were included in some system boundaries. The production of feed (O'Mara 2011; Koch et al. 2013; Opio et al. 2013; Ripoll-Bosch et al. 2011, 2013; Williams et al. 2006, 2012) and forage were inside the system boundaries of some studies (Jones et al. 2014; Kanyarushoki et al. 2008; 2010; Ripoll-Bosch et al. 2011; Williams et al. 2006). Mekonnen and Hoekstra (2012) considered amount of feed consumed per animal category, per production system and per country; while Atzori et al. (2013a) used purchased feeds, animal categories, land use, soil management and biomass available (pasture or hay). Some system boundaries included livestock husbandry systems and agricultural use of livestock waste/manure (Head et al. 2011; Jones et al. 2014; Liang et al. 2013; Williams et al. 2006). Some studies (Kanyarushoki et al. 2008, 2010; Opio et al. 2013) included the entire chain for goat/sheep milk and goat/sheep meat, from farm gate to retail entrance gate. Weiss and Leip (2012) and Mekonnen and Hoekstra (2012) included farm gates and slaughter, others (Gac et al. 2012; Koch et al. 2013; Williams et al. 2012) only the farm gates. Ledgard et al. (2010) calculated the GHG emissions across the full life cycle of meat, from farm to consumption and consumer waste stages. Peters et al. (2010b) assessed the environmental impact of the delivery of meat to the meat processing works' product gate for wholesale distribution; Head et al. (2011) included the entire food chain for 98 different animal products from farm to the supermarket. Authors usually clarified the processes excluded in the definition of the system boundaries of their studies:

- the post-farm dairy chain of goat/sheep sector (Atzori et al. 2013a, 2013b; Audsley and Wilkinson 2012; Biswas et al. 2010; Gac et al. 2012; Koch et al. 2013; Liang et al. 2013; O'Mara 2011; Ripoll-Bosch et al. 2011, 2013; Jones et al. 2014; Zonderland-Thomassen et al. 2012; Williams et al. 2006, 2012);
- the production of medicines (Gac et al. 2012; Head et al. 2011; Opio et al. 2013; Ripoll-Bosch et al. 2013; Zonderland-Thomassen et al. 2012) and their use (Kanyarushoki et al. 2008);

- machinery and buildings (Gac et al. 2012; Ripoll-Bosch et al. 2013; Head et al. 2011; Jones et al. 2014; Kanyarushoki et al. 2008, 2010; Zonderland-Thomassen et al. 2012);
- in many papers, the impacts associated with the land use change, biomass burning, biological fixation, emission from non-N fertilisers and lime;
- emissions from processing, transport, packaging, retail, consumption (Opio et al. 2013; Weiss and Leip 2012), consumer transport (Ledgard et al. 2010) and waste from the products (Head et al. 2011; Opio et al. 2013; Weiss and Leip 2012).

There were generally different reasons for the exclusions. Exclusions are motivated by the low entity of some impacts, as well as by the high degree of uncertainty in the data (Head et al. 2011), or by limitations in the availability of emission data (Jones et al. 2014; Opio et al. 2013). Secondly, exclusions may have been motivated by lack of methodology or consensus on the quantification approach (Opio et al. 2013). Finally, exclusions were also made due to a lack of appropriate characterisation factors (Kanyarushoki et al. 2008, 2010).

Among the selected studies, some considered different types of breeding and breeding systems. Williams et al. (2012) calculated the environmental impact of lamb meat taking into account the different sizes of the breeds and consequent feed requirements, different types of land and consequent yields of grass, and different rates of lamb growth and ewe productivity. Ripoll-Bosch et al. (2011, 2013) and Gac et al. (2012) studied low-mid and highly intensive productive systems, while other studies showed the environmental impact of the conventional and organic systems (Head et al. 2011; Peters et al. 2010b; Williams et al. 2006). Other scholars (Audsley and Wilkinson 2012; Head et al. 2011; Kanyarushoki et al. 2008, 2010; Mekonnen and Hoekstra 2012; Opio et al. 2013; Peters et al. 2010b; Williams et al. 2006) compared the environmental impact of ruminants and small ruminants; others (Michael 2011) compared non-ruminant animal species and ruminants. The time boundaries were specified in all studies reviewed, although due to a lack of availability of data, sources usually referred to different years. Some studies made reference to 1 year, others studies referred to 1 year but used average data (Kanyarushoki et al. 2008; 2010; Liang et al. 2013; Mekonnen and Hoekstra 2012; Zonderland-Thomassen et al. 2012), some used data from two (Peters et al. 2010b) or more years (Koch et al. 2013).

Concerning the system boundaries definition, a major problem in the livestock environmental assessments occurs when farms have surfaces designated for fodder production. In this situation, a holistic integrated approach and system expansion are needed to assess the environmental impact of both vegetable and livestock production cycles, and this approach increases the complexity of the analysis. The consideration of vegetable the production cycle to support animal nutrition requires a global assessment which considers the different use of farm land and related issues (different crops, rotation), as well as the effects of land use changes. This global assessment, even if more suitable from a conceptual perspective, makes things more difficult and increases the amount of data, the complexity of calculations, the assumptions required, and the uncertainty of results. Analysis of the literature suggests that greater attention must be devoted to the specification of geographical and time boundaries of the studies, especially considering the relevance of spatial and temporal dimensions in livestock management and environmental impact. Finally, future developments in system boundaries were linked to availability and quality of data, dealt with in the following paragraph, because the lack of data is often a reason to omit some processes from the system boundaries.

5.5.2.4 Availability and Quality of Data

Availability and quality of data is one of the most critical issues when applying LCA approaches. The literature review and the comparative analysis reveal that there is a need for further development towards more complete and reliable data.

As illustrated below, the selected papers adopted different approaches and assumptions with reference to the data used in the analysis.

Some authors used both data on farm activities and data from databases. The databases used were, for example the Ecoinvent (Michael 2011; Opio et al. 2013; Ripoll-Bosch et al. 2013; Head et al. 2011; Kanyarushoki et al. 2008, 2010; Williams et al. 2006) and the SimaPro (Biswas et al. 2010; Koch et al. 2013; Michael 2011;Williams et al. 2006).

Many papers took data from the literature. The list of data sourced from the literature is very long and varies between studies. Some took data related to the animal husbandry system (Atzori et al. 2013a, b; Biswas et al. 2010; Head et al. 2011; Liang et al. 2013; Mekonnen and Hoekstra 2012; O'Mara 2011; Opio et al. 2013 Williams et al. 2006, 2012), manure management (Opio et al. 2013) and enteric fermentation (Atzori et al. 2013a; Gac et al. 2012; Head et al. 2011; Jones et al. 2014; Liang et al. 2013; O'Mara 2011; Weiss and Leip 2012; Williams et al. 2006, 2012). Data from other literature was used for the emission of N_2O (Atzori et al. 2013b; Head et al. 2011; Liang et al. 2013, O'Mara 2011; Ripoll-Bosch et al. 2011, 2013; Williams et al. 2006, 2012), carbon dioxide CO_2 (Atzori et al. 2013a; Audsley and Wilkinson 2012; Head et al. 2011; Ledgard et al. 2010; Ripoll-Bosch et al. 2011, 2013; Weiss and Leip 2012), and the emission factor for carbon/ solid storage (Gac et al. 2012; Liang et al. 2013). Other papers used literature data on GHG emissions from the production (Head et al. 2011; O'Mara 2011; Ripoll-Bosch et al. 2013) and application of pesticides and herbicides (Gac et al. 2012; Head et al. 2011; Ripoll-Bosch et al. 2013), and fertiliser (Gac et al. 2012; O'Mara 2011; Head et al. 2011; Ripoll-Bosch et al. 2013). Some authors used data on CH_4 emissions (Atzori et al. 2013a, b; Audsley and Wilkinson 2012; Biswas et al. 2010; Head et al. 2011; Jones et al. 2014; Liang et al. 2013; O'Mara 2011; Ripoll-Bosch et al. 2013), on the deforestation (Opio et al. 2013), on crops (Ripoll-Bosch et al. 2013; Head et al. 2011; Weiss and Leip 2012; Williams et al. 2006, 2012), and feed production (Mekonnen and Hoekstra 2012; O'Mara 2011; Opio et al. 2013; Ripoll-Bosch et al. 2013). Data on slaughter (Head et al. 2011; Weiss and Leip 2012) and land use (Head et al. 2011; O' Mara 2011; Weiss and Leip 2012) also comes from the literature. Other authors (Chatterton et al. 2012) did not specify in detail data sources. Due to the variability of farming practices, soils and climate, it was often difficult to construct a realistic "national average" production system. For this reason, Koch et al. (2013) created several LCI datasets for the same product, for different farming practices or regions, and made different data quality controls. Due to the lack of some data, some studies dealt with the problem of uncertainty, especially of GHG emissions. The uncertainty of GHG emissions in the agricultural sector is due to numerous complex factors, such as a high variability in emission factors, especially in N_2O emissions from agricultural soils (Weiss and Leip 2012). Liang et al. (2013) adjusted GHG coefficients related to a China-specific situation; while the CAPRI database (Weiss and Leip 2012) applies an internal procedure to correct data automatically, filling data gaps or removing data errors, such as statistical outliers or implausible breaks in a time series. In Williams et al. (2006), the measurements of pollutants were all associated with errors and the authors reduced uncertainty in results by aggregating components. In Opio et al. (2013) uncertainties were associated with the variables used in the calculation of emission factors, in estimates of activity data (e.g. animal populations and herd parameters) and assumptions made. The analyses of uncertainty were based on the Monte Carlo (MC) simulation approach, which enables an investigation into how input uncertainty propagates through the lifecycle emissions model.

In the study of Mekonnen and Hoekstra (2012), there were several uncertainties in the quantification of the water footprint of animals and animal products, due to a lack of data, so that many assumptions were made: for example, when using aggregated data taken from official statistics or by combining different data sources from statistics and literature there is an assumption of similarity, with data that is not country-specific and/or not system-specific, that may reduce the credibility and comparability of results. The above review reported that data problems arise in both the agricultural and the following phases of the product chain (for example, transport, manufacturing, and packaging, which are often excluded from the system boundaries). The obvious suggestion to develop the database for the future goes hand in hand with the suggestion to include a sensitivity check of data quality in the studies.

5.5.2.5 Allocation Methods

Allocation describes how "input" and "output" are shared between the product studied and co-products. Co-product handling is a crucial issue because it could have a significant effect on the final LCA results (Flysjö et al. 2011). Allocation can be complex because of multiple output from processes, and of multiple use of output. For example, with reference to sheep and goats, you can consider multiple joint productions and co-products such as milk, meat production, and wool. The choice of allocation method, as well as possibly affecting results, should be evaluated together with the scope of the assessment and the functional unit used. As discussed later, economic allocation is the most frequent approach because it reflects the value of the products to society and the driving forces for production; it is related to the economic value of the co-products, taking into consideration the relative incidence of single joint production compared to total revenue of farms. With price fluctuations and spatial variability, the economic allocation could be different in time and space. Moreover, the relative importance of production can change when it is expressed in livestock units or area, so making comparisons very difficult. The problem of allocation is linked to the type of LCA: a physical approach, as proposed by ISO 14044, is preferred in consequential LCA, while an economic approach could be suitable in attributional LCA (Weiss and Leip 2012). In the LCA studies reviewed, product allocation is often based on the economic values of co-products (Biswas et al. 2010; Gac et al. 2012; Head et al. 2011; Jones et al. 2014; Kanyarushoki et al. 2008, 2010; Ledgard et al. 2010; Opio et al. 2013; Ripoll-Bosch et al. 2013; Zonderland-Thomassen et al. 2012; Williams et al. 2006). In Weiss and Leip (2012) allocation was based on the nitrogen content in the products, and other considered biophysical allocation was based on the amount of feed consumed (Zonderland-Thomassen et al. 2012); on the protein content (Opio et al. 2013), on the metabolic energy required to produce each co-product (Koch et al. 2013), or on the relative mass (Peters et al. 2010b). Sometimes biophysical allocation was used together with economic allocation (Gac et al. 2012; Koch et al. 2013; Ledgard et al. 2010; Michael 2011; Opio et al. 2013; Zonderland-Thomassen et al. 2012). Even with the variety of approaches used in the reviewed papers, the allocation procedure does not pose methodological problems. It is suggested that studies consider the opportunity to adopt mixed allocation rules and, most of all, to test the variability of results in a sensitivity analysis.

5.5.2.6 Life Cycle Impact Assessment (LCIA)

The LCA studies presented different stages of LCIA. Some used several impact categories to show results, as reported in Table 5.5. Other authors used a single impact category to analyse the environmental performance of sheep and goats in terms of CF or WF (see Table 5.4). In many studies, the authors described the evaluation method used. The IPCC 2006 method was the most used by authors, mainly addressing GHG emissions. Kanyarushoki et al. (2008) used CML 2001 and Cumulative Energy Demand. Williams et al. (2006) followed the IPCC 2001 method using timescales of 20, 100 and 500 years, and the CML method. Some scholars used other methods, such as CML and the IPCC, 2007 method (Williams et al.2012), IPCC (2007) (Gac et al. 2012; O'Mara 2011) and the ReCiPe (hierarchic) method (Head et al. 2011). Koch et al. (2013) used different calculation methods for each category impact; for example, they used IPCC, 2006 to evaluate Greenhouse gas emissions, the Recipe method to assess water quality and CML2002 for resource depletion.

The most commonly considered environmental impact categories are listed in Table 5.5. Firstly, it is important to highlight the global warming potential, which was the aim of many papers in the review. Eutrophication and acidification potential (Head et al. 2011; Kanyarushoki et al. 2008, 2010; Williams et al. 2006, 2012) and finally, land occupation (Head et al. 2011; Kanyarushoki et al. 2008; 2010;

Table 5.5 Impact categories considered in the reviewed studies

	Impact categories and Indicators																
References	EP	GWP	ODP	AP	POCP	HTP	FAETP	MAETP	TETP	ARU	CC	PF	IR	LO	LT	LU	Non-renewable energy use
Williams et al. (2006)	✓	✓	–	✓	–	–	–	–	–	✓	–	–	–	–	–	✓	✓
Kanyarushoki et al. (2008)	✓	–	–	✓	–	–	–	–	✓	–	✓	–	–	✓	–	–	✓
Kanyarushoki et al. (2010)	✓	–	–	✓	–	–	–	–	✓	–	✓	–	–	✓	–	–	✓
Head et al. (2011)	✓	–	✓	✓	✓	✓	✓	✓	✓	–	✓	✓	✓	✓	✓	–	–
Michael (2011)[a]	✓	✓	✓	✓	–	–	–	–	–	–	–	–	–	–	–	–	✓
Chatterton et al. (2012)[b]	✓	✓	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Williams et al. (2012)	✓	✓	–	✓	–	–	–	–	–	✓	–	–	–	✓	–	–	✓
Koch et al. (2013)[c]	✓	✓	✓	✓	–	✓	–	–	–	–	✓	–	✓	✓	✓	✓	–

Legend: *EP* eutrophication potential, *GWP* global warming potential, *ODP* ozone layer depletion potential, *AP* acidification potential, *POCP* photochemical oxidation, *HTP* human toxicity potential, *FAETP* fresh water ecotoxicity, *MAETP* marine ecotoxicity, *TETP* terrestrial ecotoxicity, *ARU* abiotic resources used, *CC* climate change, *PF* particulate matter formation, *IR* ionising radiation, *LO, LT, LU* land occupation, transformation and use, *NREU* non-renewable energy use

[a] Michael (2011) also considered water use (litres)

[b] Chatterton et al. (2012) also evaluated soil erosion, pesticides, N leaching, ammonia, faecal contamination, *Chryptosporidium* and total area used

[c] More impact categories than those specified in the Table are reported in the paper

Williams et al. 2012) were less considered. Chatterton et al. (2012) evaluated the impact of the livestock sector in terms of soil erosion, pesticides, eutrophication, N leaching, greenhouse gas emissions, ammonia, faecal contamination, *Chryptosporidium* and total area used. The impact categories considered by Michael (2011) were water use (litres), energy use (MJ), global warming potential (CO_2 equivalent), ozone depletion potential (CFC−11 equivalent), acidification potential (SO_2 equivalent) and eutrophication potential (PO_4).

Previous considerations lead to the argument that LCIA is an issue on which methodological problems occur, asking for a future advances. Three aspects deserve the most attention: land use and land use change, water assessment, and carbon storage, impact categories that are particularly important in the environmental assessment of livestock sectors.

5.5.2.7 Interpretation and Tools Supporting the Interpretation Analysis

The ISO standard distinguishes some elements that should be considered in the interpretation phase: (i) identification of the significant issues based on the results of the LCI and LCIA phases; (ii) evaluation that considers completeness, sensitivity and consistency checks; (iii) conclusions, limitations, and recommendations.

1. Identification of the significant issues based on the results of the LCI and LCIA phases. All reviewed studies reported information on the interpretation phase, and it was possible to identify the significant environmental issues. Most studies assessed the whole system's impacts, others showed the most impactful steps, impact categories, impactful substances or materials.
2. Evaluation that considers completeness, sensitivity and consistency checks. As far as completeness, the reviewed studies quite often declared exclusions and recognised their limitations, sometimes considered in a sensitivity analysis or through consistency checks. Firstly, the complexity of LCA methodology applied to the agricultural sector would ask for a methodological innovation to integrate the multifunctionality of agriculture in the LCA analysis. Multifunctionality recognises that agriculture also contributes non-tradable goods, such as environmental and landscape services (Kanyarushoki et al. 2010; Ripoll-Bosch et al. 2013). For this reason, Ripoll-Bosch et al. (2013) considered the cultural ecosystem services provided as co-products: beyond the primary function of producing animal products, the sheep farming systems in Spain that they studied also provide other services or public goods, such as landscape conservation, cultural heritage, preservation of biodiversity, or fire prevention. Secondly, because of system heterogeneity, the use of mixed data (at farm level and at national/international scale; from field and from database) and the methodological assumptions, have effects on the results. Even if methodologically accurate, the nature of estimated results is sometimes recognised but not supported with measures to appreciate the difference between real and potential impacts. According to the research goals, different aspects are tested through a sensitivity analysis. Some authors presented

different scenarios to investigate how varying the results affected sheep management (Atzori et al. 2013a; Chatterton et al. 2012; Jones et al. 2014; Ripoll-Bosch et al. 2011, 2013; Williams et al. 2012), the functional unit (Kanyarushoki et al. 2008, 2010), or the allocation rules (Biswas et al. 2010; Gac et al. 2012; Michael 2011; Zonderland-Thomassen et al. 2012). Kanyarushoki et al. (2008, 2010) compared cow and goat specialised dairy farms in two French regions, and investigated how varying the results affected the functional unit. In terms of hectares of land occupied, goat farms had a higher impact (Kanyarushoki et al. 2008, 2010), except in climate change (Kanyarushoki et al. 2010). As far as a sensitivity analysis related to the allocation rule, some studies considered the effect of price fluctuation (Biswas et al. 2010). Others changed the allocation method to show differences in results between economic and mass-based allocation, in some cases concluding that their results are highly dependent on this methodological choice (Gac et al. 2012), in other cases showing that few differences emerged (Michael 2011; Zonderland-Thomassen et al. 2012).

3. Conclusions, limitations, and recommendations. The reviewed studies quite often recognised their limitations, as discussed in the previous point, but only sometimes performed a sensitivity analysis or estimated errors. The papers often made proposals regarding recommendations and mitigation strategies, even if not all mitigation strategies were site-specific, as more appropriate (Nicholson et al. 2001; Gerber et al. 2010; Blake et al. 2004). Michael (2011) made methodological recommendations, suggesting that the LCA methodology should be improved to enable appropriate recognition and to focus on products with special properties (e.g. lactose content) other than fats and protein in milk from dairy sheep and dairy goats. According to Liang et al. (2013) relevant strategies that should be considered to reduce GHG emissions from the livestock sector are related to improving rearing technologies, breeding, and developing a large-scale biogas industry. In O'Mara (2011), the mitigation potential for enteric CH_4 emissions was considered as three issues: improved feeding practices, use of specific agents and diet additives, and management changes and improved animal breeding. Ledgard et al. (2010) proposed to create tools for emission mitigation, such as the minimisation of enteric fermentation methane through breeding or vaccines, and the reduction of nitrous oxide emissions through soil additives and nitrogen management practices. Lipson et al. (2011) proposed interventions both to reduce methane production and to improve the efficiency of water used by goats. Firstly, they suggested managing grazing to reduce methane production by encouraging goats to consume younger, more easily digestible forage. Secondly, they suggested improving the efficiency of water used by goats both by using water-efficient feed crops that can increase the productive efficiency of livestock water use, and fodder trees and forage legumes that also reduce erosion and improve transpiration, soil structure and soil fertility. Other mitigation strategies proposed in the paper refer to genetic selection, animal breeding and vaccination, to increase feed conversion or to reduce enteric methane emissions. Authors, citing some literature, noted that strategies to mitigate the environmental impact of livestock production may come with some risks. For

example, increased dietary reliance on crop residues in order to increase the water use efficiency of ruminant livestock may be simultaneously counterproductive to the goal of reducing greenhouse gas emissions because ruminant consumption of residual crop material increases enteric methane production during digestion. Whatever the strategies, it is necessary to evaluate emissions from livestock on temporal and spatial scales, to identify problems and trends, and to prevent environmental degradation (Liang et al. 2013). As shown, the selected papers usually reported mitigation strategies but it was not very common to find a sensitivity analysis to check for invariability of the results toward changes in strategies or hypothesis. The proposed mitigation refers to the results of other studies or suggests future research developments. Because of the sensitivity of LCA results, a stronger international standardisation of procedures and methodological advances are necessary.

5.5.2.8 Comparative Analysis of the Different Types of Breeding, Final and Processed Products

According to the specific goal, scope and system boundaries adopted in the selected papers, many different types of comparison result:

- different types of breeding and breeding system, between ruminants and small ruminants (Audsley and Wilkinson 2012; Head et al. 2011; Kanyarushoki et al. 2008, 2010; Mekonnen and Hoekstra 2012; Opio et al. 2013; Peters et al. 2010b; Williams et al. 2006);
- between ruminant and non-ruminant animal species (Michael 2011);
- between conventional and organic systems (Head et al. 2011; Peters et al. 2010b; Williams et al. 2006);
- between low-mid and highly intensive productive systems (Ripoll-Bosch et al. 2011, 2013; Gac et al. 2012);
- between different sizes of breeds and consequent feed requirements, different types of land and consequent yields of grass (and management requirements), different rates of lamb growth and ewe productivity (Williams et al. 2012).

5.5.2.9 Critical Review

In order to assess the scientific and technical validity of the study and improve its credibility, a critical review (CR) should be carried out by an external independent panel of experts, following international (the ISO 14040-series), or other national, product specific or case-specific standards. The literature review provides evidence that a CR of experts is not generally applied, both as a simple peer review of the final report (apart from the journal review process), or as a more integrated quality assurance process. This is not specific to the scientific literature about the sheep and goat sector, but is general to all the field applications, although it plays an important role in the quality assurance of LCA studies. All the reported studies appear to include the phases required by the specific LCA standard. Some difficulties could be described involving the completeness of information given by papers about the content of LCA steps, which might be useful to a CR. These difficulties mostly concern the critical review of the inventory analysis (adequacy of data and its validation, calculation and sensitivity analysis) and the critical review of the interpretation phase a (data quality assessment and a sensitivity analysis). The process of critical review is one aspect that needs a solution for the credibility of the future development of LCA and LCT approaches (Klöpffer, 2013; Weidema et al. 2013). The ongoing revision of ISO TS 14071 could become the basis for improving the review process of life cycle based standards.

5.6 Pig Production

Nine Life Cycle Assessment (LCA) studies were selected from peer-reviewed scientific journals and scientific reports. All these studies were aimed at assessing the environmental loads of pig production and at highlighting the hotspots in the production chain. The studies published in the last 10 years were selected (Table 5.6).

The studies refer principally to northern Europe and to the production of small pigs slaughtered to obtain fresh meat whereas no information is reported for larger pigs slaughtered for meat suitable for derived edible products that represent the main goal of the pig production chain in southern Europe. Principally, the studies selected evaluated environmental loads related to the pig production chain until the farm gate (Cederberg and Flysjo 2004; Basset-Mens and van der Werf 2005). Some of them accounted for subsequent stages, such as plant processing (Reckmann et al. 2013), and retail (Dalgaard et al. 2007). Some studies were a cradle to grave contribution wherein meat consumption and waste disposal were included in the cycle (Kingstone et al. 2009; Thoma et al. 2011). Some studies compared organic versus non-organic production (Cederberg and Flysjo 2004; Basset-Mens and van der Werf 2005; Williams et al. 2006) or pork meat with other sources of protein such as tofu (Håkansson et al. 2005). No studies were combined with evaluation of economic impacts (e.g. life cycle costing, net present value, etc.).

5.6.1 Goal and Scope

Although specific goals and scope differed among studies, they all aimed principally to investigate the environmental performance of different pig production systems in present or future scenarios. The environmental loads of different pig farms system such as indoor vs. outdoor, slatted floor vs. bedding and compound vs. liquid-fed were assessed (Kingstone et al. 2009). The greenhouse gas emissions (GHG) associated with pork meat production until consumer was evaluated for the USA (Thoma et al. 2011). They assessed only the global warming potential (GWP) because it is indicative of opportunities for improved energy efficiencies

Table 5.6 Summary of the reviewed studies on LCA pig production

Reference	FU	Main boundaries	Geographical areas	Time boundaries
Basset-Mens and van der Werf (2005)	1 kg of live weight; 1 ha	Cradle to farm gate	France	1996–2001
Cederberg and Flysjö (2004)	1 kg of bone and fat free meat; 1 ha	Cradle to farm gate	Sweden	*10–20 years future scenarios*
Dalgaard et al. (2007)	1 kg of carcass weight	Cradle to delivered final retail destination	Danish	*1995 and 2015*
Håkansson et al. (2005)	20 g of complete proteins	Cradle to final consumption including waste disposal	Sweden	*Not specified*
Kingstone et al. (2009)	1 kg of slaughter pork consumed	Cradle to final consumption including waste disposal	British	2007
Reckmann et al. (2013)	1 kg of slaughter weight	Cradle to slaughter gate	Germany	2010/11
Stephen (2011)	1 kg of live weight	Cradle to farm gate	British	*Not specified*
Thoma et al. (2011)	4 oz boneless meat	Cradle to final consumption including waste disposal	USA	2008–09
Williams et al. (2006)	1 kg of carcass weight	Cradle to farm gate	British	*Not specified*

or conservation and moreover the authors believed that this impact could act as a useful baseline level of GHG which would be beneficial if voluntary carbon trading markets become viable in the future. The environmental impacts of producing grower-finisher pigs (12–105 kg) were evaluated with different diet scenarios (Stephen 2011). The diet scenarios were: conventional soya-based diet, homegrown bean-based diet, homegrown pea-based diet and homegrown lupin-based diet. A Danish study compared the environmental impact of Danish pork in 2005 with that produced in 1995 (base scenarios) and considered different scenarios for the year 2015. Additionally, the same study also compared the Danish results with those of pork produced in Great Britain and the Netherlands. It evaluated the effect on the environmental profile of pork production of the improvement in the number of weaning piglets for sow and in finishing feed conversion rate (Dalgaard et al. 2007). Different future pig production systems were evaluated (Cederberg and Flysjo 2004) when different aspects of sustainability were prioritised. Production in the future scenario was focussed on animal welfare, environmental care and high quality products at low prices. The LCA was used to assess how the management of different pig production systems impacted on the environment (Basset-Mens and van der Werf 2005). The systems compared were conventional good agricultural practice, a French quality label scenario and a French organic scenario. A German study assessed the environmental impacts of pork production, highlighting the hotspots in the pork cycle as well as the performance of a sensitivity analysis to determine whether different methodological and input parameters impacted on results (Reckmann et al. 2013). Pork meat and Tofu were compared when LCA was aimed to assess the environmental load of different sources of protein (Håkansson et al. 2005).

5.6.2 Functional Unit

The functional unit (FU) was not homogeneous among the studies because it was analysed at different stages of the pig production chain. The FU was 1 kg of live weight evaluated at the farm gate (Cederberg and Flysjo 2004; Basset-Mens and van der Werf 2005). Other studies evaluated the FU at the gate of the slaughterhouse (Reckmann et al. 2013). The FU was 1 kg of bone and fat-free meat when the aim of the study was to evaluate the final function at consumer of edible parts of pig meat. This was because the amount of meat consumed does not reflect the slaughter meat since the consumer does not eat chop bones and usually not the fat (Thoma et al. 2011). In the LCA used to measure the environmental load of British pork consumption the FU was set as 1 kg of pork product as consumed by the consumers (Kingstone et al. 2009). No studies were found that had chosen the nutritional property of meat such as protein or energy content as the FU. One hectare of arable land was used in studies that compared different production systems, organic versus conventional, and it reflected the function of non-market goods such as environmental services on local scale (Cederberg and Flysjo 2004; Basset-Mens and van der Werf 2005).

5.6.3 System Boundaries

System boundaries varied with the scope of the researches. Thus, for each of the reviewed LCAs the system boundaries began and finished at different stages of the production chain. Crop and feed production, pig housing (including enteric fermentation and manure management) were considered when the functional unit was 1 kg of live weight at the farm gate (Basset-Mens and van der Werf 2005; Williams et al. 2006) or 1 kg of slaughter weight estimated at the abattoirs when slaughter operations were included (Reckmann et al. 2013) or 1 kg of slaughter weight evaluated at the distribution depots when transport from slaughterhouse to retail was also considered (Dalgaard et al. 2007). In cradle-to-grave studies the contribution of feed and pork production, delivery to processor, processing, packaging (included production of raw materials and ultimate disposal), distribution, retail, consumption and waste disposal were considered (Kingstone et al. 2009; Thoma et al. 2011). The data for the crop and feed production were related to the amount of feed needed to meet pigs' dietary requirements in the different stages of stock production. Different feed compositions were considered among the studies and the main components were soy and sunflower meal, wheat, barley, fish meal and corn (Basset-Mens and van der Werf 2005; Delgaard et al. 2007; Reckmann et al. 2013). The animal production phase comprised breeding, weaning and fattening phases. Indoor or outdoor pig housing systems were compared for non-organic sow and weaning stages, whereas fattening stock were always modelled a as entirely housed. In organic systems, the phases of breeding, weaning and finishing were modelled as an outdoors system (Williams et al. 2006). Different slaughter weights were considered as this aspect affects the time that pigs remain at the piggery and influences the environmental burden associated with feed consumption, manure production and related operations (Stephen 2011). A British study accounted for manufacturing, maintenance and housing of capital goods such as vehicles, building and machinery (Williams et al. 2006). Generally, veterinary input for insemination or consumption of medicals was excluded.

5.6.4 Availability and Quality of Data

Data were provided from different sources between studies. Primary data for inventories of crop and feed production were provided by feed factories and used in a German study including the use of fertilisers, fossil fuels and other resources. (Reckmann et al. 2013). In a Danish study (Dalgaard et al. 2007) the LCA data on barley, heat and electricity were from the LCA food database (www.LCAfood.dk), data on soy meal imported from Argentina were derived by previous study. The total amount of feed consumed by the pig during its life was calculated on the basis of data from BPEX (British Pig Executive). The content of the feed was based on recipes of feed mixtures from a Danish feed company. Data on energy use and disposal of animal by-products at the abattoir were derived from the Green Accounts from Horsens slaughterhouse (2007) and from the processor of animal by-products (DAKA 2007). In a US study the raw data were provided by industry experts and standard pork industry handbooks. Regionally specific data for feed crops were taken from the farm extension service and from the National Agricultural Statistical. Additional input data for fuels and electricity consumption for crop production were obtained from the technical literature, state agricultural extension services, the US Department of Energy, the USDA, and other academic institutions. Transport emissions from producer to processor and from processor to distributor were calculated from information provided by industrial sources (Thoma et al. 2011). In a French study (Basset-Mens and van der Werf 2005) production and delivery data of inputs for crop production were derived in accordance with Nemecek and Heil (2001). The BUWAL 250 database (BUWAL 1996) was used to assess road and sea transport loads. Data associated with building construction were from Kanyarushoki (2001). Ammonia emissions from field were estimated according to ECETOC (1994). NH_3, N_2O and CH_4 emissions were treated according to IPCC (1996) and UNECE (1999). In a British study (Williams et al. 2006) data were obtained from disparate sources. Many data on farm management, productivity and typical inputs were taken from standard texts. Values for fertiliser use and manure composition also came from Defra's RB209 (MAFF 2000) and Surveys of British Fertiliser Practice (Defra 2001–05). Data on pesticide use came from annual pesticide surveys. Gaseous emissions of ammonia, nitrous oxide and methane came mainly from the UK's national inventories, which also supplied some activity data (proportion of manure spread on arable and grassland). Other production data came from the expertise of the project team, the scientific literature and the Ecoinvent LCA database.

5.6.5 Allocation of Burdens to Co-products

The division or extension of the processes is referred to in the ISO standardised guidelines (ISO 14044) as a priority in order to avoid allocation. When a system under study produces more useful outputs and it is impossible to divide it into subsystems the environmental loads must be allocated correctly to the co-products. The allocation may be done on the basis of economic, physical or functional properties (price, mass, or protein contained). Total loads in the studies analysed were assigned mainly to products and co-products in proportion to total revenue or weight. Some crops and animals produce more than one product, for example oil and meal from soybean or milk and meat in dairy farms. As reported previously, the studies analysed here analysed only fresh pig meat and not derivates or co-products, so all burdens were allocated solely to the functional unit. When co-products were generated in feed production or slaughter operations the economic allocation was applied in five studies. The protein source used in pig feed management (soy and rapeseed meal) with the co-product generated by the extraction (soy and rapeseed oil) was allocated by economic value. Moreover, a sensitive analysis was performed to verify the outcomes when all environmental loads were charged on soy and rapeseed meal or when mass allocation was used between oil and meal (Cederberg and Flysjo 2004). In Reckmann et al. (2013), the economic allocations for soy meal and soy oil were 66.3 and 33.7 % respectively. For finishers and sows, where culled sows represent the co-products of pig production, resource use and emissions were allocated with an economic value of 6–7 % and 93–94 % for sows and finishers, respectively (Basset-Mens and van der Werf 2005). Pig systems produce prime meat from finishers, but culling breeding stock (sows and boars) also produces meat. This meat is mainly consumed as processed foods and its quality is generally considered lower and reflected in its lower price, typically less than 25 % of the price of prime meat. On this basis the burdens for the lower economic value of secondary meat were allocated and indicated a reduction of the potential output of prime meat of less than 5 % (Williams et al. 2006). At the processing gate slaughter meat and waste such as offal and blood are also produced. It was reported that these waste products were used as pet food or fertiliser, digested anaerobically or sent to landfill. The allocation of these co-products is generally very difficult because the precise type (quality) and amount are generally not known. The allocation ratio of these co-products was calculated using economic US census data relating to other species. An allocation ratio was used that assigned 89 and

11 % of the greenhouse gas burden to the meat processing and rendering operations respectively (Thoma et al. 2011). Moreover, the same authors allocated the GHG emissions related to retail processes such as energy use and refrigerant loss as a function of pork meat mass sold. Finally, for the consumer phase they allocated by pork meat mass emissions related to the transport from retail to home, refrigeration, cooking, food loss and waste disposal.

5.6.6 Life Cycle Impact Assessment (LCIA)

The environmental impacts analysed differed among the studies. Global warming potential (GWP_{100}) evaluated on the basis of a 100-year time horizon, eutrophication potential (EP), and acidification potential (AP) were the commonest ones analysed. Some studies also evaluated other impact categories such as photochemical ozone formation (Cederberg and Flysjo 2004), terrestrial ecotoxicity (Basset-Mens and van der Werf 2005), and ozone depletion and photochemical smog (Delgaard et al. 2007). Moreover, in some cases the consumption of resources such as land use, primary energy use, abiotic resource use and pesticide use were also counted (Basset-Mens and van der Werf 2005; Hakansson et al. 2005; Williams et al. 2006; Kingstone et al. 2009). Only one study assessed a single impact (GWP) and the GWP equivalents were adopted from IPCC 2007 (Thoma et al. 2011). The EDIP method (Wenzel et al. 1997. Version. 2.03) was used for the LCIA, whereas the characterisation factors for methane and nitrous oxide were chosen according to IPCC 2001 (Dalgaard et al. 2007). In a French study (Basset-Mens and van der Werf 2005) the characterisation factors for the impact category were adopted in agreement with Guinée et al. (2002). Eco-indicator'99 (H), a model that displays 11 impact categories, was used for the impact assessment but only GWP, land and fossil fuel use were considered in the LCA (Hakansson et al. 2005). The main GWP sources, expressed as kg of carbon dioxide equivalent, were nitrous oxide and methane, both from feed production and pig housing, and carbon dioxide from fossil fuel. For the EP characterisation the major chemical compounds included were nitrate and phosphate leaching in the water and ammonia emission in the air. The EP was quantified mainly as a phosphate equivalent but it was also expressed as a nitrate equivalent (Delgaard et al. 2007) or kg of O_2-eqv./FU (Cederberg and Flysjo 2004). Ammonia (NH_3) and sulphur dioxide (SO_2) emitted respectively from the agricultural phase (field and housing) and fossil fuel combustion were considered in the assessment of acidification potential. The AP was reported as sulphur dioxide equivalent and in one case as mol H^+/g (Cederberg and Flysjo 2004). It must be noted that, when reported, the characterisation factors adopted for the different impact categories differed among studies. Methane was accorded 21 (Basset-Mens and van der Werf 2005), 23 (Dalgaard et al. 2007) or 25 (Thoma et al. 2011) kg CO_2 equivalent for GWP_{100}. Nitrous oxide was treated as 296 (Dalgaard et al. 2007), 298 (Thoma et al. 2011) or 310 (Basset-Mens and van der Werf 2005) kg CO_2 equivalent for GWP_{100}. For eutrophication potential 1 kg NH_3 and 1 kg NO_3 were treated as 0.44 and 0.43 kg PO_4 equivalents (Williams et al. 2006) or 0.35 and 0.1 kg PO_4

equivalents (Basset-Mens and van der Werf 2005). Water depletion was not considered in any of the studies taken into consideration.

5.6.7 The Interpretative Analysis

Farm operations, including feed production and animal housing (animal and manure management), were the biggest hotspots in the pork chain for most of the impacts analysed. The GWP was mainly affected by emission of nitrous oxide from crops, by nitrous oxide and methane emitted from manure management and by methane emitted from enteric fermentation. The contributions of the different GHG to the total GWP were 38, 33 and 29 % for CO_2, N_2O and CH_4 respectively. The feeding stage was the greatest source of CO_2 emissions (82 %), whereas pig housing and slaughterhouse stages accounted for 5 and 13 % respectively. N_2O emissions were mainly related to feed production (95 %), and the majority of CH_4 (93 %) was related to the pig housing stage. The feeding, housing and slaughterhouse stages accounted for 63, 30 and 7 % to total GWP, respectively. Within the pig housing operations, the finishing stocks were indicated as the main contributors to GWP (Reckmann et al. 2013). When the LCA was a cradle-to-grave assessment the slaughtering and retail operations were minor sources of GHG emissions. The consumption phase that included transport, refrigerator, cooking and waste disposal was reported to account 20 (Kingstone et al. 2009) and 30 % (Thoma et al. 2011) of total GWP.

The eutrophication potential was related for the 52 % to feed production, 40 % to pigs housing and 8 % to slaughterhouse processes (Reckmann et al. 2013). A British study reported that indoor pork production on slatted floors was associated with 96 % of eutrophication potential, whereas all the processes from the farm gate to consumption and final waste disposal accounted for only 4 %. AP was 0.18 kg SO_2 equivalent for the farming phase compared with an overall impact per kg of meat produced of 0.19 kg SO_2 equivalent (Kingstone et al. 2009). In a Danish study it was reported that the greatest contribution to eutrophication was from nitrate leaching (62 %) followed by ammonia emission (32 %). Nitrate and ammonia were related to crop operations, pig housing and manure application (Dalgaard et al. 2007). The main contribution to AP derived from animal housing (76 %) and feed production (23 %). AP was related mainly to the ammonia emitted from manure/slurry in the housing, during storage and after field fertiliser application. The pigs housing and feed production accounted for 87 and 13 % of the total ammonia emitted, respectively. Total ammonia was related to the 93 % of the total acidification potential. With regard to stock, the fattening stage was identified as the greatest source of AP compared with the sow and weaning stages (Reckmann et al. 2013). High feed efficiency (less feed/FU) and greater grain yield were associated with a lower release of nitrifying substances (Cederberg and Flysjo 2004). The EP analysed for different scenarios in French pork production showed that for kg of pork it was higher in the organic scenario; on the other hand, it was lower in the organic scenarios for hectares. Acidification potential was highest both for kg of pig and for hectares in the good agricultural practice scenario (Basset-Mens and van der Werf 2005).

5.6.8 Critical Review

All the studies reviewed herein agree that, compared with the use of data provided by a bibliography, the improvement in the availability of direct data relating to the different stages of production, in particular for the main feed used, could improve assessments. Results reported for GWP of pork production in the USA evaluated at the farm gate were within the range reported in the literature. The slaughter operations, packaging and transport made a lower contribution. On the other hand, retail refrigeration, transport to home and cooking operations were significant contributors to the overall carbon footprint. The authors reported that the two main factors affecting the carbon footprint were the change in manure management from deep pit to anaerobic lagoon, and the allocation method (economic) for feed co-products. Finally, they indicated that greater sustainability could be achieved for GHG reductions associated with technologies that capture or convert the methane from anaerobic lagoons (Thoma et al. 2011). Hotspot and marginal improvement were discussed principally for eutrophication and acidification potential because the emissions of nitrate to water and ammonia to air have a direct and immediate impact on a regional scale (Basset-Mens and van der Werf 2005). The authors indicated that eutrophication potential could be reduced with optimal nitrogen use and the introduction of catch crops in the rotation adopted for feed production. Ammonia could be reduced by nutritional and manure management strategies. Reduction of the protein intake or improvement of their utilisation by the animals could reduce the amount of nitrogen in manure. Emissions of NH_3 from animal housing could be reduced by better control of microclimatic conditions (lower air velocity and temperature), improved frequency of manure removal and covering of slurry stores. Moreover, fast and effective incorporation into the soil could minimise the NH_3 emissions during manure application (Basset-Mens and van der Werf 2005). In British pork production improvement in environmental sustainability could be achieved mainly in the pig farm phase. The farming methods of loose housing and outdoor breeding make a significantly higher contribution to eutrophication and acidification than pigs raised indoors on slatted flooring. Moreover, the impacts could be reduced with a greater feed efficiency, a high number of pigs per litter and correct manure management. Environmental improvement from farm gate to grave could be achieved with high energy efficiency at the abattoirs (energy and heat) and at home with the use of AA rated appliances (refrigerator and cooker) and at retail with optimal refrigeration (Kingstone et al. 2009). Greater protein content in feed was associated with higher emissions or nitrous oxide, ammonia and nitrate from housing, manure storage and manure field application. A sensitive analysis indicated that when protein content in the fattening feed mixtures was decreased from 18 to 16 % the global warming, eutrophication and acidification potential decreased by 2, 5 and 7 % respectively (Dalgaard et al. 2007). The same authors highlighted that the concept of food miles is often used incorrectly. Indeed, the overall GWP for kg of British pork was no lower than the kg of Danish pork, which included the transport of meat from the Danish slaughterhouse to Great Britain (transport accounted for less than 1 % of total GWP). On the other hand, because of the higher productivity efficiencies in Denmark, the eutrophication and acidification potentials were lower than those for the British pork. Therefore, the idea of choosing "home products" as more environmentally friendly in view of the shorter distance from farm to retail is not justifiable if feed efficiency, high production and correct manure management are not encouraged. A Swedish study reported a greater feed consumption (+ 14 %) in the "animal welfare" scenario compared with the "environment" scenario. The higher feed consumption for kg of pork produced was because of lower piglets/sow production and the higher feed intake related to animal movements in outdoor systems. Therefore, these data suggest that production systems which guarantee a better degree of animal welfare may lead to greater feed consumption and lower efficiency of resources utilisation. Moreover, the impossibility of using synthetic amino acid in organic systems implies a higher concentration of protein in the ration and a greater amount of nitrogen in the excreta as a consequence. Finally, the authors suggest the introduction of ammonia filters in the pig house ventilation system to reduce ammonia emissions would have a significant impact on eutrophication and acidification potential (Cederberg and Flysjo 2004).

5.7 Poultry

In the following, a description of the main aspects of this sector at the international and European levels is presented. Then 11 international LCA studies on poultry production published in peer-reviewed journals, scientific reports or international conference proceedings are analysed (Table 5.7). All the LCA applications to poultry production systems published in the last 6 years excluded the work of Williams et al. (2006) that we found critical were selected for this review. Methodological problems connected with the application of LCA in this sector are examined, starting with a critical comparative analysis of the LCA case studies. Finally, the environmental hotspots are identified in order to develop possible solutions to the problems presented.

5.7.1 The Poultry Sector: Main Aspects

In 2010 poultry production worldwide reached about 79 million t, with an increase in production of 5.8 % over the previous year. According to FAO data, the main producers are the USA, China, the European Union and Brazil, which together deliver as much as 77 % of total production. The production of poultry meat in the EU 27 in 2010 increased by 4.3 % to just over 12 million t. The main producing countries are: Germany, the UK, Italy, France and Spain, all of whom saw production increase substantially compared with previous years. The main production in the European Union is represented by chicken meat and the proportion further increased (+ 3.7 %)

Table 5.7 List of references included in the literature review and their main characteristics

Reference	FU	Method	Main boundaries	Geographical areas	Time boundaries
Wiedemann et al. (2010)	1 kg of carcass weight 1 kg of live weight	LCA	Cradle to meat industry gate	Australia	1 year
Pardo et al. (2012)	0.6 kg of sliced chicken breast fillet	LCA	Cradle to grave	Spain	*Not specified*
Bengtsson and Seddon (2013)	1 t of roast chicken	LCA	Cradle to retailer	Australia	1 year
Leinonen et al. (2012)	1 t of carcass weight	LCA	Cradle to farm gate	UK	*Not specified*
Davis and Sonesson (2008)	1 meal (chicken- based) per person	LCA	Cradle to grave	Sweden	*Not specified*
Pelletier (2008)	1 t of live weight kg of CO_2 eq per kg of product	LCA	Cradle to farm gate	USA	*Not specified*
MacLeod et al. (2013)	(CW or eggs) and kg of CO_2 eq per kg of protein	LCA	Cradle to retail gate	World	*Not specified*
Cederberg et al. (2009)	1 kg of carcass weight	LCA for CF	Cradle to farm gate	Sweden	*15 years*
Katajajuuri et al. (2008)	1 t of honey marinated sliced broiler chicken fillet	LCA	Cradle to slaughterhouse gate	Finland	*Not specified*
Pelletier et al. (2013)	For example, kg CO_2-eq per 1000 tonne of eggs	LCA for CF	Cradle to slaughterhouse gate	USA	*1 year*
Williams et al. (2006)	1 kg of carcass weight	LCA	Cradle to farm gate	UK	*Not specified*

in 2010, comprising about 80 % of the total poultry meat. The turkey production increased by 6 %, whereas the duck production remained stable (+ 0.6 %). At the national level the slaughter poultry (including wild game) recorded a growth of 1.8 % over the previous year, with 548.191 million animals slaughtered, according to the ISTAT data for 2010. The chickens are the most prevalent, accounting for 89.66 % of the total, followed by turkey with 5.16 % and wild game with 3.67 %. The Veneto region in Italy slaughtered as much as 42.9 % of total poultry meat, followed by Emilia Romagna with 19.6 % and Lombardy with11.7 %.

5.7.2 Literature Review on LCA Application to Poultry Production

5.7.2.1 Goal and Scope

The poultry supply chain is recognised by several authors and researchers as the most environmentally efficient among the different meat production systems. For this reason, only a few studies have focussed on the assessment of environmental loads generated up to the farm gate (Pelletier 2008; Cederberg et al. 2009; Leinonen et al. 2012) and most target the assessment of the whole supply chain (or different post-farm activities) in order to identify the hotspots and strategies to improve the environmental sustainability of poultry production and consumption in post-farm processes (Williams et al. 2006; Katajajuuri et al. 2008; Davis and Sonesson 2008; Wiedemann et al. 2010; Pardo et al. 2012; Bengtsson and Seddon 2013). The environmental impact assessment of different chicken production systems was the goal of recent studies by Boggia et al. (2010), who compared the conventional with the free-range system; the first two also included the organic production system. One differs from the other two production systems regarding the use of organic feed (Leinonen et al. 2012). In particular, Boggia et al. (2010) compared the conventional boiler production system with two organic production systems called “organic” and “organic plus”, that differ for the restrictive requirements in terms of animal welfare considered in the second system.

5.7.2.2 Functional Unit

The functional unit (FU) was not homogeneous among the studies because it was analysed at different stages of the chicken production chain. The most common FU used in the reviewed studies was either 1 t of carcass weight (Leinonen et al. 2012; Wiedemann et al. 2010) or 1 kg live weight (Pelletier 2008; Wiedemann et al. 2010) or 1 kg of product at a number of different endpoints in the supply chain (Pardo et al. 2012; Bengtsson and Seddon 2013). Only a few studies used different FUs related to downstream processes. Davis and Sonesonn (2008) estimated the environmental impacts of two different chicken-based meals in order to identify the opportunities for environmental load reduction in the consumption phase. Similarly, Pardo et al. (2012) and Katajajuuri et al. (2008) used FUs related to further processed chicken meat, respectively 0.6 kg of sliced chicken breast fillet packaged in modified atmosphere and 1 kg of carcass weight of marinated breast fillet, in order to verify the environmental sustainability of post-farm processes. MacLeod et al. (2013) assessed the impacts in kg of CO_2-eq per kg of protein content in order to make the results comparable among different livestock products.

5.7.2.3 System Boundaries

Usually the studies related to the environmental impact estimation of poultry productions consider the cradle to farm gate system boundaries (SB). This, according to Bengtsson and Seddon (2013), is because of the lack of sufficiently detailed information in the cradle to retail or consumer supply chains. Hence, the reviewed studies were divided between those that analysed the supply chain only to the farm gate (Leinonen et al. 2012; Cederberg et al. 2009, Pelletier 2008), those that included slaughter processes (Katajajuuri et al. 2008; Williams et al. 2006), and those that included the other downstream processes (packaging, distribution, transport to retailer, etc.) to the use and disposal phases (Pardo et al. 2012; Davis and Sonesson 2008; Wiedemann et al. 2012; Bengtsson and Seddon 2013). Although all the studies include the upstream processes from the raw material extractions, the studies including post-farm processes usually simplify the input/output flows related to the agricultural phase. Wiedemann et al. (2010) did not include soil carbon fluxes in their analysis or the use of irrigation in wheat production; Davis and Sonesson (2008) did not consider the pesticides used in feed crop production because of missing data. This was also the case for capital goods and other emission sources (cleaning materials, waste treatments, etc.) that are usually not included in environmental impact assessment of the entire supply chain.

5.7.2.4 Availability and Quality of Data

Data were provided by three main sources in these studies, national inventory studies, simulated supply chain studies and industry data studies, depending on the scope of the studies. Cederberg et al. (2009), Pelletier (2008),Williams et al. (2006) and Wiedemann et al. (2010), performed national inventory studies, aiming to provide results that were representative of the country in question. All these studies collected data from a wide range of sources: national statistics, literature, direct involvement of industry and commercial stakeholders. Bengtsson and Seddon (2013) used data collected directly from an Australian industry that covered all the phases of the supply chain of chicken production. The same approach was used by Katajajuuri et al. (2008), who investigated a simulated Finnish chicken supply chain using data from the literature and commercial facilities. Leinonen et al. (2012) used an approach that applied a structural and mechanistic model to assess energy, materials, animal performance, crop production and nutrient input/output flows for the UK broiler industry. Davis and Sonesson' study (2008) was based only on previous LCA studies and explored the effects of improving sustainability measures in the post-farm phases of integrated supply chains. Most of the studies used IPCC methods for estimating GHG emissions arising from several sources within the poultry production chain: the use of energy and the handling of manure and wastes (MacLeod et al. 2013; Wiedemann et al. 2010; Leinonen et al. 2012; Cederberg et al. 2009; Williams et al. 2006). Only two studies used different sources for assessing the emissions arising from the poultry supply chain (Pelletier 2008; Davis and Sonesson 2008) The first used data from previous poultry GHG and the latter used a model for Swedish supply chains' input-output assessment (System Analysis for Food and Transport—SAFT). Only a few studies collected data from different chicken rearing systems. Conventional, free-range and organic farming were compared by Wiedemann et al. (2010) and Leinonen et al. (2012); Bengston and Seddon (2013) and Boggia et al. (2010) compared only conventional and free-range systems. The only relevant difference between the three systems is related to the higher feed consumption and manure production of free-range systems (Leinonen et al. 2012); a further difference in the organic system is the use of organic feed for animal rearing.

5.7.2.5 Allocation of Burdens to Co-products

Co-products were identified at three points along the poultry supply chains. The first allocation point was the production of meat from breeding hens and eggs in the breeding system. The second allocation point was the production of litter and meat chickens at the grow-out farm. The third allocation point was at the slaughterhause, between primary and secondary products. The allocation methodologies used in the reviewed studies varied according to their specific goal and scope. The most frequently used method was economic allocation, which resulted in a much larger share of the impacts being allocated to meat production than other less valuable co-products (Bengtsson and Seddon 2013; Davis and Sonesson 2008). The reason for using the economic allocation was that it reflects the objective of the industry that optimises the products and co-products to achieve the highest economic return (Bengtsson and Seddon 2013). Katajajuuri et al. (2008) used meat mass to allocate the environmental impacts between co-products and Pelletier (2008) and Pelletier et al. (2013) used a gross energy content allocation criterion (mass-adjusted gross chemical content) in order to reflect the real biological flows and the associated environmental impacts. For the second allocation point described above, almost all the studies used the system expansion methodology for litter (manure and bedding) nutrients, accounting for manufacture and application of synthetic fertilisers and including all the emissions arising from the use of the fertilisers (e.g. Pelletier 2008; Wiedemann et al. 2010). The lack of data on the quantities of manure transferred from livestock production into arable production systems in Sweden forced Cedeberg et al. (2009) to avoid allocation by distributing all the resources used and related emissions from manure to chicken primary products. Davis and Sonesson (2008), who focussed on the analysis of chicken meal consumption habits, used system expansion to allocate the environmental impacts between chicken meal production and wastes produced after the consumption phase. They used system expansion replacing oil and coal (50/50) with wastes produced at the end of the chicken-based meals chain in heat production.

5.7.2.6 Life Cycle Impact Assessment (LCIA)

The most common areas of environmental impacts contribution analysed in the reviewed studies were energy use, GHG emission, ozone depletion, water use and those impact categories closely related to the feed production phase (MacLeod et al. 2013; Leinonen et al. 2012; Wiedemann et al. 2010; Pelletier et al. 2013; Pardo et al. 2012; Pelletier 2008). The methods used for assessing the environmental impacts of the chicken supply chain were different. For energy use impact assessment the CED (cumulative energy demand) was the most common among the reviewed studies (Wiedemann et al. 2010; Pardo et al. 2012; Pelletier et al. 2013; Pelletier 2008). The impacts related to GHG emissions were quantified by an IPCC (2006) Tier 2-type approach (Cederberg et al. 2009; Pelletier et al. 2013; MacLeod et al. 2013) or assessment of their contribution to climate changes with the GWP as indicator (Leinonen et al. 2012; Wiedemann et al. 2012). Some studies used complex impact assessment methodology: CML2- Baseline 2000 or Recipe 2008 (Pardo et al. 2012, Pelletier 2008) or different impact categories (Bengtsson and Seddon 2013; Davis and Sonesson 2008; Prudêncio da Silva et al. 2008; Katajajuuri et al. 2008). On a global scale, MacLeod et al. (2013) found that chicken supply chains produce 58 million t of eggs and 72 million t CW (Carcass Weight) annually and related GHG emission of 606 million t CO_2-eq; average emission intensity ranges from 5.4 kg CO_2-eq/kg CW for meat and 3.7 kg CO_2-eq/kg eggs. A similarly impressive result was obtained by Katajajuuri el al. (2008), who analysed the environmental impacts of 1 t of marinated and sliced broiler fillet in Finland and found a GWP of 3,635 kg of CO_2-eq Very low values were found by Boggia et al. (2010), however, who identified GHG emissions that varied between 0.66 and 0.70 kg of CO_2-eq per kg/kg of broiler meat. The organic rearing system was found to have the greatest impact in terms of GHG emissions. Indeed, Wiedemann et al. (2010) found that this system produced 2.86 kg of CO_2-eq/kg of CW, more than the 2.38 and 1.89 CO_2-eq/kg of CW found for the free-range and conventional systems respectively. However, organic systems resulted in a lower environmental impact in terms of energy use (12.8 MJ/kg CW) than free-range and conventional systems (respectively 16.8 MJ/kg CW and 20.4 MJ/kg CW). Different results were reported by Leinonen et al. (2012), who found the organic system had the greatest impact in terms of EP (eutrophication potential), primary energy use and land occupation than the conventional system, that recorded lower impacts for GWP (global warming potential) and AP (acidification potential). The differences between the two studies are because of the different FUs. Indeed, the latter used 1 t of CW as FU, implying a smaller number of chicks but a longer productive cycle than that of the conventional system. The longer cycle meant a higher amount of feed needed in organic system with feed production that resulted in all the analysed studies as the most impactful phase in the poultry supply chain.

5.7.2.7 Interpretation

The most relevant areas of environmental impact in the poultry supply chain were, for all the analysed studies, feed production followed by grow-out housing, meat processing and breeding. These results can be ascribed to fuel, energy and fertiliser use in feed production and to manure production and management in the grow-out phase (Katajajuuri et al. 2008; Davis and Sonesson 2008; Wiedemann et al. 2010; MacLeod et al. 2013; Pelletier et al. 2013; Bengtsson and Seddon 2013). The contributions from feed production ranged from ~45 to 82.4 % (Katajajuuri et al. 2008; Pelletier 2008). When different impact categories were considered, feed production was found to be responsible for ozone depletion, acidification potential and eutrophication potential (Pelletier et al. 2008). In particular, Davis and Sonesson (2008) found that almost 90 % of eutrophying emissions originated from feed production and that ammonia (NH_4) emitted from farm activities was the main culprit responsible for acidification. When the overall environmental impacts of the feed production phase were considered, nitrous oxide was found to be the dominant emission source, ranging from 49 to 59 % (Wiedemann et al. 2010). Cederberg et al. (2009) found that carbon dioxide contributed 39–47 % of the overall emissions throughout the supply chain. Analysing the poultry supply chain from cradle to meat processing gate, Bengtsson and Seddon (2013) identified the grow-out phase as the most relevant in terms of global warming and non-renewable energy depletion; the meat processing phase contributed only 10–20 % of the overall environmental impact for more than 10 impact categories analysed, mainly energy consumption, water use and wastewater production. Among the different meat production systems, poultry meat production appears to be the most environmentally efficient because of several factors (identified by Pelletier et al. 2008) in a protein energy return on investment[4] (EROI) equal to 17.7 % compared with reported values for other livestock production systems such as beef (2.5 %), sheep (1.8 %) and pigs (7.1 %) (Pimentel 2004). The most relevant were identified by Williams et al. (2006) in very high feed conversion and high daily gain.

5.7.3 Critical Analysis

The reviewed studies agree that the use of primary data for the different production phases, in particular those related to feed production, could reduce the uncertainty

[4] EROI is a dimensionless index used to compare the relative efficiency of energy use per unit protein produced by different food systems.

of impact assessment. At the same time, almost all of the studies identify the chicken meat and egg production chains as the most efficient of the livestock production systems from the environmental perspective. The agricultural phase is recognised as the most relevant in terms of environmental emission generation, not only for conventional production systems but also for free-range and organic systems. Results reported for free-range and organic grow-out systems are almost the same as for the conventional ones, as regards the longer chick production cycle, the higher mortality rate and consequently the higher daily feed intake and manure production. Moreover, chicken production (meat and eggs) is defined as having the highest feed conversion ratio (FCR) and thus constitutes more sustainable livestock production. These data, considering the high proportion of environmental impacts allocated to the agricultural phase, suggest that the adoption of low impact agricultural practices by feed ingredient suppliers (which, for example, reduces the use of energy-intensive synthetic fertilisers and emissions resulting from their application) can significantly affect the environmental performance of chicken products. MacLeod et al. (2013), who analysed GHG emissions from the poultry sector on a global scale, found room for environmental impact reduction in several areas: reduction of land use changes (LUC) and improving efficiency in fertilisation management and in energy use both on-farm and off-farm. Davis and Sonesson (2008), analysing two meal types based on chicken in Sweden, found a promising emissions reduction strategy in terms of replacing oil and coal in heat energy production with the wastes generated at the end of life of the supply chain. The same authors recommend a shift in consumption habits, i.e. increasing the consumption of poultry-derived proteins because they have a lower environmental impact than other animal protein sources. This proposal seems difficult to achieve, and furthermore, as suggested by the same authors, requires a balance between the increase of the impacts because of increased demand for poultry protein and the reduction of the production of other livestock sectors.

Conclusions and Lessons Learned

This environmental impact assessment of the livestock sector presents some critical issues that may occur regardless of the methodologies used, but that in the case of LCA or LCA-based methodologies make the evaluation extremely complex. The complexity of approaches, data requirements, and model specifications has become so high that some standardisation is necessary to make things more credible and comparable. The variability in the results of all the livestock supply chains is caused by the difference in the production systems and methodological choices (functional unit, system boundaries, allocation method, etc.). For instance, in beef production systems a very wide variety, ranging from very intensive to very extensive (Nijdam et al. 2012), was observed; in dairy farms, which generally produce more than one product, the whole impact of dairy activities should be shared and allocated between all of them and the environmental performance of their processed products also depends on the use of milk supplied by different farms with different rearing systems. Of special interest in LCA analysis of the livestock production systems was the definition of the functional unit (FU), or, rather, the unit respecting which the environmental impacts are defined. The choice of a “corrected functional unit”, such as fat and protein or energy, could be an efficient approach which takes into account the nutritional value of livestock products and allows the comparison of the results of different studies. Livestock products differ in terms of production techniques and economic values, protein content and live weight (Nguyen et al. 2012a). Thus, the use of more complex FUs is mandatory for studies focussed on the evaluation of the environmental and economic impacts of the whole beef supply chain (Weidema et al. 2009) or comparison of different livestock sectors (beef, pigs, chickens, sheep and goats, etc.) (de Vries and de Boer 2010). The choice of FU is critical, as pointed out in Sect. 1.4, for studies addressing environmental impacts and load allocation of milk and meat in beef production (Cederberg and Stadig 2003). A common characteristic of all the analysed studies is the heterogeneity in system boundaries’ (SB) definition. Besides the variety and the complexity of livestock transformation processes, a relevant critical methodological point for LCA analysis, the inclusion of crop production (fodder especially) in rearing systems’ impact assessment is a critical and debated question. Meat and milk production systems are characterised, moreover, by a high number of co-products and by-products, let alone the production of both meat and milk. Almost all the studies reviewed consider the cradle to farm gate life cycle and exclude capital goods from the analysis. However, the environmental impact from capital goods has been included in some recent publications which found that capital goods contribute significantly to the total impact of agricultural production systems (Blengini and Busto 2009; Frischknecht et al. 2007). As regards different methods of impact assessment and classification there are several approaches (often IPCC 2007; EDIP; CML; CED; Impact 2000+ and Eco-indicator 99) that are choosen according to the goal and scope of the studies and their effectiveness in showing results. The phase with the greatest impact, in all the studies, is animal rearing; enteric CH_4, NH_3 and N from animal excreta are the major culprits responsible for environmental loads. The land use impact category is particularly relevant for beef and dairy production, which has the highest impact compared with other meat production systems (pigs and chickens). Data availability remains a long-standing problem and is hard to solve, as witnessed by scientific studies dedicated to system definition and inventory construction. A more complete picture of the environmental impacts (some of which have not been adequately addressed so far) and of the phases along the whole chain should be included as improvements for future research. A strong interaction between research experts and economic organisations (e.g. farmer’s associations) could make the LCA methodology useful in the decision-making process connected to the definition of an environmental chain strategy. This interaction is useful for many reasons: to support LCA data requirements, improving and expanding databases; to support the standardisation process and levels; to stress the main gaps in current knowledge on which future research and developments should be focussed.

From a methodological perspective, there are many studies oriented to the evaluation of environmental impacts; only a few of them (Weidema et al. 2009; Basarab et al. 2010; Van Middelaar et al. 2011; Oishi et al. 2013) combine LCA with the evaluation of economic impacts (e.g. life cycle costing, net present value, value added, etc.). A few studies combine LCA with farm simulation models (Beauchemin et al. 2010; Leip et al. 2010; Clarke et al. 2012), since they can give useful information for the improvement of rearing systems and related environmental impacts.

Only Weidema et al. (2009), using a hybrid methodology combining macroeconomic data (input/output tables) with the emissions generated by the analysed productive processes (Suh et al. 2004), evaluate the whole life cycle from cradle to fork. They include the transformation, marketing and use phases in the SB.

In LCA analysis of the livestock sector, economic allocation is the most frequently used method although the ISO standards recommend physical or biological criteria (carcass weight, protein content, etc.) in preference (Yan et al. 2011). Indeed, De Vries and de Boer (2010) and Nguyen et al. (2012a) used these allocation criteria to identify studies to include in their review of the environmental impacts of different livestock productions and to assess the environmental impacts of four beef farming systems. Moreover, economic allocation methodology does not account for the environmental benefit produced by the milk system with the reduction of biological methane and ammonia emissions (Cederberg and Stadig 2003). Thus, the application of system expansion is preferable. When a system expansion is applied, for example to dairy and beef production systems, it is assumed that the meat from both the culled dairy cow and the raised dairy calf replaces beef meat produced in a cow-calf system. The choice of meat and also the production system used to obtain this by-product is crucial because the amount of environmental impact from beef production depends on it (Flysjö et al. 2012). However, these allocation methods could be avoided, according to Weis and Leip (2012), who suggest allocating input and output flows of the processes for raising and fattening young animals for meat, and dividing the activities of dairy and suckling cows for milk production into the raising of young animals during pregnancy (which is allocated for meat) and the production of milk. The management of by-products, in LCA analysis, is another critical point. For livestock production, this is the case of manure because of its dual simultaneous effect (Garnett 2009): manure increases the nutrients in the soil and also the soil's carbon sequestration potential (FAO 2001). It was estimated that, globally, almost 22 % of the total nitrogen and 38 % of the total phosphate used for agriculture productions derive from animal excreta, of which half come from beef production (UNEP, n.d.). At the same time, manure, according to the report "Livestock's Long Shadow—Environmental Issues and Options" (FAO 2006), is responsible for N_2O and CH_4 emissions, contributing 5 % to global GHG emissions. In the majority of the studies analysed, manure management is considered as a means for the production of organic fertiliser that, depending on its relative nitrogen content,[5] is used as a substitute for chemical fertiliser (e.g. Casey

[5] Manure composition and thus its rate of chemical nitrogen fertiliser substitution are assessed by different methodologies related to the analysis of the physiological mechanisms of animals and diet composition (Ogino et al. 2004; Pelletier et al. 2010; Basarab et al. 2010; Leip et al. 2010).

and Holden 2006; Beauchemin et al. 2010; Nguyen et al. 2010; Basarab et al. 2010; Beauchemin et al. 2011). This methodological approach allows the authors to assess the environmental impacts allocated to livestock production better without leaving out the manure use impacts (linked to acidification, eutrophication and GHG emissions); but Garnett (2009) considers it incomplete, because as a natural source of nitrogen, manure reduces the need for chemical fertiliser production and transport. According to Weidema et al. (2008) and Leip et al. (2010), manure must be considered in any system expansion approach aspiring to perform good LCA analysis. This means considering the impacts of manure fertilisation on the entire chemical N fertiliser supply chain, defining the level of substitution of chemical nitrogen fertilisers by manure (it varies from 20 to 60 % depending whether manure is spread during grazing or collected from stables) (Nguyen et al. 2010). Many LCA studies on livestock production consider the land use impact categories only in terms of the m^2 required annually for livestock production (Cederberg and Stadig 2003; Cederberg et al. 2009; Doreau et al. 2011; Ridoutt et al. 2011; Flysjö et al. 2012). This category is particularly relevant for beef production, which has higher impact than certain other meat production systems (pigs or chickens). In particular, as regards meat and dairy production, to obtain 1 kg of beef meat some 27–49 m^2 of land are needed (de Vries and de Boer 2010). The high values of land required for beef production are related to two factors: the low efficiency of feed conversion rate and the lowest number of annual progeny compared with pigs and chickens. Within the beef production system, the impact on the land use category is higher for suckler-cows than the system in which the herds are a co-product of milk (de Vries and de Boer 2010). However, despite the importance of including GHG emissions because of land use changes (Flysjö et al. 2012), there is no consensus on how to include those emissions in environmental impact estimates. Three methods which include land use and land use changes in LCA of milk production were analysed by Flysjö et al. (2012), who clearly showed how GHG emission estimates differed depending on the methodology used. Many LCA analyses deal with this problem by identifying the required land to produce a specific amount of output in a given period of time (Lindeijer 2000). This type of information is useful for evaluating land use efficiency, but, according to Nguyen et al. (2010), there are other issues that need to be considered: "the opportunity cost of land" (Garnett 2009, pp. 493), or rather the cost of the land if it was used for other purposes, and the potential land use change that derives from an increase in demand for land and land products. The opportunity cost of land use has been estimated in terms of emitted CO_2 as between 2.8 and 2.2 kg CO_2/m^2 year depending on whether it was converted to crop production or grassland (Nguyen et al. 2010) In another study, Nguyen et al. (2012b), using this indicator, found a potential impact reduction on GW between 20 and 48 % if grasslands were converted to forests rather than to annual crops. Roer et al. (2013) considering four sub-processes in the life cycle of bovine meat and milk production (concentrate, forage, cattle rearing and others) found that the GWP of meat production varies from 17.7 to 18.4 CO_2-eq/kg of carcass weight. This is also the only impact category which depends on cattle rearing and accounts for 45–48 % of the total GWP of the whole life cycle.

The methodological innovations emerging from this review seek to limit this huge variability by focusing on a combination of models from representative livestock farms and related emissions assessment with the LCA analysis. A thorough review of these methodological approaches can be found in Crosson et al. (2011), who summarise the GHG emissions per kg of product in 35 whole farm modelling studies (from 31 published papers) of beef and dairy cattle production systems. Beauchemin et al. (2010, 2011) use the HOLOS model, which is based on IPCC methodology, to assess CH_4, N_2O and CO_2 emissions at the farm level. This model accounts for all the emissions linked to the beef production supply chain (fertiliser and herbicide production and transport, feed production, etc.). Modelled results were used in LCA to assess the environmental impacts of beef production in Canada. Bonesmo et al. (2013) used the HOLOS model adapted to the Norwegian situation (HOLOSNor) to assess the GHG emission intensity of Norwegian dairy and beef production systems. They found that the main culprits responsible for GHG emissions per kg of carcass weight were, in order of relevance: soil's nitrous oxide emissions, indirect energy use, soil C loss and enteric methane which was not significantly correlated to the variation in total GHG emissions per kg of carcass weight (Bonesmo et al. 2013). The same approach was used in a GGELS project (Leip et al. 2010), in which the CAPRI model was used to define the six livestock systems of EU−27 (Loudjani et al. 2010). The same method has been used to assess the GHG emissions and removals in the whole EU livestock sector (at regional scale) including: methane, nitrous oxide, carbon dioxide and also land use and land use changes (Weiss and Leip 2012) and CAPRI model results were used as input and output of the LCI. Oishi et al. (2013) evaluated the economic and environmental impacts resulting from changes in the age of animals at slaughter and in diet in the cow-calf system (race Japanese Black) in Japan, using as indicators the actualised net income and the environmental impacts from an LCA analysis. The input for economic evaluation was based on the continuous coupling system that was found (Oishi et al. 2011) to be the most economic and efficient one. The LCA analysis was built on the system presented by Ogino et al. (2007); a cradle to farm gate system with 1 kg of total live weight as FU. Then, the LCA results were normalised and the relative contribution of each category to the environmental impact of the whole system aggregated into a single dimensionless indicator, as suggested by Brentrup et al. (2004), for comparison with the economic indicator. Finally, a multi-criteria analysis was used to aggregate global warming (GWP), acidification potential (AP) and eutrophication potential (EP) impact categories following the approach suggested by Hermann et al. (2007). The results of this complex study showed that increasing culling parity to an economically efficient level can reduce the total environmental impact; changes in diet have no effect on environmental and economic impact (Oishi et al. 2011). Capper (2011) also pointed out that reducing time-to-slaughter can represent an option for decreasing CO_2-eq emissions per unit of beef because of the lifetime dilution of maintenance energy costs.

The combination of a bioeconomic model for livestock management with partial LCA (Carbon Footprint, Ecological Footprint, etc.) in order to assess the environmental impacts of livestock production systems represents an innovative approach to the environmental impact assessment for this sector. One example is the use of GBSM (Grange Beef System Model) with a partial LCA analysis in order to assess the GHG emissions of beef production systems The integration of farm management models with LCA analysis has also been suggested (e.g. Beauchemin et al. 2011; Oishi et al. 2011; Foley et al. 2011; Clarke et al. 2012), and some studies use other impact assessment methodologies to quantify the environmental loads produced by livestock production. All these attempts, which are in line with Place and Mitloehner (2012), represent efforts to account for the complex biogeochemical processes that occur within the rumen of cattle fed on different diets and also to account for varying management strategies such as age-to-slaughter, which can meaningfully alter the environmental load per unit of beef.

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