CENTRO INTERNACIONAL DE ESTUDOS
DE DOUTORAMENTO E AVANZADOS
DA USC (CIEDUS)

TESE DE DOUTORAMENTO

SUSTAINABILITY DIAGNOSIS OF ALTERNATIVE TECHNOLOGIES FOR THE MANAGEMENT OF AGRI-FOOD AND URBAN WASTE

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ESCOLA DE DOUTORAMENTO INTERNACIONAL

PROGRAMA DE DOUTORAMENTO EN ENXEÑARÍA QUÍMICA E AMBIENTAL

SANTIAGO DE COMPOSTELA

2018

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Sustainability diagnosis of alternative technologies for the management of agri-food and urban waste

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Sustainability diagnosis of alternative technologies for the management of agri-food and urban waste

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AGRADECIMIENTOS

La presente tesis se ha podido llevar a cabo gracias al soporte financiero proporcionado por la agrupación estratégica CRETUS (AGRUP2015/2) así como diversos proyectos de investigación: ManureEcoMine (603744), BBVA 2015 "Innovación y sostenibilidad ambiental de residuos alimentarios urbanos en Smart Cities" (BBVA 2015-PO027) y COST Action (ES1202).

Sin embargo, ha sido la contribución personal de todos aquellos que me han acompañado en el camino la que ha supuesto el mayor apoyo. Es por ello que me gustaría agradecer, en primer lugar, a mis directoras de tesis María Teresa Moreira y Sara González, la oportunidad que me brindaron al confiar en mí para desarrollar esta tesis, su paciencia y dedicación a lo largo de los últimos casi cuatro años de mi carrera académica. Se merece también una mención particular el profesor Gumersindo Feijoo, por su inestimable colaboración aún sin estar directamente vinculado al desarrollo de la tesis.

Asimismo, me gustaría expresar mi gratitud a Evina Katsou y Jesús Boschmonart Rives por haber dedicado también parte de su ocupado tiempo (horas, días, semanas o incluso meses) a acogerme con cariño y guiarme en mi carrera (de fondo) investigadora. Muchas gracias a ambos.

Por supuesto, muchas gracias también a todos los compañeros y compañeras del Biogroup con los que he podido compartir algún momento durante estos últimos años. En especial, muchas gracias a Paula, Andrés y Chechu (y también a las nuevas incorporaciones del último año) por vuestra buena predisposición para acogerme en un grupo donde el buen humor está siempre asegurado, aún sin haber podido compartir todos los descansos "oficiales" para el café que me hubiesen gustado. Y, por supuesto, muchísimas gracias a Xela por nuestros momentos de desconexión mutuos y su apoyo en las etapas más duras, especialmente en estos últimos meses.

Llegados a este punto, no podía olvidarme de todos mis compañeros de la línea de ACV que en algún u otro momento han colaborado para que pueda recordar mi etapa de tesis con cariño. También a aquellos que han terminado ya su ciclo en el grupo, como Paula y Pedro, que me tendieron la mano en mis inicios cuando todavía me estaba adentrando en el apasionante mundo del análisis ambiental; y, por supuesto, muchísimas gracias a María, porque aun habiendo compartido sólo unos meses juntas, todavía hoy echo de menos su alegría y compañía en el día a día.

Y, como no, inmensas gracias a todos los compañeros que desde hace ya unos meses han hecho del laboratorio 2.1 de la ETSE un entorno de trabajo realmente agradable. Muchas gracias a Andrea, Sara, Xavi e Iana por estar siempre dispuestos a dar lo mejor de vosotros mismos en beneficio del grupo; y también a las incorporaciones más actuales, Antonio y Alba, a los que deseo mucha suerte en su recién estrenada carrera investigadora. Por último, pero no menos importante, mi reconocimiento especial a Lucía, por ayudarme y aconsejarme desde el primer al último día de esta tesis, siempre con una sonrisa y sin pedir nada a cambio.

Y ya para terminar, MUCHAS GRACIAS: a Celeste, por demostrarme el significado de la amistad verdadera; a Alberto, por llegar a mi vida para ayudarme a cumplir mis sueños y convertirse en el compañero de viaje ideal; a mi hermana María, por su ejemplo de fortaleza y tenacidad y su inquebrantable protección y confianza en mí (incluso más que yo misma); y a mis padres, a los que respeto y admiro profundamente, por su apoyo incondicional y su eterno ejemplo de superación ante los obstáculos en el camino. Sin vosotros, esta tesis no hubiese sido posible. ¡Muchas gracias de corazón a todos!

OUTLINE OF THE THESIS

Chapter 4. Environmental assessment of livestock husbandry: cow dairy sector

Chapter 5. Environmental assessment of livestock husbandry: pork sector

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Abstract (English)

Global population growth, rising incomes and urbanisation are joining forces to pose serious challenges to the waste management sector, while the natural resources to support such provision of services are often more limited. This situation is also responsible for the serious pressures on the environment as a whole, highlighting the need to establish sustainable production and consumption patterns, while mitigating their related effects and ensuring environmental protection and human health.

In this context, the main objective of this doctoral thesis was to evaluate the environmental sustainability of different alternatives for the management of waste coming from both agricultural and urban sectors, including conventional practices and advanced technologies. To this aim, the principles of the LCA methodology were commonly applied to some particular waste management situations (case studies), in combination with other internationally recognised assessment tools, such as the AHP method for multi-criteria decision analysis, among others. Paying attention to the agricultural framework, pork and dairy sectors were assessed in detail, focusing not only on the processes responsible for the generation of waste (mainly manure), but also on the previous stages and their possible connection with more advanced waste management strategies. Different treatment alternatives for the recovery of energy and materials from waste generated in urban environments, including developing regions and more developed areas, were also evaluated. In this case, social participation and economic viability were also integrated with environmental outcomes to respond to the constraints of waste management decision-making.

According to the results, it can be concluded that many impacts on the environment have already been avoided and/or minimised due to advances in solid waste treatment and valorisation. However, nowadays, waste management remains a critical issue for the scientific community and the society as a whole. It is for this reason that the range of existing technologies should continue to expand, as well as mitigation strategies, with the aim of preserving environmental quality and protect the public welfare.

Keywords: livestock waste, Municipal Solid Waste (MSW), environmental assessment, sustainability.

Resumen (Spanish)

El crecimiento global de la población, el aumento de los ingresos y la urbanización han unido fuerzas para plantear serios desafíos en el sector de gestión de residuos, mientras que los recursos naturales necesarios para satisfacer esta provisión de servicios a menudo son más limitados. Esta situación es responsable, además, de las graves presiones sobre el medio ambiente, haciendo patente la necesidad de establecer patrones de producción y consumo sostenibles, a la vez que se mitigan sus efectos derivados y se garantiza la protección ambiental y la salud humana.

En este contexto, el objetivo principal de la presente tesis ha sido evaluar la sostenibilidad ambiental de diferentes alternativas de gestión de residuos procedentes de los sectores agrícola y urbano, incluyendo prácticas más convencionales junto con tecnologías avanzadas. Para ello, los principios de la metodología ACV fueron comúnmente aplicados a diferentes situaciones particulares (casos de estudio) de gestión de residuos, en combinación con otras herramientas de evaluación internacionalmente reconocidas, destacando el método AHP para la decisión multi-criterio, entre otras. Atendiendo a los entornos agrícolas, los sectores lácteo y porcino fueron evaluados en profundidad, centrando la atención no sólo en los procesos responsables últimos de la generación de residuos (principalmente estiércol), sino también en las etapas previas y su potencial vinculación con las estrategias avanzadas de gestión posteriores. De forma análoga, se evaluaron también diferentes alternativas de tratamiento para la recuperación energética y material a partir de los residuos generados en entornos urbanos, incluyendo regiones en desarrollo y áreas más desarrolladas. En este caso, la participación social y la viabilidad económica se integraron también conjuntamente con los resultados ambientales para resolver las limitaciones de la toma de decisión para la gestión de los residuos.

De acuerdo a los resultados obtenidos, se puede concluir que muchos de los impactos en el medio ambiente ya han sido evitados y/o minimizados gracias a los avances recientemente desarrollados en el tratamiento y valorización de los residuos sólidos. Sin embargo, a día de hoy, la gestión de estos residuos continúa suponiendo un desafío para la comunidad científica y la sociedad en su conjunto. Es por ello que el espectro de tecnologías debería seguir aumentando, así como las estrategias de mitigación, con el objetivo de preservar la calidad ambiental y proteger el bienestar público.

Palabras clave: residuos ganaderos, residuos sólidos urbanos (RSU), análisis ambiental, sostenibilidad.

Resumo (Galician)

O crecemento global da poboación, o aumento dos ingresos e a urbanización uniron forzas para supoñer serios desafíos ao sector da xestión de residuos, mentres que os recursos naturais necesario para satisfacer esta provisión de servizos a miúdo son máis limitados. Esta situación é responsable, ademais, das graves presións sobre o medio ambiente, facendo patente a necesidade de establecer patróns de produción e consumo sostibles, á vez que se mitigan os seus efectos derivados e se garante a protección ambiental e a saúde humana.

Neste contexto, o obxectivo principal da presente tese foi avaliar a sostenibilidade ambiental de diferentes alternativas de xestión de residuos procedentes dos sectores agrícola e urbano, incluíndo prácticas máis convencionais xunto con tecnoloxías avanzadas. Para iso, os principios da metodoloxía ACV foron comunmente aplicados a diferentes situacións particulares (casos de estudo) de xestión de residuos, en combinación con outras ferramentas de avaliación internacionalmente recoñecidas, destacando o método AHP para la decisión multi-criterio, entre outras. Atendendo aos ambientes agrícolas, os sectores lácteo e porcino foron avaliados en profundidade, centrando a atención non só nos procesos responsables últimos da xeración de residuos (principalmente estrume), senón tamén nas etapas previas e a súa potencial vinculación coas estratexias avanzadas de xestión posteriores. De forma análoga, avaliáronse tamén diferentes alternativas de tratamento para a recuperación enerxética e material a partir dos residuos xerados en ambientes urbanos, incluíndo rexións en desenvolvemento e áreas máis desenvolvidas. Neste caso, a participación social e a viabilidade económica integráronse tamén conxuntamente cos resultados ambientais para resolver as limitación da toma de decisión para a xestión dos residuos.

De acordo aos resultados obtidos, pódese concluír que moitos dos impactos no medio ambiente xa foron evitados e/ou minimizados gracias aos avances recentemente desenvolvidos no tratamento e valorización dos residuos sólidos. Sen embargo, a día de hoxe, a xestión destes residuos continúa supondo un desafío para a comunidade científica e a sociedade no seu conxunto. É por iso que o espectro de tecnoloxías debería seguir aumentando, así como as estratexias de mitigación, co obxectivo de preservar a calidade ambiental e protexer o benestar público.

Palabras chave: residuos gandeiros, residuos sólidos urbanos (RSU), análise ambiental, sostenibilidade.

SECTION I

CONTEXTUALISATION

CHAPTER 1. INTRODUCTION

Summary

Global population growth, rising incomes and urbanisation are joining forces to pose serious challenges to the food and agriculture systems, while the natural resources to support such provision of services are often more limited. Moreover, this situation is responsible for the severe pressures on the environment as a whole, requiring approaches that increase production while mitigating the related effects and ensuring food security and human health.

In this context, progress in waste treatment and mitigation, both in agriculture and in urban frameworks, can have a crucial impact on future prospects. As a result, a number of waste management technologies have been developed in recent decades, mainly in developed regions, but also in developing economies. However, despite the many plans devised to date, only some of them have already been implemented and established on a large scale. In this sense, while some countries continue working on economically viable solutions, others are subjected to legal limitations on environmental challenges.

Therefore, although significant progress has been made in waste management to date, further research is still needed to design economically viable and socially responsible strategies, but without compromising their respect for the environment and natural resources. This philosophy has guided current European legislation, thinking of the potential role of treatment systems in the conversion of waste into value-added products.

CHAPTER 1 - INDEX

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1.1 WASTE GENERATION AND THE ENVIRONMENT

Waste can be defined as any substance or object that the holder discards, intends to discard or is required to discard (EU Directive, 2008). According to the list published by the Commission Decision 2000/532/EC, the different types of waste can be classified into twenty main groups on the basis of factors such as the composition, origin and potential hazardous characteristics of the waste.

In this context, municipal wastes and similar commercial, industrial and institutional wastes including separately collected fractions are grouped together on the basis of European standards (Commission Decision, 2000). Among them, Municipal Solid Waste (MSW) is closely related to the activity of the general public and municipalities, comprising household waste and similar waste sources (Kawai and Tasaki, 2016; OECD, 2013).

Similarly, agricultural waste refers not only to residues directly produced in farming activities, but also includes waste from animal husbandry as a whole (Gerber et al., 2013). It is for this reason that this type of waste can be linked to wastes from horticultural, hunting, fishing and aquacultural primary production, food preparation and processing, in agreement with European classification (Commission Decision, 2000).

1.1.1 Agricultural framework

Agriculture plays a key role in global environmental issues, including greenhouse gas (GHG) emissions and climate change, water pollution and biodiversity losses, among others (Gerber et al., 2013). Within agriculture, the growing demand for agri-food products is changing the relationship between the livestock sector and the environment (FAO, 2009; FAO 2016); consequently, the environmental impacts of this sector have increased in recent times (Gerber et al., 2013). Among them, the poor management of livestock effluents has directly impacted on many environmental compartments, including air and soil pollution, and the subsequent transfer to surface and groundwater resources (FAO, 2009; Martinez et al., 2009).

- **Soil degradation**

Animal wastes (i.e. manure) have traditionally applied as soil amendments, as well as to enhance its physical properties, such as structure and moisture retention (Martinez et al., 2009). However, over application of manure can lead to the accumulation of both nutrients and heavy metals, responsible for damage to animal health and the environment (Martinez et al., 2009; Wang et al., 2016). Additionally, the main consequences from overloaded soils are directly related to their interaction with other environmental compartments: air and water (Martinez et al., 2009). Thus, they play a critical role in the retention, transformation and disposal of both gaseous and soluble compounds (Martinez et al., 2009; Wang et al., 2016). On the other hand, however, agricultural soils can also act as a sink for GHGs, partially offsetting their negative effects on climate change (Martinez et al., 2009).

- **Air quality and climate change**

Livestock activities contribute to air pollution and climate change through direct and indirect GHG emissions throughout the entire livestock production chain (FAO, 2009). However, most gaseous pollutants come from manure storage and other management activities (Ogbuewu et al., 2012). In fact, emissions from animal feed account for about 45% of the sector emissions mainly linked to the production and application of both organic and mineral fertilisers in agricultural soils (Gerber et al., 2013). Similarly, enteric fermentation and manure management (excluding application) are also responsible for major burdens at the farm level; however, the impacts from processing and distribution stages are primarily related to the production and use of fossil fuels (Gerber et al., 2013). Focusing on the relative contributions of the compounds, about 44% of emissions from the livestock sector are delivered in the form of methane (CH_4); the remaining part is shared between dinitrogen monoxide (N_2O) and carbon dioxide (CO_2), with contributions of about 29% and 27%, respectively (Gerber et al., 2013). Minor emissions can be attributed to hydrofluorocarbons (HFCs) on a global (Gerber et al., 2013).

However, on the contrary, the livestock sector also has substantial potential to contribute to climate change mitigation, as many of these impacts are increasingly susceptible to be avoided and/or reduced (Gerber et al., 2013). Realising this potential will require the promotion of research and development of changes in alternative stages of the livestock supply chain, particularly important in animal feed and manure management (see epigraph 1.2. Waste management: current strategies).

- **Water depletion and pollution**

The livestock sector accounts for about 8% of global water use, mainly related to irrigation steps of farming activities (FAO, 2009). In this sense, although water usage may differ from one livestock system to another, intensive production often results in much higher water consumption compared to extensive practices (FAO, 2009).

In addition, livestock production can also contribute to water pollution through leaching and nutrient runoff from soils (Martinez et al., 2009; Ogbuewu et al., 2012). As aforementioned, animal manure has been commonly spread on crops and pastures to fertilise agricultural soils and enrich their nutrient content (Ogbuewu et al., 2012). However, when excess manure is applied, nutrients can be lost through leaching and runoff to groundwater and surface watercourses (de Vries et al., 2015; Ogbuewu et al., 2012). Consequently, these water sources are polluted by a high concentration of nutrients (such as nitrates), leading to both environmental and human health problems (Ogbuewu et al., 2012). In this regard, nitrogen (N) and phosphorus (P) have been found as key elements in agricultural activities, accounting for the greatest potential for causing water pollution (de Vries et al., 2015).

Aware of this reality, the European community has taken on a great concern in recent decades, developing regulations to mitigate water depletion and improve water quality (Groundwater Directive, 2006; Nitrates Directive, 1991; WFD, 2000). In this regard, the Nitrates Directive (1991) emerged as one of the earliest legislative instruments to protect water quality in Europe by promoting the use of good agricultural practices to prevent nitrates from agricultural sources. Within the framework of this Directive, most Member

States designated those territories that could be affected by high nitrate levels as Nitrate Vulnerable Zones (NVZ), while other countries decided to provide an identical level of protection throughout their territory (Figure 1.1).

Figure 1.1. NVZ in European countries. Reference year: 2015 (JRC, 2015).

Moreover, Member States had to establish codes of good practices for farmers, together with specific action programmes, which were regularly reviewed to ensure their effectiveness in line with the objectives proposed by the Directive (European Commission, 2010). The main measures include periods when fertilisation is not allowed, minimum manure storage capacity and rules to control nutrient application in vulnerable soils (European Commission, 2010). In this sense, a common limit of 170 kg of nitrogen (from manure) applied per hectare and year was considered for all countries to comply with the provisions of the Nitrates Directive; however, the level of requirement for the other variables may vary between countries (European Commission, 2010). Overall results show good progress towards cleaner water, although agriculture remains a major source of water-related problems (European Commission, 2010).

- **Heavy metals contamination**

Some heavy metals, such as copper (Cu) and zinc (Zn), can be considered as essential minerals in feed ingredients (Ogbuewu et al., 2012). However, it had become common practice in intensive livestock to supplement the diet of animals with additional mineral mixtures, leading to oversupply. Since heavy metals are largely excreted in manure, these compounds can accumulate in soils and leach into watercourses (Ogbuewu et al., 2012; Wang et al., 2016). In the same vein, special attention should also be paid to other trace contaminants (such as antibiotics and veterinary drugs), which may also be responsible for significant negative effects on human health and ecosystems (Ogbuewu et al., 2012)

1.1.2 Urban framework

Consumption patterns have changed dramatically in recent decades, resulting in an increasing solid material stock and, consequently, in the growing generation of solid waste, especially in urban areas (Castaldi, 2014). Its volume and composition may vary greatly among different regions and economic scenarios; however, food waste tends to be the largest fraction, followed by plastics, paper and cardboard (Castaldi, 2014; Kawai and Tasaki, 2016). MSW generation worldwide is around 1.3 billion tonnes per year and, despite the reduction in generation rates in OECD (Organisation for Economic Co-operation and Development) countries, an increase of about 2.2 billion tonnes is expected by 2025 (Castaldi, 2014; Kawai and Tasaki, 2016). Consequently, its improper management remains responsible for potential risks to the surrounding environment.

- **Air quality and climate change**

Large amounts of GHG can be generated due to the decomposition of degradable waste from the indiscriminate disposal of MSW in landfills, acting as a common source of local environmental pollution (Alam and Ahmade, 2013; Lee et al., 2017). This problem is particularly acute in developing regions; only a few landfills in low-income countries comply with environmental standards, so that GHGs (mainly CH_4) emitted to air contribute significantly to climate change (Alam and Ahmade, 2013). Rapid urbanisation in some of these areas makes the situation even more difficult to address (Alam and Ahmade, 2013). Focusing on developing countries, other inadequate treatment systems, such as uncontrolled burning and incineration processes, can also have an impact on air quality, albeit on a smaller scale (Bhada-Tata and Hoornweg, 2012).

- **Soil and water pollution**

Liquid leachate also poses a critical local challenge (Alam and Ahmade, 2013). When landfills are not properly sealed, leachate can escape into surface and ground watercourses, leading to potential environmental impacts (Bhada-Tata and Hoornweg, 2012). In this regard, many modern landfills in developed regions have integrated measures to avoid the generation of leachate and favour evaporation to the detriment of their infiltration in soils (Alam and Ahmade, 2013).

- **Human health and biodiversity loss**

The light fraction can easily disperse from waste containers to surrounding areas, acting as a hazard to both wildlife and domestic species (Barnes et al., 2009). In addition, the high land requirements for the implementation of alternative management schemes (mainly landfills) result in the degradation of the natural environment (Amritha and Anilkumar, 2016; Liu et al., 2015). This can contribute to the extinction of several species and, therefore, the loss of biodiversity in the target area.

Similarly, uncollected waste can block sewers and public watercourses, which may result in both stagnant water and/or floods during dry and rainy seasons, respectively (Alam and Ahmade, 2013). Consequently, disease vectors could develop in the latter case, posing additional risks to public health (Bhada-Tata and Hoornweg, 2012).

1.2 WASTE MANAGEMENT: CURRENT STRATEGIES

1.2.1 Agricultural framework

In view of the above, it was demonstrated that the interaction of the livestock sector with the environment is complex and highly dependent on the location and the particular management practices (FAO, 2009). In this context, three main challenges need to be addressed towards a sustainable future involving modern livestock practices (Martinez et al., 2009): (i) policy rules on responsible intensification of livestock production; (ii) mitigation of environmental impacts due to the overexploitation of natural resources; and (iii) implementation of innovative livestock manure treatment schemes in developed countries and their adaptation to developing regions.

Regarding the latter, advanced treatment strategies can play a key role by providing a more flexible approach to conventional direct land application, while addressing specific problems such as malodours and/or nitrogen emissions (de Vries et al., 2015). They involve physical, chemical, mechanical or biological processes and their combination with each other.

- **Thermochemical conversion**

The main thermochemical processes for manure management include pyrolysis, gasification and liquefaction, where renewable energy is obtained as the end product (Figure 1.2). Heat can be also obtained from the **combustion** of manure; however, since energy storage is not possible in this case and waste ash is not properly recycled, combustion is not a priority option (Cantrell et al., 2008).

Pyrolysis takes the organic fraction of manure to obtain a mixture of char and bio-oil by applying heat (400–800 °C) in a non-oxygen atmosphere (Brownsort et al., 2009a,b; Roy and Dias, 2017). Char can be then used as a green energy source, as well as soil amendment and carbon sink (Qambrani et al., 2017; Quian et al., 2015); moreover, it can be applied in adsorption processes as substitute of conventional activated carbon (Qambrani et al., 2017; Quian et al., 2015). In this way, char production provide farmers economic and environmental credits (Cantrell et al., 2008). Compared to pyrolysis, **direct liquefaction** takes place at lower temperature (250–300 °C) in a pressurised atmosphere (Cantrell et al., 2008; Huang and Yuan, 2015). In this case, livestock manure is converted into bio-oil as primary product, with high conversion rates (Huang and Yuan, 2015; Ocfemia et al., 2006). Finally, focusing on **dry gasification**, air, oxygen or steam are used as reaction medium to convert the organic matter of manure into a gaseous mixture of low molecular weight, non-condensable hydrocarbon gases, together with small fractions of char as by-product (Qambrani et al., 2017). However, for this purpose, higher temperatures (700-800 °C) at atmospheric pressure should be used (Cantrell et al., 2008).

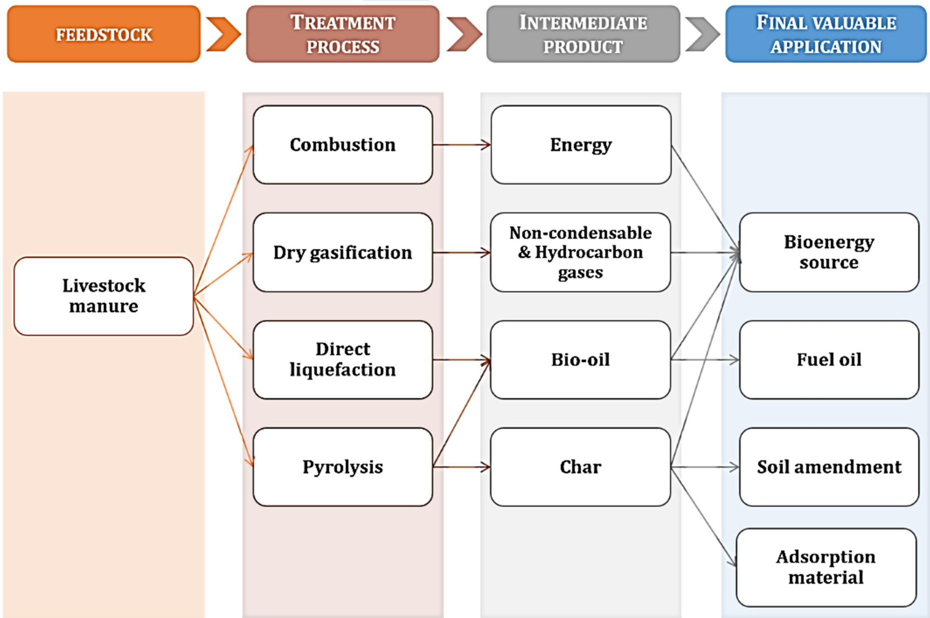

Figure 1.2. Overview of thermochemical conversion processes: intermediate products and final potential applications (adapted from Cantrell et al., 2008).

- **Biological conversion**

Anaerobic digestion (AD) has dominated the biological treatment of renewable resources in recent decades (Figure 1.3). In this process, anaerobic microorganisms are used to break down the organic fraction of waste and produce biogas (CH_4 and CO_2) as bioenergy source, along with digestate (Cantrell et al., 2008; Nasir et al., 2012). Different types of livestock manure are commonly used as feedstock, either alone or mixed with other organic sources (anaerobic co-digestion, AcoD). Three main stages can be distinguished (Li et al., 2011): hydrolysis, fermentation and methanogenesis. During hydrolysis, the complex compounds are broken down into soluble compounds, available for fermentative bacteria; they are responsible for converting them into alcohols, acetic acid and volatile fatty acids (VFAs), along with a mixture of H_2 and CO_2 (off-gas). Finally, these intermediate products are metabolised by methanogenic microorganisms into the final biogas stream. The efficiency of all stages and the yield of biogas depend to a large extent on operating conditions, such as pH and temperature (Ward et al., 2008). In this regard, AD can be carried out at different temperature ranges; mesophilic digestion takes places within 20–45 °C, while thermophilic digestion requires higher temperature conditions around 45-60 °C (Cantrell et al., 2008; Ward et al., 2008).

However, in the latter case, ammonia (NH_3) inhibition may occur during the digestion of livestock waste (Bousek et al., 2016). Various alternatives have been studied to solve this problem, although with a clear supremacy of ammonia stripping-absorption (Bousek et al., 2016; Laureni et al., 2012). In this process, digestate is percolated with gas, in such a way that dissolved gases (in this case, NH_3) are released and removed from the liquid phase (digestate) (Bousek et al., 2016). The spent ammonia digestate is recycled to the digester, while the released compounds (mainly NH_3) are removed from the gaseous stream in a scrubbing unit (Drosg et al., 2015; Melse et al., 2009). A valuable ammonia-rich compound is obtained, with potential use as a fertiliser in agriculture; the cleaned gas can eventually be reused in the stripping process (Drosg et al., 2015).

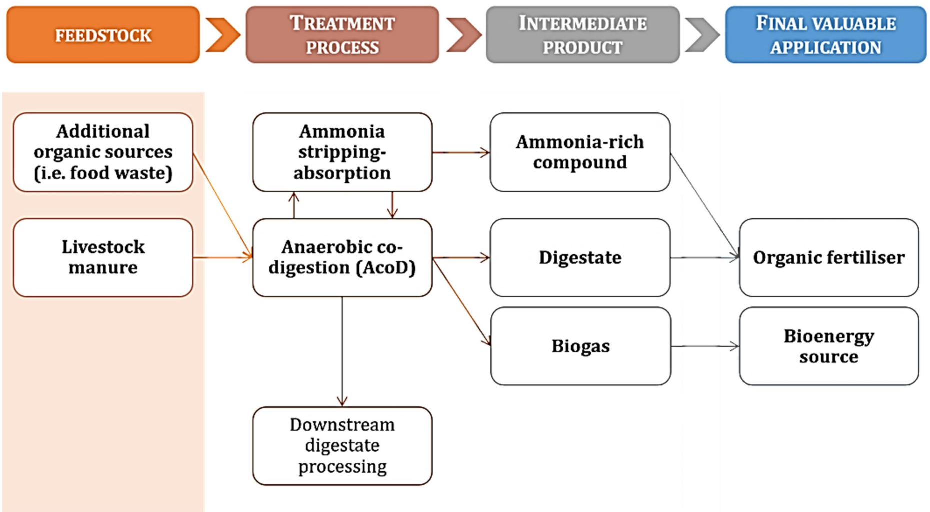

Figure 1.3. Overview of biological conversion processes: intermediate products and final potential applications (adapted from Cantrell et al., 2008).

However, since AD process does not affect the nitrogen content of digestate (only its mineralisation), its direct use as organic fertilisers still face constraints in the field (analogous to manure application), especially in areas with high nutrient surplus (Magrí et al., 2013; Nitrates Directive, 1991; Scaglione et al., 2013). Moreover, digestate has high water content, so that the feasibility of its direct application is also limited by transport requirements (Drosg et al., 2015; Magrí et al., 2013).

Accordingly, several downstream technologies have been recently enhanced to translate digestate into alternative added-value products (Figure 1.4). Initially, digestate is usually sent to a **solid/liquid (S/L) separation** step, in such a way that both upgraded fractions may be individually handled (Hjorth et al., 2010). Several technologies can be used for this purpose; however, decanter centrifuges are commonly selected, in which separation takes place by centrifugal force (Drosg et al., 2015). Additionally, this technology is enabled to provide a satisfactory degree of separation of the solid fraction with a larger phosphorus content; this will be essential when a consecutive **membrane separation** must be carried out (Drosg et al., 2015).

Typically, the solid digestate can be then used for fertilisation purposes (with shorter transport distances), as well as aerobically stabilised (composting) to obtain compost (Magrí et al., 2013; Martinez et al., 2009). During **composting**, the organic matter is aerobically decomposed, resulting in an increase in the temperature (60-70 °C) of the final product (i.e. compost); composting can thus facilitate the conversion of livestock waste into a hygienic fertiliser that can be applied to agricultural soils with lower risk of pathogens and weed seeds (Ogbuewu et al., 2012).

Regarding liquid digestate, it can also be used for irrigation, as well as be further processed to minimise nutrients (N, P) surpluses, where necessary (Magrí et al., 2013). In this regard, two main approaches can be followed: removal and recovery. Nitrogen removal mainly involves biological treatments based on the conversion of ammonium (NH_4^+) into nitrogen gas (N_2) discharged into the atmosphere (Magrí et al., 2013; Scaglione et al., 2013). Traditional biological nitrogen removal (BNR) generally encompasses **nitrification/denitrification** technologies: autotrophic nitrifying bacteria aerobically convert ammonia (NH_3) into nitrite (NO_2^-) and nitrate (NO_3^-), while anoxic denitrification from NO_3^- to NO_2^- and N_2 is carried out by heterotrophic bacteria (Rajagopal and Béline, 2011); therefore, NO_2^- is an intermediate in both phases. The main disadvantage of this conventional process lies in the need for additional input from external carbon sources (such as methanol and acetate), along with high oxygen requirements (Rajagopal and Béline, 2011). In order to address these deficiencies, advanced BNR processes such as **nitritation/denitritation** and **partial nitritation/anammox (PNA)**, have been found suitable for digestate processing (Magrí et al., 2013). With lower aeration rates and demand for carbon sources, these technologies have emerged as more cost effective alternatives (Ali et al., 2016; Castro-Barros et al., 2015; Scaglione et al., 2013, 2015). However, related N_2O emissions become a critical issue from an environmental perspective (Pijuan et al., 2014; Scaglione et al., 2013, 2015). Accordingly, several strategies on aeration supply are being evaluated in response to minimising N_2O emissions from BNR processes (Desloover et al., 2012; Rajagopal and Béline, 2011).

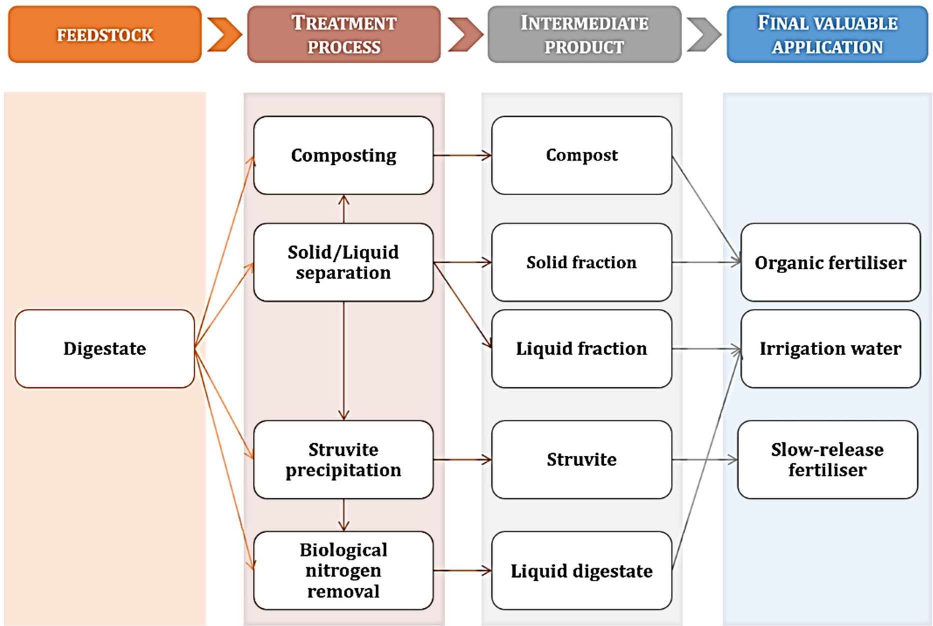

Figure 1.4. General description of alternative downstream digestate processing: intermediate products and potential end-use applications.

Moreover, strategies focused on nutrient recovery have been proposed based on the generation of separate nutrient-rich flows as potential green fertilisers (Magrí et al., 2013). Among them, **struvite precipitation** has gained significant relevance in recent times. In this process, NH_4^+ and phosphate (PO_4^{3-}) in the digestate react with magnesium (Mg^{2+}) to generate magnesium ammonium phosphate ($NH_4MgPO_4{\cdot}6H_2O$), commonly known as struvite. Since the availability of the different components must be ensured to guarantee success in struvite precipitation, Mg^{2+} is commonly added in excess; an external source of PO_4^{3-} may also be necessary (Drosg et al., 2015; Magrí et al., 2013). Therefore, the main disadvantage of struvite precipitation is attributed to the large amount of chemicals used, which increases its operational cost and related environmental impacts (Drosg et al., 2015). However, conveying relevant macronutrients, struvite can be used as a slow-release fertiliser, which partially offset its limitations (Liu et al., 2013; Magrí et al., 2013).

1.2.2 Urban framework

As mentioned above, the proper management of urban waste remains a priority issue towards the creation of sustainable communities capable of managing resources efficiently, ensuring social prosperity and environmental protection (Pires et al., 2011). In this regard, the waste sector has defined a generally accepted hierarchy of best practices for solid waste management (Figure 1.5). This pyramidal structure was developed based on the minimisation of related environmental impacts, so that the most environmental-friendly options rank in the top (Castaldi, 2014). Accordingly, significant efforts are being made to reduce waste generation rates in order to avoid waste disposal and promote the re-use of materials, followed by recycling (Castaldi, 2014). Subsequently, the remaining steps involving waste treatment downstream are considered; however, in this case, the hierarchy establishes discrimination between the available management alternatives, giving priority to the potential recovery of materials and energy sources (Castaldi, 2014).

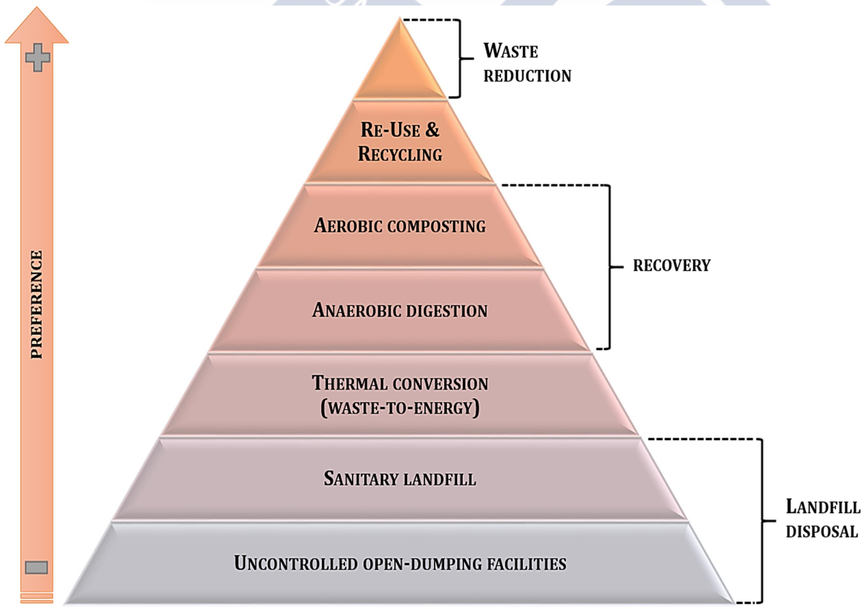

Figure 1.5. Solid waste management hierarchy (adapted from Castaldi et al., 2014).

- **Waste reduction**

Strategies focused on waste reduction aim to minimise the generation of MSW at source, either by re-designing products and production chains or by modifying consumption patterns.

- **Re-use and recycling**

The main advantages of re-use and recycling are the lower quantities of waste disposed of for treatment and the subsequent return of materials to the economy (Bhada-Tata and Hoornweg, 2012). The promotion of these practices has allowed the recovery of a significant fraction of MSW in developing regions, with the consequent reduction of the environmental impacts that this entails.

- **Waste recovery: aerobic composting and anaerobic digestion**

During **AD** (without oxygen), the organic fraction of MSW is treated in closed containers with the consequent generation of biogas, and subsequent CH_4 emissions (Bhada-Tata and Hoornweg, 2012). This biogas can be captured to be used as fuel to produce heat and/or electricity. Moreover, digestate is also produced as a co-product of the process, which can be applied as organic fertiliser in agricultural activities.

On the other hand, **aerobic composting** (with oxygen) is generally less complex and expensive than anaerobic treatment (Bhada-Tata and Hoornweg, 2012). It is developed either in open windrows or enclosed containers to produce compost, also used as organic fertiliser, analogous to digestate. Moreover, in this case, compost can also act as a soil amendment, to increase the organic matter content of soils as well as improve their water retention capacity (Martínez-Blanco et al., 2011). However, it should be noted that composting requires a separation stage at source to ensure the success of the process, otherwise contamination can lead to unusable products (Castaldi, 2014).

- **Thermal conversion**

When waste recovery through biological treatment is not possible, two other alternatives could be considered: landfilling and thermal conversion.

However, **waste-to-energy (WTE) systems** are considered preferable, with the aim of recovering energy from waste fractions rather than directly disposing of them in landfills (Bhada-Tata and Hoornweg, 2012; Castaldi, 2014). WTE technologies are capable of reducing MSW volumes by 90%, so that if residual ash is also used, they can be assumed as nearly zero-waste solutions (Castaldi, 2014).

Among the alternatives available, **incineration (with energy recovery)**, combustion and gasification have proven their safe and reliable performance (Castaldi, 2014). In contrast, incineration without energy recovery and open-cast combustion systems are particularly discouraged, mainly because of the consequences of air pollution (Bhada-Tata and Hoornweg, 2012).

- **Landfill disposal**

Landfills should be understood as the common MSW landfill in the absence or insufficiency of more sustainable treatment technologies. As the final destination of waste, they must be properly designed to ensure both the environment and human health. In this regard, modern **sanitary landfills** integrate landfill gas recovery from anaerobic decomposition of organic waste to reduce GHG emissions to the atmosphere, including energy recovery, and properly collect and prevent leachate migration into watercourses. (Bhada-Tata and Hoornweg, 2012; Castaldi, 2014). However, large investments are required to meet these environmental-friendly conditions (Castaldi, 2014).

Accordingly, more primitive landfills continue to exist around the world, equipped with primitive gas collection systems but without an energy recovery option (Castaldi, 2014). In this way, they are able to reduce CH_4 emissions and related burdens, which implies a valuable environmental advantage over **uncontrolled open-dumping facilities**, which are in the last hierarchical position (Castaldi, 2014). Unfortunately, such landfills remain a relevant option in many developing regions; however, their economic benefits are declining relevance to shift towards more sustainable alternatives including controlled and sanitary landfills (Malinauskaite et al., 2017).

Conversely, MSW generated in developed countries is increasingly sent for recycling and/or WTE processing, decreasing their waste-diversion rates to landfills (Eurostat, 2016). Within European boundaries, most countries account for less than 50% landfilling, while some of them (including Austria, Belgium, Germany, Denmark, Finland, The Netherlands, Norway and Sweden) have already reached lower levels of around 5% or less (Eurostat, 2016); in this case, incineration and recycling are the prevailing technologies today, although biological treatment (composting and anaerobic digestion) plays also an important role (Figure 1.6). This is consistent with the proposed updating of European legislation and prioritisation of waste management (EU Directive, 2008).

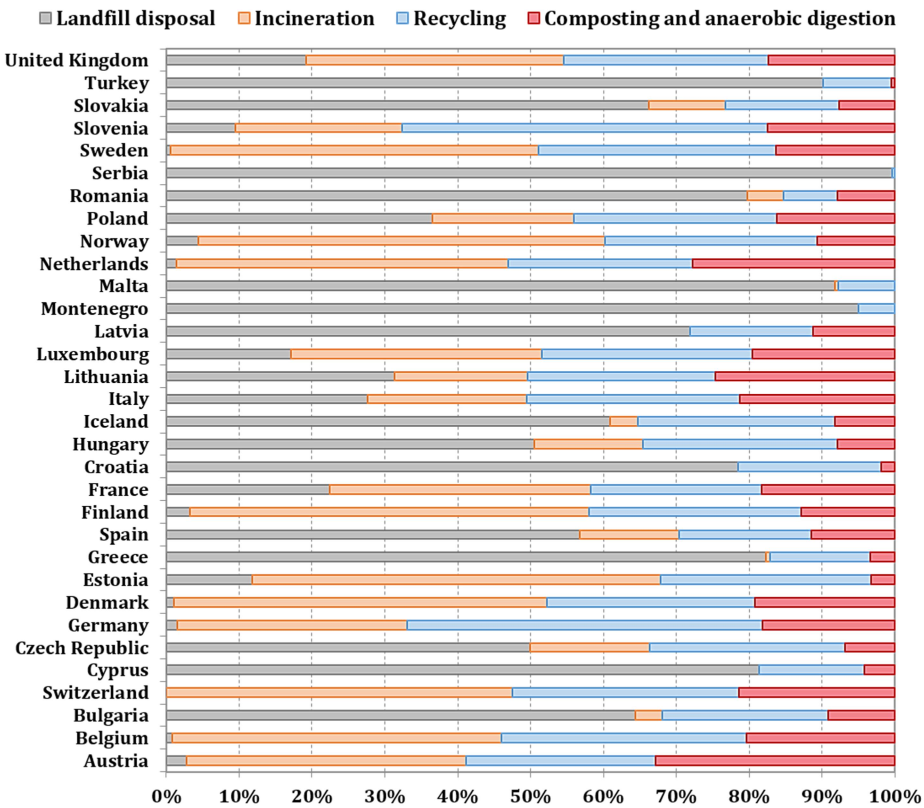

Figure 1.6. Relative distribution of the main alternatives for MSW management in Europe (Eurostat, 2016).

1.3 WASTE MANAGEMENT: TOWARDS MORE SUSTAINABLE PRACTICES

On the basis of the above facts, it could be concluded that many impacts on the environment have already been avoided, reduced or delayed due to advances in waste treatment and recovery. However, in the 21st century, more needs to be done in the area of waste management, as it remains a critical issue worldwide. Accordingly, the range of existing technologies should continue to expand, as well as mitigation strategies, with the aim of preserving environmental quality in the not-too-distant future. This evolution towards a more sustainable framework will allow public society and competent authorities to meet common waste management needs with the maximum ecological potential.

In this context, current environmental policy within European boundaries (Directive 2008/98/EC, recently updated) advocates the creation of sustainable circular economies involving waste management (Gregson and Crang, 2015). Indeed, it seeks for understanding wastes as secondary resources in European manufacturing, according to their possibilities (Gregson and Crang, 2015; Witjes and Lozano, 2016). Parallel to the growing consciousness of the potential value of waste, there is a greater awareness of the environmental consequences associated with inadequate waste management (Castaldi et al., 2014). Indeed, the evaluation of the potential related impacts has become mandatory in Europe; special attention is paid to GHG emissions and energy recovery because of their close relationship to long-term environmental concerns related to energy use and climate change (Pires et al., 2011).

Social participation and economic feasibility are also promoted in European legislation (EU Directive, 2008). The integration of a socio-economic perspective with environmentally sustainable practices is expected to help reduce waste generation and decouple its impact from social evolution (Pires et al., 2011).

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CHAPTER 2. ENVIRONMENTAL AND SUSTAINABILITY ASSESSMENT TOOLS

Summary

Sustainable development has recently become a priority issue for society and its institutions. In a context of continuous technological and socio-economic development, together with population growth and the intensification of anthropogenic activities, new patterns of production and consumption need to be defined to make responsible behaviour towards the environment and future generations a reality. As a result, several methodologies have been developed in recent decades to bring together environmental protection, economic development and societal equity.

In line with the above, the main goal of this chapter was to provide an overview of the different methodological tools available for environmental analysis and sustainability assessment of agri-food systems and waste management, paying special attention to the Life Cycle Assessment (LCA) perspective and the principles of the Analytical Hierarchy Process (AHP). LCA is a globally accepted and standardised methodology for environmental analysis of the entire life cycle of a product or process, whereas the AHP method can be defined as a multi-criteria tool that allows complex decision-making problem to be addressed by integrating environmental outcomes with additional complementary criteria (such as economic and social indicators). In addition, the Carbon Footprint (CF) and Water Footprint (WF) guidelines were also addressed as more precise tools to tackle climate change and water use assessment, respectively, in line with the growing interest in their potential negative contributions to desired sustainable development. Finally, at the end of this chapter, the key objectives of the thesis and its structuring into sections and chapters are presented.

CHAPTER 2 - INDEX

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2.1 SUSTAINABLE DEVELOPMENT: ROOTS AND CONCEPT

Literature shows that the roots of sustainability can be traced back to ancient times (Du Pisani, 2006). However, the steady increase in population growth after the Industrial Revolution, linked to more intensive consumption patterns, strengthened social awareness of the potential dangers of primary resources depletion, compromising living standards of both present and future generations (Du Pisani, 2006).

Sustainable development thinking had its origins at the United Nations Conference on the Human Environment, held in Stockholm in 1972, as the first in a series of international conferences focusing specifically on the threat of environmental concerns (Paul, 2008; Du Pisani, 2006; Whitfield, 2015). At this conference, the experts exposed the need for future development to be sustainable, so that it focuses not only on economic and social issues, but also on its linkages with the use of natural resources and its effect on the environment (Paul, 2008; Du Pisani, 2006). In addition, this conference also played a key role in the creation of the United Nations Environment Program (UNEP), responsible for leading and encouraging environmental stewardship under the premises of sustainability (Paul, 2008).

However, it was not until 1987 that sustainable development as a concept began to gain momentum through the Brundtland Report, entitled "Our Common Future" (WCED, 1987). This report was developed in the frame of the World Commission on Environment and Development (WCED) – also known as the Brundtland Commission due to its leader Gro Harlem Brundtland – founded in 1983 by the United Nations (Paul, 2008; Du Pisani, 2006; Whitfield, 2015). It provided the original and comprehensive standard definition of sustainability development (WCED, 1987): "sustainable development is the development that meets the needs of the present without compromising the ability of future generations to meet their own needs". Moreover, the Brundtland Report first introduced the need for the global integration of economic development, environmental protection and social participation (WCED, 1987; Whitfield, 2015). Indeed, this report expressed the conviction of the simultaneous co-existence of these three components, universally accepted as the three fundamental pillars of sustainability (Du

Pisani, 2006; Sikdar, 2003; WCED, 1987): environmental assessment, economic development and societal equity (Figure 2.1).

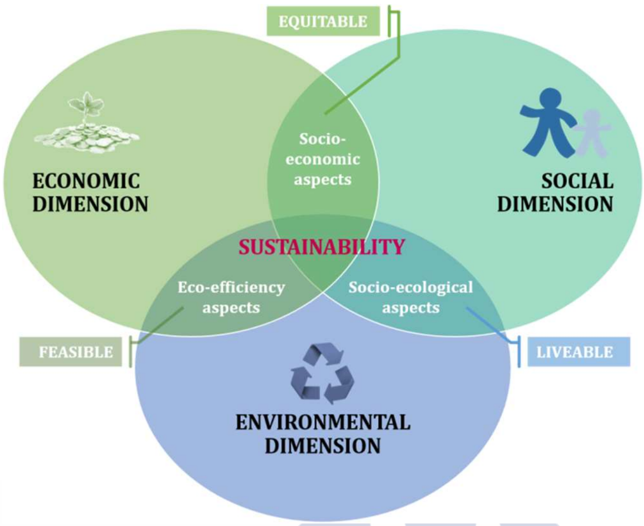

Figure 2.1. The three pillars of sustainability (adapted from Lozano (2008) and Sikdar (2003)).

In the decades since then, new international conferences and strategies have been developed, most of which have been based on the principles defined in the Brundtland Report (Paul, 2008; WCED, 1987; Whitfield, 2015): the concept of “needs” (with priority given to poor areas) and the concept of “limits” imposed by technological and social perspectives on the capacity of the environment to meet present and future needs of world population.

2.2 Life Cycle Thinking (LCT)

Sustainable development is now on the political and business agenda (European Commission, 2010a). In this context, the Life Cycle Thinking (LCT) has become as a key complementary approach to modern environmental policies and decision-making, involving both government and business support (European Commission, 2010a).

Many people around the world are already aware that products result in environmental impacts and resource consumption; however, their quantification over the entire life cycle of such products is a relatively new concept (SMM Coalition, 2014). In this sense, LCT focuses on going beyond traditional manufacturing processes to recognise the relevance of potential impacts at each stage of a product's life, from resources extraction to end-of-life management (Figure 2.2) (SMM Coalition, 2014; UNEP/SETAC Life Cycle Thinking, 2012).

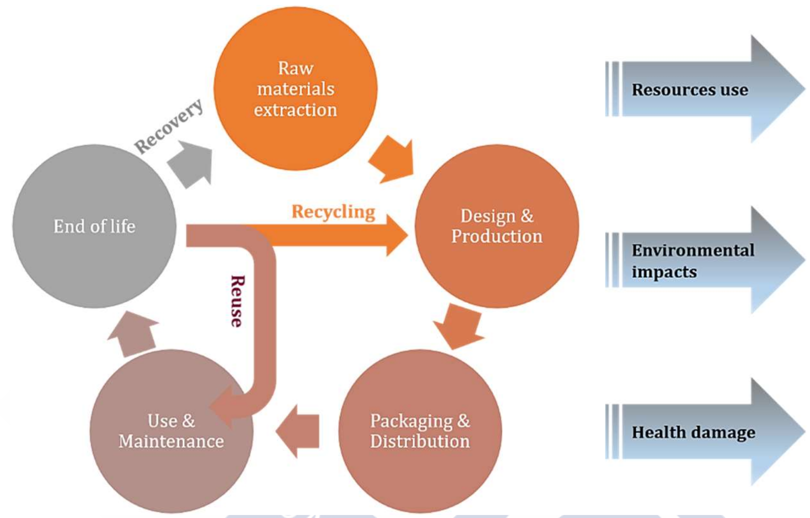

Figure 2.2. Typical life-cycle flowchart of a product (adapted from the UNEP/SETAC Life Cycle Initiative website: www.lifecycleinitiative.org)

LCT can be particularly useful in making sustainable decisions when considering the entire life cycle of a product, since focusing on one stage or another can lead to misleading outcomes (SMM Coalition, 2014). Moreover, LCT aims to reduce resource use and emissions to the environment associated with a product as well as to improve its socio-economic performance throughout its life cycle (UNEP/SETAC Life Cycle Thinking, 2012); accordingly, it may facilitate relationships between the economic, social and environmental dimensions within an organisation and across its entire value chain (European Commission, 2010b; UNEP/SETAC Life Cycle Thinking, 2012; Wolf et al., 2012).

However, while LCT is a philosophy, several life-cycle approaches and tools have been developed over the last decades that enable this way of thinking to be applied (UNEP/SETAC Life Cycle Thinking, 2012). While they may differ in methodological principles (such as data collection requirements and impact indicators to be assessed) and intended users, they all support the life-cycle thinking towards a more sustainable framework. The main methodological tools used in the present thesis are explained in detail in the following sections.

2.3 LIFE CYCLE ASSESSMENT (LCA)

The first studies that delve into the life-cycle aspects of products and materials date back to the 1950s, when several US companies began considering the full life cycle of their products as part of their cost-accounting exercises (SMM Coalition, 2014). However, it was not until the late 1960s when The Coca-Cola Company conducted what is often considered the first life-cycle based analysis to assess the consumption of resources and the environmental burdens associated with the packaging of its beverages (EEA, 1997; SMM Coalition, 2014). In the 1970s, other companies and countries followed its example, also in Europe, by initially giving greater priority to energy balances (which coincided with the oil crisis) and attributing importance to waste management issues (EEA, 1997). Finally, in the mid-1980s and early 1990s, life-cycle assessment gained real relevance, involving a wide variety of industries and sectors (EEA, 1997). In parallel, several organisations began to concentrate their efforts on developing a consistent framework in the field, which led to several guidelines and draft standards on life-cycle assessment (EEA, 1997). In this sense, the method standardisation came through the Code of Practice developed by the Society of Environmental Toxicology and Chemistry (SETAC) in 1993 (European Commission, 2010b).

Life-cycle practices have advanced significantly since them, moving towards a formalised and well-recognised approach in the scientific community today, named Life Cycle Assessment (LCA) methodology (SMM Coalition, 2014). It can be defined as a tool for analysing the different environmental impacts associated with each stage of the life cycle of a product or process, from the extraction and acquisition of raw materials up to the disposal and/or management of the different waste streams (ISO 14040, 2006). Therefore, LCA seeks to quantify all physical exchanges with the environment along all the entire productive chain, including both inputs (natural resources, land and energy) and outputs (such as emissions to air, water and soil) (European Commission, 2010b; ISO 14040, 2006; Wolf et al., 2012).

The International Organization for Standardization (ISO) has developed different standards aimed at the Environmental Administration or Management, including the ISO 14040 series for LCA (ISO 14040, 2006; ISO 14044, 2006). These standards describe the four main phases of a LCA, as well as the elements to consider for each of them (Figure 2.3): goal and scope definition, inventory analysis, impact assessment and interpretation. However, LCA is an iterative technique, so individual phases can use each other´s results to contribute to the completeness and consistency of the study and the reported results (ISO 14040, 2006; Wolf et al., 2012).

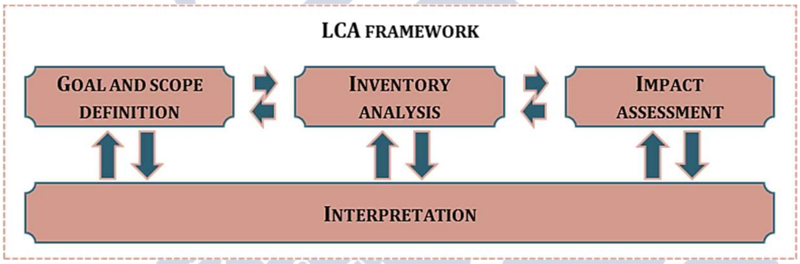

Figure 2.3. Phases of an LCA study (adapted from ISO 14040, 2006).

It is important to note that although ISO standards provide a general framework to conduct an LCA, they do not go into technical details (European Commission, 2010b; Wolf et al., 2012). As a result, other documents and handbooks have been published to provide additional guidance to ensure quality assurance and consistency, such as The International Reference Life Cycle Data System (ILCD) Handbook and Data Network[1] as well as the European Reference Life Cycle Database (ELCD)[2]. The former were developed to provide an authoritative basis to support availability of quality-assured data and methods (in line with ISO standards), while the latter provides

[1] http://eplca.jrc.ec.europa.eu/LCDN/

[2] http://eplca.jrc.ec.europa.eu/ELCD3/

reliable datasets that can also be used as an input to the ILCD Data Network (European Commission, 2010a,b; Wolf et al., 2012).

2.3.1 Goal and scope definition

In this first phase of the LCA study, the product to be analysed should be defined together with the objectives to be achieved,, as well as the decision-context (modelling approach) and intended audience and applications of the study (ISO 14040, 2006; Wolf et al., 2012). The scope of the system is also established at this stage. In defining the scope of a system, the following elements should be included in accordance with the target goal (ISO 14040, 2006; Wolf et al., 2012): the system to be studied, its functions and functional unit (FU), the system boundaries and life cycle stages to be covered, the environmental impacts to be investigated, the assessment methods to be applied, the allocation procedures, data quality requirements, assumptions and limitations.

- **Function and functional unit**

The system under study may have several possible functions, and the one selected as the basis for the analysis will depend on the goal and scope considered. The FU can be defined as a measure of the performance of the functional outputs of the product system (ISO 14040, 2006). The primary purpose of an FU is to provide a reference to which the inputs and outputs are related. This reference is necessary to ensure comparability of LCA results; indeed, the feasible comparison of LCA results is particularly critical when evaluating different systems to ensure that such comparisons are made on a common basis (ISO 14040, 2006).

- **System boundaries**

The system boundaries determine the scope of the system studied. In general, all life cycle stages, unit processes and flows should be considered when establishing the system boundaries, including the raw materials acquisition, inputs and outputs in the main sequence processing (including primary and secondary products), distribution and transportation, fuel and energy requirements, recovery of used products, waste disposal and other additional operations (ISO 14040, 2006). However, sometimes the stages that are expected not to be significant can be cut-off to focus effort on obtaining more reliable data for the relevant processes and elementary flows (European Commission, 2010a). Cut-off criteria must be carefully defined to achieve the right balance between not having incomplete data (which limits the appropriateness of the results) and not increasing overall uncertainty due to the use of inaccurate information (European Commission, 2010a). In any case, the system boundaries initially defined may be reconsidered at a later stage in an iterative analysis (ISO 14040, 2006).

- **Modelling approach**

The modelling principles are closely linked to the decision-context of the study, and two main alternatives are used in LCA practice (European Commission, 2010a): attributional (the most widely used) and consequential modelling. The attributional model refers to the actual supply chain of the product (foreground system), along with its use and end-of-life stages. In this way, it is assumed that the system is embedded into a static technosphere, which makes it possible to estimate the potential environmental impacts of the system throughout its life cycle (also upstream and downstream processes). Instead, the consequential approach aims to evaluate the implications of the interaction of the foreground system with other systems in the market (including the background processes of the system). Therefore, the consequential model analyses a hypothetic supply chain in a dynamic technosfere that is reacting to its consequences.

Although the attributional modelling has so far been the most widely used by the research community, the consequential approach is sometimes preferred to better reflect market constraints and its potential over the system (European Commission, 2010a).

- **Allocation rules**

Few systems lead to a single output or are based on the linearity of raw material inputs and outputs; in fact, most of them supply more than one product or even recycle intermediate or discarded products in the process (European Commission, 2010b; ISO 14040, 2006). They are defined as "multifunctional" systems. However, in most LCA studies, the interest only makes sense in one of the products obtained. For this reason, particular attention should be paid to the allocation of potential burdens and credits involved in multiple products and/or recycling processes of the entire system (ISO 14040, 2006). In this sense, different approaches can be used to solve multifunctionality.

In accordance with ISO standards, allocation should be avoided, whenever possible (ISO 14044, 2006). To this aim, two main alternatives are proposed (European Commission, 2010b): (i) to divide the system into different subsystems to collect the inventory data related to each of them or (ii) to expand the overall system to include additional functions related to the co-products ("system expansion approach"). The latter involves either adding another function that is not intended to make the system comparable ("system expansion itself") or subtracting non-required functions by substituting them with the ones that are replaced ("substitution by system expansion").

Conversely, when allocation is not avoidable, ISO standards propose splitting up inputs and outputs between the co-functions/co-products of the system according to some allocation criteria reflecting the link between them (e.g. mass, energy content, market price), while giving priority to the underlying physical relationships (European Commission, 2010; ISO 14044, 2006).

2.3.2 Life Cycle Inventory (LCI) analysis

The Life Cycle Inventory (LCI) analysis mainly includes data collection and calculation procedures for quantifying inputs and outputs relevant to the life cycle of the system to be assessed together with data on background processes (Wolf et al., 2012). It is, therefore, the phase that requires the greatest efforts and resources in a LCA study, involving the collection and modelling of several flows (European Commission, 2010b):

- Elementary flows, including the use of resources as well as releases to air and discharges to water and/or soil.
- Products flows, including both goods and services obtained as a product of the system.

- Waste flows, such as wastewater and solid/liquid wastes, which are subject to management processes to ensure environmental-friendly performance.

The LCI should be conducted in accordance with the previous goal and scope definition, although it may be revisited after preliminary analysis (ISO 14040, 2006). In other words, as more and more knowledge and data are acquired, new data requirements or limitations may be identified, which implies an update in data collection. However, a first validation of data is already carried out at this phase (Wolf et al., 2012).

2.3.3 Life Cycle Impact Assessment (LCIA)

The Life Cycle Impact Assessment (LCIA) aims at evaluating the significance of potential environmental impacts using the results of the LCI analysis (ISO 14040, 2006; ISO 14044, 2006). In general, this process involves associating inventory data with specific impact indicators related to human health, environment and resource depletion (European Commission, 2010b). The level of detail, selection of impacts and methodologies used should be in line with the goal and scope definition.

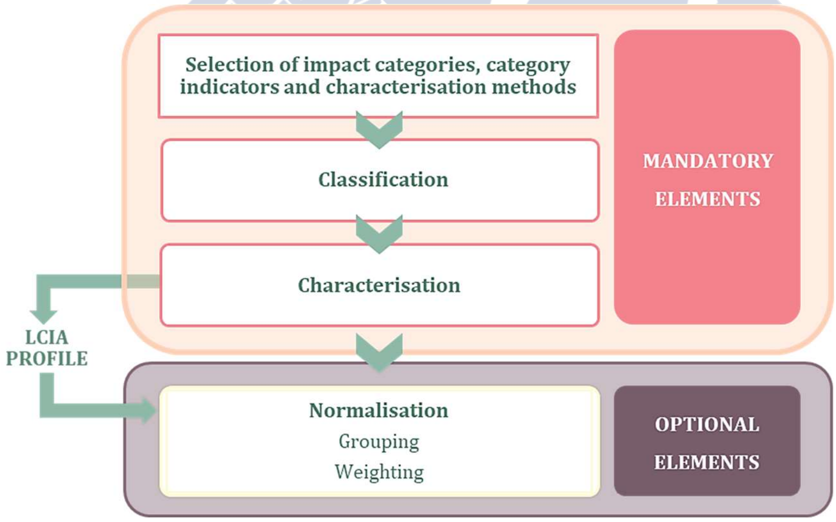

Figure 2.4. Steps of the LCIA phase (adapted from ISO 14040, 2006).

In accordance with the official LCA standards (European Commission, 2010b; ISO 14044, 2006), this evaluation phase should include three main mandatory steps (Figure 2.4): selection of impact categories and characterisation methods, classification and characterisation. Additional steps – normalisation, grouping and weighting – can be developed optionally at a later stage.

- **Selection of impact categories and characterisation methods**

As aforementioned, the environmental impact categories to be considered in the LCIA, as well as the corresponding characterisation methods, will be defined in the earliest phases, according to the goal and scope of the LCA study. To date, several common characterisation factors and methods have been developed. They can be classified into midpoint and endpoint level: while the former includes a greater number of impact categories (around 10) and provides more accurate results, the latter focuses on the three areas of protection – human health, environment, resource depletion – commonly used for endpoint assessment. Moreover, the selection of the impact categories will be in line with the choice of the characterisation method in the study, aiming to fulfil international acceptance (European Commission, 2010b).

According to the standards, several LCA impact categories are recommended, such as climate change, ozone depletion, human toxicity, photochemical ozone formation, acidification, eutrophication, ecotoxicity, land use and resource depletion, as well as areas of protection: human health, natural environment and natural resources. However, each specific study will focus on those categories that are relevant based on the basis of related inputs and outputs (European Commission, 2010b).

- **Classification and characterisation**

The classification step consists of the assignment of LCI results to one or more impact categories and/or indicators, while the estimation of the final environmental results is conducted during the characterisation step in accordance with the model previously selected for the calculations (ISO 14044, 2006). To this aim, inventory data of the different flows must be linearly multiplied with the impact factors provided by the characterisation model. Aggregated results are associated with each impact category and a global numerical indicator is obtained. The methodological mechanism used in LCIA for the "climate change" impact category is presented in Figure 2.5 as an example.

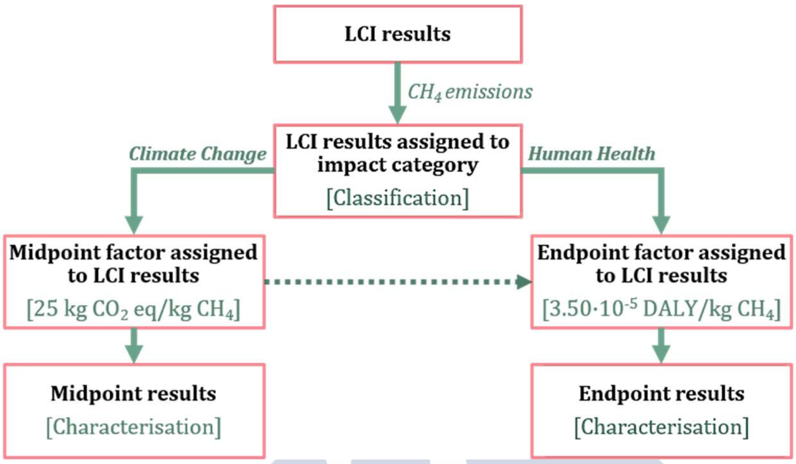

Figure 2.5. Concept and methodological mechanism applied in LCIA (adapted from European Commission (2010b) and ISO 14044 (2006)).

- **Normalisation**

Normalisation is based on the calculation of the relative magnitude of the results attributed to each category, divided by a selected reference value (ISO 14044, 2006). This optional phase seeks to help to better understand the relative relevance of each impact category in the product system under study (ISO 14044, 2006).

- **Grouping**

Grouping is based on the aggregation of different impact categories into one or more sets, including also the definition of priority rankings according to subjective value-choices.

- **Weighting**

Weighting is defined as the process of converting the results of the different impact categories using numerical factors in line with value-choices criteria (ISO 14044, 2006). Therefore, the weighting phase is based on subjective knowledge rather than available scientific knowledge, so that different results can be obtained for the same indicator. Sensitivity analysis may therefore be appropriate to assess the impact on the results of the different value-choices and weighting criteria.

- **LCIA mechanisms: ReCiPe method**

Among the different methodologies developed to date, the ReCiPe method can be defined as the most updated alternative that allows the LCA-approach to be flexible and uniform (Goedkoop et al., 2013). According to LCA experts, this method provides a common framework in which midpoint and endpoint levels can be evaluated, as opposed to previous alternatives, such as CML and Eco-indicator, which are based on either midpoint or endpoint indicators, respectively (Goedkoop et al., 2013).

The ReCiPe method was first developed in 2008, although an updated 2016 version is already available in the literature (Huijbregts et al., 2016). It provides characterisation factors representative of the global scale, not only of the European framework, but also maintains the possibility of implementing characterisation factors at a country and/or continental scale in some specific categories (Huijbregts et al., 2016). Figure 2.6 shows an overview of the methodological mechanisms of the ReCiPe method, comprising eighteen midpoint indicators linked to three additional endpoint indicators. Converting midpoints to endpoints can simplify the interpretation of the environmental results, although each aggregation step may also be responsible for greater uncertainties in a LCA study (Huijbregts et al., 2016). In this sense, the ReCiPe method addresses the inherent uncertainties through three different perspectives based mainly on time horizon criteria (Goedkoop et al., 2013; Huijbregts et al., 2016): individualist (I), hierarchist (H) and egalitarian (E). Perspective I is based on the short-term interest (20 year-time horizon), indisputable impacts and technological optimism regarding human adaptation. Perspective H is based on the most common time frame of the policy principles (100 year-time horizon). Finally, perspective E refers to the most precautionary situation, taking into account the longest time horizon (100,000 years).

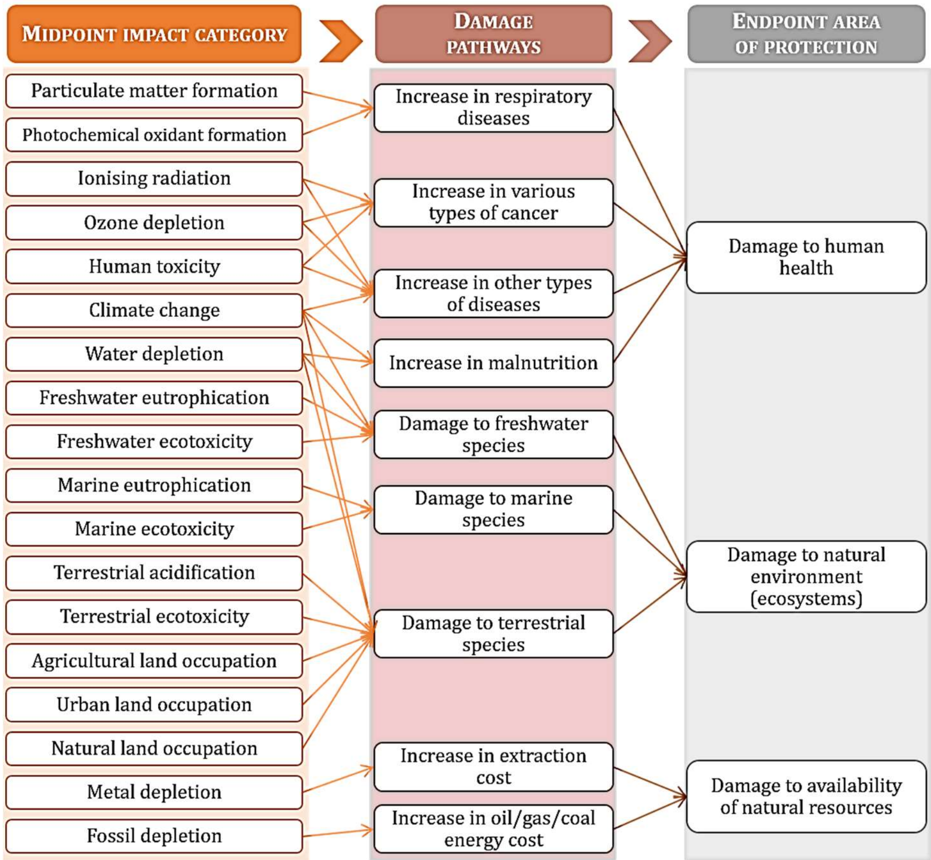

Figure 2.6. Environmental mechanisms in the ReCiPe method (adapted from Huijbregts et al., 2016).

A detailed overview of midpoint impact categories included in the ReCiPe method is reported in Table 2.1:

Table 2.1. Overview of midpoint categories in ReCiPe 2008 (Huijbregts et al., 2016).

Impact category	**Acronym**	**Unit**	**Indicator**
Climate change	CC	kg CO_2 eq	Infrared radiative forcing increase due to GHGs emission
Ozone depletion	OD	kg CFC-11 eq	Decrease in stratospheric ozone concentration
Terrestrial acidification	TA	kg SO_2 eq	Proton increase in natural soils due to changes in acid deposition
Freshwater eutrophication	FE	kg P eq	Phosphorous increase in freshwater
Marine eutrophication	ME	kg N eq	Nitrogen increase in marine water
Human toxicity	HT	kg 1,4-DB eq	Risk increase of both cancer and non-cancer diseases incidence
Photochemical oxidant formation	POF	kg NMVOC	Increase in tropospheric ozone concentration
Particulate matter formation	PMF	kg PM_{10} eq	Increase in the human population intake of fine particulate matter
Terrestrial ecotoxicity	TET	kg 1,4-DB eq	Hazard-weighted increase in natural soils
Freshwater ecotoxicity	FET	kg 1,4-DB eq	Hazard-weighted increase in freshwater
Marine ecotoxicity	MET	kg 1,4-DB eq	Hazard-weighted increase in marine water
Ionising radiation	IR	kg Bq eq	Impact of radioactive substances and other radiation sources
Agricultural land occupation	ALO	$m^2 \cdot yr$	Use and conversion of agricultural land transformation
Urban land occupation	ULO	$m^2 \cdot yr$	Use and conversion of urban land transformation
Natural land transformation	NLT	m^2	Use and conversion of natural land transformation
Water depletion	WD	m^3	Increase in water consumption by population
Metal depletion	MD	kg Fe eq	Increase in the consumption of non-renewable resources limiting their future availability
Fossil depletion	FD	kg oil eq	

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2.3.4 Interpretation of the results

Finally, the interpretation phase is carried out based on the combination of the major findings from the LCI and the LCIA stages (ISO 14040, 2006). Moreover, this phase may include sensitivity, consistency and uncertainty analyses in order to ensure the reliability of the results (Wolf et al., 2012). In this way, they are expected to serve as a basis for conclusions and recommendations to decision-makers, in line with the goal and scope of the study.

2.4 CARBON FOOTPRINT (CF)

Climate change resulting from anthropogenic activity has been identified as one of the greatest challenges facing by society in recent times, with major implications for both human and natural ecosystems (GHG Protocol, 2011). In this context, the Carbon Footprint (CF) has gained recognition as an indicator specifically focused on measuring the contribution of goods and services to climate change (European Commission, 2010b). Thus, this indicator is based on the estimation of the amount of direct and indirect GHGs emitted into the environment in terms of kg of carbon dioxide equivalent (kg CO_2 eq) (ISO 14067, 2013).

Over the last few decades, several methodologies have been developed that include GHG measurement and CF calculation, with the aim of addressing climate change mitigation within the frameworks of organisations (i.e. ISO 14064 and ISO 14069) and products (i.e. PAS 2050 and ISO 14067). Among them, the GHG Protocol[3] was the early leader to provide companies and organisations with an internationally accepted methodology to help quantify GHG emissions associated with their operations, as well as potential reduction opportunities (GHG Protocol, 2004).

It was developed in the late 1990s through a partnership between the World Resources Institute (WRI) and the World Business Council for Sustainable Development (WBCSD), as an extensive and inclusive but effective methodology to account for not only direct but also indirect GHG

[3] http://www.ghgprotocol.org/

emissions from external sources such as raw materials extraction and related transport (GHG Protocol, 2011). Since then, the GHG Protocol initiative has continued to work on the development of new and updated guidelines, based on the same multi-stakeholder approach followed in existing standards, with the participation from businesses, policymakers and academia, among other experts (GHG Protocol, 2011). In this regard, this international standard has served as the basis for the design of other tools, such as ISO 14064-1 and ISO 14069, to better incorporate GHG impacts into business decision-making (ISO 14064-1, 2012; ISO 14069, 2015); similarly, GHG Protocol principles have also been taken into consideration in the development of the latest ISO standards covering the CF of product life cycle chains (ISO 14067, 2013).

In this context, the ISO 14067 has recently been recognised as the main standard for calculating product-based CFs, with the aim of unifying the different impact assessment tools proposed by the scientific community to date (ISO 14067, 2013). It is based on the ISO 14020 and 14040 series, paying attention to specific principles, requirements and guidelines for the quantification and reporting of the CF of products (ISO 14067, 2013). In this way, the CF method can be seen as a simplified LCA study where priority is given to GHG emissions with an effect on one single impact category: climate change (Table 2.1). Accordingly, the four phases of LCA – goal and scope definition, LCI, LCIA and interpretation – must also be conducted to determine the CF of a product throughout its entire life cycle (see Section 2.3 LCA).

2.5 WATER FOOTPRINT (WF)

Human activities are responsible for consumption and pollution of large volumes of water (WWAP, 2009). However, until recently, little attention had been paid to sustainable water management in accordance with water consumption patterns and pollution charges across the life cycle productive chains (Hoekstra et al., 2011). It has been in recent years that some research studies have shown that a better understanding of the relationship between products and their water requirements can provide the basis for better management of global trade on water resources use worldwide (Hoekstra and Champagain, 2008; Jeswani and Azapagic, 2011).

In this context, several methodologies and indicators began to be developed to evaluate water usage and potential damage on freshwater resources, ecosystems and human health (Jeswani and Azapagic, 2011). Among them, the concept of Water Footprint (WF) has gained growing interest, as a comprehensive indicator of freshwater resources appropriation, next to the traditional and restricted measure of water depletion (Hoekstra et al., 2011). It can be defined as the overall indicator of the volume of water consumed for a product, both directly and indirectly, due to its production process (Figure 2.7). The direct water consumption of a product includes the amount of water used and/or contaminated during the manufacturing process, as well as the water present in the product itself as an ingredient; the indirect consumption corresponds to the water required for the production of the different raw materials.

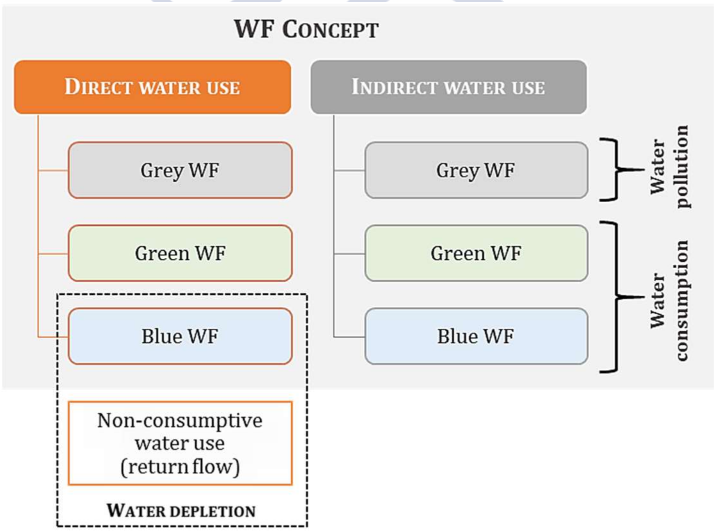

Figure 2.7. Components of the WF concept in comparison with the traditional water depletion measure (adapted from Hoekstra et al., 2011).

The WF is expressed in units of water consumed (m^3 of water, for example) and can be divided into three different variables (colours): green, blue and grey. Green WF is related to rainwater evaporated or incorporated in the products and is particularly important for agricultural products. Blue WF includes the water that has been sourced from surface or groundwater resources and is either evaporated, incorporated into the product or extracted from one water body and returned to another, or at a different time. Finally, grey WF is related to water quality as well as its possible contamination and includes the amount of fresh water required to dilute the content of pollutants in the streams to meet specific water quality standards (Franke et al., 2013).

Different standards based on the concept of WF have been developed under different approaches. In this regard, the Water Footprint Network (WFN)[4] provided a global standard for "water footprint assessment" covering a comprehensive set of definitions and methods for water footprint accounting (Hoekstra et al., 2011). According to this method, a full WF assessment consists of a four-phase process, according to the LCA perspective: (i) setting goals and scope, (ii) accounting, (iii) sustainability assessment and (iv) response formulation. The aim is to clarify the main objective of the WF study, while the scope defines its spatial and temporal scale. The accounting phase focuses on data collection to estimate the WF of the relevant processes involved in the study. After accounting, sustainability assessment is conducted to evaluate whether water use is environmentally sustainable and resource efficient; the social and economic perspective must also be included in this phase. Finally, response strategies and/or policies should be formulated to reduce the WF and improve the sustainability of the system under assessment. However, it is important to highlight that the method of four phases proposed by the WFN is more of a guideline than a mandatory standard, so that some phases may not be necessary in all WF studies (Hoekstra et al., 2011). In this sense, both system boundaries and cut-off criteria should be defined in the first phase of goal and scope.

2.6 Analytical Hierarchy Process (AHP) method

As aforementioned, sustainability has become a key issue in any decision-making situation, involving the co-existence of environmental, economic and social perspectives as the three main pillars of sustainable development (WCED, 1987; Zamagni et al., 2015). In this sense, the LCA

[4] http://waterfootprint.org/

principles are considered to provide valuable support for integrating the environmental approach into the sustainable design of products and services, in accordance with international environmental policies, using life cycle thinking as reference framework (Zamagni et al., 2015). However, other tools have been found that can be combined with LCA studies for more extensive assessment, involving both economic and social perspectives, among others aspects (UNEP/SETAC Life Cycle Initiative, 2011; Zamagni et al., 2015).

In this context, the Multi-Criteria Decision Analysis (MCDA) has emerged as an important support decision tool that allows the comparison and assessment of quantitative and qualitative elements in a systematic and consistent way, through the use of real data and subjective expert decisions (Achillas et al., 2013; Soltani et al., 2015). Thus, several MCDA methods have been developed in recent years as a means to help decision-makers select the optimal decision among different alternatives (Achillas et al., 2013). Among them, the Analytical Hierarchy Process (AHP) methodology has been accepted by the international scientific community as a robust and flexible multi-criteria decision-making tool for dealing with complex decision problems, so it has been applied for a wide range of applications (Vaidya and Kumar, 2006; Soltani et al., 2015).

The AHP method was developed by Thomas Saaty in 1980, based on three fundamental principles (Saaty, 1980): (i) breaking down the problem, (ii) pair-wise comparison of the various alternatives and (iii) synthesis of preferences. The first step of the AHP analysis consists in subdividing the multi-level decision-making problem into a hierarchy with unidirectional hierarchical relationships between levels (Figure 2.8). At the first level, the goal of the analysis is defined; the second level aggregates and combines the selected criteria and sub-criteria, which contribute to the main goal of the study; finally, the third level suggests the alternatives available for application.

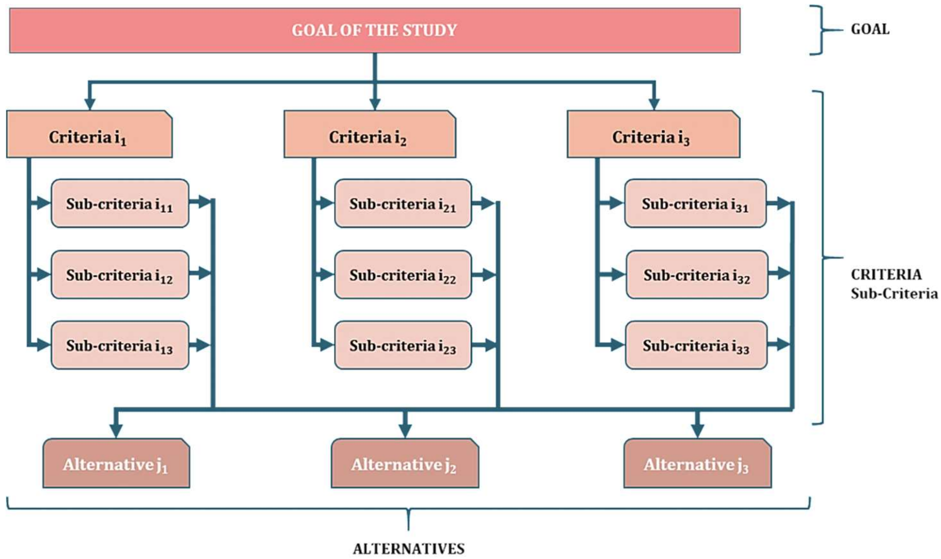

Figure 2.8. Example of hierarchical structure in the AHP methodology (Saaty, 1980).

Once the hierarchy is constructed, the evaluation phase is conducted based on pair-wise comparison principles (Saaty, 1980): all alternatives should be compared pair-wise in terms of their contribution to their control criteria. The relative importance values are determined on a 9-point scale, the so-called "Saaty's Fundamental Scale" (Table 2.2):

Table 2.2. Saaty's Fundamental Scale for AHP preference (Saaty, 1980).

Preference number	Definition	Explanation
1	Equal importance	Two activities contribute equally to the objective
3	Moderate importance	Slight favour of one activity over another
5	Strong importance	Strong favour of one activity over another
7	Very strong importance	Dominance of one activity over another is demonstrated
9	Absolute importance	Extreme favour of one activity over another
2,4,6,8	Intermediate values	

Note: if activity i has one of the above non-zero numbers assigned to it when compared with activity *j*, then *j* has the reciprocal value when compared with *i*.

The numerical judgements established at each level of the hierarchy make up the pair-wise matrices (Table 2.3). The same procedure is followed for criteria and sub-criteria in terms of their relevance to their upper level criterion; analogous pair-wise matrices are obtained (Table 2.4).

Table 2.3. Example of a pair-wise comparison matrix for hypothetical alternatives in relation to each specific criteria and/or sub-criteria.

	Alternative j_1	**Alternative j_2**	**Alternative j_3**
Alternative j_1	1	1/5	1/4
Alternative j_2	5	1	1
Alternative j_3	4	1	1
	CRITERIA i = 1, 2, ... n		

Table 2.4. Example of a pair-wise comparison matrix for hypothetical criteria.

	Criteria i_1	**Criteria i_2**	**Criteria i_3**
Criteria i_1	1	3	1
Criteria i_2	1/3	1	1/3
Criteria i_3	4	1	1

Once comparison matrices have been created, the relative weights of the elements of each level with respect to an element in the adjacent upper level are computed as the components of the normalised eigenvector (also called priority vector) associated with the largest eigenvalue of the comparison matrix. It should be noted that the eigenvector method provides a natural measure of consistency to ensure the reliability of the comparisons considered.

Finally, composite weights are determined by aggregating the weights throughout the hierarchy. This is done by following a path from the top of the hierarchy down to each alternative at the lowest level, and multiplying the weights along each segment of the path. The outcome of this aggregation is an overall priority vector that allows a relative priority ranking among the different competing alternatives to be established (Table 2.5). Consequently, the alternative with higher eigenvector will prevail over the others.

Table 2.5. Example of a hypothetical overall priority vector.

	Overall Priority Vector
Alternative j_1	0.344
Alternative j_2	0.197
Alternative j_3	0.459

According to the example shown in Table 2.5, the alternative j3 would be the best option. However, it is important to note that the final priorities of the alternatives depend to a large extent on the weights associated with the general criteria and sub-criteria (Achillas et al., 2013); thus, minor changes in the relative weights can therefore cause major changes in the final ranking. For this reason, a sensitivity analysis can be helpful to be performed on the final outcome with the aim of providing information on the stability of the ranking initially defined (Achillas et al., 2013; Soltani et al., 2015).

2.7 ENVIRONMENTAL ASSESSMENT TOOLS APPLIED TO AGRICULTURAL SYSTEMS: LIVESTOCK HUSBANDRY AND WASTE MANAGEMENT

The intensification of food production regimes has been promoted over the past decades, while the demand for food products has continued to grow, leading to countless increasing problems of food safety and environmental damage (Gerber et al., 2013; Roy et al., 2006; Steinfeld et al., 2006). Consequently, consumers have recently demanded a shift towards more sustainable patterns within the food sector (Notarnicola et al., 2012).

In this context, special attention has been given to the livestock sector because of its dominant relevance in the food industry (de Vries and de Boer, 2010; Roy et al., 2006; Steinfeld et al., 2006). Indeed, it is responsible for about 18% of anthropogenic GHG emissions, as well as one of the most relevant drivers of water consumption and pollution, among other environmental problems (FAO, 2009; Steinfeld et al., 2006; Hoekstra, 2014).

Dairy and meat (especially pork) products account for the greatest socio-economic relevance (Alexandratos and Bruinsma, 2012; FAO, 2011; FAOSTAT, 2016), but at the expense of major environmental impacts (Weidema et al., 2008; Gerber et al., 2013). It is for this reason that the environmental performance of several pork and dairy production systems has been extensively assessed in literature to date (Nguyen et al., 2010; Philippe and Nicks, 2014; Yan et al., 2011). The principles of the LCA methodology have been followed predominantly as a basis for the environmental evaluation, with special attention to some common indicators involving eutrophication, acidification and resources depletion concerns (Baldini et al., 2017; McAuliffe, 2016). Similarly, most of these studies aimed to focus only on those burdens related to GHG emissions (carbon footprint approach), demonstrating the relevance of climate change impacts to sustainable production (de Léis et al., 2015; Gerber et al., 2011; Groen et al., 2016; Nguyen et al., 2010; Philippe and Nicks, 2014). In contrast, most authors omitted other interesting environmental indicators, such as water use (water footprint approach), arguing several uncertainties and limitations in data collection (McAuliffe, 2016; Ran et al., 2016).

The review of the available works in the field proves that there is still no consensus on methodological choices and assumptions, making it difficult to perform comparisons between similar studies (de Vries and de Boer, 2010; McAuliffe et al., 2016; Yan et al., 2011; Zehetmeier et al., 2014). In this sense, while some authors focus their attention on some specific stages of the production chain (especially feed production and animal husbandry at farm), other studies extend their boundaries to include life cycle processes of the entire system. This variability also leads to different products and/or sub-products, as well as potential environmental credits due to avoided processes (when applicable), which affected the choice of the functional unit and allocation approach (Reckmann et al., 2012; Zehetmeier et al., 2014).

However, regardless of methodological disparities, available studies reported common conclusions in terms of *hotspots* activities, noting the production of feed sources as the main contributor to most impact categories assessed in livestock systems followed by the animal husbandry stage, mainly due to related manure management practices. In this sense, specific studies can also be found in the literature to assess the environmental impacts of the production of different animal feed sources (Colombini et al., 2015; García-Lunay et al., 2014; Williams et al., 2014); many of them focus on the cultivation of some cereals crops as elementary ingredients in the livestock diets (Bacenetti et al., 2014, 2015; Gallego et al., 2011; González-García et al., 2016; Wang et al., 2014). According to these studies, the environmental consequences from these systems are mainly related to diesel consumption in agricultural activities, production and application of fertilisers and related emissions.

Greater attention has also been paid to the management of the waste generated at farm facilities (mainly livestock manure), changing the waste disposal model to its valorisation as a source of nutrients (Bayo et al., 2012; Lopez-Ridaura et al., 2009). In this way, manure has come to be seen as an organic fertiliser increasing the chemical and physical properties of agricultural soils; however, manure application can also be responsible of adverse environmental impacts (Groen et al., 2016; Nguyen et al., 2010; Oenema et al., 2007). For that reason, more recent studies have been carried out in search of treatment strategies that make it possible to exploit the nutritional contribution of animal waste without compromising long-term stability in the environment.

In this regard, several LCA studies involving alternative manure post-treatment technologies can be found. However, most of them focus on the AD process and the subsequent use of digestate (Bacenetti et al., 2016; Lijó et al., 2014), while only a few go beyond these conventional strategies (Brockmann et al., 2014; Karakashev et al., 2008). AD has the advantage of removing organic matter from manure, as well as producing renewable energy from biogas generation; however, it does not remove nutrients, so that additional digestate post-treatment is also required (Nasir et al., 2012). The technical feasibility of these innovative strategies has already been evaluated, but there is limited evidence on the potential environmental advantages of nutrient recovery in digestate management (Fernandez-Lopez et al., 2016; Pardo et al., 2017; Rehl and Müller, 2011; Rodríguez-Verde et al., 2014).

2.8 ENVIRONMENTAL AND SUSTAINABILITY ASSESSMENT TOOLS APPLIED TO URBAN SYSTEMS: MSW GENERATION AND MANAGEMENT

Industrial and socio-economic development, together with population growth and more intensive anthropogenic activities – including higher rates of food consumption – has led to the generation of steadily increasing volumes of MSW (Laurent et al., 2014a; Vergara and Tchobanoglous, 2012). This is why the proper management of this waste has become a key issue worldwide, with the objective of adequately mitigating its adverse effects on human health and the quality of ecosystems (Laurent et al., 2014b; Soltani et al., 2015).

This problem is of particular concern in less developed regions, where MSW treatment is still in its infancy (Bezama et al., 2007; Wilson et al., 2015). Few environmental studies can be found in literature focused on the state-of-the-art and/or potential improvements involving waste management in these areas. According to them, inappropriate practices remain responsible for major environmental problems (Orazbayev et al., 2013; Othman et al., 2013). This highlights the need to delve into this issue, moving towards a more responsible and environmental-friendly framework in terms of MSW management in developing regions.

Fortunately, MSW management has already been investigated in detail in recent times by developed areas. Indeed, numerous authors have applied the LCA method to assess the environmental profile of alternative management strategies in different developed regions worldwide (Achillas et al., 2013; Castaldi et al., 2014; Vergara and Tchobanoglous, 2012). According to these studies, conventional management practices have been gradually improved, seeking more efficient treatment of biodegradable waste and increasing recycling rates. In this way, the traditional consideration of waste has progressively shifted from a waste to a valuable resource for society (Laurent et al., 2014b). However, the most recent outcomes on waste management continue to encourage the economically developed world to continue working, in line with the current policy initiatives proposed by the authorities for environmental protection (Laurent et al., 2014a,b).

Moreover, the sustainable management of MSW goes beyond quantifying and addressing environmental impacts, since the economic and social perspectives also provide a valuable input for identifying appropriate solutions for designing transition strategies (Achillas et al., 2013; Soltani et al., 2015). In this context, several authors have analysed the efforts made by developed and developing regions to move towards more sustainable alternatives based on a balanced three-pillars approach (Soltani et al., 2015). In this sense, MCDA has been commonly used as a support tool to integrate economic and social criteria in decision-making related to environmental analysis (Achillas et al., 2013). Indeed, numerous research works have addressed the AHP method linked to the LCA approach to identify the most sustainable alternative among alternative MSW management schemes (Contreras et al., 2008; Dong et al., 2014; Stypka et al., 2016; Tarmundi et al., 2011; Yan and Nixon, 2015). According to these studies, developed countries preferred anaerobic digestion and new gasification technologies to more conventional practices, while incineration remains the most desirable option in developing economies.

However, it was found that the prioritisation of alternative technologies depended to a large extent on the distribution of weight between criteria and stakeholder opinion, so that each case should be analysed individually (Contreras et al., 2008; Soltani et al., 2015). This demonstrates the need to continue working to implement the potential added value of this combined approach, especially in those similar areas that are still at an early stage of MSW management.

2.9 THESIS OUTLINE: OBJECTIVES AND STRUCTURE

The main goal of this doctoral thesis was to evaluate the environmental sustainability of conventional and innovative technologies for the management of different types of waste, both in agriculture and urban frameworks. Additionally, the entire life cycle chain was evaluated in agricultural systems, linking the upstream stages responsible for livestock waste generation to downstream strategies, while socio-economic drivers were integrated with environmental issues in the urban sector.

With this in mind, the present document has been structured into four main sections, with their respective chapters, as shown in Figure 2.9.

Section I: Contextualisation. This section aims to provide potential readers with an overall vision of the state-of-the art of the problem to be addressed in this thesis, as well as the methodological support used. In this way, Chapter 1 focuses on the state-of-the art on the field, including sources of waste generation, current management technologies and potential valorisation strategies. Chapter 2 delves into the fundamental principles of the different methodological tools applied throughout the subsequent chapters, especially in terms of environmental (LCA) and sustainability (AHP) assessment.

Section II: Agricultural framework. This section seeks to focus attention on current and advanced alternatives for the management of waste from livestock systems in both the pig and dairy sectors. To this aim, not only were the downstream processes that involved waste treatment environmentally evaluated, but also the entire productive chain up to farm facilities where waste is generated during livestock husbandry. As a result, Chapter 3 analysed the environmental profile of several winter and summer cereals in different cultivation regimes as priority ingredients in most feed mixtures and diets. Moreover, Chapter 4 and Chapter 5 were developed to identify the environmental burdens associated with milk and pork production, respectively, with special attention to the impacts of climate change (CF perspective) and water use concerns (WF approach). Finally, Chapter 6 analysed alternative integrated configurations based on the eco-innovative principles of sustainability, aimed at translating animal manure into an important opportunity for reuse as nutrient source for soils. Two case studies were evaluated in leading European locations (Spain and The Netherlands) with intensive cattle and pig production.

Section III: Urban framework. As in the previous Section II, waste management practices and possible improvement alternatives were evaluated in Section III, but drawing attention to the urban framework. In this sense, the main objective of Chapter 7 was to evaluate the environmental profile of MSW management in developed and developing regions, following the LCA approach. To this aim, two case studies were conducted in Spain and Kazakhstan as representatives of conductive and non-conductive conditions. Additionally, to complement this work, the AHP principles were combined with LCA results to compare the sustainability of different MSW management models in a developed region. In this way, economic, social and environmental indicators were integrated together to establish a ranking of sustainable priorities including the options available for waste management.

Section IV: Conclusions. Finally, Chapter 9 summarised the main findings, results and conclusions of the thesis.

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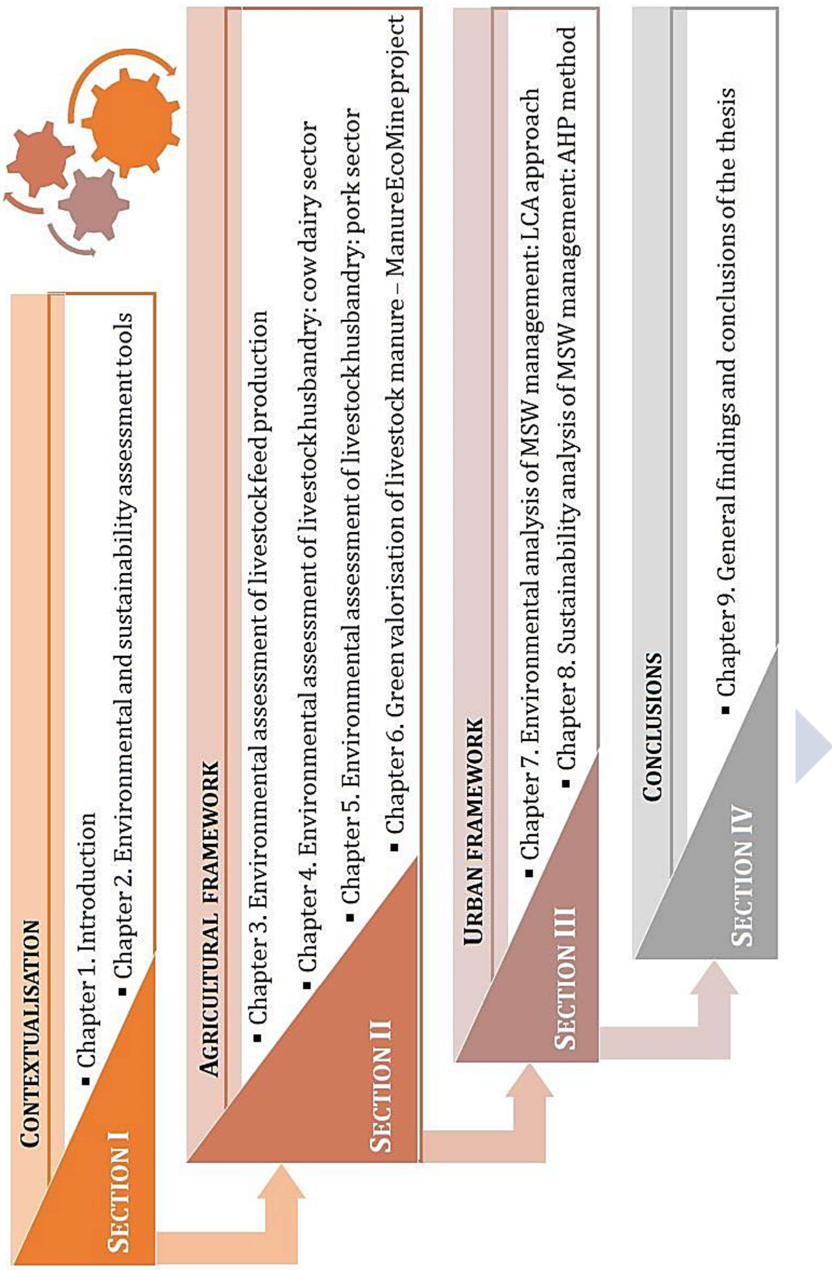

Figure 2.9. Outline of the present doctoral thesis.

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SECTION II

AGRICULTURAL FRAMEWORK

CHAPTER 3. ENVIRONMENTAL ASSESSMENT OF LIVESTOCK FEED PRODUCTION

Summary

In recent decades, current agricultural production practices have been shown to be responsible for increasing environmental damage caused by the use of limiting resources and the emission of pollutants into air, water and soil. In this context, animal feed production has been identified as the major contributor within the agricultural sector, making it crucial to determine its environmental impacts and mitigation alternatives.

In this chapter, the LCA methodology was used to estimate the environmental impacts and to identify the most critical stages (hotspots) of cultivation of the most widespread cereals crops in the Lombardy region (Northern Italy) for animal feed. Different varieties and cultivation regimes that included two summer crops (maize and sorghum) and four winter crops (wheat, triticale, barley and rye) were compared to determine the alternatives with the highest and lowest impacts per kg of crude protein (mass-based FU).

According to the results, the hotspots include field emissions, agricultural activities and production of agrochemicals (fertilisers and herbicides) regardless of the cultivation system considered. Moreover, the comparative assessment shows that rye and maize classes 600-700 are the most favourable environmental options among winter and summer cereals, respectively. They present less intensive agricultural activities as well as higher biomass yields compared to the other alternatives. However, a sensitivity analysis shows that the classification of cropping systems according to their environmental impacts can change significantly when selecting a land-based FU. Therefore, it should be noted that the choice of the best alternative could depend to a large extent on the FU considered as a basis for calculation. The main findings of the present study are expected to be useful in promoting more sustainable practices in the target area and related locations.

CHAPTER 3 - INDEX

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3.1 INTRODUCTION TO LIVESTOCK FEED PRODUCTION

In recent times, the world has faced growing demand for feed and food, as well as increasing concern for environmental sustainability (Nikkhah et al., 2015; Notarnicola et al., 2012; Roy et al., 2009). Indeed, current agricultural production systems have been identified as a major contributor to environmental damage: they are responsible for significant emissions of pollutants into air, water and soil, as does growing competition for scarce resources such as land, water, minerals and fossil fuels (de Vries and de Boer, 2010, Nikkhah et al., 2015). In this context, livestock feed production can be considered one of the leading contributors to the environmental impact of the agricultural sector (de Vries and de Boer, 2010; Mogensen et al., 2014). In fact, current feed production systems contribute up to 64% of the total GHG released into the environment from agricultural production (Mogensen et al., 2014), together with the emission of other polluting substances such as nitrates, phosphates, sulphur oxides or ammonia from the application of agrochemicals and use of machinery (Bellarby et al., 2008; Notarnicola et al., 2012; Reay et al., 2012).

More specifically, the cultivation of cereal crops and the energy requirements for their transformation into animal feed are directly related to the major environmental consequences (Mogensen et al., 2014). Cereals have been traditionally cultivated under extensive agriculture, following farming directives that aim at optimising the use of internal inputs while reducing external ones such as fertilisers and pesticides (Nemecek et al., 2011). Conversely, the lower yields per unit of land associated with these systems imply larger requirements for arable land (Mözner et al., 2012). Considering that the global demand for agricultural products is expected to double in the coming decades (Baudron and Giller, 2014), intensive production is becoming particularly important (Mözner et al., 2012). Intensive systems achieve higher yields than conventional or extensive ones, but require a higher degree of mechanisation and a wide range of chemicals (Mözner et al., 2012). Consequently, guaranteeing sufficient growth in primary production while limiting the environmental impacts of cereal production systems is one of the priorities of international policy agendas for the agricultural sector (Alexandratos and Bruinsma, 2012; Bell et al., 2014).

In this sense, the LCA methodology has been widely used to assess the environmental impacts of several farming activities, although greater attention has been paid to the cultivation of some cereal crops (such as wheat and maize, among others) as basic ingredients in livestock diets (Fallahpour et al., 2012; Gallego et al., 2011; Niero et al., 2015; Roer et al., 2012; Wang et al., 2007, 2014). However, there is scarcely any information so far involving similar studies in the Lombardy region (Northern Italy), though it has been recognised as a leading area for the cultivation of cereals and forages with high potential yields, due to its favourable soil, climatic conditions and water availability (Carrosio, 2013; MATTM & MIPAAF, 2010). Among them, the development of cereals such as maize (*Zea mays* L.) and wheat (*Triticum* spp. L.) has been dominating other crop varieties (Bacenetti et al., 2014; Negri et al., 2014a,b). Nevertheless, due to recent revision of the European Union's Common Agricultural Policy and the "greening" issue (aim of protecting and improving biodiversity, as well as making food production more sustainable from an environmental perspective; Singh et al., 2014), triticale (*Triticosecale* Wittmack), barley (*Hordeum vulgare* L.), rye (*Secale cereale* L.) and sorghum (*Sorghum* spp.) play, now and in the future, an important role in animal feeding (MATTM & MIPAAF, 2010). These cereals can be grown in two types of cropping systems: single and double cropping (Bacenetti et al., 2014). In single cropping, only one summer crop (usually maize or sorghum hybrids with a long growing cycle of more than 125 days) is cultivated, while double cropping adds a winter cereal (barley, rye, wheat or triticale) before the summer crop (such as maize hybrids with a short crop cycle or sorghum). Therefore, double cropping systems arise in response to calls for intensified production while avoiding the potential consequences of expanding arable land (Borchers et al., 2014). However, despite an increased production capacity, field operations and inputs requirements are usually higher for double cropping systems, leading to higher economic costs and environmental impacts (Bacenetti et al., 2014, 2015; Borchers et al., 2014).

With this in mind, the main purpose of this Chapter 3 was to evaluate the environmental effects caused by the cultivation of several widely consumed cereal crops in Europe for feed production from the LCA perspective. The plantations under study were located in the Po Valley (Lombardy region) in response to the limited evidence of related environmental studies in this region and other similar climatic areas to date. Indeed, although analogous cropping systems for both feed and energy purposes have already been evaluated (Bacenetti et al., 2014, 2015; González-García et al., 2013, 2016), a comprehensive analysis focusing on livestock feeding in this region has not yet been addressed. The assessment included the identification of the most critical stages (commonly referred to as hotspots) throughout the life cycle of each cereal crop; in addition, since they can be complementary to each other, a comparative assessment was also carried out to determine which cereal is responsible for the highest and lowest environmental impacts.

3.2 LIVESTOCK FEED PRODUCTION: SUMMER CEREALS

3.2.1 Goal and scope definition

This section focuses on the environmental analysis and comparison of the profile of two main summer crops: maize and sorghum. In total, seven cropping systems were considered, taking into account five different varieties for maize (300, 400, 500, 600, 700) as well as one hybrid for the production of sorghum silage under both single and double cropping.

As aforementioned, all these cultivations are located in the Po Valley area (Figure 3.1), in the Lombardy region (Northern Italy), recognised as one of the most important agricultural areas in Europe, with a large number of livestock farms and agro-industries (Carrosio, 2013).

Figure 3.1. Location of the Po Valley in the Lombardy region (Northern Italy).

The study was performed from a cradle-to-farm gate perspective, including all agricultural processes, from field preparation to biomass harvesting; further stages such as biomass conversion, consumption and final disposal of waste were excluded from the assessment. Moreover, background processes that include inputs production (e.g. seeds, agrochemicals and fuels) and their transport to cropping systems, as well as the production, use and final disposal of agricultural machinery were also considered. Finally, emissions to air, water and soil due to the production and use of agrochemicals (fertilisers and herbicides) and fuels were also estimated.

- **Functional unit**

A FU must be defined as a reference for comparison in this study. In this sense, alternative FUs can be found in published works involving agricultural systems, although most of them are based on mass production (Fallahpour et al., 2012; Niero et al., 2015; Wang et al., 2014) and land occupation (Goglio et al., 2012; Roer et al., 2012). However, for several years now, Europe has needed high-quality sources of vegetable protein to replace animal-derived ones in livestock feed (Baumgartner et al., 2008). According to the literature, both legumes and cereal crops are important sources of vegetable proteins (Baumgartner et al., 2008; Jensen et al., 2010; Voisin et al., 2014); however, given that relatively little land is devoted to legume production in Europe, cereal crops can be considered the most suitable alternative to meet this need and foster the increased use of vegetable proteins (Voisin et al., 2014).

Table 3.1. Main properties of the summer crops under evaluation: variety, biomass yield, moisture and crude protein content.

Cereal crop	**Variety/ Hybrid**	**Biomass yield (t_{wb}[a]/ha)**	**Moisture (%)**	**Crude protein (% dry basis)**
Maize	Class 300	6.71[b]	15.0	8.00
	Class 400	9.27[b]	15.0	8.30
	Class 500	12.7[b]	15.0	8.10
	Class 600	14.8[b]	15.0	8.70
	Class 700	14.0[b]	15.0	7.90
Sorghum	Sweet caroline	83.8[c]	68.7	9.00
		51.4[c]	69.2	9.00

[a] wb = wet basis; [b] $t_{grain}\cdot ha^{-1}$; [c] $t_{biomass}\cdot ha^{-1}$

Consequently, the crude protein content (1 kg of crude protein in biomass) was considered as the basis (FU) for estimating the environmental impacts of the different cropping systems assessed. The average protein proportions and moisture contents of maize and sorghum are reported in Table 3.1.

- **Systems description**

The different maize varieties (classes) are cultivated in single cropping, whereas sorghum is cultivated in both single and double cropping systems. However, all of them can be structured into three main stages: (S1) field preparation and sowing, (S2) crop growth and (S3) biomass harvesting and storage. The cultivation of maize requires six months, while sorghum only requires four and five months for single and double cropping, respectively. A detailed description of each cropping system is reported below.

Maize

The different maize classes are defined by their vegetative cycle (FAO cycle) that determines the time from the birth of the plant to its physiological maturity (Jugenheimer, 1958); however, physiological maturity does not coincide with agronomic maturity (14 – 18% moisture content of the grain), i.e., when the grain is ready to be harvested (Llanos Company, 1984). The FAO cycle is expressed in a number from 100 to 900, depending on the earliness of maize (Llanos Company, 1984): (i) 100 – 300 very early cycle, (ii) 400 – 500 early cycle, (iii) 600 – 700 middle cycle and (iv) 800 – 900 late cycle. The choice of the maize variety will depend on climatic conditions and the time available for plant growth (Llanos Company, 1984). In this study, five different classes of maize were considered: 300, 400, 500, 600 and 700; other maize hybrids with a longer cropping cycle (i.e. maize classes 800 or 900) cannot be grown in Italy, since they would not have enough time to mature under Italian climatic conditions.

However, regardless of the maize variety, all the cropping systems share the same main stages (Figure 3.2). The first stage consists of the preparation of the land and subsequent sowing of the selected variety (S1). For this purpose, the soils are fertilised in May with digestate at a rate of 40 tonnes wet basis (t_{wb})/ha for classes 300, 400 and 500, and at a rate of 85 t_{wb}/ha for classes 600 and 700 (Table 3.2). After organic fertilisation, also in May, the soil is ploughed, harrowed and sown with a seeding range from 16 to 18 kg seeds/ha, depending on the maize class. In addition, for classes 300, 400 and 500, one single mineral fertilisation with potassium and phosphorous-based fertilisers is required between ploughing and harrowing. During crop growth (S2), herbicide control is carried out by applying 4 kg/ha of Lumax; additional chemical weed control is also required in June, using 1 kg/ha of Dual (once for maize classes 300, 400 and 500 and twice for maize classes 600 and 700). During the same period, hoeing and mineral fertilisation with urea is performed. In months of July and August following fertilisation, irrigation is conducted four to six times, depending on the maize class (Table 3.2), with a water volume of 800 m^3/ha for each irrigation. In September, biomass is harvested (S3) and grain is separated for feed purposes, while straw is bailed before its distribution. There were large differences in the biomass yield among the five different classes of maize (Table 3.1), while a moisture content of 15% and 30% was considered for grain and straw, respectively.

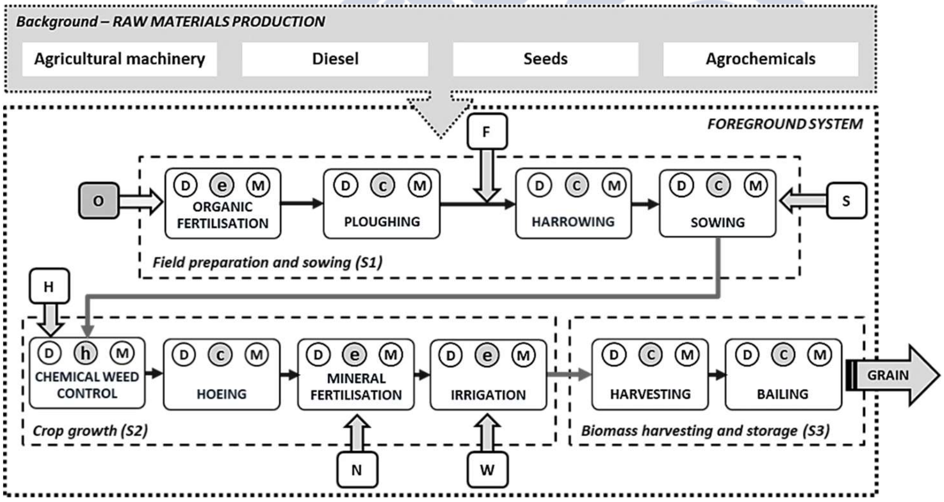

Figure 3.2. Scheme of the system boundaries for the cultivation of maize varieties. Key: D = diesel fuel production; M = agricultural machinery production and maintenance; O = organic fertiliser (digestate) production – excluded; S = seed production; H = herbicide production; N = nitrogen fertiliser production; e = field emissions and discharges from fertilising; c = combustion emissions; h = herbicide emissions.

Table 3.2. Field operations and inventory data (per ha) for the cultivation of maize varieties (300, 400, 500, 600, 700).

Stage	Activity	Time (month)	Tractor (A)	Operative machine			A + B	Input rates
			Weight (kg)	Type	Weight (kg)	Effective work (ha/h)	Diesel use (kg/ha)	
Field preparation & Sowing	Organic fertilisation	May	6500	Manure spreader	2500	2.50[a] 2.20[b]	4.71[a] 5.76[b]	45 t/ha Digestate[a,b] 85 t/ha Digestate[a,c]
	Ploughing	May	8500	Ploughshare	2000	0.80	22.6	-
	Mineral fertilisation[b]	May	5050	Fertiliser spreader	350	3.00	3.17	100 kg/ha P 100 kg/ha K
	Harrowing	May	5050	Rotary harrow	1000	0.50	24.2	-
	Sowing	May	6500	Precision seed drill	900	1.50	10.3	16-18 kg seeds/ha
Crop growth	Chemcial weed control	May	4580	Spaying machine	500	3.00	3.32	4 kg/ha Lumax
	Chemcial weed control	June	4580	Spaying machine	500	3.00	3.32	1 kg/ha Dual (S-metolachlor)
	Chemcial weed control[c]	June	4580	Spaying machine	500	3.00	3.32	1 kg/ha Dual (S-metolachlor)
	Hoeing	June	4850	Hoeing machine	500	3.00	2.75	-
	Mineral fertilisation	June	5050	Fertiliser spreader	350	3.00	3.17	200 kg/ha Urea 60 kg/ha Urea
	Irrigation	July-August	4450	Pump	400	0.83	10.1	8 m³/ha
Biomass harvesting & Storage	Harvesting	Sept	14,000	Combine harvester	-	2.00[c] 2.50[f]	15.8; 17.0; 19.7; 20.1; 19.8	-
	Bailing	Sept	0.004[e]	Bailing machine	0.0071[e]	N.D.	0.89[e]	-

[a]Digestate composition: 5.36% dry matter; 0.4% total N; 0.2% N-NH_4; [b]Maize classes 300/400/500; [c]Maize clases 600/700; [d]Maize classes 300/400 (x4), Maize class 500 (x5), Maize classes 600/700 (x6).; [e]Per bale: N.D.: no available data.

Sorghum

Under single cropping, field preparation and sowing (S1) of sorghum begin in April with the application of pig slurry at 100 t_{wb}/ha followed by ploughing, harrowing and sowing at a rate of 20 kg seeds/ha. During crop growth (S2), weeds are chemically controlled in April by spraying 3 kg aclonifen/ha. Mineral fertilisation is also required in June; so 100 kg of urea/ha is applied. Sorghum is irrigated in June and July in two stages (800 m^3/ha each). Finally, biomass is harvested and ensiled (S3) in August, with a biomass yield of 83.8 t_{wb}/ha and a moisture content of 68.7% (see Table 3.1). Double cropping has the same operations as single cropping, but as sorghum is preceded by a winter cereal, the timing of agricultural activities differs from that of single cropping (Table 3.3). Thus, field preparation, sowing and herbicide application occur in June, while mineral fertilisation and irrigation take place in July and July-August, respectively. Biomass is harvested and ensiled in September, with biomass yield of 51.4 t_{wb}/ha and a moisture content of 69.2% (see Table 3.1).

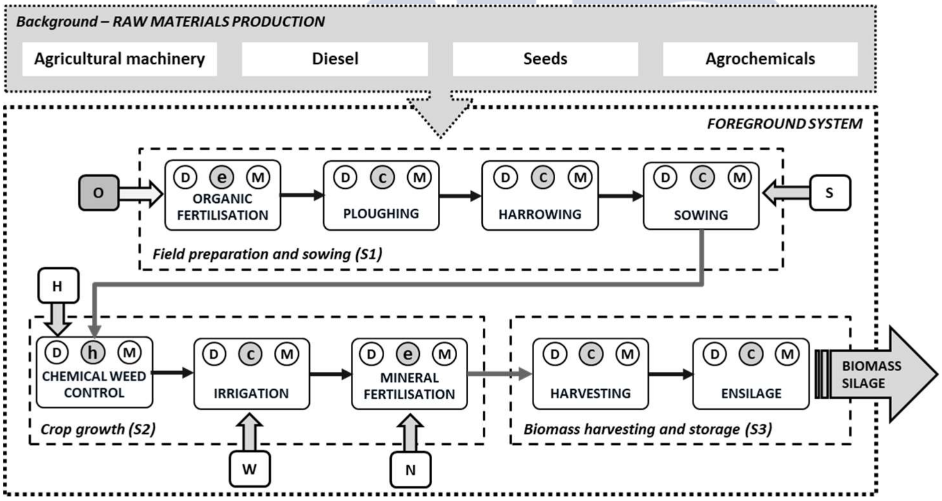

Figure 3.3. Scheme of the system boundaries for the cultivation of sorghum (identical for single and double cropping). Key: D = diesel fuel production; M = agricultural machinery production and maintenance; O = organic fertiliser (pig slurry) production – excluded; S = seed production; H = herbicide production; N = nitrogen fertiliser production; W = water; e = field emissions and discharges from fertilising; c = combustion emissions; h = herbicide emissions.

Table 3.3. Field operations and inventory data (per ha) for the cultivation of sorghum.

Stage Activity		Time (month)	Tractor (A)	Operative machine			A + B	Input rates
			Weight (kg)	Type	Weight (kg)	Effective work (ha/h)	Diesel use (kg/ha)	
Field preparation & Sowing	Organic fertilisation	April[a] June[b]	6500	Slurry spreader	2500	0.26	43.7	100 t_{wb}/ha Pig slurry
	Ploughing	April[a] June[b]	6500	Ripper	1150	0.77	22.0	-
	Harrowing	April[a] June[b]	6500	Rotary harrow	550	0.56	19.0	-
	Sowing	April[a] June[b]	4000	Pneumatic seed drill	1200	2.94	6.30	20 kg seeds/ha
Crop growth	Chemcial weed control	April[a] June[b]	4000	Spaying machine	500	3.03	6.20	3 kg/ha Aclonifen
	Irrigation	June[a]; July[a,b] August[b]	6500	Pump	250	0.83	20.2	800 m^3/ha for each intervention
	Mineral fertilisation	June[a] July[b]	4000	Fertiliser spreader	450	2.50	3.80	100 kg/ha Urea
Biomass harvesting & Storage	Harvesting	August[a] September[b]	-	Forage harvester	16500	0.91	39.5	-
	Ensilage	August[a] September[b]	5050	Font loader	450	0.50	0.44[c]	-

[a] Single crop regime; [b] Double crop regime; c kg/ton (Bacenetti and Fusi, 2015).

- **Allocation rules**

Different products and co-products can be distinguished in the different systems under evaluation. Thus, ensiled biomass is the only product obtained from sorghum cultivation, so it was not necessary to apply allocation rules and all the environmental impacts were assigned to biomass production. On the contrary, although grain is the main product in the different maize systems, the straw obtained also has an economic value (for the generation of energy in boilers and animal bedding) and is therefore not a waste. Accordingly, the allocation of environmental burdens was required, so that an allocation based on economic criteria was selected as base case. The resulting allocation percentages for grain and straw for each maize variety are displayed in Table 3.4.

Table 3.4. Economic allocation factors for grain and straw for each cropping systems involving maize varieties.

Cropping system		**Biomass yield (t_{wb}/ha)**	**Price (€/t_{wb})**	**Allocation factors (%)**
Maize Class 300	Grain	6.71	247	72.7
	Straw	8.59	72.5	27.3
Maize Class 400	Grain	9.27	247	72.1
	Straw	12.2	72.5	27.9
Maize Class 500	Grain	12.7	247	74.1
	Straw	15.1	72.5	25.9
Maize Class 600	Grain	14.8	247	74.0
	Straw	17.7	72.5	26.0
Maize Class 700	Grain	14.0	247	74.9
	Straw	16.0	72.5	25.1

Moreover, pig slurry and digestate (from anaerobic digestion of livestock manure) were assumed to be used as organic fertilisers in the different cropping systems. However, since they were considered as co-products of the pork production chain and agro-industrial sector, respectively, the environmental impacts of their production were not allocated to the current systems. On the contrary, diffuse emissions from their use in fertilisation activities were included within the system boundaries.

3.2.2 LCI analysis

LCI data was collected in this study representing an area of 2.6 ha for sorghum, while two major farms were assessed for maize: one (15 ha of total cultivated area) provided information on maize classes 600 and 700, and the other one (33 ha of total cultivated area) was evaluated for the production of maize classes 300, 400 and 500 (also with wheat and triticale cultivation).

For both maize and sorghum and their varieties, inventory data on agricultural inputs (such as machinery, labour hours, agrochemicals and fuels and water requirements) were obtained directly from personal interviewers and questionnaires filled out by farmers (see Tables 3.2 – 3.3). However, secondary data were used to estimate background information about the production of such agricultural inputs. In this sense, the ecoinvent® database v3.2 was used as a priority basis (Althaus et al., 2007; Nemecek and Käggi, 2007). Similarly, emissions from fuel combustion in agricultural processes (i.e. fertiliser and herbicide application, ploughing, harrowing, sowing, harvesting and ensiling/bailing), as well as transport activities, were also taken from the ecoinvent® database (Dones et al., 2007; Spielmann et al., 2007).

As aforementioned, the environmental burdens related to the production of pig slurry and digestate fall outside the system boundaries, since they were considered as co-products of previous processes. However, their storage, application and diffuse emissions associated were included, along with field emissions from the use of additional mineral fertilisers. Thus, nitrogen emissions in terms of N_2O, NH_3 and leached NO_3^- were estimated according to the emission factors provided by the Intergovernmental Panel on Climate Change (IPCC, 2006). In this regard, Tier 2 method was followed by combination of primary information and defaults emissions factors (IPCC, 2006). Phosphate emissions were also considered on the basis of an applied ratio of 0.01 kg P-PO_4^{3-}/kg of applied P proposed by Rossier (1998). Table 3.5 summarises the emissions resulting from the management of organic and mineral fertilisers in both maize and sorghum cropping systems.

Table 3.5. Estimation of nitrogen (IPCC, 2006) and phosphorous (Rossier, 1998) emissions (per cultivated ha) from the management of organic and mineral fertilisers in maize and sorghum cropping systems.

	Maize Classes 300/400/500		Maize Classes 600/700		Sorghum	
Emissions	**Digestate**	**Urea**	**Digestate**	**Urea**	**Pig slurry**	**Ammonium nitrate**
Storage						
N_2O (kg/ha)	0.03	-	0.06	-	1.58	-
NH_3 (kg/ha)	2.93	-	5.53	-	153	-
NO_3^- (kg/ha)	4.27	-	8.07	-	223	-
Application						
N_2O (kg/ha)	0.11	1.48	0.22	0.44	5.94	0.74
NH_3 (kg/ha)	1.77	11.4	3.32	3.42	91.8	5.71
NO_3^- (kg/ha)	9.61	125	18.1	37.5	502	62.4
PO_4^{3-} (kg/ha)	0.02	-	0.04	-	0.83	-

Similarly, direct emissions to the environment from herbicide application to soils were also estimated using the distribution factors available in the literature; accordingly, it was assumed that about 85% of the herbicide applied enter the soil, while only 10% was assumed to be emitted to air and less than 10% to water sources (Althaus et al., 2007; Audsley et al., 1997; Margni et al., 2002; Wang et al., 2007).

Finally, the carbon content of agricultural soils depends to a large extent on factors such as management practices, climate or previous cropping regimes in such agricultural area. Therefore, in absence of reliable data, no changes in the overall carbon content were considered, also in line with related studies in the literature (González-García et al., 2013).

3.2.3 Impact assessment

Among the stages defined within the LCIA phase of the standardized LCA methodology, the classification and characterisation stages were undertaken in this study (ISO 14044, 2006). The characterisation factors of the ReCiPe Midpoint (H) v1.12 method were applied (Goedkoop et al., 2013a) to the following impact potentials: CC, OD, TA, FE, ME, HT, POF, TET, FET, MET, WD, FD, which were considered to be most affected by agricultural activities. In contrast, land use changes were not considered within the framework of the study, mainly because no significant biodiversity losses or landscape impacts were expected since fields in all the cropping systems were used only for agriculture (Souza et al., 2015). Moreover, the choice of this midpoint methodology and impact categories was consistent with previous studies on agricultural products in the literature (Bacenetti et al., 2014; Fusi et al., 2014; González-García et al., 2013; Mogensen et al., 2014; Niero et al., 2015). SimaPro v8.2 software was used for the computational implementation of the inventories (Goedkoop et al., 2013b).

3.2.4 Results and discussion

The results obtained in relation to the environmental burdens associated with the different cropping systems are discussed below. The agricultural processes or activities considered throughout the life cycle of each system were grouped into five contributing factors: field emissions, agrochemicals production, agricultural activities, seeds production and transport activities, with the aim of facilitating the analysis and the identification of the hotspots.

Field emissions include the effect caused by the direct emissions to air from the application of fertilisers (mineral and organic) and herbicides to the soil during the cultivation of the different cereals. In the **agrochemical production**, all environmental burdens associated with the manufacturing processes of pesticides and inorganic fertilisers used are considered. As previously reported, the production of the organic fertilisers (pig slurry or digestate) was not included within the system boundaries. **Agricultural activities** are directly related to the production and use of diesel in agricultural activities, as well as the production, use and maintenance of agricultural machines. These agricultural operations include organic fertilisation, ploughing, harrowing, sowing, chemical weed control, mineral fertilisation, irrigation, harvesting and bailing/ensilage steps. In case of maize, hoeing should be also considered. **Seeds production** takes into account the environmental burdens derived from the production of seeds used in the sowing process. **Transport activities** factor includes all emissions from transport.

- **Maize cultivation**

Tables 3.6 and 3.7 report the characterisation results (per FU) of the cropping systems of maize classes 300-400-500 and 600-700, respectively, because they share analogous cultivation practices and inputs ratios. Accordingly, the differences among the different systems are substantial. The maize class 300 is the variety with the greatest impacts in all categories, followed by the maize class 400. In contrast, the maize classes 600 and 700 are responsible for the best environmental results in almost all categories. Indeed, they show environmental impacts ranging from 5% to 73% (depending on the category) of those related to class 300. The favourable results for these varieties of maize are related to the higher biomass yields compared to the other maize classes, as well as the lower requirements of fertilisation. Maize production capacity increases with the growth cycle, so that the yield of biomass is higher in the less early maize varieties.

Table 3.6. Characterisation results per FU (1 kg of crude protein in biomass) of the maize clasess 300, 400 and 500.

		Maize varieties			
Impact category	**Units**	**Class 300**	**Class 400**	**Class 500**	**Average values**
CC	kg CO_2 eq	5.13	3.56	2.79	3.83±0.97
OD	mg CFC-11 eq	0.75	0.52	0.41	0.56±0.14
TA	kg SO_2 eq	0.10	0.07	0.06	0.08±0.02
FE	g P eq	2.80	1.93	1.50	2.08±0.54
ME	g N eq	4.92	2.65	2.65	3.41±1.07
HT	kg 1,4-DB eq	2.57	1.77	1.39	1.91±0.49
POF	kg NMVOC	0.02	0.02	0.01	$0.02 \pm 5.77 \cdot 10^{-3}$
TET	kg 1,4-DB eq	0.03	0.02	0.06	0.04±0.02
FET	kg 1,4-DB eq	0.12	0.08	0.11	0.10±0.02
MET	kg 1,4-DB eq	0.09	0.07	0.06	0.07±0.01
WD	m^3	5.23	3.65	3.48	4.12±0.79
FD	kg oil eq	1.56	1.09	0.85	1.17±0.29

Table 3.7. Characterisation results per FU (1 kg of crude protein in biomass) of the maize clasess 600 and 700.

		Maize varieties		
Impact category	**Units**	**Class 600**	**Class 700**	**Average values**
CC	kg CO_2 eq	0.97	1.13	1.05±0.08
OD	mg CFC-11 eq	0.14	0.16	0.15±0.01
TA	kg SO_2 eq	0.03	0.03	0.03±0.00
FE	g P eq	0.16	0.18	0.17±0.01
ME	g N eq	1.55	1.79	1.67±0.12
HT	kg 1,4-DB eq	0.27	0.32	0.30±0.03
POF	g NMVOC	7.44	8.74	8.09±0.65
TET	g 1,4-DB eq	10.4	12.2	11.3±0.90
FET	g 1,4-DB eq	23.9	28.2	26.1±2.15
MET	g 1,4-DB eq	14.5	17.1	15.8±1.30
WD	m^3	3.26	3.84	3.55±0.29
FD	kg oil eq	0.26	0.31	0.29±0.03

Figure 3.4 shows the contributions to the different impact categories from each contributing factor in the cultivation of maize classes 300, 400 and 500. They all share the same agricultural activities and agrochemicals requirements, with only a slight difference in relation to the number of irrigation steps (Table 3.2). Moreover, the biomass yield is also different (Table 3.1), so that the maize class with the lowest yield coincides with that of the worst environmental profile. However, negligible differences can be found in the relative influence of the factors contributing to each impact category. Accordingly, three critical factors for the three maize varieties (300-400-500) can be distinguished: field emissions, agrochemical production and agricultural activities. Field emissions are relevant in terms of CC, TA, ME, TET and FET with higher contributions, up to 97%. The agricultural activities contribute considerably to CC, OD, HT, POF, FET, MET and FD: 15% - 53% depending on the category. It is mainly due to the steps of ploughing, harrowing, irrigation and harvesting, where diesel fuel consumption by agricultural machinery is higher and, therefore, the emissions from

combustion are remarkable. Moreover, the agricultural activities are decisive in terms of WD due to the irrigation stage, in which water consumption is significant (around 3,200 m^3/ha). The production of agrochemicals plays a critical role in CC, OD, TA, FE, POF, FD and all toxicity categories (with contributions exceeding 30% in all these categories), mainly due to the production of nitrogen-based fertilisers, since the manufacturing processes of these agrochemicals imply high energy consumption and, therefore, the emissions derived have a considerable environment impact. Seeds production shows an important effect (over 17%) on ME, mainly due to the emissions from the agrochemicals used during seed production. However, transport activities do not have a remarkable effect on the environmental results.

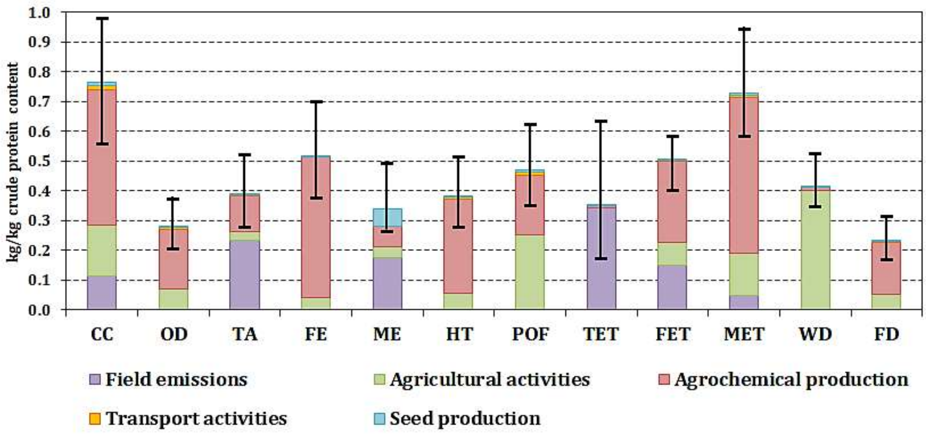

Figure 3.4. Mean contribution (per FU) to each impact category involving the cultivation of maize classes 300, 400 and 500. Errors bars indicate maximum and minimum values. Key: CC (kg CO_2 eq/5); OD (kg CFC-11 eq*5·10^5); TA (kg SO_2 eq*5); FE (kg N eq*250); ME (kg P eq*100); HT (kg 1,4-DB eq/5); POF (kg NMVOC*25); TET (kg 1,4-DB eq*10); FET (kg 1,4-DB eq*5); MET (kg 1,4-DB eq*10); WD (kg H_2O/1·10^4); FD (kg oil eq/5).

Similarly, Figure 3.5 shows the relative contributions involved in the cultivation of maize classes 600 and 700. Both cropping systems share the same agricultural activities and agrochemical requirements, while they only differ in the amount of seeds sown and the biomass yield (Table 3.1). Therefore, the relative influence of the different factors on the categories can be considered as analogous. Again, field emissions, agrochemical production and agricultural activities were identified as the three critical factors.

However, in this case, the effect of the agrochemical production is limited when compared to the environmental profiles of maize classes 300, 400 and 500. This is due to the lower requirements for mineral fertilisers in the maize classes 600 and 700 (60 kg urea/ha as opposed to 200 kg/ha) and the need for an additional fertilisation step with phosphorus and potassium-based fertilisers in the other maize varieties. Moreover, seeds production gains relevance in terms of ME, while transport activities are still not relevant in any of the impact categories.

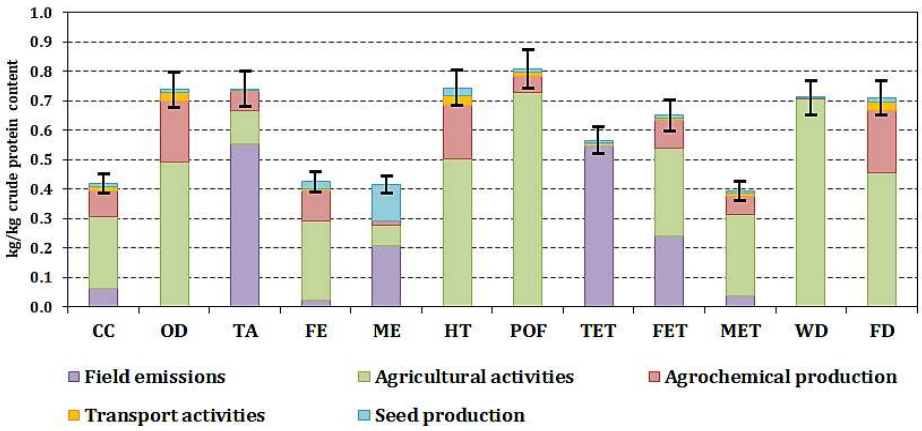

Figure 3.5. Mean contribution (per FU) to each impact category involving the cultivation of maize classes 600 and 700. Errors bars indicate maximum and minimum values. Key: CC (kg CO_2 eq/2.5); OD (kg CFC-11 eq*5·10^6); TA (kg SO_2 eq*25); FE (kg N eq*2500); ME (kg P eq*250); HT (kg 1,4-DB eq/2.5); POF (kg NMVOC*100); TET (kg 1,4-DB eq*50); FET (kg 1,4-DB eq*25); MET (kg 1,4-DB eq*25); WD (kg H_2O/5·10^3); FD (kg oil eq*2.5).

- **Sorghum cultivation**

Table 3.8 shows the environmental results (per FU) for both single and double cropping systems for sorghum. According to the results, single cropping of sorghum has approximately 40% lower impacts than those of double cropping in all impact categories. Since agricultural activities and input requirements (in terms of organic and mineral fertilisers, herbicides, seeds and fossil fuels) are identical for both scenarios, the differences in the results are due exclusively to differences in biomass yields (51.4 t_{wb}/ha vs 83.8 t_{wb}/ha for double and single cropping systems, respectively) (Table 3.1).

Table 3.8. Characterisation results per FU (1 kg of crude protein in biomass) of sorghum in the two evaluated cropping systems.

Impact category	Units	Double crop	Single crop	Reduction single crop relative to double crop (%)
CC	kg CO_2 eq	7.38	4.45	39.7
OD	mg CFC-11 eq	0.17	0.10	41.2
TA	kg SO_2 eq	0.44	0.26	40.9
FE	g P eq	0.41	0.25	39.0
ME	g N eq	17.0	10.2	40.0
HT	kg 1,4-DB eq	0.37	0.22	40.5
POF	g NMVOC	10.2	6.13	39.9
TET	g 1,4-DB eq	2.21	1.33	39.8
FET	g 1,4-DB eq	14.5	8.74	39.7
MET	g 1,4-DB eq	11.5	6.90	40.0
WD	m^3	1.14	0.69	39.5
FD	kg oil eq	0.39	0.23	41.0

However, the environmental results of both agricultural systems can be considered similar, given the relative influence of environmental factors on the overall results (Figure 3.6).

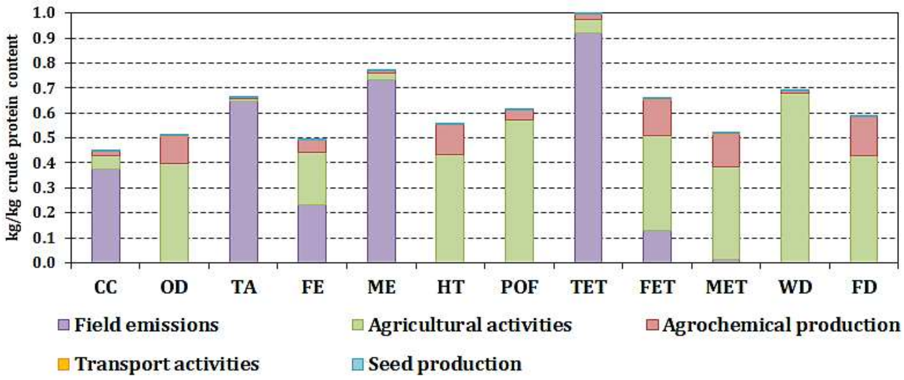

Figure 3.6. Contributions (per FU) to each impact category involving the cultivation of sorghum under both single and double cropping regimes. Single cropping results are shown as an example. Key: CC (kg CO_2 eq/10); OD (kg CFC-11 eq*5·10^6); TA (kg SO_2 eq*2.5); FE (kg N eq*2000); ME (k g P eq*75); HT (kg 1,4-DB eq*2.5); POF (kg NMVOC*100); TET (kg 1,4-DB eq*750); FET (kg 1,4-DB eq*75); MET (kg 1,4-DB eq*75); WD (kg H_2O/1000); FD (kg oil eq*2.5).

As for maize varieties, three of them stand out: field emissions, agricultural activities and agrochemical production. However, in this case, field emissions play a critical role (over 47%) in CC, TA, FE, ME and TET, mainly due to emissions from fertiliser applications. Agricultural activities strongly influence OD, FE, HT, POF, FET, MET, WD and FD (with contributions ranging from 43 to 99%); harvesting, organic fertilisation and irrigation, followed by ploughing and ensilage, are the most influential agricultural activities. Finally, the production of agrochemicals contributes to OD, HT, FET, MET and FD (up to 20%), mainly due to urea production. Transport activities and seed production contribute little to each impact category.

- **Comparative assessment**

Figure 3.7 shows the overall comparison of the environmental results per impact category according to the mean values previously determined for the different cropping systems and varieties; single cropping regime was considered for sorghum cultivation (Figures 3.4 – 3.6).

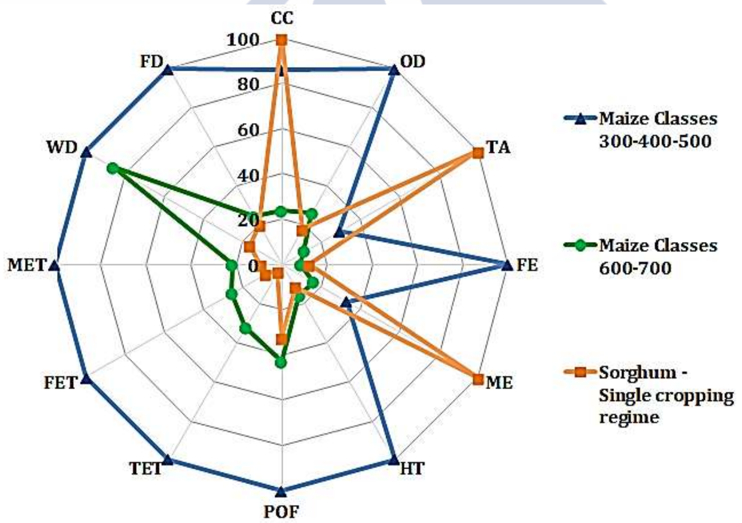

Figure 3.7. Comparative environmental performance (per FU) for the different summer cropping systems taking into account the mean values for the different maize systems and single cropping regime for sorghum cultivation. A "spider-web" graph with a 0-100 scale (from null to the highest environmental impact) depicts the relative values of environmental impacts for the different cropping systems.

According to the figure, maize classes 300, 400 and 500 are the cropping systems with the greatest environmental impacts in most categories. This is mainly due to the requirement of more intensive agricultural practices (especially in terms of water usage and herbicides application) as well as much lower biomass yields. However, special attention should be paid to CC, TA and ME, since the cultivation of sorghum is severely penalised due to the increased use of organic sources in fertilisation activities. Thus, between 45 – 85 t_{wb}/ha of digestate are applied to soils in the different maize cropping systems, while around 100 t_{wb}/ha of pig slurry, with much higher nitrogen content, is used for the production of sorghum. This leads to higher related emissions in terms of N_2O, NH_3 and NO_3^- to the environment, which has a critical effect on such particular categories. Finally, maize classes 600 and 700 are responsible for an intermediate profile, although with unfavourable performance in WD due to the additional irrigation steps compared to the other cropping systems.

- **Sensitivity analysis: alternative FUs**

As mentioned above, the choice of a particular FU has a significant influence on the environmental performance and, therefore, the ranking of priorities in a comparative study. With this in mind, two additional FUs were evaluated to ensure the reliability of the results. Three main functions and FUs are frequently used in agricultural systems (Nemecek et al., 2011): (i) the land management function measured by cultivated hectare per year, (ii) the financial function expressed per currency unit and (iii) the productive function quantified by physical units. However, in many agricultural LCA studies, both mass-based and land-based FUs prevail over other choices. As a result, the different summer systems were evaluated and compared on the basis of 1 ha of cultivated land and 1 ton of biomass (dry basis).

According to the results, the environmental performance of the different cropping systems would be analogous to the base case when 1 ton of biomass is considered as basis for comparison (Figure 3.8a). Thus, again the maize classes 300, 400 and 500 would be the worst option, while the cultivation of sorghum would still be the best alternative, despite its higher impacts on those categories particularly dependent on carbon and nitrogen emissions

(CC, TA, ME). On the contrary, it would be possible to observe a greater difference based on land use (1 ha) criteria (Figure 3.8b). Although maize classes 300, 400 and 500 would again be the worst choice, the environmental impacts associated with the cultivation of sorghum would increase significantly relative to the base case. This would be attributed to the larger requirements of agricultural activities and agrochemicals per hectare compared to maize systems; since the greater ratios of biomass yields are not taken into consideration, they cannot offset the additional requirements. Finally, the maize classes 600 and 700 would become the worst alternatives in WD due to their outstanding water usage.

(a)

(b)

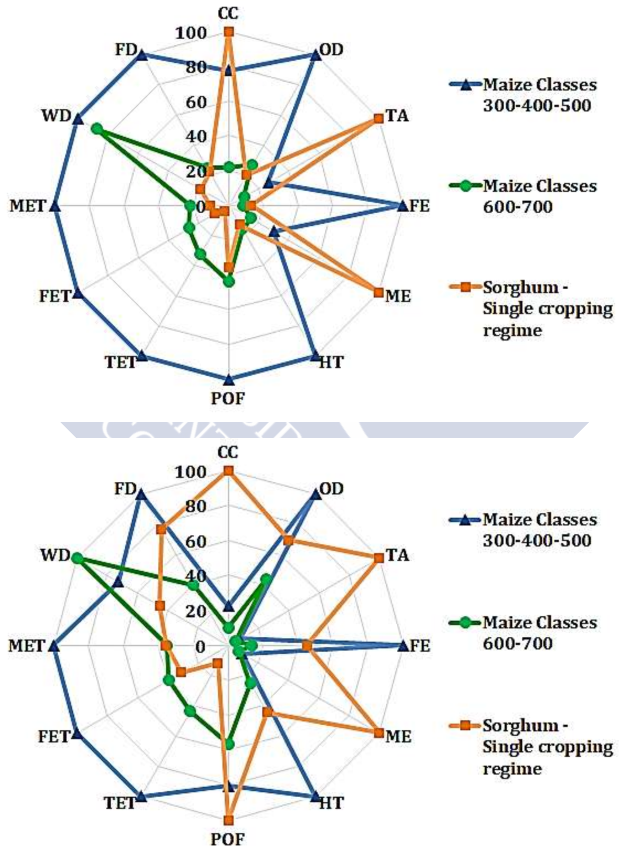

Figure 3.8. Comparative environmental performance for the different summer cropping systems taking into account two alternative FUs: (a) 1 ton (dry basis) of biomass and (b) 1 ha of cultivated land.

3.3 LIVESTOCK FEED PRODUCTION: WINTER CEREALS

3.3.1 Goal and scope definition

This section aimed to evaluate and compare the environmental impacts associated with four winter crops widely cultivated in the Po Valley (Figure 3.1) for animal feed: wheat, triticale, barley and rye. Three varieties were assessed for barley (Reni, Anemone and Alimini) and four for rye (Allawi, Satellit, Luchs and Dank Nowe), while inventory data was provided for one single variety of wheat and triticale. The environmental assessment was conducted from a cradle-to-farm gate approach (including background processes and related emissions), so that further stages from biomass conversion to final waste disposal were left out of the scope of the study.

- **Functional unit**

The crude protein content of the different winter cereals was again considered as the basis for calculations, according to previous studies on summer crops. Thus, the FU was defined as 1 kg of crude protein in biomass. Note that biomass refers to both cereal grain and/or biomass silage, depending on the cropping system evaluated. Mean protein ratios and moisture content were used to estimate the environmental outcomes (Table 3.9).

Table 3.9. Main properties of the winter crops under evaluation: variety, biomass yield, moisture and crude protein content.

Cereal crop	Variety/ Hybrid	Biomass yield (t_{wb}[a]/ha)	Moisture (%)	Crude protein (% dry basis)
Wheat	-	6.55[b]	13.0	12.0
Triticale	-	6.72[b]	13.0	11.9
Barley	Reni	39.7[c]	66.7	10.3
	Anemone	34.5[c]	63.7	10.3
	Alimini	40.1[c]	67.6	10.3
Rye	Allawi	41.4[c]	68.7	10.0
	Satellit	40.3[c]	68.2	10.0
	Luchs	45.8[c]	66.6	10.0
	Dank Nowe	49.6[c]	50.4	10.0

[a] wb = wet basis; [b] $t_{grain}·ha^{-1}$; [c] $t_{biomass}·ha^{-1}$

- **Systems description**

As in summer systems, the cultivation of winter cereals can be divided into three main stages (Figure 3.9): (S1) field preparation and sowing, (S2) crop growth and (S3) biomass harvesting and storage. Wheat and triticale cultivation requires eight months per year, whereas barley and rye require nine months. Specific agricultural operations and timeframe for each cropping system are detailed below.

Wheat and triticale

Wheat and triticale cultivation share the same agricultural steps (Table 3.10). Thus, the first stage (S1) entails land preparation and subsequent sowing. Initially, the soil is fertilised in September with a dose of 40 t_{wb}/ha of digestate from a nearby biogas plant. After organic fertilisation, the soil is ploughed and harrowed; finally, sowing was performed in October at a density of 35,000 seeds/ha.

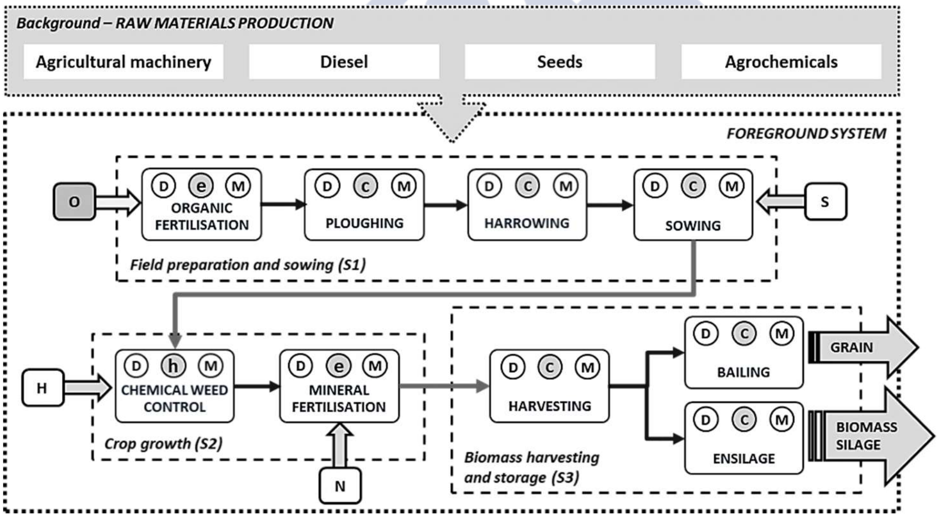

Figure 3.9. Scheme of the system boundaries for the cultivation of wheat and triticale, as well as barley and rye varieties. Key: D = diesel fuel production; M = production and maintenance of agricultural machinery; O = organic fertiliser (digestate) production – excluded; S = seed production; H = herbicide production; N = nitrogen fertiliser production; e = field emissions and discharges from fertilising; c = combustion emissions; h = herbicide emissions.

During the crop growth (S2), chemical weed control is carried out by spraying a mixture of terbutilazine and alachlor at a total rate of 5 kg/ha in one single step. Moreover, mineral fertilisers should be also applied at 60 kg/ha in two periods: ammonium nitrate in November and urea in February. When the seed reaches the suitable ripeness in early June, biomass is harvested (S3). A combine harvester is used to cut the plants and separate the grain from the straw, which is bailed for further uses. Although wheat and triticale share the same cultivation system, the biomass yields are slightly different (Table 3.9); in terms of moisture content, 13% and 15% were considered for grain and straw, respectively.

Barley

Regarding barley varieties (Table 3.11), pig slurry is applied (organic fertilisation) in September at a rate of 45 t_{wb}/ha (S1). Next, the soil is also ploughed and harrowed in September and sown in October at a seeding rate of 190 kg seeds/ha. During crop growth (S2), weeds are chemically controlled in October by spraying 2 kg bifenox/ha. Because mineral fertilisation is also required, 140 kg/ha ammonium nitrate is applied in two stages (half each in November and February). Finally, when the crop reaches the suitable ripeness in May, biomass is harvested and ensiled on the farm in bunker silos (S3). The three barley varieties have similar cultivation processes but different biomass yields and moisture contents (see Table 3.9).

Rye

Rye cultivation shares analogous operations to barley systems, but the timeframe is not exactly the same (Table 3.12). The field preparation and sowing stage (S1) for rye begins in September with the application of 45 t_{wb}/ha of pig slurry. The soil is then ploughed, harrowed and sown in October at a rate of 190 kg seeds/ha. During crop growth, weeds are chemically controlled in October with the application of 2 kg/ha of a herbicide mixture based on cloripalid, MCPA and fluroxipir. As mineral fertilisation is also required, 60 kg/ha of nitrogen as urea is applied in February/March in a single stage. Finally, biomass is harvested and ensiled in early May (S3). Again, although the four varieties of rye share the same cultivation process, they differ in biomass yields and moisture contents (see Table 3.9).

Table 3.10. Field operations and inventory data (per ha) for the cultivation of wheat and triticale.

Stage	Activity	Time (month)	Tractor (A)	Operative machine			A + B	Input rates
			Weight (kg)	Type	Weight (kg)	Effective work (ha/h)	Diesel use (kg/ha)	
Field preparation & Sowing	Organic fertilisation	September	6500	Manure spreader	2500	2.50	4.71	40 t_{wb}/ha Digestate[a]
	Ploughing	September	8500	Plougshare	2000	0.80	22.6	-
	Harrowing	September	5050	Rotary harrow	1000	0.50	24.2	-
	Sowing	October	6500	Seeder	800	0.70	8.53	35,000 seeds/ha
Crop growth	Chemcial weed control	October	4580	Spaying machine	500	3.00	3.32	5 kg/ha Terbutilazine + Alachlor
	Mineral fertilisation	November	5050	Fertiliser spreader	350	3.00	3.17	60 kg/ha Ammonium nitrate
Biomass harvesting & Storage	Harvesting	June	14,000	Combine harvester	-	2.50	17.3	-
	Bailing	June	0.004[b]	Bailing machine	0.0071[b]	N.D.	0.89[b]	-

[a] Digestate composition: 5.36% dry matter, 0.4% total N, 0.2% N-NH_4, 0.08%P, 0.314%K; [b] Per bale; N.D.: no available data.

Table 3.11. Field operations and inventory data (per ha) for the cultivation of barley varieties (Reni, Anemone, Alimini).

Stage	Activity	Time (month)	Tractor (A)	Operative machine			A + B	Input rates
			Weight (kg)	Type	Weight (kg)	Effective work (ha/h)	Diesel use (kg/ha)	
Field preparation & Sowing	Organic fertilisation	September	5050	Slurry spreader	2000	0.30	29.8	40 t_{wb}/ha Pig slurry
	Ploughing	September	7300	Plougshare	2000	0.90	25.0	-
	Harrowing	September	7300	Rotary harrow	1800	0.83	22.3	-
	Sowing	October	5050	Seeder	900	1.00	4.60	190 kg seeds/ha
Crop growth	Chemcial weed control	October	4450	Spaying machine	600	3.03	2.70	2 kg/ha Bifenox[a]
	Mineral fertilisation	November & February	6850	Fertiliser spreader	500	7.69	2.90	140 kg/ha Ammonium nitrate[b]
Biomass harvesting & Storage	Harvesting	May	-	Forage harvester	13,000	1.00	33.4	-
	Ensilage	May	5050	Front loader	450	0.50	0.44[c]	-

[a] Herbicide application needed 1 year every 3 years; [b] 70 kg/ha needed for each fertilisation stage; kg/ton (Bacenetti and Fusi, 2015).

Table 3.12. Field operations and inventory data (per ha) for the cultivation of rye varieties(Allawi, Satellit, Luchs, Dnak Nowe).

Stage	Activity	Time (month)	Tractor (A)	Operative machine			A + B	Input rates
			Weight (kg)	Type	Weight (kg)	Effective work (ha/h)	Diesel use (kg/ha)	
Field preparation & Sowing	Organic fertilisation	September	5050	Slurry spreader	2000	0.30	30.2	45 t_{wb}/ha Pig slurry
	Ploughing	October	7300	Plougshare	2000	0.90	23.7	-
	Harrowing	October	7300	Rotary harrow	1800	1.00	18.9	-
	Sowing	October	5050	Seeder	900	1.00	5.50	190 kg seeds/ha
Crop growth	Chemcial weed control	October	4450	Spaying machine	600	3.03	3.00	2 kg/ha Chloripalid + MCPA + Fluroxipir
	Mineral fertilisation	February/ March	6850	Fertiliser spreader	500	7.69	3.10	128 kg/ha Urea
Biomass harvesting & Storage	Harvesting	Early May	-	Forage harvester	13,000	1.00	35.7	-
	Ensilage	Early May	5050	Front loader	450	0.50	0.44[c]	-

[a] kg/ton (Bacenetti and Fusi, 2015).

- **Allocation rules**

As in the case of summer cereals, different products and co-products can be obtained in the different winter systems under evaluation. Since all the biomass is ensiled together in both barley and rye systems, all the environmental burdens were allocated directly to biomass production. In contrast, in cropping systems involving wheat and triticale, the main product is the grain harvested, although the remaining biomass (i.e. straw) has other applications on the market as well. Therefore, although further processing of straw is beyond the scope of this study, allocation rules should be applied to determine the environmental impacts assigned to the grain, which is entirely dedicated to animal feed (Table 3.13).

Table 3.13. Economic allocation factors for grain and straw for each cropping systems involving wheat and triticale cultivation.

Cropping system		Biomass yield (t_{wb}/ha)	Price (€/t_{wb})	Allocation factors (%)
Wheat	Grain	6.55	283	84.1
	Straw	6.99	50.0	15.9
Triticale	Grain	6.72	271	83.0
	Straw	7.45	50.0	17.0

Regarding the different organic fertilisers used (that is, pig slurry and digestate), they were assumed as co-products of previous systems (as described in section 3.2.1), so that the environmental impacts associated with their production should not be taken into consideration; not so for related emissions due to their application in soils for the cultivation of the winter cereals assessed.

3.3.2 LCI analysis

Inventory data was collected representing an area of 15.3 and 14.0 ha for barley and rye, respectively, while a sizable farm of 33.0 ha of total cultivated area was evaluated for the production of both wheat and triticale (in combination with maize cultivation).

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Similar to summer systems assessed, the life cycle inventory data for direct agricultural inputs were obtained through surveys and questionnaires fulfilled by farmers (Table 3.10 – Table 3.12). Moreover, analogous data sources were also used to compile background inventory (see epigraph 3.2.2). Thus, the ecoinvent® database was primarily consulted to estimate the environmental burdens associated with the production of the different agricultural inputs (such as fertilisers and herbicides) as well as combustion emissions from fuel use in agricultural activities and transport (Althaus et al., 2007; Dones et al., 2007; Nemecek and Käggi, 2007; Spielmann et al., 2007). Field emissions from the application of mineral and organic fertilisers were also estimated according to the Tier 2 method from IPCC guidelines (IPCC, 2006), while a ratio of 0.01 kg P-PO_4^{3-}/kg of applied P proposed by Rossier (1998) was considered for phosphate emissions (Table 3.14). Finally, the distribution factors published in the literature were applied to estimate direct emissions from herbicide application to the different environmental compartments (Althaus et al., 2007; Audsley et al., 1997; Margni et al., 2002; Wang et al., 2007): soil (≥85%), air (10%) and water (≤10%). Note that no changes in soil carbon content were assumed, in line with previous studies on summer cereals (see epigraph 3.2.2).

Table 3.14. Estimation of nitrogen (IPCC, 2006) and phosphorous (Rossier, 1998) emissions (per cultivated ha) from the management of organic and mineral fertilisers in the different cropping systems involving wheat, triticale, barley and rye cultivation.

	Wheat/Triticale		**Barley**		**Rye**	
Emissions	**Digestate**	**Ammonium nitrate/ Urea**	**Pig slurry**	**Ammonium nitrate**	**Pig slurry**	**Urea**
Storage						
N_2O (kg/ha)	0.03	-	0.71	-	0.71	-
NH_3 (kg/ha)	2.60	-	68.8	-	68.8	-
NO_3^- (kg/ha)	3.80	-	100	-	100	-
Application						
N_2O (kg/ha)	0.10	0.77	2.67	0.77	2.67	0.94
NH_3 (kg/ha)	7.53	5.97	41.3	5.95	41.3	7.29
NO_3^- (kg/ha)	8.54	65.4	226	65.1	226	79.7
PO_4^{3-} (kg/ha)	0.02	-	0.37	-	0.37	-

3.3.3 Impact assessment

Again, the characterisation factors of the ReCiPe Midpoint (H) v1.12 method were used (Goedkoop et al., 2013a) in combination with the SimaPro v8.2 software (Goedkoop et al., 2013b) to estimate the potential impacts. According to similar studies (and also in line with the previous section involving summer crop systems), the following impact categories were considered to be the most relevant ones for agricultural systems (Bacenetti et al., 2014; Fusi et al., 2014; González-García et al., 2013; Mogensen et al., 2014; Niero et al., 2015): CC, OD, TA, FE, ME, HT, POF, TET, FET, MET, WD, FD.

3.3.4 Results and discussion

- **Wheat and triticale**

Table 3.15 shows the environmental results associated with wheat and triticale cultivation. It should be noted that, since both cereals share the same cropping system (Table 3.10) and differ slightly in the biomass yields (Table 3.9), their related environmental profiles are remarkably similar.

Table 3.15. Characterisation results per FU (1 kg of crude protein in biomass) of the production of wheat and triticale.

Impact category	Units	Wheat	Triticale
CC	kg CO_2 eq	1.97	1.91
OD	mg CFC-11 eq	0.40	0.38
TA	kg SO_2 eq	0.04	0.04
FE	g P eq	0.25	0.31
ME	g N eq	1.79	1.74
HT	kg 1,4-DB eq	0.44	0.42
POF	g NMVOC	8.78	8.52
TET	g 1,4-DB eq	5.39	5.22
FET	g 1,4-DB eq	17.7	17.1
MET	g 1,4-DB eq	14.6	14.1
WD	m^3	0.03	27.0
FD	kg oil eq	0.43	0.41

The same applies to the relative effect of the different factors contributing to both systems. Thus, according to Figure 3.10, field emissions, agricultural activities and agrochemical production are three of the most important activities, as in previous summer cropping systems. As explained earlier in this chapter, field emissions include emissions from the application of fertilisers and herbicides, while agrochemical production includes impacts related to their manufacturing processes; similarly, agricultural activities integrate the potential impacts related to the production of fuels and agricultural machinery, as well as their use during cultivation activities.

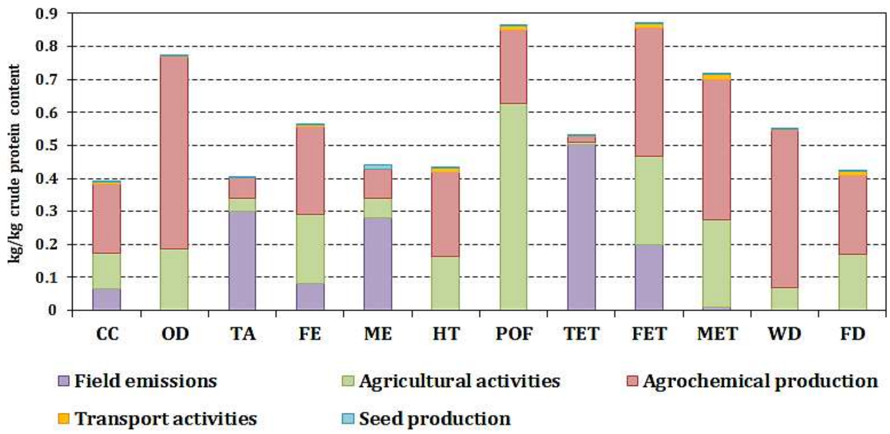

Figure 3.10. Contributions (per FU) to each impact category involving the cultivation of wheat and triticale. Key: CC (kg CO_2 eq/10); OD (kg CFC-11 eq*2·10^6); TA (kg SO_2 eq*10); FE (kg N eq*2000); ME (kg P eq*250); HT (kg 1,4-DB eq*2.5); POF (kg NMVOC*100); TET (kg 1,4-DB eq*100); FET (kg 1,4-DB eq*75); MET (kg 1,4-DB eq*50); WD (kg H_2O/50); FD (kg oil eq).

Field emissions are crucial in the categories of TA, ME and TET with contributions above 64%, mainly due to nitrogen-based emissions. The agricultural activities contribute significantly to OD, FE, POF, FD and all toxicity categories, ranging from 37% to 73% depending on the category. Harvesting, harrowing and ploughing are now the stages with the greatest effects, due to the consumption of diesel and the derived combustion emissions. Regarding agrochemicals production, two nitrogen based fertilisers are used: urea and ammonium nitrate. Their production has a considerable influence on all categories (except TET), with contributions over

15%; however, the production of herbicides is the key process, which implies higher energy requirements that are detrimental to environmental performance. By contrast, seed production and transport activities do not have a remarkable effect on any of the categories.

- **Barley**

The environmental results of the different barley varieties evaluated in this study are presented in Table 3.16, as well as their average values. Accordingly, the Anemone variety of barley has the highest environmental impacts in most categories. However, the impacts among the three varieties varied by less than 5% because they have identical agricultural activities and input requirements (e.g. agrochemicals, fossil fuels, seeds), differing only in biomass yield (see Table 3.9).

Table 3.16. Characterisation results per FU (1 kg of crude protein in biomass) of the different barley cropping systems.

		Barley varieties			
Impact category	**Units**	**Reni**	**Anemone**	**Alimini**	**Average values**
CC	kg CO_2 eq	7.28	7.05	7.42	7.43±0.13
OD	mg CFC-11 eq	0.17	0.18	0.17	$0.17 \pm 5.77 \cdot 10^{-3}$
TA	kg SO_2 eq	0.22	0.23	0.22	$0.22 \pm 5.77 \cdot 10^{-3}$
FE	g P eq	0.31	0.32	0.32	$0.32 \pm 5.77 \cdot 10^{-3}$
ME	g N eq	10.5	11.0	10.7	10.7±0.21
HT	kg 1,4-DB eq	0.37	0.39	0.38	$0.38 \pm 8.16 \cdot 10^{-3}$
POF	g NMVOC	8.86	9.25	9.21	9.11±0.18
TET	g 1,4-DB eq	0.73	0.76	0.74	0.74±0.01
FET	g 1,4-DB eq	0.01	0.01	0.01	0.01±0.00
MET	g 1,4-DB eq	0.01	0.01	0.01	0.01±0.00
WD	m^3	0.01	0.01	0.01	0.01±0.00
FD	kg oil eq	0.35	0.37	0.36	$0.36 \pm 8.16 \cdot 10^{-3}$

Figure 3.11 shows the average values, maximum and minimum impacts of barley varieties. The contribution analysis again revealed that field emissions, agricultural activities and agrochemical production have the

highest influence on the environmental outcomes. Field emission contributions to CC, TA, FE and ME range from 30% (FE) to 96% (TA). Nitrogen-based emissions from organic fertilisation are responsible for the greatest impacts due to the high nitrogen content of pig slurry applied to soils. Agricultural activities have also a strong influence on OD, FE, HT, POF, FET, MET and FD, with contributions exceeding 44%. In this case, organic fertilisation has a significant contribution, together with ploughing, harrowing and harvesting, due to their higher diesel requirements (20-30 kg/ha) compared to other activities. Similarly, agrochemical production contributes considerably to OD, FE, HT, POF, FET, MET, WD and FD (with contributions of up to 77%); however, fertiliser production is responsible for the greatest impacts (more than 80%) while herbicide production has a minor contribution. It is also worth noting the strong effect of seed production in ME and TET (18% and 71%, respectively), mainly due to the emissions of agrochemicals used to produce seeds. Transport activities contribute less than 1% to each impact category.

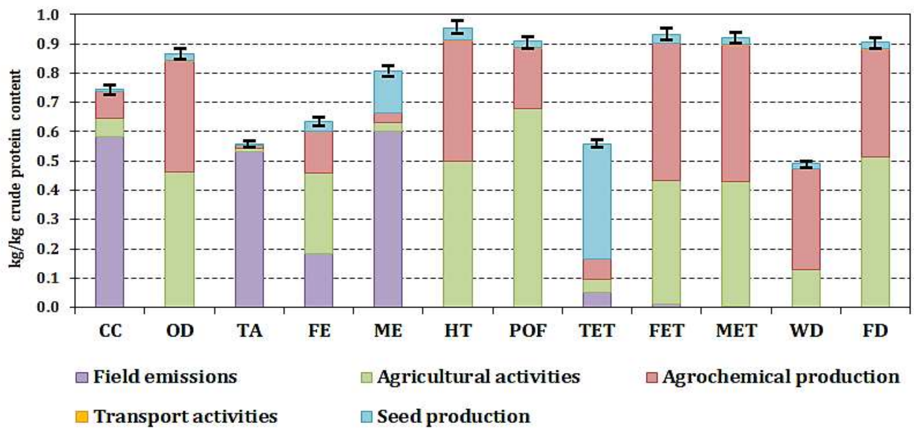

Figure 3.11. Mean contribution (per FU) to each impact category for barley production. Error bars indicate maximum and minimum values. Key: CC (kg CO_2 eq/10); OD (kg CFC-11 eq*5·10^6); TA (kg SO_2 eq*2.5); FE (kg N eq*2000); ME (kg P eq*75); HT (kg 1,4-DB eq*2.5); POF (kg NMVOC*100); TET (kg 1,4-DB eq*750); FET (kg 1,4-DB eq*75); MET (kg 1,4-DB eq*75); WD (kg H_2O/25); FD (kg oil eq*2.5).

Figure 3.11 also displays the maximum and minimum impacts for the cultivation of the barley varieties under assessment, which allows comparing the influence of the crop with the average values (Table 3.16). Thus, Anemone is above average (by approximately 2%) in all impact categories, Reni is generally below average (up to 3%) and Alimini tends to be within 2% of the average.

- **Rye**

Table 3.17 shows both the environmental impacts related to the rye cultivation systems for the four varieties and their average environmental results. The Satellit variety of rye shows the highest impacts in all the categories assessed, followed by the Allawi cropping system. In contrast, Dank Nowe has the best results regardless of impact category. Like barley, variations are directly related to differences in biomass yields, although moisture content also has an influence for rye varieties (see Table 3.9).

Table 3.17. Characterisation results per FU (1 kg of crude protein in biomass) of the different rye cropping systems.

		Rye varieties				
Impact category	**Units**	**Allawi**	**Satellit**	**Luchs**	**Dank Nowe**	**Average values**
CC	kg CO_2 eq	6.90	6.98	5.85	5.57	6.33±0.62
OD	mg CFC-11 eq	0.12	0.12	0.10	0.10	0.11±0.01
TA	kg SO_2 eq	0.23	0.23	0.19	0.18	0.21±0.02
FE	g P eq	0.28	0.28	0.24	0.22	0.26±0.02
ME	g N eq	9.97	10.4	8.70	8.28	9.34±0.87
HT	kg 1,4-DB eq	0.27	0.28	0.24	0.22	0.25±0.02
POF	g NMVOC	7.38	7.51	6.30	6.00	6.80±0.66
TET	g 1,4-DB eq	1.21	1.33	1.12	1.06	1.18±0.10
FET	g 1,4-DB eq	8.12	8.29	6.95	6.62	7.50±0.72
MET	g 1,4-DB eq	7.91	8.06	6.76	6.44	7.29±0.70
WD	m^3	0.01	0.01	0.01	0.01	0.01±0.00
FD	kg oil eq	0.28	0.29	0.24	0.23	$0.26 \pm 2.60 \cdot 10^{-3}$

Figure 3.12 shows the average environmental results associated with the cultivation of the rye varieties analysed in this study. Field emissions contribute significantly to CC, TA, FE, ME and TET, with contributions ranging from 34% (FE) to 97% (TA). Agricultural activities play a critical role in most

impact categories (OD, FE, HT, POF, FET, MET and FD) with contributions above 51%: the higher the consumption of diesel, the greater the environmental impact. The production of agrochemicals contributed 20-72% to OD, HT, FET, MET, WD and FD, mainly due to the manufacture of fertilisers; in fact, these contributions are lower than those of barley varieties because the cultivation of rye requires less mineral fertilisation. Again, as in the case of barley, seed production contributes in particular to ME and TET, while transport activities contribute less than 1% to all impact categories. Comparing the maximum and minimum impacts with the mean values of the four varieties (Figure 3.12), Allawi and Satellit are above average in all categories (by approximately 8% and 11%, respectively); whereas Luchs and Dank Nowe varieties are below average (around 7% and 12%, respectively).

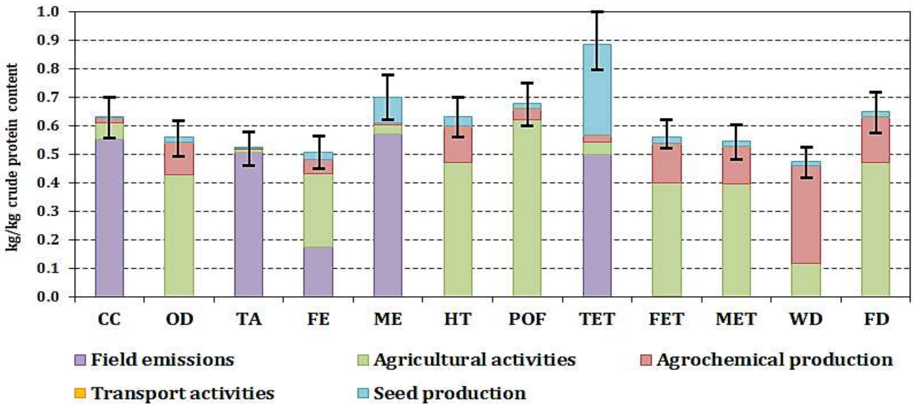

Figure 3.12. Mean contribution (per FU) to each impact category for rye production. Error bars indicate maximum and minimum values. Key: CC (kg CO_2 eq/10); OD (kg CFC-11 eq*5·10^6); TA (kg SO_2 eq*2.5); FE (kg N eq*2000); ME (kg P eq*75); HT (kg 1,4-DB eq*2.5); POF (kg NMVOC*100); TET (kg 1,4-DB eq*750); FET (kg 1,4-DB eq*75); MET (kg 1,4-DB eq*75); WD (kg H_2O/25); FD (kg oil eq*2.5).

- **Comparative assessment**

Figure 3.13 shows the overall comparison of environmental performance per impact category for different winter systems and varieties. Among winter cereals, wheat and triticale are the options with the greatest impact in most categories. These differences are mainly related to a greater need for chemical weed control, including both the production of herbicides and their subsequent application in agricultural soils. However, analogous to summer cereals, this trend changes in terms of CC, TA and ME: the large amount of organic fertilisers used (the highest of all crops compared) was the main reason for the most unfavourable results of barley varieties in these categories. The higher ratios of pig slurry applied to soils are responsible for greater nitrogen emissions, with a key role in acidification and eutrophication potentials. Finally, rye cultivation shares values close to those of barley in these cases (with similar fertilisation rates), although it was found as the winter cereal with the lowest impacts in the other impact categories.

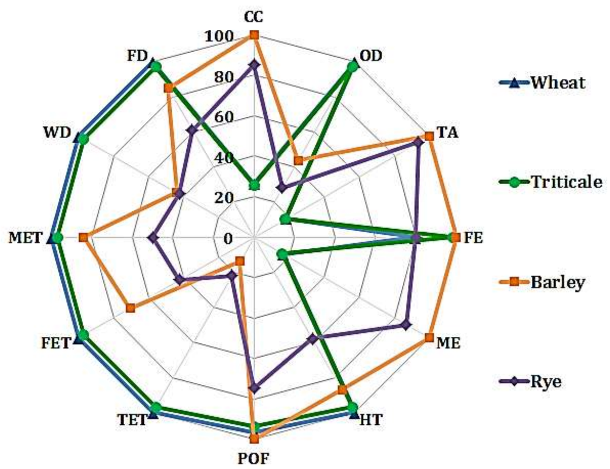

Figure 3.13. Comparative environmental performance (per FU) for the different winter cropping systems: barley and rye (mean values) together with winter and triticale.

- **Sensitivity analysis: alternative FUs**

The selection of a FU is essential to provide a basis for comparison. In this regard, a mass-based FU (1 kg of protein in biomass) was chosen as the base case to compare the different winter cereals evaluated according to their use as a protein source for animal feed. However, the application of more than one FU could further clarify the environmental performance of the cropping systems and support the prioritisation of different alternatives. Accordingly, two additional FUs were proposed (in line with the approach defined for summer cereals): (a) 1 ha of cultivated land and (b) 1 ton of biomass (dry basis).

(a)

(b)

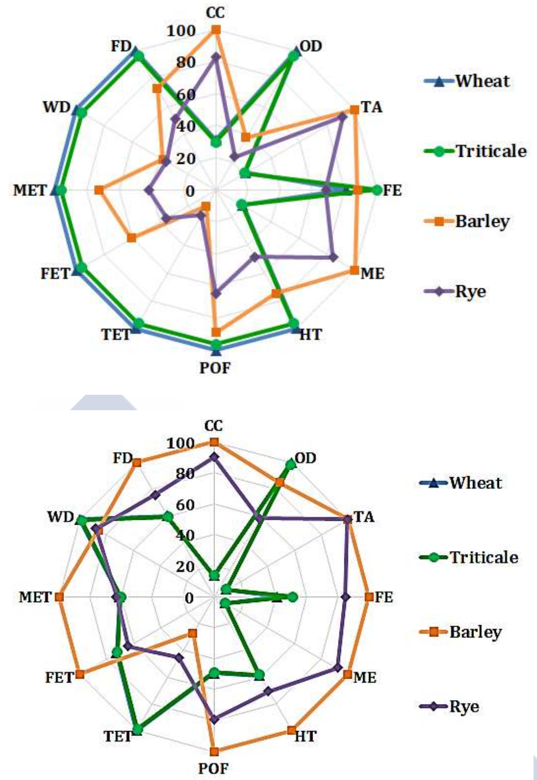

Figure 3.14. Comparative environmental performance for the different winter cropping systems taking into account two alternative FUs: (a) 1 ton (dry basis) of biomass and (b) 1 ha of cultivated land.

Figure 3.14a demonstrates that similar results can be achieved when comparing the different systems in relative terms according to mass-based FUs. Thus, again the cultivation of wheat and/or triticale would be responsible for the highest environmental impacts, except for CC, TA and ME. However, focusing on the land-based FU (1 ha), different outcomes to those obtained for a mass-based FU are obtained (Figure 3.14b). In this case, the production of barley and rye would have the worst results in most categories, in addition to its previous contribution to CC, TA and ME in the baseline situation. These results are closely linked to both agricultural practices and biomass yield. The biomass yields of barley and rye are more than 6 times higher, so that fewer inputs are needed to obtain the same production yields, reducing related impacts. However, this proportionality is not accounted for when considering a land-based FU, which leads to higher requirements per unit area (1 ha). This is the rationale behind the reductions in the environmental impacts allocated to wheat and triticale, while barley and rye gain relevance as the least environmental-friendly alternatives.

3.4 Conclusions

In this chapter, the environmental impacts of the most widespread cereal crops cultivated in the Lombardy region (Northern Italy) for animal feed were estimated and compared following the LCA approach. The aim was to identify the most environmentally friendly option among the different summer and winter cereals assessed, as well as the critical activities involved in the different cropping systems.

In view of the results, field emissions, agricultural activities and agrochemical production were identified as the factors that most contribute to the environmental impacts (hotspots), regardless of the system evaluated. Moreover, focusing on winter cereals, Reni and Dank Nowe had the most beneficial environmental results among barley and rye varieties, respectively; maize classes 500 and 600, together with single cropping for sorghum production, had the lowest impacts involving summer cereals systems.

Moreover, the overall comparative analysis revealed that rye and maize classes 600-700 (mean values) were the best options with the most favourable environmental profiles of winter and summers crops, respectively. However, a sensitivity analysis was also developed to evaluate the effect of alternative FUs on the comparative results. The reliability of the results held on different mass-based criteria, while substantial changes were registered with a land-based FU (1 ha) is considered. Therefore, the selection of the best cereal cropping system may depend to a large extent on the FU used as the basis for calculations.

The main findings of this chapter could be used as a basis for future environmental studies involving agricultural systems in both the study area and other related areas, in response to the limited data on this topic available to date in the literature.

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CHAPTER 4. ENVIRONMENTAL ASSESSMENT OF LIVESTOCK HUSBANDRY: COW DAIRY SECTOR

Summary

In recent decades, agricultural practices in the food sector have intensified with the objective to meet the growing demand for food products worldwide. This has led to severe environmental consequences, mainly due to the large amount of primary resources consumed by the livestock sector, which plays a key role in the food industry. In this sense, cow farming systems are at the forefront of livestock-related emissions, most of which come from the dairy supply chain.

In this context, the present chapter focused on the assessment of the environmental performance of a cow milk farming system, representative of the dairy sector in Northeast Spain, from a cradle-to-farm gate perspective. The LCA principles established by ISO standards were followed along with the CF guidelines proposed by the International Dairy Federation (IDF). The environmental results showed two critical contributing factors: livestock feed production and on-farm emissions from agricultural activities, with contributions above 50% in most impact categories. A comparison with other LCA studies was carried out, which confirmed the consistency of these results with the values reported in the literature for dairy systems in several countries. Additionally, WF values were also estimated according to the WFN methodology to reveal that feed and fodder production also had a predominant influence on the overall WF impacts, with contributions of up to 99%. Green WF was responsible for significant environmental burdens (about 88%) due to the impacts associated with the cultivation stage. Finally, the substitution of alfalfa by other alternative sources of protein in animal feed was also proposed and discussed because of its relevance as one of the main contributors to livestock feed. The results obtained in this chapter are expected to cooperate in the implementation of alternative practices that improve the environmental profile of dairy farms in the region.

CHAPTER 4 - INDEX

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4.1 INTRODUCTION TO LIVESTOCK HUSBANDRY: DAIRY SECTOR

As explained in Chapter 3, in recent times, the world is facing problems of food safety and environmental damage (Gerber et al., 2013; Reckmann et al., 2012, 2013). Indeed, the continued growth of the world's population has led to an increase in the demand for products in the food sector, forcing the food industry to adopt increasingly intensive practices (de Léis et al., 2015; Yan et al., 2011). In this context, concern for sustainable food production has prompted ecological considerations to be taken into account in the development of environmentally friendly production methods, as encouraged by consumers (Notarnicola et al., 2012; Pirlo et al., 2014).

Particular attention has been paid to the livestock sector due to its dominant relevance in the food industry (de Vries and de Boer, 2010; del Prado et al., 2013; Roy et al., 2009). The life cycle stages of livestock products (such as production, conservation and distribution) require large amounts of food, energy and water, resulting in negative environmental consequences (Roy et al., 2009; Steinfeld et al., 2006). As a result, the livestock sector has been identified as responsible for about 14.5% of GHG emissions (Gerber et al., 2013), as well as one of the most important indirect drivers of water consumption and pollution (Hoekstra, 2014). In this context, cattle breeding occupies a leading position with about 65% of emissions from the livestock sector, of which 20% are allocated to dairy cattle (Gerber et al., 2013); in addition, dairy products account for about 10% of the global anthropogenic eutrophication potential, as well as 6% of the acidification problems in Europe (Gerber et al., 2011; Tukker and Jansen, 2006).

The most important product in the dairy sector is milk, whose production has expanded rapidly over the past decades, especially in developing countries (Gerosa and Skoet, 2012). Moreover, milk consumption is expected to double by 2050 compared to 2000 (Alexandratos and Bruinsma, 2012), probably due to projected population growth and awareness that milk is considered a prescription for good health and an important ingredient in a nutrient-rich diet (Meneses et al., 2012). However, in line with the above, growing demand for milk has encouraged changes in production practices that may lead to less environmentally friendly systems (Battini et al., 2014; Yan et al., 2011).

Among the different assessment methods considered to evaluate the environmental burdens of milk production, the LCA methodology has been applied to a wide range of dairy products (Baldini et al., 2017; Fantin et al., 2012; Thomassen et al., 2009; Van der Werf et al., 2009; Vasilaki et al., 2016). However, to date there is little environmental information available on Spanish milk production, despite the fact that Spain ranks seventh in Europe in terms of milk production (6.8 million tonnes in 2014). Moreover, although there are several areas of relevant dairy farming activities in this country (FAOSTAT, 2014), most environmental studies were focused on Northwest Spain (Del Prado et al., 2013; González-García et al., 2013c; Meneses et al., 2012), while other regions are still pending to be fully evaluated.

Catalonia – in Northeast (NE) Spain – ranks fourth in the Spanish regions for the dairy industry with a total production of about 758,000 tonnes of raw milk based predominantly on confined feedlot regimes (MAPAMA, 2017). In a recent report, Vasilaki et al. (2016) led the evaluation of the dairy sector in this area, but focused only on the CF and WF of various types of yogurt. Under this premise, this Chapter 4 aims to evaluate the environmental burdens of a dairy cow system in Catalonia, focusing on the production of raw milk at the farm stage and extending the framework of the study by integrating additional impact indicators: acidification, eutrophication and resources depletion. In this way, it is expected that the main outcomes provide further information on critical activities (hotspots) in the production chain, as well as to be useful in proposing alternative practices that improve the environmental profile of the target system.

4.2 MILK PRODUCTION IN SPAIN: A CASE STUDY IN CATALONIA

This chapter followed the principles of the LCA methodology (ISO 14040, 2006; ISO 14044, 2006) and the CF guidelines defined by the International Dairy Federation (IDF, 2010), along with the WFN method for water assessment (Hoekstra et al., 2011).

4.2.1 Goal and scope definition

The environmental study was performed in a typical dairy farm located in Osona (Catalonia – Figure 4.1). This region is recognised as one of the most significant livestock production areas in Spain, especially for milk production, with a high density of farms, husbandry activity and related industries such as slaughterhouses, food factories and sub-suppliers (GenCat, 2010).

The study was conducted through a cradle-to-farm gate perspective: from raw materials up to the point when raw milk is ready to leave the farm. Further stages of dairy processing were excluded from the assessment because of their minor contribution in line with previous LCA studies involving dairy products (Daneshi et al., 2014; Fantin et al., 2012; González-García et al., 2013a,b; Vasilaki et al., 2016).

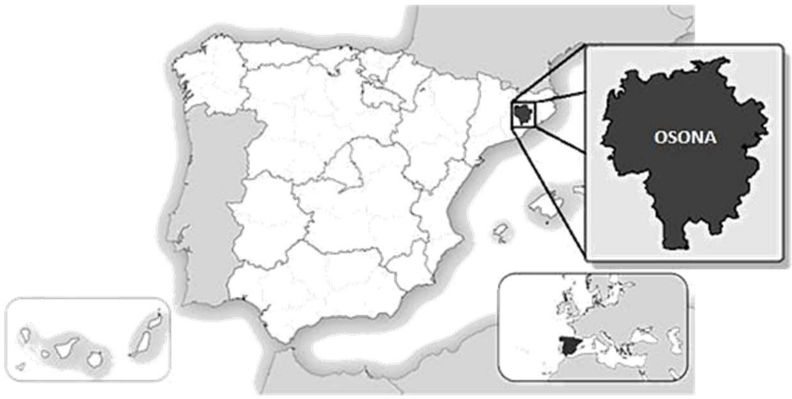

Figure 4.1. Location of Osona in Catalonia (NE Spain).

- **Functional unit**

Although mass-based and volume-based FUs have prevailed in LCA studies regarding milk production systems to date, the use of a quality corrected FU that includes the mass and nutrient content of the product is recently being considered as a better approach (Fantin et al., 2012). Accordingly, 1 kg of fat and protein corrected milk (FPCM) at farm gate was taken as the reference unit (FU) in this study, following the recommendations of the IDF guidelines for dairy farming systems (IDF, 2010). The raw milk weight was converted to FPCM (kg/yr) using the following equation (IDF, 2010):

$$\mathrm{FPMC} = \mathrm{P} \times [0.1226 \times \mathrm{FT} + 0.0776 \times \mathrm{TP} + 0.2534] \qquad (1)$$

Where P = production (kg/yr); FT = fat content (%); TP = true protein[1] content (%). It should be noted that 1 kg of FPCM is equivalent to approximately 1 kg of raw milk according to the outcome of Equation (1), considering the standard fat and protein content in milk (4% fat and 3.3% true protein) stated by IDF guidelines (IDF, 2010).

- **System description**

In line with above, the farming system under study encompasses all the relevant processes related to the production of cow feed (on-farm and off-farm) and other farm inputs, transport activities, animal husbandry, raw milk production and waste management. Moreover, emissions and discharges to soil, air and water derived from the entire farming system were also considered (Figure 4.2).

The characteristics of the farm are common to typical dairy farms in NE Spain. The total area of the farm is 74.7 ha, of which 55.3 ha are used for cereal cultivation, while the rest is occupied by the facilities of the farm (including breeding barns, milking rooms, sheds for equipment and agricultural machinery). The dairy farm itself comprises a herd of 1,180 animals, including dairy cows (400 heads), dry cows (80 heads), heifers (500 heads) and calves (200 heads). Detailed information on the characteristics of the herd is given in Table 4.1. An average number of 400 births per year and replacement rate of culled cows of 34% are registered. The main products of the farm are raw milk, meat (beef) and cereals (grain barley and oat silage); the beef output includes surplus heifers, calves and culled cows for meat.

[1] True protein = crude protein – NPN (non-protein nitrogen)

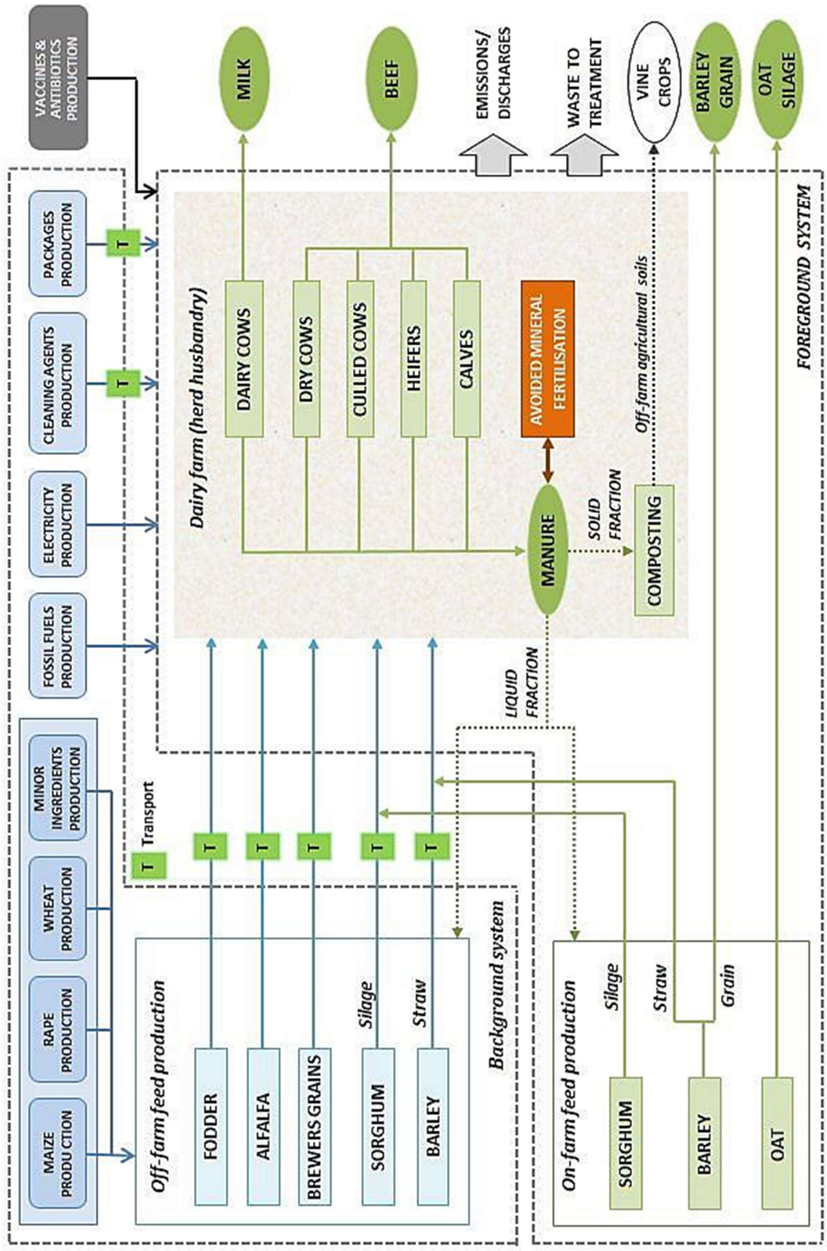

Figure 4.2. Scheme of the system boundaries for the cow dairy system. Key: Brown box includes avoided impacts due to both mineral fertiliser production and further application on soils. Grey box represents excluded processes.

Table 4.1. Characteristics of herd in the dairy farm under evaluation.

Parameter	Units	Dairy cows	Dry cows	Heifers	Calves
Animal breed	-	Frisona	Frisona	Frisona	Frisona
Herd size	heads/yr	400	80	500	200
Milk production	kg milk/day	26.7	-	-	-
DE[a]	%	50.0	50.0	50.0	50.0
GE[b]	MJ/day	570	320	196	53.0

[a] Feed digestability; [b] Gross energy.

The animal feed is rich in silage and concentrate feed (fodder), although diets vary widely depending on the type of animal (Table 4.2). The fodder fed to the cattle consists mainly of maize meal (56.7%), rapeseed meal (22.9%) and wheat bran (20.4%). Both water and feed are supplied to herd by troughs.

Table 4.2. Main ingredients in the diet of each type of animal in the herd.

Ingredient (kg/ head·day)	Dairy cows	Dry cows	Calves/ Heifers
Sorghum silage	24.0	22.6	22.6
Alfalfa (pellets)	2.50	-	-
Barley straw	1.30	4.00	4.00
Fodder	12.5	1.60	1.60
Brewers grains	5.00	-	-

As far as cereal cultivation is concerned, three different crops are grown on the farm premises (on-farm) for feeding purposes: sorghum, barley and oat. Double-crop system regimes are applied: sorghum is grown for six months of the year (June – November) followed by barley or oat cultivation (for the period between November and June). Thus, 55.3 ha are cultivated with sorghum, while only 21.0 ha are intended for barley and 34.3 ha for oat. Both sorghum silage and barley straw are used as animal feed whereas barley grain and oat silage are sold as surplus.

The manufacture of products used for cleaning activities was considered, as well as the management of the waste generated. Similarly, the elimination of pharmaceuticals (vaccines and antibiotics) was also included in the study, but no inventory data on their production was available. The production and consumption of diesel and electricity were also taken into account.

Solid and liquid manure are valorised due to their nutrient content. The liquid fraction is directly used as organic fertiliser in on-farm fields destined to cereal cultivation whereas the solid fraction is initially composted before its use in off-farm vine crops. In both cases, emissions and discharges from manure management (including storage, composting and application) were included, together with the environmental credits related to its use as an organic fertiliser (avoided mineral fertilisation) in on-farm activities. In contrast, manure impacts and/or credits were left out WF assessment. Within European boundaries, Nitrates Directive (1991) defines areas affected by nitrate pollution as vulnerable areas. In this context, several NVZ can be found in Catalonia. Accordingly, the Decree 139/2009 was developed with the aim of regulating the management practices of manure and other fertilisers, as well as preventing and reducing the pollution of water bodies caused by nitrates from agricultural activities in the region (Catalunya, 2009). To this aim, several variables such as type of agricultural soil, irrigation activities and fertilisation activities were taken into consideration, although significant progress has been made in this issue. With this in mind, it was considered appropriate to assume a neutral effect from cow manure valorisation on water results.

Finally, the construction and maintenance of the infrastructure was not considered within the system boundaries, as its contribution may be considered negligible according to similar studies in the literature (Castanheira et al., 2010; De Léis et al., 2015; Hoekstra et al., 2011).

- **Allocation rules**

The entire farming system assessed in this chapter can be considered as a typical multi-output system, in which four main products can be obtained: (raw) milk, beef, barley grain and oat silage. In this regard, an economic allocation approach was followed, using historical market prices recently registered in Spain (MAPAMA, 2014). This choice would be in agreement with most LCA studies in the dairy industry worldwide (Castanheira et al., 2010;

González-García et al., 2013a; Pirlo et al., 2014; Yan et al., 2011). The resulting allocation percentages for the different outputs are presented in Table 4.3.

Similarly, it should be noted that barley grain is also a by-product of the barley cropping system, so that the related environmental burdens should therefore be distributed between the grain and the straw. A mass-based partition was applied in this case, resulting in factors of 58.7% and 41.3% allocated to barley grain and straw, respectively.

Table 4.3. Economic allocation factors for the different products in the farming system.

Product	Production (t/year)	Price (€/kg)	Allocation factors (%)
Raw milk	3900	0.35	76.0
Beef	64.0	3.50	12.5
Barley grain	542	0.18	5.4
Oat silage	644	0.17	6.1

As aforementioned, although the excess (about 17% of liquid slurry and 100% of solid fraction) of manure generated at the farm is typically delivered to other nearby farmlands, only a fraction of this manure is used as organic fertiliser for the cultivation of on-farm cereals. Therefore, in the latter case, the environmental impacts derived from its valorisation at farm must also be taken into consideration in the study; a system expansion approach was applied on this issue. In this regard, manure was assumed as a substitute for mineral fertilisers, so the avoided impacts related to its production, storage and subsequent application were also included, in line with the substitution ratio defined in literature (de Vries et al., 2011, 2012; Nguyen et al., 2010; Sommer and Birkmose, 2007). Accordingly, the mineral fertiliser equivalent (MFE) for N from liquid manure can be close to 75%, implying that every 100 kg of N applied to a crop in the form of livestock manure should replace 75 kg of N from other mineral fertilisers (Nguyen et al., 2010).

4.2.2 LCI analysis

With the aim of ensuring data quality and reducing uncertainties, most of the LCI data used in this study was primary data obtained through surveys and questionnaires fulfilled by farmers for the 2013/2014 farming season.

The questionnaires compiled information on the size of farm facilities, herd composition and characteristics, origin and production stages of feed inputs, feed rations, outputs production, resources consumption (water, energy, diesel…) and waste management. Global LCI data per FU (1 kg of FPMC at the farm gate) are displayed in Table 4.4.

Table 4.4. LCI data per FU (1 kg of FPMC at farm gate) for the global system.

Inputs/Outputs	Amount	Units
Inputs from technosphere		
Animal feed		
Sorghum silage (on-farm)	720	g
Sorghum silage (off-farm)	1.40	kg
Barley straw (on-farm)	139	g
Barley straw (off-farm)	168	g
Fodder	554	g
Alfalfa	93.6	g
Cereals cultivation		
Oat silage (on-farm)	165	g
Barley grain (on-farm)	139	g
Cleaning products		
Detergent (sodium chloride)	0.08	g
Acid solution	0.01	g
Disinfectant	0.03	g
Kraft paper	0.72	g
Packaging material		
Silage plastic (polyethylene)	0.04	g
Fossil fuels		
Machinery lubricant oil	0.05	g
Diesel	2.29	g
On-farm energy use		
Electricity	22.3	Wh
Transport		
Van/Lorry	172	kg·km
Inputs from environment		
Water	2.67	L

Table 4.4 (cont.). LCI data per FU (1 kg of FPMC at farm gate) for the global system.

Inputs/Outputs	Amount	Units
Outputs to technosphere		
Products		
Raw milk	1.00	kg
Beef (meat)	16.5	g
Barley grain	139	g
Oat silage	165	g
Co-products		
Liquid manure[a]	0.80	L
Solid manure[b]	1.37	L
Waste to treatment		
Solid waste	0.49	g
Wastewater	2.67	L
Avoided fertiliser production		
N from manure	13.7	g
Outputs to environment		
Air emissions		
CH_4 – Enteric fermentation	44.6	g
CH_4 – Manure management	6.35	g
N_2O – Manure storage	0.74	g
N_2O – Manure application	0.50	g
NH_3 – Manure storage	13.2	g
NH_3 – Manure application	1.09	g
Water emissions		
NO_3^- – Manure storage	59.7	g
NO_3^- – Manure application	6.65	g
Avoided fertiliser application		
N_2O	0.49	g
NH_3	0.39	g
NO_3^-	4.23	g

[a] N content: 3.50 g N/kg liquid manure; [b] N content: 5.57 g N/kg solid manure.

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As regards on-farm cereals (sorghum, barley and oat), all the activities related to their cultivation, from field preparation to biomass harvesting, were also taken into consideration in the study. Moreover, the production and use of inputs (e.g., seeds, agrochemicals and diesel), distribution to the farms, emissions from agrochemicals application and diesel use, as well as the production, maintenance, use and disposal of agricultural machinery, were also considered. Specific inventory information (Table 4.5) related to activities performed in these cropping systems was taken from a previous case study developed by González-García et al. (2015). Similarly, the different stages involved in alfalfa cultivation and fodder production were obtained from Gallego et al. (2011) and ecoinvent® database (Nemecek and Käggi, 2007), respectively. In this way, agricultural activities were adapted taking into account the primary information provided by growers about inputs, diesel and energy requirements.

Table 4.5. Field operations timeline and main inventory data (per ha) for the cultivation of on-farm cereals (González-García et al., 2015).

Cereal	Cultivation season	Agricultural activities	Diesel use (L/h)	Additional comments
Sorghum	June-November	Organic fertilisation	18.0	170 kg N[a]
		Ploughing	25.0	-
		Sowing	20.0	-
		Chemical weed control	10.0	1.2 L MCPA
		Harvesting	200	-
Barley	November-June	Organic fertilisation	18.0	170 kg N[a]
		Ploughing	19.0	-
		Sowing	19.0	-
		Chemical weed control	10.0	1.5 L Herbicide[b]
		Harvesting	200	-
Oat	November-June	Organic fertilisation	18.0	170 kg N[a]
		Ploughing	19.0	-
		Sowing	19.0	-
		Harvesting	200	-

[a] Cattle manure (liquid fraction); [b] Banvel Triple: 10% w/v Dicamba; 26.5% w/v MCPA; 19.6% w/v 2,4-D.

Secondary data from literature and commercial databases were also used to complete the background inventory. Thus, data on electricity generation (Spanish profile), diesel production, as well as the manufacture of agrochemicals, cleaning products and packaging material were also taken from the ecoinvent® database (Althaus et al., 2007; Dones et al., 2007; Hischier, 2007; Spielmann et al., 2007; Wernet et al., 2016). Moreover, combustion emissions associated with fuel use by agricultural machinery and transport activities were also compiled from the ecoinvent® database (Nemecek and Käggi, 2007; Spielmann et al., 2007; Wernet et al., 2016).

Direct emissions into the atmosphere resulting from dairy cow farming activities were also estimated. CH_4 emissions from both enteric fermentation and manure management (storage and further field application) were calculated according to the emissions factors provided by the IPCC (2006). To this aim, Tier 2 method was applied combining the default emission factors (Y_m, %DE, EF_T) with the primary data collected from questionnaires, including herd composition and characteristics, animal diet, live weight and average weight gain per cattle head. Direct and indirect N emissions from manure storage and application were also estimated; however, both Tier 1 and Tier 2 methods were followed in parallel, depending on the availability of reliable information (IPCC, 2006). In this case, data on herd composition, manure generation and N content was required in combination to the default emission factors provided by the methods (EF_3 and EF_4) for N_2O, NH_3 and NO_3^-. Finally, Tier 1 method from IPCC guidelines was also applied to estimate the avoided emissions and discharges associated to the avoided production and use of mineral fertilisers due to the fertilising capacity of (organic) manure excreted by the herd.

Focusing on the WF assessment, primary inventory data collected from questionnaires was prioritised to analyse water requirements throughout the different stages of the whole system (Table 4.4). The direct water consumption was mainly related to the water demand from cattle and cleaning activities. Indirect water consumption was also estimated by including water usage from energy requirements, animal feed (fodder and feeding crops) production – either inside or outside farm boundaries – as well as water in waste streams (mainly liquid manure). However, secondary data are necessary in the selection of the characterisation factors, due to the lack of accurate information on WF associated with the compounds and processes evaluated.

Table 4.6. Secondary data sources used for WF calculations.

Inventory data	Data sources
Animal feed	Mekonnen and Hoekstra (2010)
Sorghum silage	FAOSTAT crop code: 83
Barley	FAOSTAT crop code: 44
Oat silage	FAOSTAT crop code: 75
Maize (fodder)	FAOSTAT crop code: 56
Rape meal (fodder)	FAOSTAT crop code: 270
Wheat (fodder)	FAOSTAT crop code: 15
Waste disposal	Franke et al. (2013)
Wastewater	Directive 91/271/EEC (1991)
Energy use	Ecoinvent® database (Dones et al., 2007)
Electricity	Spanish country mix
Cleaning products	Ecoinvent® database (Althaus et al., 2007; Hischier et al., 2007)
Detergent	Sodium chloride, brine solution, at plant/RER U
Acid solution	Sulphuric acid, liquid, at plant/RER U
Disinfectant	Phosphoric acid, industrial grade, 85% in water, at plant/RER U
	Potassium hydroxide, at regional storage/RER U
	Sodium hypochlorite, 15% in water, at plant/RER U
Kraft paper	Kraft paper, bleaches, at plant/RER U
Packaging material	Ecoinvent® database (Hischier et al., 2007)
Silage plastic	Polyethylene, LPDE, granulate, at plant/RER U
Transport	Ecoinvent® database (Spielmann et al., 2007)

In this regard, Mekonnen and Hoekstra (2010) provide valuable information on the WF of imported crops and derived cereal products. The following raw materials for animal feeding were considered in the study (Table 4.4): sorghum silage, barley (grain and straw), oat silage, alfalfa and brewers grains, together with maize, rapeseed meal and wheat (bran) as main ingredients of fodder mixtures. Similarly, WF ratios in literature were applied to estimate the grey WF results (Franke et al., 2013). Thus, the maximum acceptable concentration (C_{max}) established by the Directive 91/271/EEC (1991) was considered as a reference; a constant value of 0 mg/L for the natural concentration in the receiving water body (C_{nat}) was used for calculations. Finally, the life cycle information from the ecoinvent® database (Althaus et al., 2007; Hischier et al., 2007; Dones et al., 2007) was used to complete the background inventory, mainly involving chemical and energy-based requirements of the system. Detailed information on secondary data sources used in WF calculations is included in Table 4.6.

4.2.3 Impact assessment

The characterisation factors reported by the ReCiPe Midpoint (H) 1.12 method were considered (Goedkoop et al., 2013a), and the following impact categories were assessed, identified as the most representative ones of the dairy production chain (Castanheira et al., 2010; González-García et al., 2013b; Yan et al., 2011): CC, TA, FE, ME, WD and FD. SimaPro v8.2 software was used for the computational implementation of the inventories (Goedkoop et al., 2013b). In contrast, no land-use changed was considered in this study. Indeed, the IDF guidelines (IDF, 2010) stated that when land use changed more than 20 years before the assessment (on or after 1 January 1990), emissions from land use change could be assumed to have occurred earlier. Since the dairy system under assessment was founded in 1975 (consolidating its activity 10 years later), it was considered that all the agricultural area related to milk production at the farm was cultivated before 1990, and therefore no land-use change should be considered. The same consideration was taken for the animal feed purchased outside the farm facilities (assuming that the agricultural land was also cultivated before 1990).

Similarly, soil carbon sequestration was not included, also in accordance with the IDF guidelines (IDF, 2010): no consideration of carbon abatement is recommended mainly due to the lack of reliable scientific data at the world level. Likewise, other published works stated that arable land cultivated for more than 20 years could be considered in equilibrium in terms of changes in soil organic matter (carbon) (Daneshi et al., 2014). Finally, the biogenic CO_2

delivered during biomass growth by photosynthesis was quantified, but not calculated since its characterisation factor is zero (Goedkoop et al, 2013a).

The three WF components were estimated and evaluated individually: green WF (evapotranspiration of rainwater from the field), blue WF (water sourced from surface or groundwater resources) and grey WF (freshwater volume required to assimilate pollution loads).

4.2.4 LCA results

The global LCA results (in absolute terms) are reported in Table 4.7 for each impact category, while the relative influence of the different activities and processes involved throughout the life cycle of the cow dairy system are displayed in Figure 4.3. Similar to Chapter 3 above, such processes were grouped into different contributing factors: on-farm emissions, animal feeding, surplus cereals, transport activities, fossil fuels, electricity use, waste treatment and avoided processes.

Table 4.7. LCA results per FU (1 kg of FPMC at farm gate) of the global system.

Impact category	Units	Global LCA results
CC	kg CO_2 eq	1.32
TA	g SO_2 eq	27.8
FE	mg P eq	37.1
ME	g N eq	2.85
WD	m^3	0.52
FD	kg oil eq	41.5

On-farm emissions comprise those related to enteric fermentation during herd confinement, manure management and application to agricultural soils. All the environmental burdens derived from the production of the cattle feed were considered in **animal feeding**; not only those cereals cultivated on-farm (sorghum and barley straw) but also the imported ingredients. Similarly, **surplus cereals** factor refers to the environmental impacts associated with the cultivation of those cereals (barley grain and oat silage) that are exported from the farm. **Transport activities** include all the environmental burdens derived from both the supply of inputs and the transfer of waste to management centres. **Fossil fuels** factor represents the impacts from the use of diesel and lubricant oil for the operation and maintenance of the machinery at the farm, respectively. **Electricity use** includes the environmental burdens related to the production of the energy consumed for the farm activities, which is taken from the national grid. **Waste treatment** considers the emissions and discharges from the treatment of the residues (wastewater and solid waste) generated at the farm. Finally, **avoided processes** comprise the environmental credits related to the non-use of mineral fertilisers for cereal cultivation. As aforementioned, the use of the manure excreted by the herd as organic fertiliser avoids the production and subsequent application of a certain amount of N and, therefore, the production of nitrogen-based mineral fertilisers, thus contributing favourably to the environment. The manufacture of cleaning products was responsible for a negligible effect (below 1%) in all categories, so that it was not included in the results.

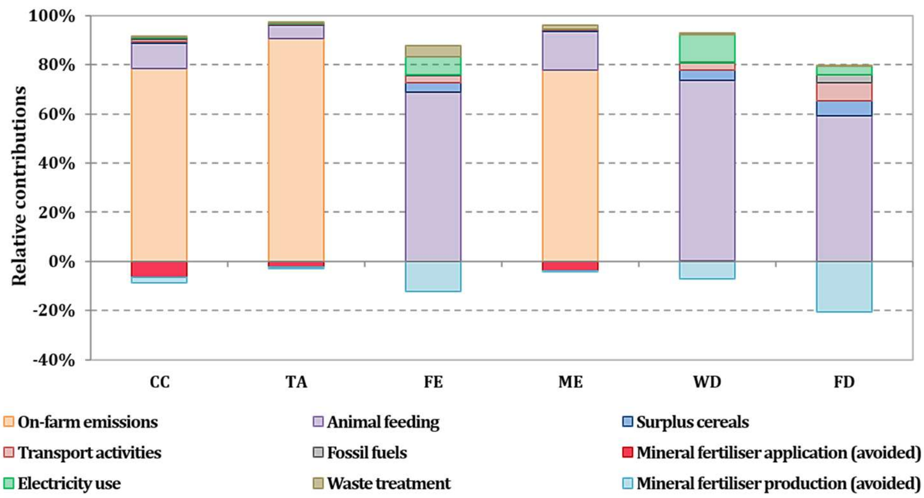

Figure 4.3. Relative contributions from the different processes of the global system. Note: positive values (above x-axis) represent environmental impacts while negative results (below x-axis) make reference to environmental credits.

Two main critical contributing factors can be identified: animal feed production and on-farm emissions. While other processes involving the cultivation of surplus cereals, electricity use and transportation also have environmental impact, their relevance is limited compared to the contributions of the ones aforementioned. Moreover, minor impacts can be observed due to the consumption of fossil fuels and the final treatment of waste.

On-farm emissions play a critical role in CC, TA and ME (with values above 75%); direct CH_4 and N_2O emissions amount to 85% and 15% of CC, respectively. Similarly, indirect N emissions in terms of NH_3 and NO_3^- also show a substantial effect: NH_3 volatilisation accounts for more than 90% of TA while NO_3^- leaching contributes to 47% of ME. These results highlight the role of enteric fermentation and manure management as relevant hotspots in the environmental profile of the system.

Animal feed is critical in many of the impact categories assessed, such as FE, WD and FD, with values ranging from 59% to 85%. In response to these results, a specific evaluation was carried out for feed production with the aim of identifying the most critical steps in the production of the cattle feed. Figure 4.4 shows the environmental contributions of the production of the different ingredients to the overall animal feeding process.

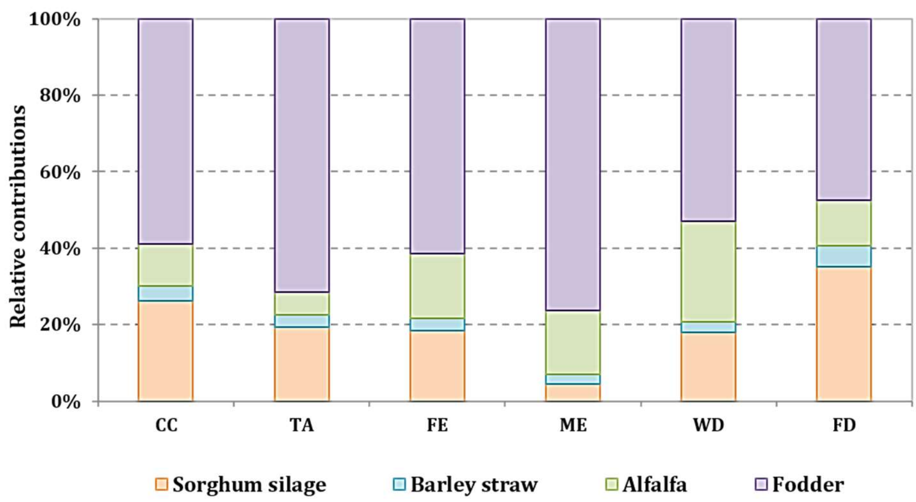

Figure 4.4. Relative contributions of the production of the different ingredients involved in the animal diets at farm.

According to the results shown in the figure, the production of the fodder is the main contributor to the environmental burdens in all categories. Fodder is based on maize, wheat bran and rapeseed meal, the latter having the greatest impact. Emissions from both the application of fertilisers and the combustion of diesel fuel used in agricultural machinery are mainly responsible for the loads associated with the rapeseed meal cultivation. Among the other ingredients in animal feed, the production of alfalfa and sorghum silage is also responsible for relevant contributions: while the drying and irrigation stages imply the greatest impacts in the former case, emissions from fertilisation activities are responsible for the greatest negative effects for most of the impact categories in the latter.

4.2.5 WF results

Table 4.8 shows the WF results for the global system, as well as disaggregated rates for the two main life cycle stages: animal feed production and dairy farm. The first comprises all the activities involved in the production of both on- and off-farm ingredients (cereals and fodder mixtures), while the latter integrates all other activities, including transport from and to the farm. According to the results, feed/fodder production stands as the most critical processes (hotspots) in the WF impacts, with contributions of about 99%, while dairy farm activities have less influence (about 1%). This would be directly associated with increased water requirements during the cultivation of different cereals, especially in the case of sorghum silage.

Table 4.8. WF results per FU (1 kg of FPMC at farm gate) from the global system.

Life cycle stage	WF results (m^3)				Relative contribution (%)
	Green WF	**Blue WF**	**Grey WF**	**Total WF**	
Animal feed production	7.589	0.583	0.442	8.614	99.99
Dairy farm activities	0.000	$1.2 \cdot 10^{-3}$	0.000	$1.2 \cdot 10^{-3}$	0.01
Global cow dairy system	7.589	0.584	0.442	8.615	100
Relative contribution (%)	88.09	6.78	5.13	100	

The environmental impacts related to green, blue and grey WF are also displayed individually (Table 4.8). Green WF is the main responsible for the outstanding influence of the production of animal feed, accounting for 88% of the contributions at this stage, mainly due to the effect of rainwater evapotranspiration from agricultural soils. This impact is especially relevant concerning sorghum cultivation, with about 80% of the total green WF impacts. Blue and grey WFs have a minor influence, with contributions of 7% and 5%, respectively. When focusing on the dairy farm itself, the entire (100%) WF impacts correspond to the blue contribution. It can be related to the critical role of the transportation of the different inputs to the farm facilities. Consequently, both green and grey WFs have negligible impact on the environmental results of the dairy farm stage.

4.2.6 Discussion

- **Comparative assessment: WD vs. WF**

Both WD and WF were evaluated within the framework of the present chapter, including the environmental evaluation of a dairy farm. In this regard, the following results were obtained: 8.65 m^3 (Table 4.8) and 0.52 m^3 (Table 4.7) for WF and WD, respectively. Based on these values, a much higher value can be reported for the WF indicator compared to the WD results. The underlying reason lies in the different approaches considered in the definition of both concepts. WD focuses mainly on surface or groundwater resources, causing water scarcity due to water evaporation and/or its use as an input in production processes. Therefore, WD could be directly related to the blue WF concept.

This conclusion is in line with the results presented in this study: when WD (0.52 m^3) and blue WF (0.58 m^3) results are compared pairwise, similar rates can be found. However, the WF results also include the environmental impacts associated with the water from precipitation that is evaporated or uptaken by plants (green WF), as well as the fresh water required to meet specific water quality standards (grey WF). The sum of these additional indicators provides a comprehensive picture of water use by delineating a broader scope of the study (Hoekstra et al., 2011). Consequently, the WF results always lay above the values reported by the WD indicator. This effect becomes particularly noticeable in productive processes where agricultural, horticultural and forestry inputs are involved due to the corresponding major contribution of the green WF (Hoekstra et al., 2011).

- **Comparison with related LCA studies**

As aforementioned, both on-farm emissions and animal feed production stood as the environmental hotspots of the target system (Figure 4.3). Moreover, Figure 4.4 shows that fodder production is the most significant contributor to the environmental impact of feed production, followed by alfalfa and sorghum silage cultivation. These results are in line with those of other similar studies in the dairy sector. As an example, González-García et al. (2013a,b) analysed different representative Portuguese schemes on dairy products and identified on-farm emissions from enteric fermentation manure management as the main drivers of environmental impact, along with burdens related to feed (especially fodder) production. Analogous results were reported by Daneshi et al. (2014), Meneses et al. (2012) and Thomassen et al. (2008) for dairy production systems in Iran, Spain and The Netherlands, respectively.

In addition, concerns about climate change were found to be assessed primarily in most of the studies focusing on dairy products in the literature, followed by acidification and eutrophication potentials (Yan et al., 2011). This may be due to the remarkable influence of on-farm emissions – reported from a hotspot on milk production – on this indicator in terms of CH_4 and N_2O delivered to the atmosphere. In this regard, Table 4.9 shows a brief description of the studies collected from the literature with special attention to CC (otherwise CF) for comparative analysis. Being aware that differences in methodological options may give uncertainty to the analysis, only LCA studies that shared similar assumptions of FU and allocation rules were considered for comparison. Additionally, the characterisation values were recalculated per kg of FPCM at farm gate, where necessary.

In contrast, lower differences in acidification potential between these studies were identified (Table 4.9) by using different characterisation factors despite similar NH_3 emissions as the main contributor: a ratio of 2.45 kg SO_2/kg NH_3 was used according to the ReCiPe method, while 1.60 kg SO_2/kg

NH_3 was applied by Castanheira et al. (2010) according to Huijbregts (1999), as an example. However, discrepancies became more evident (around 50%) when Thomassen et al. (2008, 2009) apply the factor proposed by Heijungs et al. (1992) (1.88 kg SO_2/kg NH_3) combined to lower NH_3 emission rates; these authors apply the characterisation data collected from previous studies in the literature, rather than the estimations proposed by the IPCC guidelines. However, the quantitative validation of these estimations is undoubtedly difficult due to the lack of detailed inventory information provided by the authors.

Table 4.9. Characteristics of the LCA studies considered for comparison and related results involving CC and TA impacts per FU (1 kg of FPCM at farm gate).

Study	Country	Allocation approach	CC (kg CO_2 eq)	TA (g SO_2 eq)
This study	Spain	Economic	1.32	27.8
Casey and Holden (2005)	Iran	Economic	1.30	-
Thomassen et al. (2008)	The Netherlands	Economic	1.40	9.50
Thomassen et al. (2008)	The Netherlands	Economic	1.36	11.2
Castanheira et al. (2010)	Portugal	Economic	1.02	20.4
González-García et al. (2013a)	Portugal	Mass	0.74	19.0
Gollnow et al. (2014)	Australia	Bio-physical	1.11	-

It was therefore not possible to avoid the variability inherent in the application of alternative characterisation factors in the absence of similar studies using the ReCiPe method in the calculations, with a critical effect on the eutrophication results. ReCiPe divides the eutrophication potential into two impact categories: FE and ME, expressed as equivalent emissions of P and N, respectively. Conversely, characterisation methods applied in previous studies (mainly CML 2001 method, Guinée et al., 2001) combine all eutrophication impacts in only one impact category, named as eutrophication potential and expressed as equivalent emissions of PO_4^{-3} or NO_3^-, depending on the method. As no conversion factors are available to inter-relate alternative methods under reliable conditions, a comparative analysis in terms of eutrophication results has remained outside the scope of the study.

However, leaving aside alternative methodological choices to focus attention on substances and emission rates, it could be concluded that the results from our study are consistent with related LCA studies available in the literature. It is for this reason that they were considered suitable as basis for the formulation of actions to upgrade the environmental profile of cow dairy production systems, at least on acidification and climate change mitigation.

- **Improvement actions**

Since animal feed production was found to contribute critically to environmental impacts, several authors have tried to find alternatives to current diets of the dairy sector that improves their environmental profiles (Hospido et al., 2003; Gollnow et al., 2014; Iribarren et al., 2011; Pirlo et al., 2014). Hospido et al (2003) pointed to the need for a study of various combinations of food rations to find the most sustainable option capable of producing a lower environmental impact while maintaining protein and energy supplement requirements in the final mixture. According to these authors, the use of a more significant proportion of maize silage instead of grass silage could be an appropriate alternative to reduce the environmental burdens associated with feed production processes. Similarly, Iribarren et al. (2011) stated that, in most cases, high efficiency scores are associated with farms where feed consists primarily of maize silage and concentrate rather than grass and alfalfa silage. In addition, Salcedo (2004) highlighted the advantages of using grass silage as a protein source over other ingredients in animal diets with a similar protein content, such as alfalfa or clover silage.

With this in mind, alternative scenarios were considered in this chapter based on the use of different protein sources in animal diets. Thus, since alfalfa was identified as one of the major contributors to the impacts associated with feed requirements at farm, two additional scenarios focused on the substitution of alfalfa (Base Scenario – BS) by maize silage (Scenario A – SA) and grass silage (Scenario B – SB) were evaluated. Table 4.10 provides a brief description of the crude protein content and proportions of ingredients (protein sources) delivered to the herd at farm for each scenario.

Table 4.10. Inventory data considered involving the protein content and supply rates associated with each scenario (annual basis).

Scenario	Protein source	Protein content (%)	Amount (t/yr)	Protein supply (t/yr)
BS	Alfalfa	19.5[a]	365	71.2
SA	Maize silage	8.00[b]	890	71.2
SB	Grass silage	15.0[b]	475	71.2

[a] Delgado et al. (2005); [b] González-García et al. (2013b).

According to the results (Figure 4.5), the consideration of maize silage (SA) as protein source instead of alfalfa (BS) should improve the environmental profile in most impact categories (except for TA and ME) with reductions up to 18% (in term of WD) relative to the base case. Conversely, the environmental profile should be worse in most impact categories when grass silage (SC) is considered as a protein source. Thus, increases of up to 8% in environmental burdens were registered in CC, TA, FE, ME and FD compared to the base case (BS).

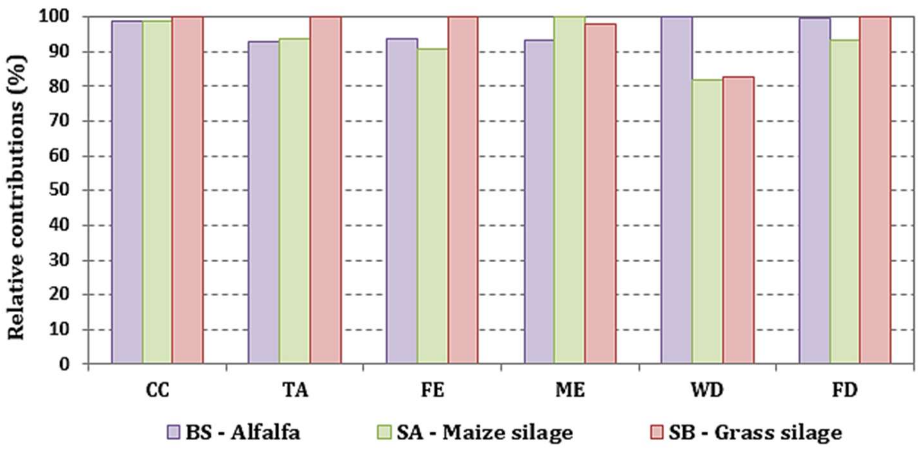

Figure 4.5. Comparative environmental results (per FU) for the different scenarios considering alternative protein source in animal diets.

These results corroborate the positive effect on the environment due to the use of maize silage as the main source of protein in herd diets, in accordance with published reports. However, the benefits of using grass silage instead of alfalfa suggested by Salcedo (2004) could not be confirmed in the present study. The expected advantages of grass silage would be attributed to a more environmentally friendly farming practice as well as the lower nitrogen losses from manure. In this comparative analysis, only the environmental burdens related to agricultural practices were taken into consideration, so the influence of the manure composition was not included. Nevertheless, the final results in which positive impacts from an environmental perspective could be found should be carefully interpreted, since climatic conditions in the area under study could prevent the viability of grass silage cultivation. Thus, the import of off-farm grass silage could involve additional burdens related to transport activities, reducing the possible environmental credits initially identified. However, future studies could be conducted with the aim of delving deeper these issues.

4.3 Conclusions

Milk is the primary product of the dairy sector and the basis for the production of other dairy products. Moreover, it is considered to be a prescription for good health and an essential ingredient for a nutrient-rich human diet. However, the high demand for milk has fostered changes in the production chain, making the dairy sector a recognised source of environmental impact, taking the leader position in overall emissions of the livestock sector. In this context, several LCA studies about dairy products have been published to date, but fewer have focused on Spanish cow milk production. In this study, a cradle-to-farm gate assessment was carried out to evaluate the environmental profile of a cow dairy system located in NE Spain.

According to the results, it could be concluded that feed and fodder production was the major responsible for the WF impacts of the global system, mainly due to the environmental burdens related to the cultivation activities. These results highlight the critical role of the indirect water consumption and pollution derived from the production of animal feed, rather than the minor influence of the direct water requirements. Moreover, when a pairwise comparison between WD and WF indicators was carried out, outstanding differences were found. However, these discrepancies are directly related to unequal assessment approaches: only blue water was considered by the WD indicator, while the overall scope of the WF concept additionally includes green and grey water values. However, data limitations hamper data collection procedures, so major improvement actions should be mainly focused on amending this weakness, to move to a more accurate and reliable WF methodology.

Moreover, animal feed production together with on-farm emissions were also found as the main environmental hotspots throughout the entire life cycle of the system, in line with other LCA studies involving dairy systems. Consequently, an additional assessment was carried out with the aim of reducing the environmental impacts related to alfalfa cultivation as one of the main contributors to environmental burdens of feed production. In this regard, two alternative ingredients instead of alfalfa were proposed as protein source in animal diets: maize silage and grass silage. The results obtained were compared with the base case, and only the potential environmental credits of maize silage usage were verified. On the contrary, more detailed analysis would be needed to confirm the advantages of prioritising grass silage ahead of alfalfa as protein source in animal feed.

The aforementioned results are intended to serve as a basis to define the pattern of environmental performance of the cow milk sector in the region compared to other similar studies in alternative locations, as well as promote greater advances on the sustainable management of related dairy products.

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CHAPTER 5. ENVIRONMENTAL ASSESSMENT OF LIVESTOCK HUSBANDRY: PORK SECTOR

Summary

Pork production plays a foremost and representative role in the Spanish food sector. However, beyond its economic benefits, conventional practices in the pork industry also involve a number of environmental impacts that need to be addressed. In this context, the environmental performance of pork production was evaluated in this chapter from a LCA perspective, in two different locations: Galicia and Catalonia. While a cradle-to-farm gate approach was followed in the former, a more comprehensive study involving slaughtering and cutting stages was developed in the latter case study. In both cases, five common impact categories (CC, TA, FE, ME, FD) were simultaneously addressed and special attention was also paid to energy efficiency and water use in Galicia and Catalonia, respectively.

On-farm emissions were proved to have a relevant effect on the global results, directly attributed to related emissions from the management of manure for fertilisation purposes. However, the outcomes of the analysis showed that animal feed (fodder) production was the main contributor to the environmental impacts in both case studies, also in agreement with published data in similar works in the literature. This would be mainly due to the burdens associated with the cultivation of cereal crops that predominantly make up the composition of the fodder, which is also responsible for major contributions to water footprint results throughout the whole supply chain in Catalonia. In this regard, the energy balance regarding Galician pork products was found to be in line with their analogous in other countries, as well as alternative protein sources.

Finally, due to its relevance on the results, strategies to mitigate the impacts of fodder production were proposed within the framework of this chapter, giving priority to the improvement of resources efficiency in alternative fodder compositions.

CHAPTER 5 - INDEX

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5.1 INTRODUCTION TO LIVESTOCK HUSBANDRY: PORK SECTOR

As noted in Chapter 4, the main challenge for the livestock sector has been to meet the growing demand for food products while reducing the related environmental impacts (Gerber et al., 2013; Notarnicola et al., 2012).

As a primary dietary source of protein and micronutrients, meat can be considered as an important element of the human diet (Davis et al., 2010; González-García et al., 2015). Indeed, its demand is growing steadily worldwide, and specifically in Europe, meat consumption has increased by 63% in the last 40 years (Ciolos et al., 2012; Davis et al., 2010). Among the main European meat varieties (beef, pork, chicken, poultry and sheep), pork is the most widely consumed, with an average annual consumption rate of 31.8 kg per capita in relation to the average consumption (64.7 kg of meat per capita) of all types of meat in Europe (Eurostat, 2013). In this context, Spain ranks second (after Germany) in the European pork sector, with 13% of total production (FAOSTAT, 2014). Moreover, in 2016 it exported 2,044,170 tons of pork products, being Catalonia responsible for 56% (1,138,129 tons) of the total volume exported (Observatori del Porcí, 2016), followed by other autonomous communities such as Aragón, Castilla La Mancha and Murcia. In fact, Catalan pork production holds around 40% of the national pork industry and 50% of pork processing activities (Observatori del Porcí, 2016). On the other hand, despite its leading role in the production of dairy products, Galicia occupies a secondary position in the Spanish meat market (Fernández et al., 2013); moreover, contrary to the meat production structure in the country, pork products do not predominate, but instead holds the third position, with around 23% of the total livestock production in Galicia (Fernández et al., 2013; Sineiro-García and Lorenzana-Fernández, 2007). However, the annual data recently published by the Consellería do Medio Rural (2017) show a significant recovery of the pork sector in recent years: the number of farm facilities and livestock heads grew throughout the region and further efforts were focused on the consolidation of fattening farms and the closed-loop production system, in line with the recent trend in the sector.

However, in parallel with its socio-economic relevance, pork production also demands large requirements of natural resources (energy and water) and generates significant waste flows at both national (Spain) and international (Europe) levels (Gerber et al., 2013; PRTR, 2014). This underlines the need to assess the environmental impacts that influence the pork supply chain: improvements in the pork products can provide competitive advantages in the global market, not only from an economic perspective, but also taking into account their environmental sustainability (Groen et al., 2016; Nguyen et al., 2010; Philippe and Nicks, 2014; Philippe et al., 2011).

In this regard, the environmental impacts of different pork production systems have been evaluated in the literature, not only in the entire European region but also outside its boundaries (Dourmad et al., 2014; González-García et al., 2015; Lehmann et al., 2011; Pelletier et al., 2010; Reckmann et al., 2012, 2013; Wiedemann et al., 2010). Although different assessment methods are available to date, the principles of the LCA methodology have been commonly followed as the basis for environmental analysis (McAuliffe, 2016). In general, the available reports identified feed production as the main contributor to the impact categories assessed, followed by pig housing, mainly due to GHG emissions coming from manure management practices. Therefore, both feed use and manure valorisation were considered key areas for improving the environmental performance of the pork supply chain, pending the definition of a common methodological framework on the field (McAuliffe, 2016; de Vries and de Boer, 2010).

With this in mind, the main goal of this Chapter 5 was to evaluate the environmental burdens associated with the pork sector in Spain from a LCA perspective. To do so, two separated pork systems representative of the sector in Galicia (Northwest Spain) and Catalonia (Northeast Spain) were assessed in detail. While special attention was paid to the farming stage in the Galician case study, a more comprehensive approach was evaluated in Catalonia, in line with its well-established role within the Spanish pork sector. Moreover, additional indicators in terms of energy efficiency (Galicia) and water use (Catalonia) were also included in the study, given their primary interest for stakeholders of the pork sector.

5.2 PORK SECTOR IN SPAIN: A CASE STUDY IN GALICIA

5.2.1 Goal and scope definition

This section focuses on the identification and evaluation of the environmental burdens associated with pork production in Galicia (NW Spain – Figure 5.1) through a LCA approach (ISO 14040, 2006; ISO 14044, 2006). To this aim, a cradle-to-farm gate perspective was followed in line with the main outcomes of similar LCA studies previously published in the literature: although most of them comprised the slaughterhouse stage within the system boundaries, the hotspots were recorded at earlier stages in all cases (González-García et al., 2015; Nguyen et al., 2011; Reckmann et al., 2013). Therefore, further stages involving slaughtering, packaging, distribution and meat consumption were excluded from the assessment.

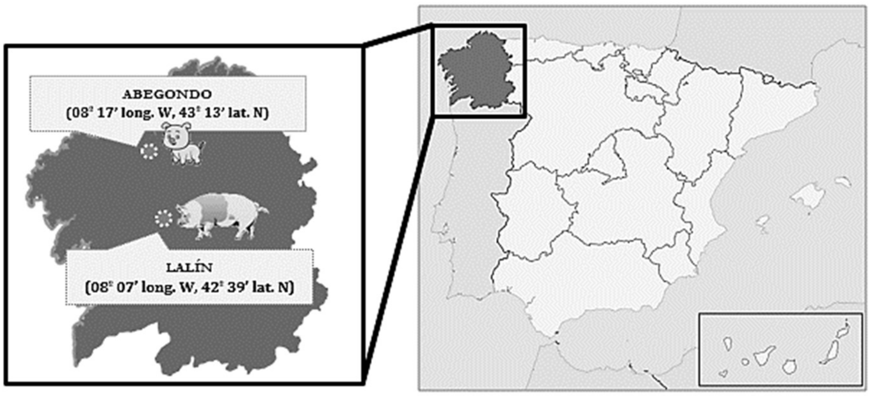

Figure 5.1. Location of the pork farming systen in Galicia (NW Spain).

Moreover, LCI data was also used to calculate the edible protein energy return on investment: ep-EROI. It is a dimensionless ratio that allows estimating the energy efficiency of a productive system, since it studies the proportionality between the energy requirements for food production and the energy that the product provides to human consumers (Vázquez-Rowe et al., 2014).

- **Functional unit**

Different choices for the selection of the FU for pork production systems can be found in the literature: kg live weight (Dourmad et al., 2014; Pelletier et al., 2010), kg carcass weight (Nguyen et al., 2010, 2011; Reckmann et al., 2013; Wiedemann et al., 2010) or kg edible protein (de Vries and de Boer, 2010). However, to sum up, mass-based FUs have been typically selected for LCA studies in this sector. Accordingly, 100 kg live weight of pork at farm gate was defined as FU for evaluation purposes, which corresponds to the average live weight of pigs before slaughterhouse in the system assessed.

- **System description**

Figure 5.2 shows the main processes involved in the system studied in this section. Thus, all activities related to crop cultivation and fodder production, distribution to the farms, gestation and lactation stages, rearing of piglets until weaning and post-weaning, as well as fattening pigs were encompassed in the study. Moreover, emissions to soil, air and water from the entire system were also taken into consideration.

The production system can be divided into two main subsystems (Figure 5.2): (S1) off-farm fodder production and (S2) pig farming. In addition, the former consists of three main stages: (S1.1) sows fodder production, (S1.2) weaning fodder production and (S1.3) fattening fodder production; likely, the farming subsystem (S2) consists of both (S2.1) a weaning farm and (S2.2) a fattening farm.

Fodder production

Off-farm fodder production (S1) comprises all the activities carried out during crop cultivation, production of additives and other fodder ingredients, transportation from cropland and final animal feed production. Thus, inputs (seeds, agrochemicals and fossil fuels) production and use together with their derived emissions are included in S1.

In addition, emissions and resources associated with energy use and water requirements directly related to the final production of fodder in the factory are also considered in this subsystem.

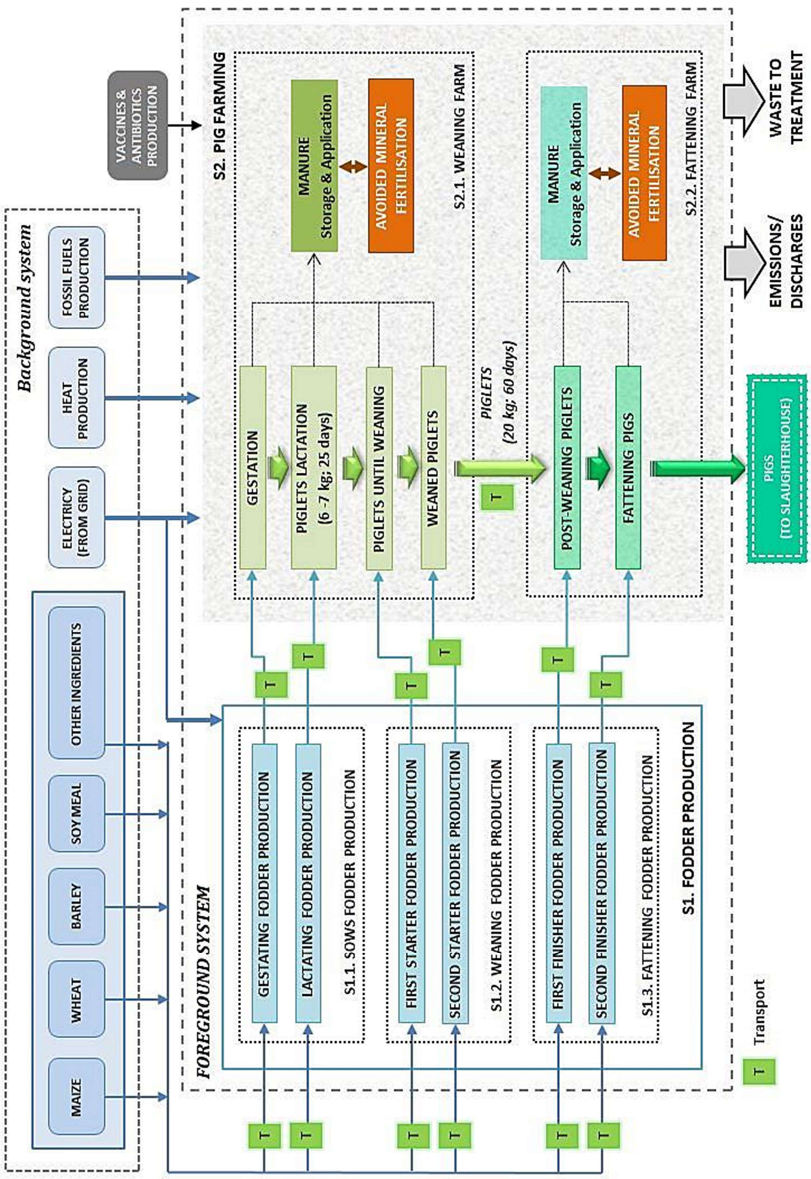

Figure 5.2. Scheme of the system boundaries. Key: Brown boxes include avoided impacts due to both mineral fertiliser production and further application on soils. Grey box represents excluded processes.

A total of six different fodder mixtures (formulated on protein content) must be supplied to pigs in accordance with their physiological stage at farm level: gestating fodder, lactating fodder, first starter fodder, second starter fodder, first finisher fodder and second finisher fodder. Detailed information on related ingredients and compositions is reported in Table 5.1.

Table 5.1. Main ingredients and composition of the different fodder mixtures used for animal feeding: first starter fodder (FSF); second starter fodder (SSF); gestating fodder (GF); lactating fodder (LF); first finisher fodder (FFF); second finisher fodder (SFF).

	Weaning farm (S2.1)				Fattening farm (S2.2)	
	FSF	**SSF**	**GF**	**LF**	**FFF**	**SFF**
Ingredients (%)						
Wheat	14.5	18.0	15.5	9.30	30.0	35.0
Maize	22.0	37.0	-	-	27.7	23.7
Barley	-	6.00	57.2	53.7	18.0	15.0
Soybeans	14.0	-	-	-	-	-
Soy meal	10.0	28.0	2.60	21.3	17.1	11.4
Soybean oil	3.80	-	-	-	-	-
Milk powder	12.0	-	-	-	-	-
Rice	5.00	-	-	-	-	-
Animal fats	2.50	5.50	2.60	1.70	2.10	2.50
Sunflower	-	-	16.1	3.80	-	-
Peas	-	-	4.10	4.90	-	-
Beet molasses	3.00	1.60	-	-	-	-
Fava beans	2.00	-	-	-	-	-
Fish meal	2.00	-	-	-	0.40	-
Rapeseed	-	-	-	-	2.00	7.00
DDGS[a]	-	-	-	-	-	3.00
Others	9.20	3.90	1.94	5.30	3.10	2.40
Composition (%)						
Crude protein	20.7	17.7	14.2	17.2	16.0	15.5
Crude cellulose	2.10	4.50	9.40	6.70	-	-
Crude fibre	-	-	-	-	3.60	3.70
Crude fat	8.80	7.10	4.00	7.20	4.50	4.80

[a] DDGS = Distiller's dried grains with solubles.

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Pig farming – Weaning farm

A farm located in Abegondo (Figure 5.1) was selected to evaluate the piglets breeding stage (S2.1). With an area of about 21 ha, this farm has a capacity of 4895 sows and 18 boards for the production of piglets of up to 20 kg of live weight. It operates in cycles to produce 10,595 piglets per breeding cycle, with an average of 9 – 10 cycles per year.

After initial insemination, sows remain housed in the gestation section for approximately 110 days and then transferred to the farrowing section, 5 days before the expected farrowing date. Sows stay during the lactation period until the piglets are ready for weaning, i.e. around 25 days of age and between 6 and 7 kg live weight (Figure 5.2). The weaning pigs are then reared until they reach the appropriate weight to leave the weaning farm for the feedlot (S2.2), approximately 60 days later with a piglet weight of 20 kg live weight. Due to the young age of the piglets, optimal ambient conditions must be provided; thus, both electrical and thermal energy consumptions arise as key factors in the weaning stage (S2.1). As far as feeding is concerned, piglets are fed for about 35 days with ratios of 105 g of first starter fodder/(piglet·day) and 415 g of second starter fodder/(piglet·day).

Pig farming – Fattening farm

After the first stage, the weaned piglets are transported (115 km away) to the fattening farm (S2.2) located in Lalín (Figure 5.1) with an area of approximately 2 ha. This second farm also operates under cycles, completing one fattening stage during each cycle. An average of 2.3 cycles is developed per year with an approximate duration of 4.5 months per cycle. Around 2,000 weaned piglets enter the farm per cycle to produce 1,960 pigs, resulting in a mortality rate of about 2%.

Weaned piglets are initially fed with a ratio of around 190 g of first finisher fodder/(weaned piglet·day). The second finisher fodder is then supplied to the pigs until the end of the cycle (100 kg live weight), with a ratio of 1.50 kg of second finisher fodder/(pig·day).

Pig farming – Common practices

Water troughs are used for water supply, which comes from an extraction well. Wastewater from both troughs and cleaning activities is sent to tanks where it is mixed with manure. This mixture is directly valorised as organic fertiliser due to its nutrient content. The agricultural land used for the application of manure from the weaning (S2.1) and fattening (S2.2) farms are 90 ha and 82.5 ha, respectively.

- **Allocation rules**

After the fattening stage, pork meat was assumed to be the main product in the system evaluated in this section, with significant commercial value as a raw material for the subsequent production of alternative pork products. However, the manure excreted by pigs (in combination with wastewater flows) can be also considered as a valuable co-product. In line with the above, it can be valorised as organic fertiliser on agricultural soils, with the aim of reducing dependence on synthetic mineral fertilisers to some extent. In this way, not only emissions and discharges derived from manure management (including storage and application) are included within the system boundaries, but also the reduction of environmental burdens due to the avoided mineral fertilisers production and application (system expansion approach).

Similar to the main assumptions in Chapter 4, the MFE for N derived from an organic source was assumed to be 75% (de Vries et al., 2011, 2012; Sommer and Birkmose, 2007). Regarding P, MFE values close to 100% can be found in the literature (Sommer et al., 2008); however, Dalgaard et al. (2006) stated that the substitution rate of P in manure should be adjusted to 97%, taking into account the potential P leaching from crops cultivation at farms.

Finally, it should be noted that allocation rules should also be defined in background processes responsible for the production of the different fodder ingredients (mainly crops). In this regard, both mass and economic allocation was considered for the distribution of the environmental impacts according to the processes selected for this purpose (see epigraph 5.2.2 involving LCI information).

5.2.2 LCI analysis

The sources used to obtain the inventory data, together with the main assumptions of the study, are described below.

Fodder production

Feed supply for each pig category was analysed and inventoried in detail. Thus, primary data on fodder production and composition was provided by a Galician factory for the 2013/2014 season (Table 5.1). Moreover, information on the transport of the ingredients from their origins was also reported by the factory. Most of the ingredients (wheat, barley and other minor components) were produced in Europe, mainly England, France and Spain, which implied assuming average distances of 1,200 km, 1,500 km and 650 km from the croplands to the fodder company. Similarly, the average transport distances between the fodder company and the pig farms were estimated at 82 km for the weaning farm (Abegondo) and 100 km for the fattening farm (Lalín). Other valuable ingredients were also imported, but beyond European borders: maize was transported from Argentina, USA and Ukraine; soymeal was shipped from Argentina, USA and Brazil. In this case, the distances between ports were calculated with a web distance calculator[1]. Lorries for road transport were considered, while bulk carriers were assumed to be used for sea transport.

However, the inventory of fodder production also included background data on crop cultivation, agrochemicals use and energy consumption. This additional information was obtained mainly from the ecoinvent® database (Althaus et al., 2007; Nemecek and Käggi, 2007; Wernet et al., 2016), except for soy cultivation, whose background data was taken and adapted from the LCA Food DK database (Nielsen et al., 2003).

Pig farming

Inventory data per FU (100 kg live weight of pork at farm gate) corresponding to the weaning and fattening farms are summarised in Tables 5.2 and 5.3, respectively. Primary inventory data for the entire production

[1] www.vesseldistance.com (accessed November 2014).

chain (fodder supply ratios, water use, energy consumption) for both farms were obtained through surveys and questionnaires fulfilled by growers, as well as through farm visits. It should be noted that metering devices were used to record electricity requirements for the weaning farm in real time, while information on electricity consumption at the fattening farm was obtained from devices that measure free energy. Similar to fodder production, secondary data from the ecoinvent® database was also used to complete the background inventory, including energy generation, agrochemicals manufacture and combustion emissions from transportation activities (Althaus et al., 2007; Dones et al., 2007; Spielmann et al., 2007; Wernet et al., 2016).

Direct emissions to the environment from farm activities were also estimated. Thus, the Tier 1 method proposed by the IPCC (2006) was applied to calculate CH_4 emissions to air from both enteric fermentation and manure storage. For this purpose, primary data on the number of heads for each category of livestock (sows, piglets and fattening pigs) collected from both farms was used. Based on this information, together with the default emission factors provided by the method, it was possible to determine CH_4 emissions of the global system to the atmosphere. Similarly, IPCC information (IPCC, 2006) was also considered to estimate direct (N_2O) and indirect (NH_3, NO_3^-) nitrogen-based emissions derived from manure management; however, the Tier 2 method was followed as the basis for calculations in this case. Information on heads and average mass of each category of livestock as well as the volume and density of manure was required. Moreover, a two-step process was assumed for manure management: (i) storage of pig manure on the farm premises and (ii) further confinement in either tanks or earthen ponds where manure can be removed periodically. Default emissions factors (such as EF_3 and EF_4, among others) throughout the calculations were selected based on this information. Nitrogen emissions from manure application in soils as organic fertiliser were also determined by means of Tier 2 method, taking into account the same primary data. In this line, inventory data from the production and use of avoided mineral fertilisers (urea, triple superphosphate and potassium sulphate) - due to the fertilising capacity of manure - were also considered, following again the IPCC guidelines (IPCC, 2006). Finally, PO_4^{-3} emissions to water were calculated in accordance with the conversion factor of 0.01 kg P-PO_4^{-3}/kg of applied P proposed by Rossier (1998).

Finally, the following information was considered for the energy balance in terms of ep-EROI (FAO, 2007; ODAFF, 2014): 57% yield of carcass pork per kg live weight, 11.2% of protein content per kg of carcass pork and an energy density of 16.73 MJ per kg of protein. Accordingly, an energy output of 1.07 MJ per kg live weight was assumed as the basis for the calculations.

Table 5.2. LCI data per FU (100 kg live weight of pork at farm gate) corresponding to the weaning farm (S2.1).

Inputs/Outputs	Amount	Units
Inputs from environment		
Water	123	L
Inputs from technosphere		
Animal feed		
First starter fodder	3.74	kg
Second starter fodder	14.6	kg
Gestating fodder	2.11	kg
Lactating fodder	1.25	kg
On-farm energy use		
Heat	13.0	kWh
Electricity - Illumination	1.16	kWh
Electricity - Ventilation	0.09	kWh
Outputs to technosphere		
Products and co-products		
Weaned piglets to fattening farm	20.0	kg
Pig manure[a]	370	kg
Avoided fertiliser production		
N from manure	1.77	kg
P from manure	0.30	kg

[a] Pig manure composition: 5.04 g N/kg manure; 0.83 g P/kg manure.

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Table 5.2 (cont.). LCI data per FU (100 kg live weight of pork at farm gate) corresponding to the weaning farm (S2.1).

Inputs/Outputs	Amount	Units
Outputs to environment		
Air emissions		
CH_4 – Enteric fermentation	150	g
CH_4 – Manure management	1.01	kg
N_2O	100	g
NH_3	2.23	kg
Water emissions		
NO_3^-	4.80	kg
PO_4^{-3}	9.40	g
Avoided fertiliser application		
N_2O	28.0	g
NH_3	250	g
NO_3^-	2.35	kg
PO_4^{-3}	9.10	g

Table 5.3. LCI data per FU (100 kg live weight of pork at farm gate) corresponding to the fatterning farm (S2.2).

Inputs/Outputs	Amount	Units
Inputs from environment		
Water	1236	L
Inputs from technosphere		
Animal feed		
First finisher fodder	25.5	kg
Second finisher fodder	199	kg
On-farm energy use		
Electricity	2.45	kWh
Fuel oil	0.32	kg
Chemicals		
Quicklime	38.3	g
Transport		
Lorry	1.64	t·km

Table 5.3 (cont.). LCI data per FU (100 kg live weight of pork at farm gate) corresponding to the fatterning farm (S2.2).

Inputs/Outputs	Amount	Units
Outputs to technosphere		
Products and co-products		
Slaughter pigs	100	kg
Pig manure[a]	903	kg
Avoided fertiliser production		
N from manure	4.33	kg
P from manure	0.73	kg
Outputs to environment		
Air emissions		
CH_4 – Enteric fermentation	555	g
CH_4 – Manure management	3.70	kg
N_2O	250	g
NH_3	5.43	kg
Water emissions		
NO_3^-	11.7	kg
PO_4^{-3}	23.0	g
Avoided fertiliser application		
N_2O	68.0	g
NH_3	530	g
NO_3^-	5.75	kg
PO_4^{-3}	22.0	g

[a] Pig manure composition: 5.04 g N/kg manure; 0.83 g P/kg manure.

5.2.3 Impact assessment

The following impact categories were assessed according to previous studies (Reckmann et al., 2012): CC, TA, FE and ME. The results were also expressed in terms of FD in order to identify the use of non-renewable resources throughout the life cycle system. Characterisation factors reported by the ReCipE Midpoint (H) 1.12 method were considered for all impact categories (Goedkoop et al., 2013a), and SimaPro v8.2 software was used for the computational implementation of the inventories (Goedkoop et al., 2013b).

Additionally, the results of ep-EROI were also evaluated in this section. The rationale behind this is that the combination of this ratio with the LCA methodology allows reducing uncertainties in the calculations (Vázquez-Rowe et al., 2014). The ep-EROI sheds light on the efficiency of food systems taking into account the energy that food provides to consumers and the energy requirements for food production (Pimentel and Pimentel, 2003; Tyedmers, 2004; Vázquez-Rowe et al., 2014). Thus, it was estimated by dividing the protein energy output of the pork obtained in the system and the energy inputs related to the rearing up to farm gate (pigs to the slaughterhouse). The embodied energy in pork (energy output) was calculated based of the protein content per edible portion per FU (100 kg live weight of pig at farm gate) using LCI data. In terms of energy inputs, the Cumulative Energy Demand (CED) indicator proposed by VDI-Richtlinien (1997) was selected to determine the energy requirements of the global system. The CED represents the primary energy used throughout the entire life cycle of a product, including the energy consumed in extraction, manufacturing and disposal activities (VDI-Richtlinien, 1997).

5.2.4 Results

- **ep-EROI results**

An ep-EROI value of 2.28% was obtained based on the global energy (renewable and non-renewable energy) requirements of the system, while an ep-EROI of 7.31% was registered only taking into account the consumption of non-renewable energy. However, it should be noted that only ep-EROI results based on non-renewable energy requirements could be compared with literature data, due to the absence of studies based on total energy consumption as an energy input. In this regard, Table 5.4 provides a summary of the available ep-EROI results reported in similar studies in the literature (Pimentel and Pimentel, 2003; Iribarren et al., 2010; Pelletier et al., 2010; Vázquez-Rowe et al., 2013).

Table 5.4. ep-EROI values (per kg live weight pork) in the present study in comparison with other studies based on the production of pork and other protein sources.

Study	Country	Protein source	Energy input (MJ)	ep-EROI (%)
This study	Spain	Pork	47.0[a]	2.28[b]
			14.6[a]	7.31[c]
Pimentel and Pimentel (2003)	USA	Pork	N.D.	7.10
Pelletier et al. (2010)	USA	Pork	11.9	8.96
Iribarren et al. (2010)	Spain	Mussels	N.D.	6.90
Vázquez-Rowe et al. (2013)	Spain	Patagonian grenadier	N.D.	10.4

[a] CED results; [b] Renewable and non-renewable energy requirements; [c] Non-renewable energy requirements; N.D.: no available data.

According to the results, the ep-EROI value (7.31%) is in line with those obtained for pork production in other countries (United States), ranging from 7.10% to 8.96% (Table 5.4). However, it is important to highlight the liability when comparing diverse studies conducted in different countries, because the discrepancies in the calculation assumptions (boneless meat and carcass meat as the basis for calculation) limit the comparability of the ep-EROI results. Consequently, data adjustments in terms of energy requirements and pork energy content are necessary for a reliable assessment. Regarding other protein sources within Spain, the literature suggests that livestock products, i.e. pork and milk, show also close ep-EROI results. Similar scores can be also found for aquaculture products such as mussels (6.90%), while slightly higher results are recorded for other fishing species such as Patagonian grenadier (10.4%) based on trawling techniques.

- **LCA results**

Table 5.5 shows the characterisation results for all impact categories for the entire system; the environmental impacts related to the weaning and fattening farms are also presented individually. It should be noted that the environmental burdens arising from fodder production (S1) were included in the total rates associated with each farm, as appropriate. This means that both sows fodder and weaning fodder were attributed to the weaning farm (S2.1), while fattening fodder was linked to the fattening farm (S2.2).

According to the results, fattening farm activities contributed to all impact categories (with contributions above 72%), so that the environmental burdens associated with this closure phase are decisive. These results are consistent with other LCA studies including pig rearing and may be explained by the fact that the period from weaning to slaughtering (fattening time) is longer than the piglets weaning period (Reckmann et al., 2012). Another explanation is related to the weight of the pigs throughout the different stages. The herd on fattening farm (post-weaning piglets and fattening pigs) has a higher weight, thus increasing both the need for concentrated feed and the amount of manure (Reckmann et al., 2012).

Table 5.5. Global LCA results per FU (100 kg live weight of pork at farm gate).

Impact category	Units	Weaning farm	Fattening farm	Global LCA results
CC	kg CO_2 eq	40.0	302	342
TA	kg SO_2 eq	5.09	13.5	18.6
FE	g P eq	1.50	18.0	19.5
ME	kg N eq	0.98	4.03	5.01
FD	kg oil eq	2.54	27.3	29.8

However, because severe differences were found in the environmental profile of both subsystems (weaning farm and fattening farm), an in-depth assessment of each farm was conducted separately. In this way, the most critical processes (hotspots) associated with each subsystem were identified and evaluated, as well as their impact on the environment. With this in mind, and following the same approach as in Chapter 4, these processes were grouped into several contributing factors: on-farm emissions, energy use, fodder production (including first starter, second starter, gestating and lactating fodder) and avoided processes (including avoided mineral fertiliser production and avoided mineral fertiliser application). Burdens from transportation of animal feeding were attributed to each type of fodder used.

Figure 5.3 displays the contributions to the five impact categories of each contributing process involved in the **weaning farm (S2.1)** stage. According to the results, two critical environmental factors can be distinguished: animal feed (fodder) production and on-farm emissions associated with farming activities. The contribution of on-farm emissions mainly includes farm-associated emissions (such as CH_4 emissions from enteric fermentation during herd confinement) as well as the manure management process. The environmental impacts related to fodder production (S1.1 and S1.2) were relevant in almost all impact categories (except for TA), ranging from 10% (ME) to 47% (FD) depending on the category. Similarly, on-farm emissions factor also resulted in important contributions in categories such as CC, FE and ME, with a particular influence on TA (about 87%). Conversely, it was argued that the application of manure excreted by piglets prevents the production of a certain amount of N and P and, therefore, the production of mineral fertilisers based on these compounds. In this sense, consideration of both avoided mineral fertilisers production and avoided mineral fertilisers application represents environmental credits in most categories, contributing to reductions of up to 43% (FE) of the impacts.

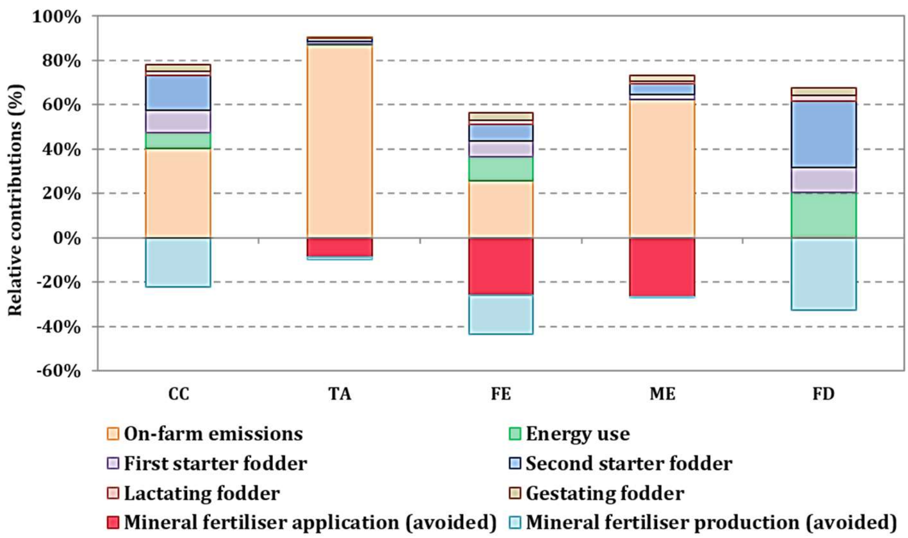

Figure 5.3. Relative contributions from the main processes involved in the weaning farm (S2.1) stage. Note: positive values (above x-axis) represent environmental impacts while negative results (below x-axis) make reference to environmental credits.

Figure 5.4 displays the contributions of the most relevant processes involved in the **fattening farm (S2.2)** stage. Accordingly, animal feed production and on-farm emissions were again identified as the main drivers of environmental impacts; however, in this case, feed production stands as the most critical process. Thus, the environmental burdens associated with fodder (S1.3) play an important role in all impact categories assessed, with contributions ranging from 10% (TA) to 83% (FD). As aforementioned, two types of fodder must be used for the fattening stage: first finisher fodder and second finisher fodder; however, the impact of the latter is clearly predominant over the former (with contributions of around 89% in all impact categories). This is mainly due to the higher ratio of second finisher fodder supplied (190 g of first finisher fodder/(weaned piglet·day) vs. 1.50 kg of second finisher fodder/(pig·day)) for the longest period of time (20 days of first finisher fodder compared to 3 months of second finisher fodder). Moreover, on-farm emissions significantly affect CC, TA and ME, with contributions of up to 81%. By contrast, both avoided mineral fertiliser production and application factors were responsible for environmental credits (up to 27%) in most impact categories, contributing to the reduction of environmental impacts.

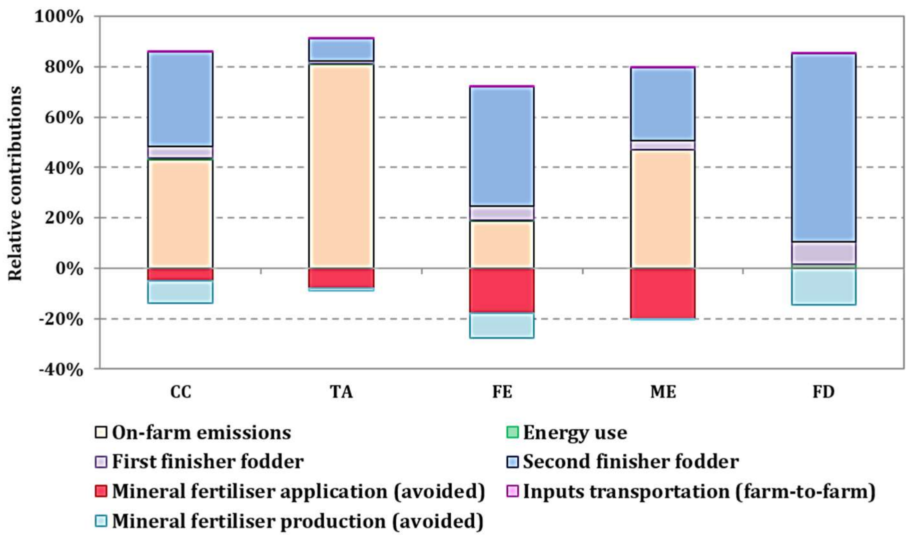

Figure 5.4. Relative contributions from the main processes involved in the fattening farm (S2.2) stage.

5.3 PORK SECTOR IN SPAIN: A CASE STUDY IN CATALONIA

5.3.1 Goal and scope definition

In this section, a detailed evaluation of the environmental profile of the pork production chain in Catalonia region (NE Spain – Figure 5.5) was carried out through the LCA perspective (ISO 14040, 2006). Additionally, WF results were also estimated to help Catalan companies in the pork sector to improve the efficient use of water resources.

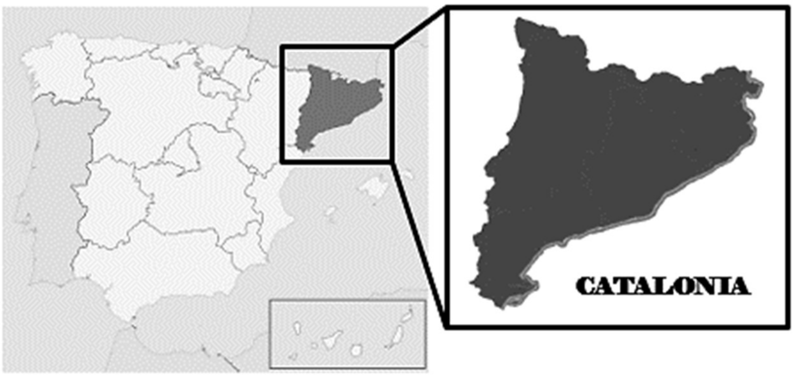

Figure 5.5. Location of the pork production system in Catalonia (NE Spain).

A cradle-to-gate assessment was conducted in this section, covering all the processes in the pork production chain up to the cutting room stage, where fresh/frozen cut pork is obtained as a product (Figure 5.6). Thus, all activities related to animal feed production, breeding and fattening pigs at farm, slaughterhouse and cutting stage were encompassed in the study; the final stages of pork processing fell out the system boundaries. This perspective goes beyond those defined in the previous section, where the scope extends from feed production to pig farm gate, in line with most LCA studies in literature (McAuliffe et al., 2016). Related emissions to soil, air and water were also considered.

Finally, according to the results, the main hotspots were identified as the basis for the proposal of alternative strategies to reduce the impacts of the Catalan pig sector and improve its environmental sustainability.

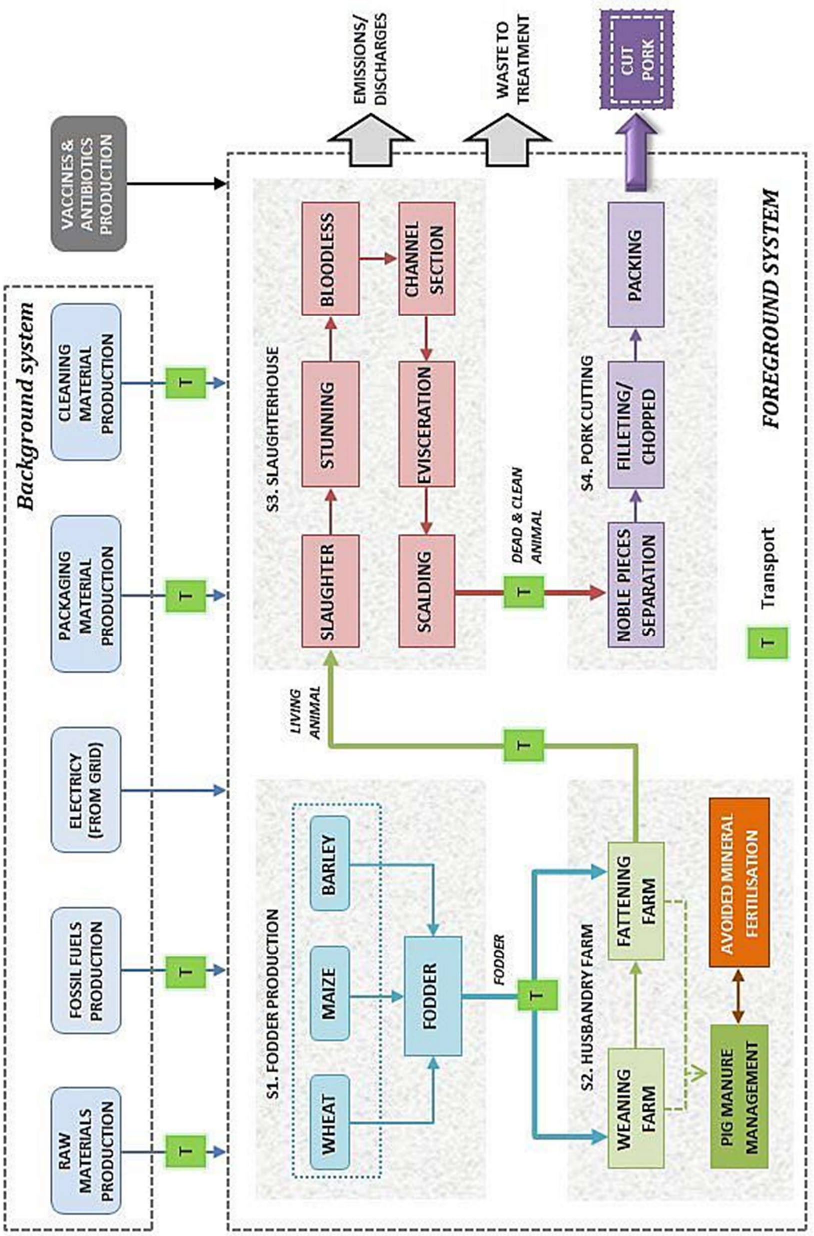

Figure 5.6. Scheme of the system boundaries for the pork production system. Key: Brown box includes avoided impacts due to both mineral fertiliser production and application on soils. Grey box represents excluded processes.

- **Functional unit**

As aforementioned, mass-based FUs prevail over other choices in most LCA studies reviewed for pork sector. In this context, a FU based on the mass flow of the product was selected as the basis for the evaluation: 1 kg of cut pork (fresh or frozen).

- **System description**

The whole system assessed in this section comprises four main subsystems (Figure 5.6): (S1) fodder production, (S2) animal husbandry on the farm, (S3) slaughterhouse and (S4) pork cutting. All the environmental burdens related to the consumption of energy and fossil fuels, raw and other auxiliary materials, water supply, transport activities and management of waste streams were considered for each life cycle of the pork chain. The information on the input and output flows attributed to each subsystem is detailed below.

Fodder production

The production of feed supplied to pigs during husbandry stage is included within the boundaries of this subsystem (S1), so that the following flows were considered: raw materials production (mainly cereal crops), water and energy consumption, transport activities for the supply of raw materials and final management of the associated waste.

Animal husbandry on the farm

All the activities related to the piglets rearing and later fattening stage are recorded in this subsystem (S2). Thus, environmental burdens associated with the maintenance and feeding of animal herd were taken into account, along with the transport of piglets from the weaning farm to the fattening farm. Carbon and nitrogen based emissions from manure management activities were also considered for assessment.

Slaughterhouse

All the processes carried out at the slaughterhouse were taken into account, from the transport of raw materials (pigs included) to the output of the carcass pork. Thus, the environmental impacts related to the consumption of raw materials, water and energy and the generation of solid wastes and wastewater in all the processes of this subsystem were considered, including: stunning and killing (slaughter), bloodless, channel section, evisceration and scalding.

Pork cutting

Similar to S3, this subsystem encompasses both the consumption of raw materials, water and energy and the generation of solid waste and wastewater from the different modules, from the reception of the dead and clear pork (including transport) to the output of the final products: fresh and/or frozen cut pork.

- **Allocation rules**

Analogous to the previous section, allocation was not considered in the system assessed here, since the different products obtained along the production chain were the only ones responsible for the impacts of each stage of the life cycle. The rationale behind that is the lower relevance of by-products (such as butter, guts and hardeners, among others) in comparison with the other main outputs, as well as their unprofitable market value. Consequently, the evaluation of the management practices and the further disposal of these by-products was not included within the system boundaries.

Similarly, the system expansion approach was again applied to take into account the valorisation of pig manure as an organic fertiliser. In this way, the emissions and discharges derived from manure management (including storage and application) as well as the environmental credits related to its use as an organic fertiliser (avoided mineral fertilisation) were included within the system boundaries.

Finally, mass allocation in the background processes was also prioritised over other alternatives, followed by economic distribution, in accordance with secondary processes in databases (see epigraph 5.3.2 involving LCI data).

5.3.2 LCI analysis

The close collaboration between the research team and different stakeholders, either companies or industries within the Catalan pork sector, was fundamental for the development of the inventory. Thus, the study has involved the participation of more than 15 representative companies from the pork production sector in Catalonia, including feed factories, pig farms, slaughterhouse and cutting rooms, and even processing facilities for the manufacture of the processed products. Taking into account individual company data for the different stages of the production chain, a global integrated inventory was drawn up for each intermediate or final product assessed throughout the life cycle of pork production. As a result, comprehensive inventory data were obtained, as well as the average results of the Catalan pork sector.

Fodder production

Fodder production was assessed and detailed inventories were carried out. To this aim, questionnaires filled in by Catalan fodder companies were used to collect primary data on fodder composition and other input needs (water and energy). However, unlike the Galician case study, no differentiated information was provided regarding alternative types of fodder supplied to pigs, so a general composition was assumed for fodder used on both weaning and fattening farms.

Therefore, wheat, maize and barley were considered the main ingredients (Table 5.6). All of them are produced in Catalonia and average transport distances of 1,100 km, 840 km and 285 km were assumed, respectively. For imported ingredients such as rapeseed meal, rye and peas, average distances of 3,800 km (Russia), 2,450 km (Poland) and 2,600 km (France) were considered, respectively. Background inventory data related to agricultural activities for the cultivation of ingredients, agrochemical inputs, electricity consumption, fuel use, packaging and waste management were obtained from the ecoinvent® database (Althaus et al., 2007; Dones et al., 2007; Nemecek and Käggi, 2007; Spielmann et al., 2007; Wernet et al., 2016).

Table 5.6. Main ingredients and composition of the fodder for animal feeding.

Ingredients (%)	Fodder (weaning and fattening farms)
Wheat	37.1
Maize	22.9
Barley	16.1
Rye	6.03
Rice	1.16
Animal fats	1.57
Peas	2.33
Beet molasses	1.18
Rapeseed meal	8.07
Others	3.56

From animal farming to pork cutting

Most of the inventory data for the other subsystems were also obtained from questionnaires filled in by the cooperating partners. In the case of farming activities (S2), such questionnaires compiled information on herd features, fodder ration and other inputs supply, consumption of resources (waste, energy, chemicals), outputs production and waste management. For the stages of slaughterhouse (S3) and cutting room (S4), additional information on the requirements for packaging materials was also included.

However, similarly to the previous fodder production subsystem (S1), secondary data taken from literature and commercial databases (mainly the ecoinvent® database) were used to complete the LCA background inventory, including electricity production or agricultural processes for the production of fodder ingredients, among others (Althaus et al., 2007; Doka et al., 2007; Dones et al., 2007; Hischier et al., 2007; Spielmann et al., 2007; Wernet et al., 2016). Moreover, emissions from enteric fermentation and manure management, both in C and N form, were calculated based on the Tier 1 method proposed by the IPCC guidelines (IPCC, 2006). The ratio proposed by Rossier (1998) was considered to estimate PO_4^{3-} emissions. Finally, the values of the WFN database were also used for the information on agricultural raw materials (see Table 5.6) in the WF calculations. Global inventory data per FU is summarised in Table 5.7.

By contrast, the impacts associated with other elements such as the production and use of veterinary drugs and other specific raw materials, were not included due to the lack of reliable information and precise characterisation factors for these compounds. However, a minor influence on the final results would be expected, as the requirements for these inputs can be considered negligible compared to the contribution of other flows that were included within the system boundaries (Dourmad at al., 2014; Reckmann et al., 2013).

Table 5.7. Inventory data from the different subsystems per FU (1 kg of cut pork).

Inputs/Outputs	Amount	Units
Inputs from environment		
Water	0.59	L
Inputs from technosphere		
Raw materials		
Water	15.7	L
Fodder	4.45	kg
Live weight pork	1.46	kg
Carcass wieght pork	1.17	kg
Cleaning products		
Soap	0.45	g
Chemicals	106	g
Packaging material		
Paper	1.02	g
Cardboard	1.87	g
Plastic	9.96	g
Wood	1.46	g
On-farm energy use		
Electricity	2.45	kWh
Natural gas	73.5	L
Fuel oil	7.26	mL
Transport		
Lorry	9.28	t·km

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Table 5.7 (cont.). Inventory data from the different subsystems per FU (1 kg of cut pork).

Inputs/Outputs	Amount	Units
Outputs to technosphere		
Products and co-products		
Cut pork	1.00	kg
Pig manure[a]	17.2	kg
Waste to treatment		
Wastewater	14.1	L
Cardboard	7.19	g
Plastic	0.57	g
Aluminium	0.46	g
Wood	0.28	g
Organic waste	88.1	g
Sewage sludge	11.1	g
Avoided fertiliser production		
N from manure	42.1	g
P from manure	5.95	g
Outputs to environment		
Air emissions		
CH_4 – Enteric fermentation	7.78	g
CH_4 – Manure management	51.9	g
N_2O	2.20	g
NH_3	71.6	g
Water emissions		
NO_3^-	0.32	kg
PO_4^{-3}	0.19	g
Avoided fertiliser application		
N_2O	0.66	g
NH_3	5.11	g
NO_3^-	0.06	kg
PO_4^{-3}	0.18	g

[a] Pig manure composition: 3.26 g N/kg manure; 0.36 g P/kg manure.

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5.3.3 Impact assessment

The characterisation factors reported by the ReCipE Midpoint (H) 1.12 method (Goedkoop et al., 2013a) were considered for the following impact categories, identified as those of primary interest for pork sector (Reckmann et al., 2012): CC, TA, FE, ME and FD. SimaPro v8.2 software was used for the computational implementation of the inventories (Goedkoop et al., 2013b). Additionally, in this section, the WF results were also evaluated as an increasingly important environmental indicator in the context of the organisations and products of the pork supply chain in Catalonia.

5.3.4 Results

- **WF results**

Global and disaggregating WF results for the different subsystems are reported in Table 5.8. According to the results, the highest WF ratios are recorded in fodder production (above 80%), mainly due to both rainwater and irrigation water used for crop cultivation. In contrast, the other subsystems slightly contribute to the entire system.

Table 5.8. WF results per FU (1 kg of cut pork) from the global system.

Life cycle stage	WF results (m^3)				Relative contribution (%)
	Green WF	**Blue WF**	**Grey WF**	**Total WF**	
Fodder production	6.11	0.93	0.70	7.74	80.6
Animal husbandry	0.00	0.58	0.00	0.58	6.00
Slaughterhouse	0.00	0.01	1.15	1.16	12.1
Pork cutting	0.00	$3.6\cdot10^{-3}$	0.12	0.12	1.30
Global pork system	6.11	1.52	1.97	9.60	100
Relative contribution (%)	63.6	15.8	20.5	100	

Focusing on each type of WF, it was found that green WF was widely attributed to fodder production, representing 79% of the contributions in this subsystem, mainly due to the effect of rain and evapotranspiration in agriculture; meanwhile, blue (12%) and grey (9%) results have a minor influence. In contrast, blue WF has a critical role in animal husbandry (100%

of contributions) while almost the entire WF ratio for slaughterhouse and pork cutting corresponds to the grey indicator, due to the presence of blood residues in the water used for cleaning activities.

- **LCA results**

The global characterisation results per FU, together with the particular contribution of the different subsystems are reported in Table 5.9 for all impact categories evaluated. In this way, it is possible to know the contribution of each preceding subsystem to the final product. According to the results, fodder production (S1) shows a major influence, with contributions ranging from 35% (TA) to 99% (FE), followed by husbandry farm (S2), with a particular effect on TA (63%) and ME (46%). Pork cutting (S4) and slaughterhouse (S3) subsystems account for up to 6% and 4% of the impacts, respectively.

Table 5.9. Global LCA results per FU (1 kg of cut pork) and relative contributions from the different subsystems to the impact categories.

Impact category	Units	S1	S2	S3	S4	Global results
CC	kg CO_2 eq	4.35	0.37	0.10	0.13	4.95
TA	g SO_2 eq	91.0	162	0.56	0.64	254
FE	g P eq	1.28	-0.03	0.01	0.02	1.28
ME	g N eq	78.9	68.1	0.14	0.35	147
FD	kg oil eq	1.21	0.08	0.09	0.06	1.44

Moreover, Figure 5.7 also shows the environmental results broken down into percentages. In this case, the contributions of the different processes and activities involved throughout the system are shown in relation to the overall results. Once again, the processes and activities were grouped into different contributing factors to facilitate the analysis: on-farm diffuse emissions, supply of raw materials, energy use, cleaning and packaging products and avoided processes. It should be noted that the supply of raw materials includes the environmental impacts related to the production of the final product from the different material inputs (mainly fodder). In this regard, all the ingredients for fodder mixtures were considered, not only locally grown ingredients (such as wheat, maize and barley) but also imported; therefore, burdens from transportation were also included in this factor.

According to Figure 5.7, the environmental impacts related to the production of raw materials (fodder and other nutritional components) play a critical role, with contributions of more than 32% regardless of the category considered. More specifically, fodder produced in the previous stage (S1) and used for on-farm feeding (S2) stands as the most damaging process (hotspot) in the environmental profile of the global system, according to other LCA studies involving pork production in the literature. This is mainly due to emissions associated with the production and application of mineral fertilisers for crop cultivation (especially wheat and rape meal) together with combustion emissions from the use of agricultural machinery. Transport activities also have an important influence on this factor, especially in terms of CC and FD (with contributions up to 38%). This can mainly be attributed to the long distances required for most inputs (especially fodder ingredients). Diffuse emissions also have a relevant contribution, especially in terms of CC, TA and ME; however, avoided processes from manure valorisation as fertiliser represent environmental credits ranging from 5% to 10%, which partially offset related impacts.

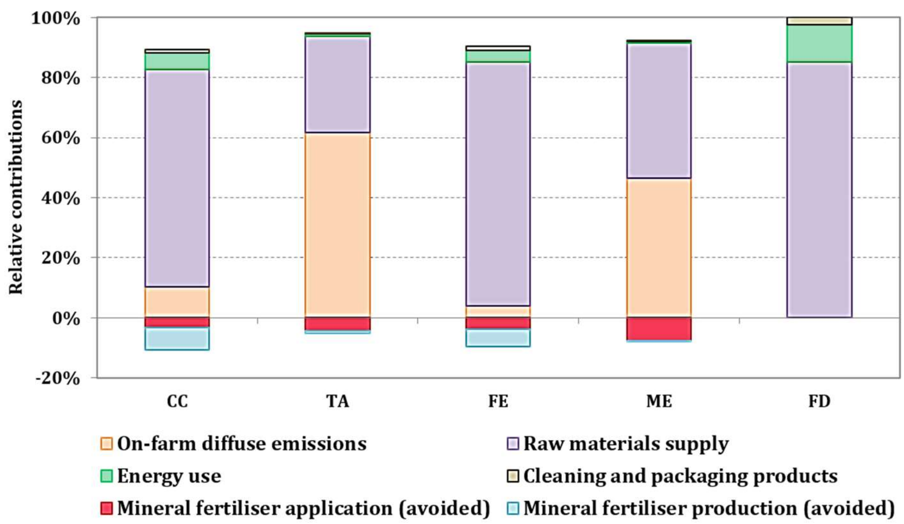

Figure 5.7. Relative contributions from the main processes involved in the global system.

Finally, energy use mainly affects to CC and FD, especially due to the high energy demand within farm facilities during husbandry stage, while minor influence can be observed from the impacts of cleaning and packaging products (below 2%).

5.4 DISCUSSION

5.4.1 Comparison with related LCA studies

In this chapter, an environmental assessment was carried out taking into consideration different stages of the pork chain, within the framework of the Galician and Catalan pork sectors. According to the results obtained, fodder production stands as the main hotspot, regardless of the impact indicator and the location considered. These outcomes are in line with the conclusions of related literature studies, which reported that animal feed production is the main process responsible for the environmental impacts of pork products (de Miguel et al., 2015; González-García et al., 2015; Dourmad et al., 2014; Reckmann et al., 2013; Nguyen et al., 2011; Pelletier et al., 2010).

Moreover, Table 5.10 shows a summary of the environmental results for all the studies involving pork production compiled from the literature for comparison. It should be noted that, although the number of LCA studies for pork production has increased in recent years, their methodological approach varies considerably. For this reason, only those studies that shared similar (mass-based) FUs and allocation rules (system expansion) were identified as suitable for comparison with the present study. In the absence of similar studies covering the entire supply chain up to the cutting room, the results of the Catalan case study had to be recalculated per kg carcass weight (slaughterhouse gate). Finally, particular attention was paid to CC and TA results, since discrepancies in the characterisation methods prevent a reliable comparative analysis involving the potential results of eutrophication.

According to the data shown in Table 5.10, similarities were found among the present study and similar works in terms of CC, with own values falling within the range of values reported in the literature per kg of carcass pork (from 3.34 to 4.08 kg CO_2 eq) and slightly above regarding results per kg of live weight pork (from 2.30 to 3.42 kg CO_2 eq). Since N_2O emissions were considered to be the main contributor to CC, the criteria used by the different authors to estimate them are considered as one of the main factor responsible for the minor differences in CC values. Thus, for example, while the ReCiPe method was applied in the present study, Dourmad et al. (2014) and Dalgaard et al. (2007) used the characterisation factors provided by the CML and EDIP methodologies, respectively, for data assessment; this leads to alternative characterisation factors being partially responsible for the major differences in the CC results.

Table 5.10. LCA studies considered for comparison and related results involving CC and TA impacts per FU (1 kg of FPCM at farn gate). Key: LW = live weight (farm gate); CW = carcass weight (slaughterhouse gate).

Study	Country	FU	CC (kg CO_2 eq)	TA (g SO_2 eq)
This study	Galicia – Spain	1 kg of LW	3.42	186
Halberg et al. (2010)	Denmark	1 kg of LW	3.30	-
Dolman et al. (2012)[a]	The Netherlands	1 kg of LW	5.46	53.0
Dourmad et al. (2014)	EU	1 kg of LW	2.30	44.0
Groen et al. (2016)	-	1 kg of LW	2.61	-
This study	Catalonia – Spain	1 kg of CW	4.26	217
Dalgaard et al. (2007)	Denmark	1 kg of CW	3.60	45.0
Nguyen et al. (2010)	EU	1 kg of CW	4.81	-
Wiedemann et al. (2010)[b]	Australia	1 kg of CW	4.30	-
González-García et al. (2015)	Portugal	1 kg of CW	3.34	22.8

[a] Economic allocation; [b] Average data have been estimated.

Regarding TA, the results obtained in the present study are above the range of values reported by other authors. Previous studies on pork production have shown that acidification potential may be directly related to

NH_3 emissions from farming activities. In this regard, the different emission factors used in LCA studies for different types of housing, manure storage and manure application techniques may be decisive for the variability of the acidification results, together with the composition of manure in terms of its nitrogen content. Moreover, NH_3 emissions were also found to be closely related to the production of fodder ingredients.

Thus, the activities developed during crop cultivation, together with fodder composition and consumption rates on the farm, also accounted for sustainable differences in TA values. In this regard, in both Galician and Catalan case studies, it was considered that pigs were fed exclusively on fodder mixtures with a high cereal content. These crops are typically grown under intensive regimes with a significant consumption of pesticides and fertilisers, either mineral or organic, with an inherent discharge of nitrogen compounds into the environment. By contrast, according to other authors (Dalgaard et al., 2007; Dourmad et al., 2014), alternative diets based on the combination of fodder with other protein-rich ingredients and mineral sources could lead to more environmentally friendly performances in terms of acidification potential.

5.4.2 Improvement actions: optimisation of fodder production

In accordance with the results aforementioned, fodder production has been shown to be the subsystem with the highest share in most impact categories. Therefore, special attention should be paid to this issue with the aim of proposing improvement actions that can reduce impact levels. In this context, eco-design has gained relevance in the fodder manufacture (Philippe et al., 2011). Integrating this concept into the feed production chain would not only allow a rigorous and verifiable environmental approach, but would also increase the added value of these fodders and differentiate them from similar products on the market (Knight and Jenkins, 2009). Thus, various proposals could be implemented, such as the use of sustainable ingredients, the production of new packaging based on biodegradable materials and the improvement of energy efficiency through the optimisation of transport logistics and distribution management (Philippe et al., 2011; Baumgartner et al., 2008).

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In this line, a wide range of research studies has focused on reducing the burdens associated with pork production to date, in which the most common proposal is related to the optimisation of fodder composition (McAuliffe, 2016). However, optimisation is limited because of the nutritional requirements that fodder must meet (Baumgartner et al., 2008). Moreover, most of the main components in current fodder formulations (such as wheat, maize or barley) are also variables with the highest potential environmental impacts (Baumgartner et al., 2008). In this sense, proposals focused on the use of alternative protein sources are currently being particularly successful, increasing the relevance of the use of synthetic feed use amino acids (FU AA) to reduce the requirements of other more environmentally harmful protein sources, especially soybean and other soy products (García-Launay et al., 2014; Ogino et al., 2013; Mosnier et al., 2011). Other alternatives based on the use of seaweed as an AA source to minimise the consumption of other protein sources are still pending research (McAuliffe, 2016).

In addition to fodder composition, the production of each ingredient could be also optimised. According to Kool et al. (2010), differences of up to 15% in GHGs emissions during the production of raw materials can be found between countries. For example, the production of wheat (the main ingredient in fodder production with an average of 573 kg/t feed) reports emissions ranging from 500 to 575 kg CO_2 eq, being The Netherlands the country with the lowest emissions per tonne, while France presents the highest ones (Kool et al., 2010). This demonstrates that the selection of the country from which raw materials importation takes place can have a crucial effect in reducing related impacts. Finally, the optimisation of the feed rate supplied to pigs can be also a remarkable variable to reduce environmental results. In this sense, recent studies (Kool et al., 2010) together with data provided by companies in the sector show that significant reductions can be achieved, not only in terms of GHG emissions but also in nitrogen excretion levels.

5.5 Conclusions

Pork is the most widely type of meat consumed in Europe but also the second largest contributor to GHG emissions from the livestock sector, which highlights the necessity to explore the potential causes. In this context, conventional practices on pork production were evaluated environmentally through a LCA perspective in two alternative locations: Galicia and Catalonia. While a cradle-to-farm gate study was carried out in the former, the whole supply chain was assessed in the latter case.

According to the results, fodder production was found to be the main environmental hotspot in both systems, followed by the emissions from both on-farm activities and manure management practices. Therefore, both animal feed production and farm activities were proved to play a key role in the environmental profile of the pork supply chain, over further processing stages involving slaughtering and cutting processes. These results were in line with the major outcomes reported by similar LCA studies involving pork production worldwide.

In response to these findings, potential improvement strategies focused on fodder production were proposed within the framework of this chapter. Accordingly, three main areas for action can be distinguished, optimising: (i) fodder composition, (ii) quantities of feed supplied to pigs and (iii) production processes, especially related to import of fodder ingredients. However, further studies should be developed to demonstrate the benefits of such proposals in specific studies and analogous systems in the pork sector.

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USC UNIVERSIDADE DE SANTIAGO DE COMPOSTELA

CHAPTER 6. GREEN VALORISATION OF LIVESTOCK MANURE – MANUREEECOMINE (MEM) PROJECT

Summary

The use of non-renewable energy sources still has a critical impact on climate change, closely linked to intensive agriculture that is highly dependent on the use of agrochemicals, mostly of mineral origin. However, manure, on the other hand, contains significant amounts of unused nutrients, which could be used effectively as organic fertilisers in agricultural activities at European level.

In this context, the ManureEcoMine (MEM) project proposed an alternative approach to exploit the fertilising potential of animal manure, in response to the related environmental impacts due to its management as waste. To this aim, an integral treatment scheme is defined within the framework of the project, based on complementing the conventional AD process with additional key stages such as solid/liquid separation, struvite precipitation and biological nitrogen removal.

Consequently, two case studies were evaluated in this chapter from an environmental point of view, focusing mainly on the management of pig slurry (The Netherlands) and cow manure (Spain). In both cases, the MEM prototype (S1) was analysed in comparison with more conventional practices as a base case (BS); in addition, the Spanish case study also considered an additional scenario involving an acidification stage prior to the innovative management stages. Comparative results showed that MEM prototypes are the most environmentally friendly scenarios in most impact categories, mainly due to the unfavourable contribution of diffuse emissions and the direct disposal of digestate in soils in conventional practices (BS). Moreover, the energy generated by biogas valorisation proportionally offset the environmental burdens regardless of the scenario and location considered; however, the avoided emissions from nutrients recovery have an additional influence on the favourable performance of MEM scenarios, from an environmental perspective.

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USC UNIVERSIDADE DE SANTIAGO DE COMPOSTELA

6.1 INTRODUCTION TO THE VALORISATION OF MANURE: MEM PROJECT

Since the so-called "oil crisis" in the 1970s, and more intensely in recent years, climate change has been a major global concern (IPCC, 2013). Accordingly, special attention has been paid to fossil fuels consumption and agricultural activities, since they are considered to be the most responsible for the high levels of GHGs emitted into the environment (IPCC, 2014; Roy et al., 2009).

In this context, energy production from renewable sources (bioenergy) plays a crucial role because of its contribution to energy security and GHG mitigation (Benoist et al, 2012; Souza et al., 2017). The use of biomass for bioenergy is considered one of the most promising sources of renewable energy, especially due to the generation of biogas from anaerobic digestion (AD) of organic matter (Benoist et al, 2012). As a result of its use as an energy source, it is estimated that biogas production in Europe could represent 25% of the bioenergy generated in the future (Nielsen and Oleskowicz-Popiel, 2007). However, biogas-based systems have advantages not only from an energy point of view, but also from the production of digestate, a nutrient-rich stream that can be used as organic fertiliser in agricultural soils (Abubaker et al., 2012).

Meanwhile, intensive agriculture, in an effort to meet the needs of growing populations, has become highly dependent on the use of mineral fertilisers to sustain both food and feed production (FAO, 2011; Garnett et al., 2013; Ward et al., 2016). The production process of this type of fertiliser results in a high carbon footprint due to the significant energy requirements for the production of N-based fertilisers, as well as the wide range of pollution related to mining activities in the extraction of P and K-based compounds (Skowronska and Filipek, 2014). Moreover, the long-term application of mineral fertilisers deprives depleted soils of organic carbon (Zhang et al., 2016). As a result, agricultural soil absorbs less carbon and becomes less productive, resulting in eutrophication and acidification impacts.

In assessing the role of livestock waste management in Europe, pigs and cows together produce about 1.27 billion tonnes of manure per year, a major source of organic carbon and untapped nutrients (Tarragó et al., 2014; Steinfeld et al., 2006). However, in many cases, the direct use of these nutrients is now environmentally restricted, given their direct relationship to environmental damage (Nitrate Directive, 1991; Steinfeld et al., 2006). In fact, most European regions with intensive cattle and pig production overlap to a large extent with NVZ (Nitrate Directive, 1991).

In this context, the ManureEcoMine (MEM)[1] project aimed at translating concerns on manure management into an important opportunity for recovery and reuse. Thus, it focused attention on an integrated approach to the treatment and reuse of animal manure in nitrate vulnerable and sensitive zones and beyond, with resource recovery and energy efficiency as key principles. To this end, high potential eco-innovative technologies already implemented in the wastewater treatment were tested to evaluate their technical feasibility and environmental performance involving livestock manure.

In this sense, the present Chapter 6 focused on the life cycle analysis conducted to demonstrate the environmental sustainability of the different scenarios proposed, as well as to identify the most environmentally friendly alternatives. Two case studies were evaluated within the framework of the project, based on the valorisation of pig and cow manure for both Dutch and Spanish locations, respectively.

6.2 MEM PROTOTYPE IN THE NETHERLANDS

6.2.1 Goal and scope definition

This section focuses attention on the environmental evaluation of the configuration proposed within the MEM framework in The Netherlands, aiming to compare its performance against more conventional practices. To this aim, a cradle-to-grave approach was followed, covering the entire life cycle of the system, from the production and transport of raw materials (where applicable) to the use of the product and the management of the final

[1] www.manureecomine.ugent.be

waste. A mixture of pig slurry (83.1%) and eco-frit (16.9%) as main substrates was considered to be fed to the system (Table 6.1).

Table 6.1. Flow and mean composition of feeding mixture in The Netherlands: dry matter (DM); Chemical Oxygen Demand (COD); Total Nitrogen (TN); Total Phosphorous (TP).

Composition	Pig slurry	Eco-frit	Feeding mixture
Flow (kg/d)	74.0	15.0	89.0
DM (kg/d)	85.5	175	101
COD (g/kg)	75.0	225	100
TN (g/kg)	6.00	7.25	6.21
$N\text{-}NH_4^+$ (g/kg)	4.50	1.70	4.03
TP (g/kg)	1.65	2.50	1.79
$P\text{-}PO_4^{3-}$ (g/kg)	1.43	2.10	1.54

- **Functional unit**

Mass and energy-based FUs are the most common practices applied in LCA studies involving AD and similar processes, depending on the specific objective of each case (Bacenetti et al., 2016; Lijó et al., 2017); similarly, waste mass-based FUs are also frequently used in LCA studies on livestock waste management (Laurent et al., 2014a,b; Vázquez-Rowe et al., 2015).

Accordingly, a mass-based FU was chosen as the basis for the calculations in this section: 89 kg of incoming waste to be treated in the system (per day). This choice is in agreement with the average flow feeding as feasible for the pilot scale configuration, common to the different management scenarios proposed for evaluation.

- **System description**

Two scenarios were assessed within the Dutch case study: the baseline scenario (BS) and the MEM prototype (S1). BS (Figure 6.1) refers to the most common practices to date involving anaerobic co-digestion (AcoD) with the generation of biogas together with the valorisation of digestate as an organic fertiliser in agricultural soils. However, S2 (Figure 6.2) represents the configuration of the Dutch pilot plant, in which the AD process is complemented with further stages focusing on the recovery and/or removal of several nutrients for their final valorisation. These stages include S/L separation, struvite precipitation and biological nitrogen removal (BNR) as key technologies; and it is assumed that the biogas produced is used as an energy source in a cogeneration (CHP) unit.

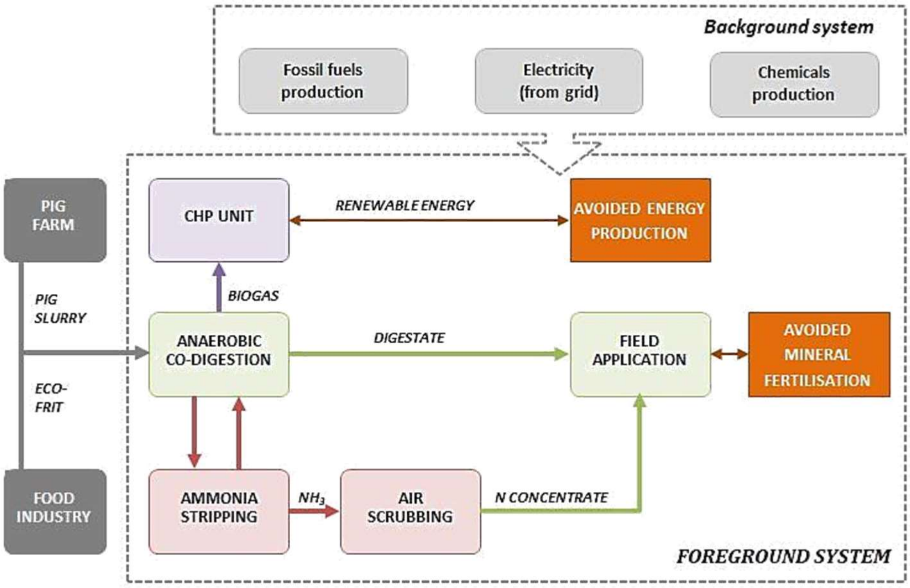

Figure 6.1. Scheme of the system boundaries corresponding to BS in The Netherlands. Key: Brown boxes include avoided impacts due to organic fertilisation and renewable energy generation. Grey boxes represent excluded processes.

It should be noted that a thermophilic digester is considered to be used in both scenarios. Thermophilic digesters are more sensitive to free ammonia toxicity than mesophilic ones. For this reason, the thermophilic anaerobic co-digestion process is linked to a stripping unit to prevent the accumulation of inhibitory levels of free ammonia in the digester. Consequently, an additional stage of air scrubbing must be also included to recovery N- and S-based compounds from the exhaust air stream from the stripping stage.

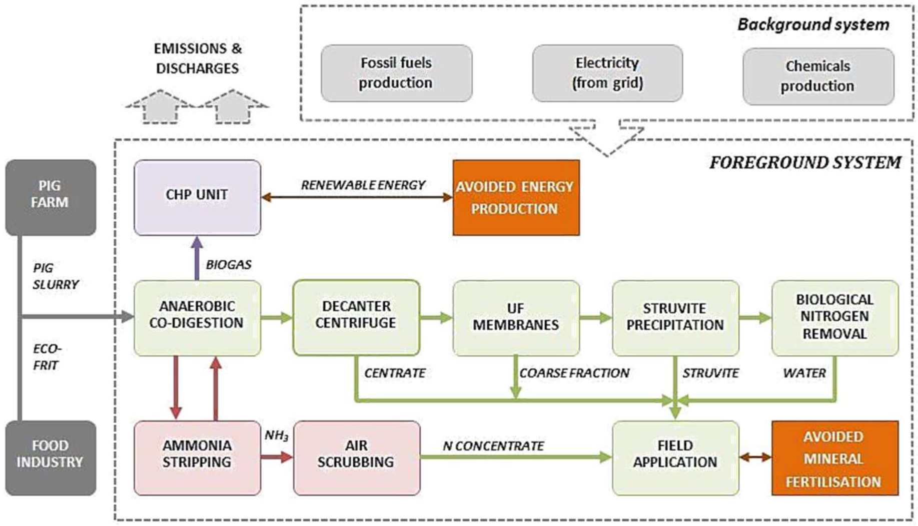

Figure 6.2. Scheme of the system boundaries for S1 in The Netherlands. Key: Brown boxes include avoided impacts due to organic fertilisation and renewable energy generation. Grey boxes represent excluded processes.

▪ Allocation rules

In this section, a system expansion strategy was followed, with the aim of avoiding the distribution of loads between the different products obtained in both scenarios. It can be assumed that different organic fertilisers are formulated from the nutrient streams recovered throughout each system. They were considered that could replace mineral fertilisers to some extent, so that credits related to the avoided production and subsequent application of such fertilisers can be attributed to the systems. In addition, the recovery of biogas as a raw material for renewable energy generation was also included within the system boundaries.

Moreover, it should be noted that the generation of both substrates (pig slurry and eco-frit) did not contribute to the overall impacts, as they were considered as residues from pig farms and food industry.

6.2.2 LCI analysis

Primary data provided by the different MEM project partners through reports, deliverables and project meetings were mainly considered, while secondary data were also used to fill gaps in the foreground inventory as well as to complete background information. In this regard, a detailed description of the inventory information and data sources for each stage for both scenarios (BS and S2) is provided below.

- **Anaerobic co-digestion**

As aforementioned, a thermophilic AcoD (at 55 °C) is carried out and two main streams are obtained: biogas and digestate. The biogas rate was estimated taking account the organic loading rate (OLR), the gas production rate (GPR), the reactor volume and the conversion ratio of about 0.50 m^3 of biogas per kg COD removed (Table 6.2). However, diffuse emissions in terms of biogas losses were also considered according to de Vries et al. (2012): 1% from the AcoD process and 0.5% for the CHP unit. Similarly, N_2O and nitrogen oxides (NO_x) emissions from the AcoD stage were also estimated: average ratios of 0.62 mg N_2O and 1.22 g NO_x per m^3 of biogas produced were assumed. Regarding digestate, two main options were considered: (BS) storage and direct application on agricultural soils and (S2) digestate management for nutrients (N, P) recovery and subsequent use as organic fertilisers.

Table 6.2. Flow and mean composition of digestate from AcoD in The Netherlands.

Composition	Feeding mixture	Biogas	Digestate
Flow (kg/d)	89.0	2.91[a,b]	88.0
DM (g/kg)	101	-	57.5
COD (g/kg)	100	-	67.5
TN (g/kg)	6.21	-	4.05
$N\text{-}NH_4^+$ (g/kg)	4.03	-	2.70
TP (g/kg)	1.79	-	1.58
$P\text{-}PO_4^{3-}$ (g/kg)	1.54	-	1.45

[a] m^3/d; [b] OLR = 4.10 kg COD/(m^3·d); GPR = 1.94 kg COD/(m^3·d); Vreactor = 3 m^3.

Biogas valorisation can be performed by means of a CHP unit to generate both electricity and heat; otherwise, CH_4 and CO_2 would be discharged directly to air. In this sense, a calorific value of 5.42 kWh/m^3 of biogas was considered, on the basis of a calorific value of 8.33 kWh/m^3 CH_4 (IDAE, 2014) and a composition of 65% CH_4 (35% CO_2). Moreover, an average electric efficiency of 33% was assumed (Pöschl et al., 2010), so that a ratio of 5.12 kWh/day of electricity was produced, of which it was assumed that around 3% (0.15 kWh/day) is needed for CHP operation. This electricity could be sold to the national grid and could potentially partially replace the demand for electricity generated by fossil sources (avoided electricity). Similarly, based on a thermal efficiency of 50% (Pöschl et al., 2010), a value of 7.76 kWh/day would be available to partially offset the heat requirements of the plant (avoided heating energy).

- **Ammonia stripping**

The stripping unit is included to avoid the accumulation of inhibitory levels of free ammonia in the digester. For this purpose, air must be supplied by blowers with an aeration rate of 75 L air/(L digestate·hour) for approximately 6 h; since around 170 kg digestate/day is treated, 76.5 m^3 of air is daily fed to the stripping tank. Accordingly, 0.24 kg N-NH_4^+ are recovered each day in the pilot plant (Table 6.3).

Table 6.3. Flow and mean composition of stripped digestate and recovery percentages from stripping and scrubbing stages in The Netherlands.

Composition	Digestate	Stripped Digestate	N recovery (%)
Flow (kg/d)	88.0	88.0	-
DM (g/kg)	57.5	57.5	-
COD (g/kg)	67.5	67.5	-
TN (g/kg)	4.05	2.30	43.2
N-NH_4^+ (g/kg)	2.70	1.70	99.4
TP (g/kg)	1.58	1.58	-
P-PO_4^{3-} (g/kg)	1.45	1.45	-

Moreover, an air scrubber unit was also used for N recovery the exhaust gas flow from the stripping stage. The scrubbed air is mainly recirculated to the top of the ammonia stripping stage in such a way as to minimise heat consumption (assumed to be negligible). The remaining air is discharged into the atmosphere and may contain traces of NH_3. According to the literature (Melse and Ogink, 2005; Melse et al., 2009; Bousek et al., 2016), an average NH_3 removal rate of 96% for the scrubber was considered (≈700 ppm of NH_3 in the inlet stream). Finally, a nitrogen concentrated fraction (hereinafter referred to as N concentrate) is obtained as a result of the reaction of NH_3 and H_2SO_4 solution (98% v/v): 1.32 kg $(NH_4)_2SO_4$ (ammonium sulphate) is produced and used for fertilisation purposes.

- **Solid/liquid separation**

S/L separation is conducted in two main stages: centrifugation and ultrafiltration (UF). Two main streams are firstly obtained from the decanter centrifuge: the centrate (liquid) and the coarse (solid) fraction (Table 6.4). The centrate is then fed to the UF membranes and two main streams are again obtained: the permeate (liquid) and the retentate (solid). Both coarse and retentate fractions can be used as organic fertilisers, so that related impacts were also taken into consideration (see epigraph Fertilisation activities).

Table 6.4. Flow, mean composition and separation percentages of the centrifugation and UF membrane stages in The Netherlands.

Composition	Centrate	Coarse fraction	SR1[a] (%)	Permeate	Retentate	SR2[a] (%)
Flow (kg/d)	74.8	13.2	15.0	56.1	18.7	25.0
DM (g/kg)	27.5	235	-	12.5	-	-
COD (g/kg)	22.5	324	71.7	7.20	68.4	76.0
TN (g/kg)	3.25	8.62	31.8	1.95	7.15	55.0
$N\text{-}NH_4^+$ (g/kg)	1.90	7.26	40.2	1.60	2.80	36.8
TP (g/kg)	0.28	8.98	85.2	0.11	0.79	71.4
$P\text{-}PO_4^{3-}$ (g/kg)	0.23	8.40	86.5	0.10	0.64	69.0

[a] SR = separation rate.

- **Struvite crystallisation**

Ammonium struvite ($MgNH_4PO_4{\cdot}6H_2O$) was assumed to be obtained from the struvite crystallisation stage. According to Equation 1, magnesium (Mg^{+2}), ammonium (NH_4^+) and phosphate (PO_4^{3-}) are required in equimolar quantities to obtain the target product (Table 6.5):

$$Mg^{+2} + NH_4^+ + PO_4^{-3} + 6\,H_2O \rightarrow MgNH_4PO_4 \cdot 6H_2O \qquad \textbf{Eq. 1}$$

NH_4^+ is sufficiently present as a substrate for struvite precipitation. However, Mg^{+2} is stoichiometrically limited in the reaction medium, so that it is additionally added to achieve a sufficiently high PO_4^{3-} removal. Accordingly, Mg^{+2} is dosed in excess of 60% (assuming there is no Mg^{+2} present in the medium) to ensure that it is not rate limiting for struvite formation: $Mg(OH)_2$ solution (53% suspension) is used for this purpose. Similar to the solid fractions of the previous S/L separation, struvite was also assumed to be applied as a fertiliser to agricultural soils (see epigraph Fertilisation activities).

Table 6.5. Flow, mean composition and separation percentages from the struvite crystallisation stage in The Netherlands.

Composition	Permeate	Struvite effluent	Struvite (fertiliser)	Separation (%)
Flow (kg/d)	56.1	56.1	$3.60{\cdot}10^{-2}$	-
DM (g/kg)	12.5	12.5	-	-
COD (g/kg)	7.20	4.80	-	-
TN (g/kg)	1.95	1.85	167	5.13
N-NH_4^+ (g/kg)	1.60	1.50	167	6.25
TP (g/kg)	0.11	$1.60{\cdot}10^{-2}$	139	84.8
P-PO_4^{3-} (g/kg)	0.10	$1.40{\cdot}10^{-2}$	128	85.8

The nucleation zone is aerated with 2 L/min up to 10 L/min for about 12 h. It is assumed that an average of 6 L/min of air is fed to the struvite crystalliser during operation. Consequently, the energy consumption of the blower operation was included in the inventory (see epigraph Energy requirements).

- **Biological nitrogen removal**

Two different technologies were initially proposed for the BNR stage: nitritation/denitritation and partial nitritation/anammox. However, as the partners' project encountered some technical problems in relation to the latter, nitritation/denitritation was considered for the LCA study.

Satisfactory performance was supposed with removal percentages of ≈100% $N\text{-}NH_4^+$ and ≈80% TKN (Table 6.6). Glycerine was assumed to be used as carbon source with an organic matter concentration of 1.2 kg COD/L; the ratio of 2.1 between the carbon source and $N\text{-}NH_4^+$ in the influent was considered for calculations. In this way, and taking into account a density of 1.26 kg/L for glycerine and a rate of 0.08 kg $N\text{-}NH_4^+$/d, it was estimated that about 0.14 L of glycerin/d was added to the reactor.

Table 6.6. Flow, mean composition and removal percentages from the BNR stage in The Netherlands.

Composition	Struvite effluent	Final effluent	Removal (%)
Flow (kg/d)	56.1	56.1	-
DM (g/kg)	12.5	-	-
COD (g/kg)	4.80	-	-
TN (g/kg)	1.85	0.39	80.0
$N\text{-}NH_4^+$ (g/kg)	1.50	0.00	100
TP (g/kg)	$1.60 \cdot 10^{-2}$	-	-
$P\text{-}PO_4^{3-}$ (g/kg)	$1.40 \cdot 10^{-2}$	-	-

Nitrogen losses in terms of N_2O emissions can be also registered from this stage. However, only ≈0.12% of influent $N\text{-}NH_4^+$ (≈0.17 g N_2O/d) was assumed to be emitted into the atmosphere by the combination of intermittent patterns of aeration, anoxic feeding and anoxic carbon dosage (lab scale experimentation). With this new strategy, the final N_2O emissions are reduced by 99% in comparison with the baseline nitrification/denitrification process (up to ≈17% N_2O emissions).

Finally, it was assumed that an aeration rate of approximately 200 L/min was required over a period of 8 hours, so it was estimated that the blowers pumped about 118 m^3 of air daily during this stage (see epigraph Energy requirements).

- **Fertilisation activities**

As explained above, nutrients can be recovered from the digestate in the BS, as well as from the different fractions obtained from the management of digestate in S1. These nutrients can be used as organic fertilisers, partially avoiding the production and use of mineral ones in the market. Consequently, not only the environmental credits related to the avoided production of such mineral fertilisers were accounted for, but also those derived from the avoided emissions due to their application to soils (Table 6.7).

In this regard, avoided nitrogen emissions to air were estimated in terms of N_2O and NH_3, in agreement with the IPCC (2006) guidelines. The Tier 2 method was followed as the basis for calculations, using both available primary data together with the default emission factors provided by the method. Moreover, NO_3^- leaching rate was also accounted (IPCC, 2006) as well as PO_4^{3-} emissions to water using the ratio of 0.01 kg P-PO_4^{3-} per kg of applied P proposed by Rossier (1998). Regarding avoided fertilisers production, a MFE ranging from 41% to 62% was considered for N according to the literature (Birkmose et al., 2007; de Vries et al., 2011, 2012); 100% was only applied in the case of N from N concentrate and mineral struvite. As regards P, an average uptake rate of 97% was considered (Dalgaard et al., 2006; de Vries et al., 2011; Rahman et al., 2014). Nevertheless, it should be highlighted that the use of these organic fertilisers also generates emissions and discharges, which were also considered (see Organic fertilisers emissions in Table 6.7). Again, the IPCC guidelines (IPCC, 2006) were applied to estimate the related environmental burdens arising from the use of recovered nutrients as organic fertilisers.

Table 6.7. Emission rates from storage and application on agricultural soils during fertilisation activities in Dutch scenarios.

Fertiliser	Flow (kg/d)	Nutrients load	Organic fertilisers emissions[a]	MFE values	Avoided mineral fertilisers emissions	Avoided mineral fertilisers production[c]
N concentrate	1.32	N = 0.24 kg/d	N_2O = 4.40 g/d NH_3 = 68.0 g/d NO_3^- = 0.37 kg/d	N = 100%	N_2O = 4.40 g/d NH_3 = 34.0 g/d NO_3^- = 0.15 kg/d	N = 0.28 kg/d
Digestate	88.0	N = 0.36 kg/d P = 0.14 kg/d	N_2O = 5.71 g/d NH_3 = 0.25 kg/d NO_3^- = 0.40 kg/d PO_4^{3-} = 4.25 g/d	N = 62% P = 97%	N_2O = 1.81 g/d NH_3 = 0.01 kg/d NO_3^- = 0.15 kg/d PO_4^{3-} = 4.12 g/d	N = 0.12 kg/d P = 0.13 kg/d
Coarse fraction	13.2	N = 0.11 kg/d P = 0.12 kg/d	N_2O = 1.82 g/d NH_3 = 79.7g/d NO_3^- = 0.13 kg/d PO_4^{3-} = 3.62 g/d	N = 41% P = 97%	N_2O = 0.38 g/d NH_3 = 2.93 g/d NO_3^- = 0.03 kg/d PO_4^{3-} = 3.51 g/d	N = 0.02 kg/d P = 0.11 kg/d
Retentate	18.7	N = 0.13 kg/d P = 14.7 g/d	N_2O = 2.14 g/d NH_3 = 94.0 g/d NO_3^- = 0.09 kg/d PO_4^{3-} = 0.45 g/d	N = 41% P = 97%	N_2O = 0.45 g/d NH_3 = 3.46 g/d NO_3^- = 0.04 kg/d PO_4^{3-} = 0.44 g/d	N = 0.03 kg/d P = 0.01 kg/d
Struvite	36.1 (g/d)	N = 5.61 g/d P = 4.57 g/d	N_2O = 0.09 g/d NH_3 = 1.36 g/d NO_3^- = 7.45 g/d PO_4^{3-} = 0.14 g/d	N = 100% P = 97%	N_2O = 0.09 g/d NH_3 = 0.68 g/d NO_3^- = 7.45 g/d PO_4^{3-} = 0.14 g/d	N = 2.30 g/d P = 4.43 g/d
Water (avoided)	56.1	N = 16.4 g/d	N_2O = 2.44 g/d NH_3 = 5.31 g/d NO_3^- = 0.03 kg/d	N = 62%[b]	N_2O = 0.21 g/d NH_3 = 1.65 g/d NO_3^- = 0.02 kg/d	N = 0.01 kg/d

[a] Emissions from both organic fertilisers application and previous storage (if necessary); [b] It has been assumed the same behaviour for both N-rich water steam (from BNR stage) and the liquid fraction of the digestate in absence of specific references; [c] The amount of avoided N mineral fertiliser was estimated as follows: [N load (kg/d) – N emissions form storage (if necessary, kg/d) x MFE/100].

CH_4 emissions from the temporary storage of some products were also taken into consideration, so that alternative emissions rates were accounted for the assessment based on the properties of the stored fractions, as reported in Table 6.8. Regarding BS, an emission factor of 0.20 kg CH_4/kg COD in the digestate was estimated in accordance with the Organic Waste Digestion Project Protocol (2009), which proposes a generic way of determining CH_4 emissions regardless of the storage period, while focusing on the COD content of the digestate and the potential CH_4 yield. This factor would also be in line with the values reported in similar studies in the literature (Gioelli et al., 2011), considering 150 days of storage time (as in the present study) and the volatile solids (VS) content of the digestate from cattle slurry (and other farmyard manure). Focusing on CH_4 emissions from the storage of both the coarse fraction (centrifuge) and the retentate (membranes), the emission factor proposed by de Vries et al. (2011) was considered according to the storage of a mixture of pig and cattle manure for 90 days (3 months).

Table 6.8. Emission factors and CH_4 emissions related to storage actitivies of alternative products in The Netherlands.

Input/Output	Storage time (days)	Emission factor	CH_4 emissions (kg/d)
Digestate	Undefined	0.20 m^3 CH_4/m^3 digestate	0.125
Coarse fraction	90	0.17 kg CH_4/t digested solid fraction	0.002
Retentate	90		0.003

- **Transportation**

Both inputs and outputs transportation was considered. As for the supply of inputs, an average distance of 100 km was assumed for pig slurry, since it will come from different farms in the south of The Netherlands whereas only 5 km were considered for eco-frit transportation (personal communication).

The distances for transport of the different outputs (to agricultural soils) were calculated taking into account the area needed for the safe application of N, in accordance with European restrictions on the N content of soils (Table 6.9). Thus, they were assumed to be directly affected by the properties of each fertiliser product, mainly in terms of their primary nutrients (mainly N) and moisture content (digestate vs. concentrated fertilisers). In this regard, 170 kg N/ha was considered to be the limit for N in NVZ under the Nitrate Directive 91/676/EEC (1991), while P fertilisation was indirectly limited by N regulation.

Table 6.9. Tranport distances estimated for the different inputs and outputs for both scenarios in The Netherlands.

Input/Output	Flow (kg/d)	N content (kg/d)	Distance (km)
Pig slurry	74.0	-	100
Eco-frit	15.0	-	5.00
Digestate	88.0	0.19	0.02
N concentrate	1.32	0.24	0.02
Coarse fraction	13.2	0.06	0.01
Retentate	18.7	0.07	0.01
Struvite	0.04	$6.00 \cdot 10^{-3}$	$6.00 \cdot 10^{-3}$
Irrigation water (avoided)	56.1	0.02	0.30

- **Energy requirements**

Energy use rates (including both electricity and heat requirements) should be also quantified within the LCA study. The heating inputs are mainly derived from the AcoD stage coupled to the stripping process to maintain the operational temperature at an adequate level. A heat transfer coefficient (C) of 0.013 kW/(°C·m) was considered for calculations in both cases, as well as an input and output temperatures of 15 °C and 55 °C, respectively, and a height of 2.20 m for the digester; similarly, the following inventory information was used in the stripper unit: height = 2.50 m, T_{input} = 55 °C, T_{output} = 66 °C.

Table 6.10. Consumption factors applied to the main electric devices in The Netherlands.

Unit	Consumption factor	Reference
Mixing devices	0.0065 kWh/m³ tank	Metcalf & Eddy (2004)
Pumps	0.0385 kWh/m³ influent	Metcalf & Eddy (2004)
Decanter centrifuge	74.3 MJ/t influent (≈20.7 kWh/t influent)	Pöschl et al. (2010)

Electricity demand is shared among the different stages of the global system, either for the operation of certain units or in terms of pumping and blowing systems. In the case of blowers, the use of electricity was estimated taking into account the primary data on their power (kW) and operating time (hours). Table 6.10 summarises the main consumption ratios used in the other cases, together with the data sources.

- **Trace contaminants**

The influence of some of the most relevant trace contaminants on the final outputs were also accounted for using the USEtox database® (Hauschild et al., 2008; Rosenbaum et al., 2008). Accordingly, Table 6.11 displays a summary of the inventory data collected for the different streams in terms of veterinary drugs, heavy metals, pesticides and mycotoxines for both scenarios.

Table 6.11. Summary of trace contaminants considered in both scenarios: BS and S1.

Trace contaminant	Type	BS (mg/d)	S2 (mg/d)
Veterinary drugs	Flumequin	2.97	0.75
	Oxytetracyclin	0.61	0.47
	Sulfadiazin	1.06	0.32
Heavy metals	Chrome (Cr)	123	106
	Cooper (Cu)	2000	1590
	Zinc (Zn)	4862	3090
Pesticides	Chlorpropham	9.68	4.90
	Ethoxyquin	-	0.62
	Imazalil	-	0.12
Mycotoxins	Zearalenon	1.38	0.68

- **Background inventory**

Finally, secondary data was used to complete the background inventory. Data on electricity generation (Dutch profile), diesel and other fossil fuels production, chemicals manufacture, combustion emissions and transportation equipment were taken from the ecoinvent® database (Althaus et al., 2007; Dones et al., 2007; Spielmann et al., 2007; Wernet et al., 2016).

6.2.3 Impact assessment

The environmental results for the different scenarios were evaluated in terms of six main indicators: CC, TA, FE, ME, HT and FT (freshwater toxicity). Characterisation factors provided by the ReCiPe Midpoint (H) method (Goedkoop et al., 2013) were used for acidification (TA) and eutrophication (FE, ME) estimations, while IPCC (2013) and USEtox databases (Hauschild et al., 2008; Rosenbaum et al., 2008) were considered for CC and toxicity calculations, respectively.

6.2.4 Results and discussion

Table 6.12 reports the comparative results of both scenarios from an environmental perspective. Accordingly, BS was found to be responsible for the largest impacts in all categories, with the exception of FE. This is mainly due to the greatest GHGs emission (CH_4 and N_2O) from storage activities (for CC, TA, ME) together with the highest concentrations of different micro-pollutants in the digestate stream (HT). On the contrary, the comparison shows that S1 is the most environmental-friendly option in most categories.

Table 6.12. Environmental results for the six impact categories evaluated for the Dutch scenarios.

Impact category	Units	BS	S1
CC	kg CO_2 eq	4.26	0.94
TA	kg SO_2 eq	0.70	0.54
FE	g P eq	-1.33	-1.12
ME	g N eq	87.1	65.2
HT	CTUh	$3.10 \cdot 10^{-6}$	$2.60 \cdot 10^{-6}$
FT	CTUe	148	108

Focusing on the relative contributions for each category (except for toxicity indicators), the different processes or activities involved in all scenarios were grouped into eight contributing factors: transport (inputs and outputs), chemicals, energy use (both electric and calorific energy), diffuse emissions (mainly from storage and fertilisation activities) and avoided processes as a result of the energy generated (avoided energy) and the nutrients recovered (avoided fertiliser production, avoided fertiliser emissions, avoided water) in the different fractions throughout the life cycle of both scenarios.

Climate change

Figure 6.3 shows diffuse emissions as one of the most important contributing factors to CC, independently of the scenario evaluated. These emissions mainly come from storage activities (CH_4 and N_2O) together with the scrubbing and BNR stages (NH_3 and N_2O, respectively). Moreover, as explained above, the impacts of these diffuse emissions from digestate storage are responsible for the worst environmental profile of BS in comparison with S1, with contributions up to 38%, of which about 62% are CH_4 emissions.

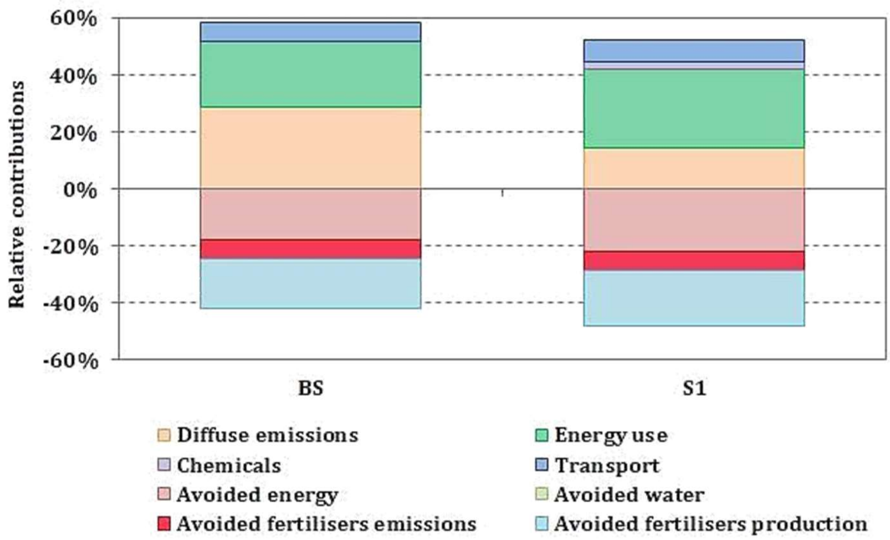

Figure 6.3. Relative contributions to CC from the main processes involved in the different Dutch scenarios.

In contrast, these emissions represent only 23% of diffuse emissions in S1, which are more relevant for both direct (N_2O) and indirect (NH_3, NO_3^-) nitrogen emissions, with contributions of around 77%. Energy requirements have also a significant influence (up to 30%), while transportation and chemicals use have minor environmental consequences.

Conversely, energy generation (avoided energy) and nutrients recovery (avoided fertilisers production) partially offset (up to 22%) the environmental burdens associated with energy use and diffuse emissions from fertilisation activities. Finally, transportation and avoided fertilisers emissions have a similar negative and positive contribution (about 6%), respectively, to the environmental profile of both scenarios.

Terrestrial acidification

Figure 6.4 demonstrates how the impacts from diffuse emissions are decisive for TA in both scenarios. In this sense, the NH_3 discharged (into air) from the scrubbing stage is responsible for up to 99% of these impacts; nitrogen emissions from other stages (such as storage and further nutrients application) also contribute to acidification problems. Avoided fertilisation emissions show the most favourable contribution to both profiles.

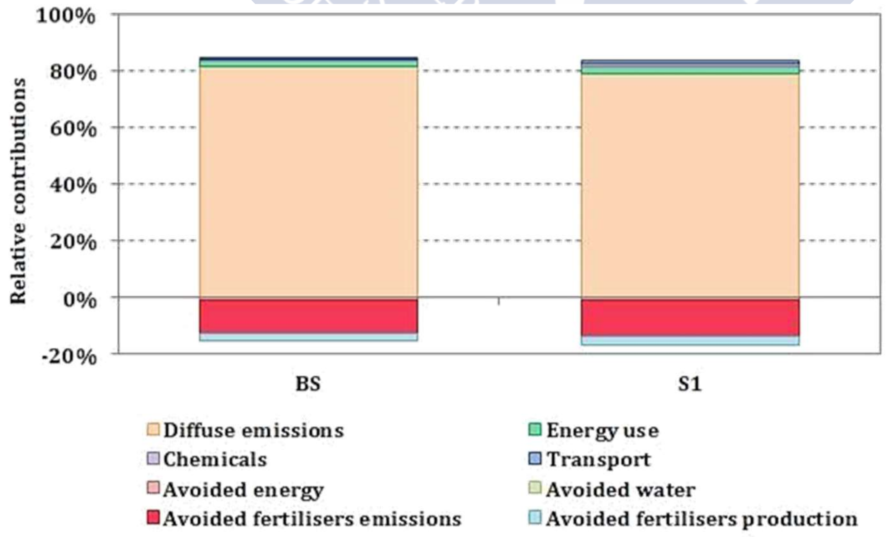

Figure 6.4. Relative contributions to TA from the main processes involved in the different Dutch scenarios.

Eutrophication

Similar performance can be found for ME compared to TA impacts, so that diffuse emissions stand out again as the main hotspot in all scenarios (Figure 6.5). However, in this case, the impacts associated with this contributing factor come mainly from NO_3^- discharged to water during fertilisation activities (about 68%). Similarly, avoided fertilisers emissions show also the most favourable contribution, although this factor gains relevance (up to 38%) in all scenarios in relation to TA profiles. The other factors are not relevant in any of the scenarios.

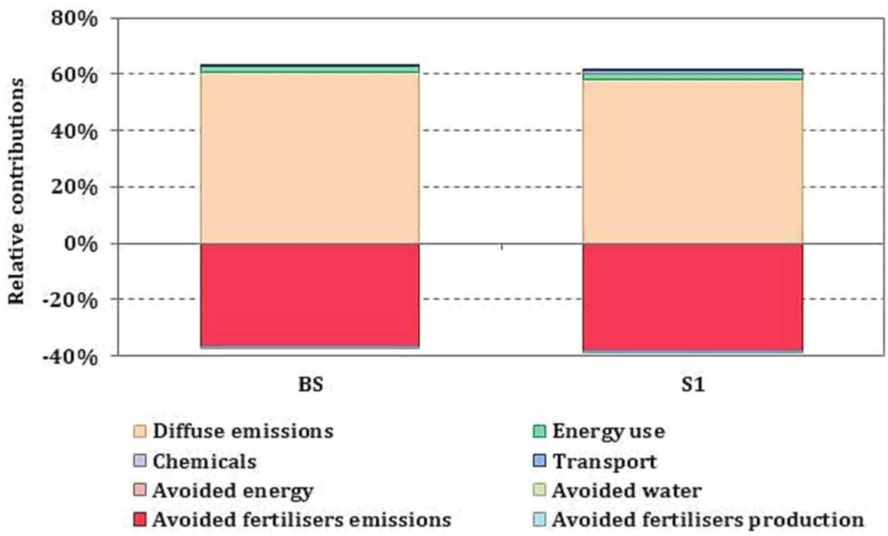

Figure 6.5. Relative contributions to ME from the main processes involved in the different Dutch scenarios.

According to Figure 6.6, different factors have a significant positive and negative influence on the environmental profiles in terms of FE. Thus, diffuse emissions and energy use have similar unfavourable effect (about 20%) in all the scenarios, while chemical use shows also a lower contribution to S1; since minimal chemicals requirements are necessary in BS, this slightly favours its environmental profile. On the other hand, most of the avoided processes equally offset the environmental impacts in both scenarios.

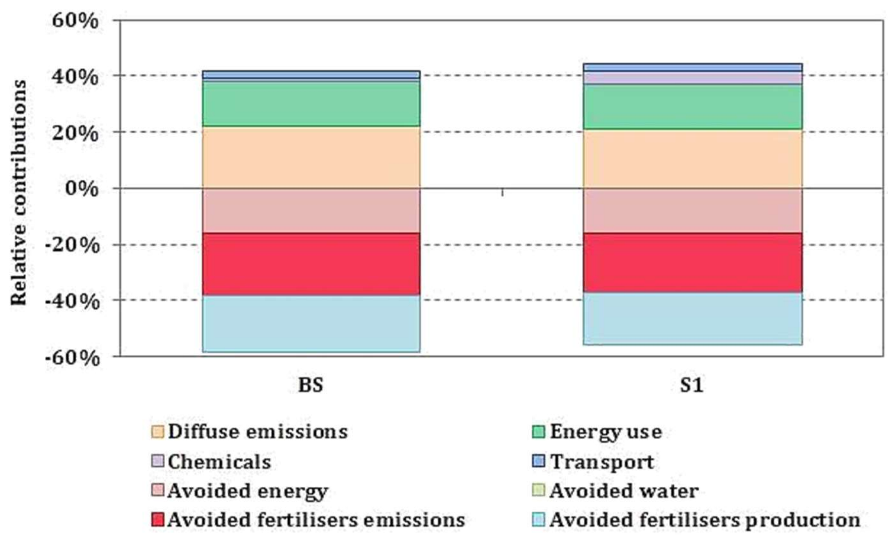

Figure 6.6. Relative contributions to FE from the main processes involved in the different Dutch scenarios.

Toxicity

In both scenarios, environmental impacts from trace contaminants in toxicity categories were found to be mainly related to the presence of mycotoxins (Zearalenon) and heavy metals, especially in terms of Cu and Zn. The influence of the latter compounds was to be expected since much higher concentrations of heavy metals were identified in the different fractions, in comparison with the other contaminants (see Table 6.11). However, additionally, more unfavourable characterisation factors associated with mycotoxins (USEtox database) result in even greatest impacts on human toxicity, shared with Zn impact.

Regarding the other contaminants, both pesticides and veterinary drugs, they were responsible for minor consequences on human and environmental health.

6.3 MEM PROTOTYPE IN SPAIN

6.3.1 Goal and scope definition

This section aims to assess and compare the environmental impacts associated with the alternative scenarios defined for the valorisation of livestock waste within the framework of the MEM project in Spain. A cradle-to-grave study was developed, from raw materials production, going through the valorisation processes to the final disposal of waste. In this sense, a mixture of three substrates is fed to the system (Table 6.13) in the Spanish pilot plant: cow manure (52.0%), pig manure (43.0%) and segregates (5.0%).

Table 6.23. Flow and mean composition of feeding mixture in Spain: dry matter (DM); Chemical Oxygen Demand (COD); Total Nitrogen (TN); Total Phosphorous (TP).

Composition	Manure mixture	Segregates	Feeding mixture
Flow (kg/d)	118.75	6.25	125
DM (kg/d)	72.9	92.2	73.9
COD (g/kg)	75.4	154	79.3
TN (g/kg)	4.50	0.70	4.36
N-NH_4^+ (g/kg)	2.09	0.06	1.99
TP (g/kg)	0.83	0.02	0.79
P-PO_4^{3-} (g/kg)	0.63	0.01	0.60

- **Functional unit**

Similar to the previous section, a mass-based FU was selected as the basis for the calculations. However, in this case, 125 kg were considered to be fed daily to the system as a variable common to all management scenarios (at pilot scale) for comparison purposes (see Table 6.13).

- **System description**

In the Spanish case study, two main scenarios were evaluated, the outlines of which are presented in Figures 6.7 – 6.9. Similar to the previous section, the base scenario (BS) represents more conventional practices based primarily on the AcoD process to produce biogas (energy source) and digestate (nutrient source); S1 (MEM Prototype) represents the design proposed in the MEM project for the Spanish scenario, in which further S/L separation, struvite precipitation and BNR are coupled to AcoD to enhance the added value of the digestate stream and potential products.

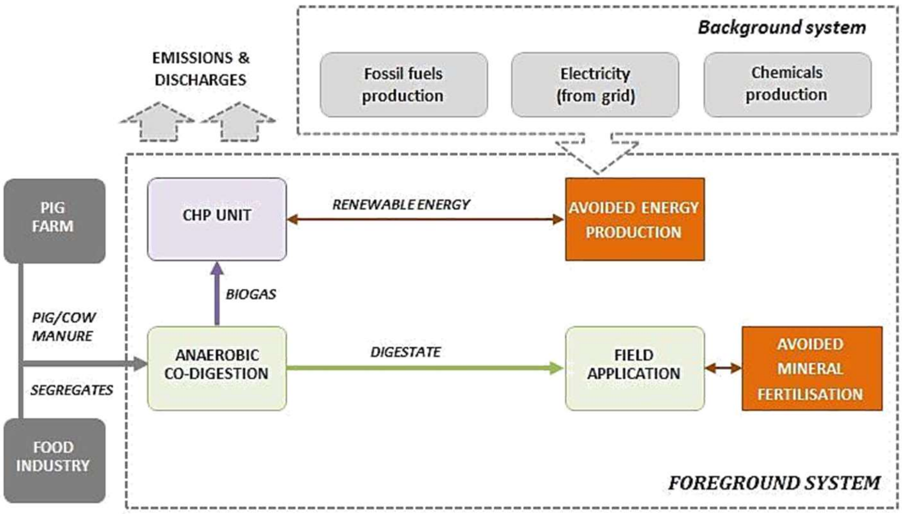

Figure 6.7. Scheme of the system boundaries corresponding to BS in Spain. Key: Brown boxes include the avoided impacts due to organic fertilisation and renewable energy generation. Grey boxes represent the excluded processes.

Additionally, a third potential scenario was also defined in Spain (Figure 6.8): MEM Acidification (S2). Its configuration is similar to that of S1, but with an acidification stage prior to centrifugation, followed by an additional S/L separation by means of UF membranes. Again, it is assumed that biogas is recovered as an energy source in a CHP unit in all scenarios.

Since a mesophilic digester is considered to be used in the Spanish configurations, the additional stages of ammonia stripping and air scrubbing are not necessary in the scenarios assessed in this section.

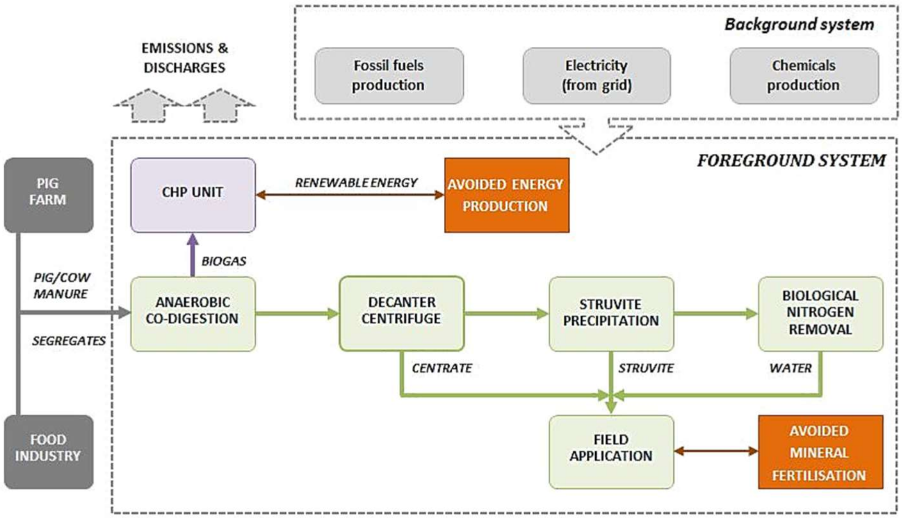

Figure 6.8. Scheme of the system boundaries corresponding to S1 in Spain. Key: Brown boxes include the avoided impacts due to organic fertilisation and renewable energy generation. Grey boxes represents excluded processes.

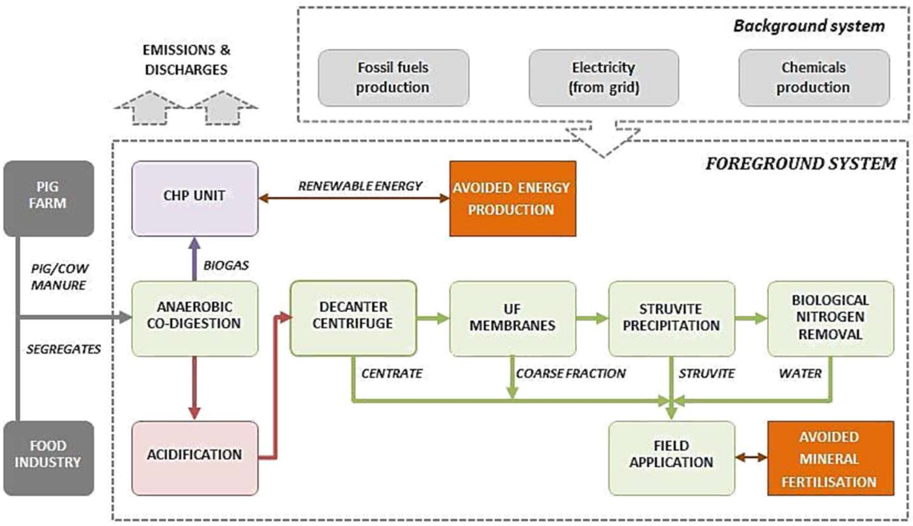

Figure 6.9. Scheme of the system boundaries corresponding to S2 in Spain. Key: Brown boxes include the avoided impacts due to organic fertilisation and renewable energy generation. Grey boxes represents excluded processes.

- **Allocation rules**

Based on the same premise as the Dutch study, a system expansion approach was applied to take into account the valorisation of the different products. This means that the analysis not only took into account the environmental impacts associated with their production processes, but also the environmental credits associated with their subsequent recovery as suppliers of energy (biogas) and nutrients (recovered fractions).

On the other hand, both livestock (cow/pig) manure and segregates were considered as residues from livestock farms and food industry, respectively, so that the impacts related to their production were not taken into account in the environmental impacts of the scenarios assessed.

6.3.2 LCI analysis

As in the case of the Dutch scenarios, primary data provided by partners throughout the project lifespan was always prioritised. However, secondary data were also used to complete the inventory, where necessary. A detailed description of the inventory information and data sources for each stage is reported in the following epigraphs.

- **Anaerobic co-digestion**

A mesophilic AcoD (at 37 °C) is carried out in Spanish scenarios to obtain two main products: biogas and digestate (Table 6.14). While direct application is assumed in BS, digestate management for nutrients recovery in different fractions is considered in the other cases (S1 and S2). Moreover, a biogas rate of 2.70 m^3/day was estimated, although it was assumed to be partially discharged (≈1.5%) to the atmosphere due to biogas losses from the digester and the CHP unit (de Vries et al., 2012). Similarly, emissions of N_2O (0.62 mg N_2O/m^3 biogas) and NO_x (1.22 g NO_x/m^3 biogas) from the biogas stream were also included in the inventory (de Vries et al., 2012).

Focusing on energy recovery, a calorific value of 5.42 kWh/m^3 biogas was considered, based on a 60% content of CH_4 in biogas with a calorific value of 8.33 kWh/m^3 CH_4 (IDAE, 2014). Moreover, electric and heat efficiencies of 33% and 50% were assumed, respectively, so that about 4.39

kWh/day of electricity is generated from organic sources (avoided electricity), as well as 6.65 kWh/day of heat (avoided heating); however, it is assumed that about 3% (0.13 kWh/d) of electricity is used in the CHP operation (Pöschl et al., 2010).

Table 6.14. Flow and mean composition of digestate from AcoD in Spain.

Composition	Feeding mixture	Biogas	Digestate
Flow (kg/d)	125	2.70[a]	122
DM (g/kg)	73.9	-	49.9
COD (g/kg)	79.3	-	42.4
TN (g/kg)	4.36	-	4.45
$N\text{-}NH_4^+$ (g/kg)	1.99	-	2.69
TP (g/kg)	0.79	-	0.75
$P\text{-}PO_4^{3-}$ (g/kg)	0.60	-	0.60

[a] m^3/d.

- **Solid/liquid separation**

S/L separation takes place in a single stage through centrifugation, in such a way that two main streams are obtained: the centrate (liquid) and the coarse (solid) fraction (Table 6.15). Related environmental burdens and credits from the use of the solid fraction as organic fertiliser are also addressed for analysis (see epigraph Fertilisation activities).

Table 6.15. Flow, mean composition and separation percentages of the centrifugarion stage in Spain.

Composition	Digestate	Centrate	Coarse fraction	Separation (%)
Flow (kg/d)	122	106	16.5	13.5
DM (g/kg)	49.9	25.6	205	-
COD (g/kg)	42.4	27.8	136	23.5
TN (g/kg)	4.45	3.92	7.87	23.8
$N\text{-}NH_4^+$ (g/kg)	2.69	2.46	4.18	20.9
TP (g/kg)	0.75	0.22	4.19	75.1
$P\text{-}PO_4^{3-}$ (g/kg)	0.60	0.12	3.70	82.5

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- **Struvite crystallisation**

Analogous to the Dutch case study, it was assumed that ammonium struvite ($MgNH_4PO_4{\cdot}6H_2O$) was obtained from the struvite precipitation stage (see Eq. 1). While NH_4^+ is sufficiently present in the reaction medium, additional Mg^{2+} must be added as $Mg(OH)_2$ at the rate of 15 L/d. Struvite is then assumed to be used as a fertiliser (Table 6.16). As far as aeration requirements are concerned, it is considered that the crystalliser is fed with an average air velocity of 6 L/min during the operation time.

Table 6.16. Flow, mean composition and separation percentages of the struvite crystallisation stage in Spain.

Composition	Centrate	Struvite effluent	Struvite (fertiliser)	Separation (%)
Flow (kg/d)	106	121	0.28	-
DM (g/kg)	25.6	22.4	-	-
COD (g/kg)	27.8	24.3	4.29	0.04
TN (g/kg)	3.92	3.40	11.4	0.78
$N\text{-}NH_4^+$ (g/kg)	2.46	2.12	13.2	1.41
TP (g/kg)	0.22	0.13	23.6	29.0
$P\text{-}PO_4^{3-}$ (g/kg)	0.12	0.06	18.9	41.0

- **Biological nitrogen removal**

Successful performance was considered in nitritation/denitritation stage with removal percentages of ≈100% and ≈62% for $N\text{-}NH_4^+$ and TKN, respectively (Table 6.17). Glycerine (1.26 kg/L) was added as an external carbon source, taking into account a ratio of 2.1 in relation to the $N\text{-}NH_4^+$ influent (0.26 kg $N\text{-}NH_4^+$/day); therefore, it was estimated that about 0.48 L of glycerin/day was added to the reactor.

Moreover, it was assumed that a daily aeration rate of approximately 1.33 kg O_2 would be pumped by blowers during this stage. Regarding nitrogen emissions, about 0.48 g N_2O/d were assumed to be released to air (≈0.12% of influent $N\text{-}NH_4^+$) during this stage.

Table 6.17. Flow, mean composition and removal percentages of the BNR stage in Spain.

Composition	Struvite effluent	Final effluent	Removal (%)
Flow (kg/d)	121	122	-
DM (g/kg)	22.4	20.3	-
COD (g/kg)	24.3	20.7	14.0
TN (g/kg)	3.40	1.28	62.0
N-NH_4^+ (g/kg)	2.12	0.00	100
TP (g/kg)	0.13	0.13	-
P-PO_4^{3-} (g/kg)	0.06	0.06	-

- **Fertilisation activities**

Diffuse emissions can be registered due to the use of recovered nutrients (fractions) in fertilisation activities (see Organic fertilisers emissions in Table 6.18); however, related avoided impacts can also be accounted for in terms of N_2O and NH_3, taking into account the Tier 2 method proposed by the IPCC guidelines (IPCC, 2006). NO_3^- leaching was also estimated (IPCC, 2006), together with PO_4^{3-} emissions to water at the ratio of 0.01 kg P- PO_4^{3-} per kg of applied P proposed by Rossier (1998). Regarding avoided fertilisers production, a MFE value ranging from 41% to 62% was assumed again for N, according to literature (Birkmose et al., 2007; de Vries et al., 2011, 2012), except for 100% applied to mineral struvite. The uptake rate of P was considered to be 97% (Dalgaard et al., 2006; de Vries et al., 2011; Rahman et al., 2014). Nevertheless, it should be noted that the use of these organic fertilisers also generates emissions and discharges, which have been also considered. Again, the IPCC guidelines (IPCC, 2006) were considered to estimate the inventory data derived from the use of recovered nutrients as organic fertilisers. Finally, CH_4 emissions from storage were also estimated (see Table 6.8), according to the emission factors reported in the literature (Gioelli et al., 2001; Organic Waste Digestion Project Protocol, 2009; de Vries et al., 2011).

Table 6.18. Emission rates from storage and application on agricultural soils during fertilisation activities in Spanish scenarios.

Fertiliser	Flow (kg/d)	Nutrients load	Organic fertilisers emissions[a]	MFE values	Avoided mineral fertilisers emissions	Avoided mineral fertilisers production[c]
Digestate	122	N = 0.54 kg/d P = 0.09 kg/d	N_2O = 8.71 g/d NH_3 = 0.38 kg/d NO_3^- = 0.62 kg/d PO_4^{3-} = 2.81 g/d	N = 62% P = 97%	N_2O = 2.75 g/d NH_3 = 0.02 kg/d NO_3^- = 0.23 kg/d PO_4^{3-} = 2.73 g/d	N = 0.18 kg/d P = 0.09 kg/d
Coarse fraction	16.5	N = 0.13 kg/d P = 0.07 kg/d	N_2O = 8.02 g/d NH_3 = 91.0 g/d NO_3^- = 0.15 kg/d PO_4^{3-} = 2.11 g/d	N = 41% P = 97%	N_2O = 0.43 g/d NH_3 = 3.35 g/d NO_3^- = 0.04 kg/d PO_4^{3-} = 2.05 g/d	N = 0.03 kg/d P = 0.07 kg/d
Struvite	0.28	N = 3.67 g/d P = 5.29 g/d	N_2O = 0.06g/d NH_3 = 0.89 g/d NO_3^- = 4.88 g/d PO_4^{3-} = 0.16 g/d	N = 100% P = 97%	N_2O = 0.06 g/d NH_3 = 0.45 g/d NO_3^- = 4.88 g/d PO_4^{3-} = 0.16 g/d	N = 3.67 g/d P = 5.13 g/d
Water (avoided)	122	N = 0.16 kg/d P = 16.4 g/d	N_2O = 2.44 g/d NH_3 = 0.04 kg/d NO_3^- = 0.21 kg/d PO_4^{3-} = 0.50 g/d	N = 62%[b] P = 97%	N_2O = 1.52 g/d NH_3 = 0.01 kg/d NO_3^- = 0.13 kg/d PO_4^{3-} = 0.50 g/d	N = 0.10 kg/d P = 0.02 kg/d

[a] Emissions from both organic fertilisers application and previous storage (if necessary); [b] It has been assumed the same behaviour for both N-rich water steam (from BNR stage) and the liquid fraction of the digestate in absence of specific references; [c] The amount of avoided N mineral fertiliser was estimated as follows: [N load (kg/d) – N emissions form storage (if necessary, kg/d) x MFE/100].

- **Transportation**

Regarding the inputs supply, both cow and pig manures were assumed to be close to the pilot plant facilities (personal communication), so that negligible distance was considered. However, an average distance of 80 km was assumed for segregates, coming from a sugar factory in NE Spain. The transport distances for the different outputs (Table 6.19) were calculated on the basis of the legal limitations on nitrogen application (170 kg N/ha) in NVZ in Europe (Nitrate Directive, 1991).

Table 6.19. Transport distances estimated for the different inputs and outputs of both scenarios in Spain.

Input/Output	Flow (kg/d)	N content (kg/d)	Distance (km)
Pig/cow manure	118.75	-	≈0.00
Eco-frit	6.25	-	80.0
Digestate	122	0.28	0.02
Coarse fraction	16.5	0.07	0.01
Struvite	0.28	$4.00 \cdot 10^{-3}$	$2.00 \cdot 10^{-3}$
Irrigation water (avoided)	122	0.16	0.02

- **Energy requirements**

Table 6.10 provides a summary of the main consumption ratios and calculations considered for the estimation of electricity needs, similar to the Dutch scenarios. For the heat inputs at the AcoD stage, a heat transfer coefficient (C) of 0.013 kW/(°C·m) was considered, together with input and output temperatures of 15 °C and 37 °C, respectively.

- **Background inventory**

Finally, secondary data taken from literature and commercial databases were used to complete the background inventory, involving electricity generation (Spanish profile), agro-chemicals production, combustion emissions from fuel use in pilot plant machinery and transportation equipment (Althaus et al., 2007; Dones et al., 2007; Spielmann et al., 2007; Wernet et al., 2016).

- **MEM Acidification (S2): additional acidification stage**

As explained above, an additional scenario is considered for the Spanish case study taking into account the acidification of digestate prior to S/L separation in the decanter centrifuge. Therefore, about 0.02 kg H_2SO_4 solution (40%) was assumed to be used for acidification purposes: 0.98 kg H_2SO_4 was estimated to be added for a digestate rate of 122 kg/day. Moreover, an additional S/L separation stage by means of UF membranes was also considered to be necessary. The following stages are analogous to the scenario without acidification (S1), so that the same assumptions as reported above can be assumed. In this sense, a summary of main inventory data for this additional scenario is shown in Annex I (Supplementary material).

6.3.3 Impact assessment

The environmental outcomes of the different scenarios were evaluated according to four environmental indicators: CC, TA, FE and ME. Contrarily to the Dutch case study, no toxicity categories were considered for the assessment in Spain, due to the absence of primary data available involving trace contaminants concentrations throughout the global configuration. Again, characterisation factors provided by ReCiPe Midpoint (H) method (Goedkoop et al., 2013) were applied for acidification (TA) and eutrophication (FE, ME) indicators, while IPCC (2013) was used for CC calculation.

6.3.4 Results and discussion

Table 6.20 reports the comparative analysis on the basis of the overall results for all the scenarios under evaluation. This table shows BS as the worst environmental alternative in most impact categories (except for FE where it reports the lowest impacts). It should be mainly due to the highest diffuse emissions registered in this scenario, especially in terms of the CH_4 emitted to the atmosphere from the storage stage of the digestate. These emissions have a significant effect on both CC and TA, while N_2O emissions are also responsible for the environmental burdens on ME. Conversely, S1 would be the most environmental-friendly alternative in all impact categories (except for FE).

Table 6.20. Environmental results for the four impact categories evaluated for the Spanish scenarios.

Impact category	Units	BS	S1	S2
CC	kg CO_2 eq	5.48	0.49	1.52
TA	kg SO_2 eq	0.88	0.28	0.43
FE	g P eq	-0.78	-0.36	-0.01
ME	g N eq	125	67.8	77.5

As in the Dutch scenarios, the following sections also show the contributions for each impact category for the three Spanish scenarios. To this aim, the different processes or activities involved were again grouped into eight contributing factors: transport, chemicals, energy use, diffuse emissions and avoided processes as a consequence of energy generation and nutrients recovery in all the scenarios.

Climate change

According to Figure 6.10, both energy use and diffuse emissions show the greatest influence on the environmental impacts of all scenarios in terms of CC consequences. As explained above, the influence of diffuse emissions is particularly significant in BS due to CH_4 emissions (0.196 kg CH_4/d) associated with digestate storage, with contributions up to 68%, although N_2O emissions also have a relevant impact (26%). It should be also stated the important effect of chemicals usage in the others scenarios, especially in S2 (around 13%) since a solution of H_2SO_4 must be used for acidification purposes.

In contrast, avoided energy due to energy generation in the CHP unit is responsible for the largest environmental credits (up to 30%) in all scenarios, followed by avoided fertiliser production (around 10%) and avoided fertilisers emissions (around 5%). The other factors (transport and avoided water) are not relevant in any of the scenarios.

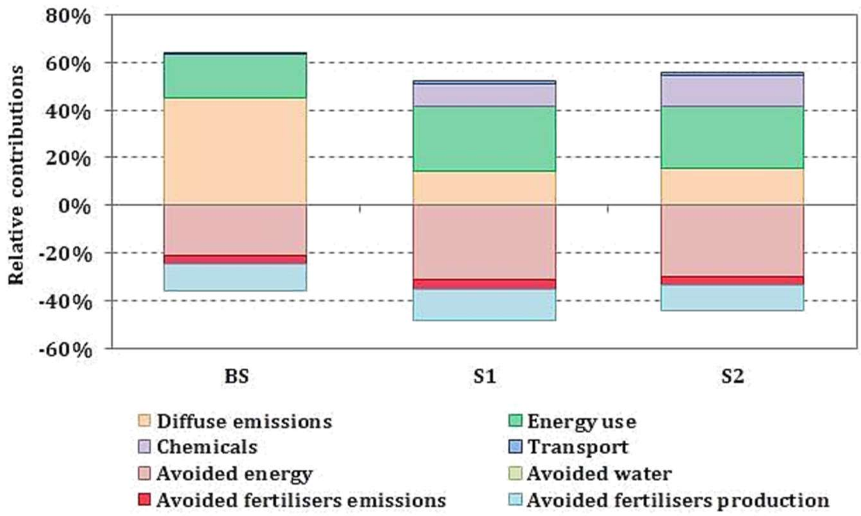

Figure 6.10. Relative contributions to CC from the main processes involved in the different Spanish scenarios.

Terrestrial acidification

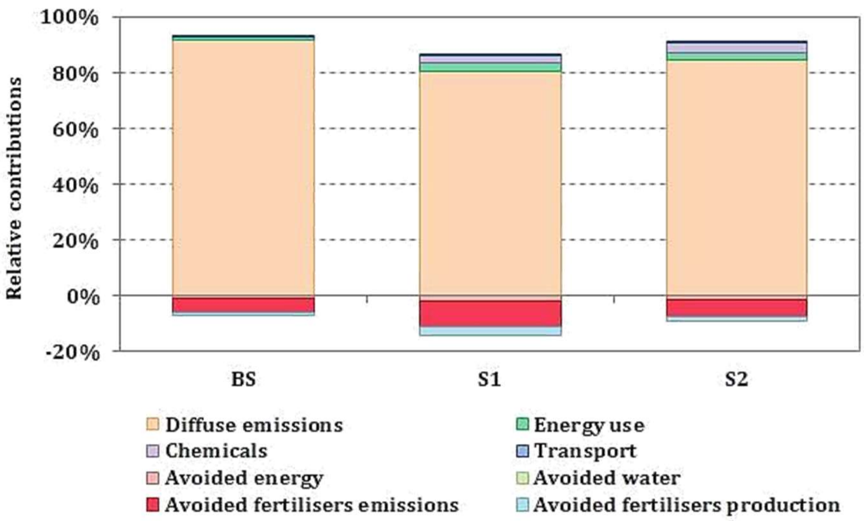

Figure 6.11. Relative contributions to TA from the main processes involved in the different Spanish scenarios.

Focusing on Figure 6.11, it could be concluded that diffuse emissions play a critical role in TA results, regardless of the scenario assessed. In fact, emissions from the storage stages (mainly N_2O emissions) and during fertilisation activities (N_2O, NH_3, NO_3^-, PO_4^{3-}) are responsible for the main environmental impacts, with contributions of around 85%. In the same line, avoided fertilisation emissions due to the use of recovered nutrients as organic fertilisers (that partially substitute mineral ones) can be found as the main credit that positively contribute to all profiles.

Eutrophication

Analogous to acidification results, diffuse emissions show a critical role on the environmental profile of the different scenarios in terms of ME (Figure 6.12). However, in this case, other factors such as chemicals use and energy consumption gain relevance (up to 10%) in all scenarios, except for BS as chemicals are not required. Similarly, avoided fertiliser emissions again have the greatest environmental benefits (above 20%), mainly due to the use of recovered organic nitrogen (from organic fractions such as the coarse fraction of centrifugation) as fertiliser in agricultural soils.

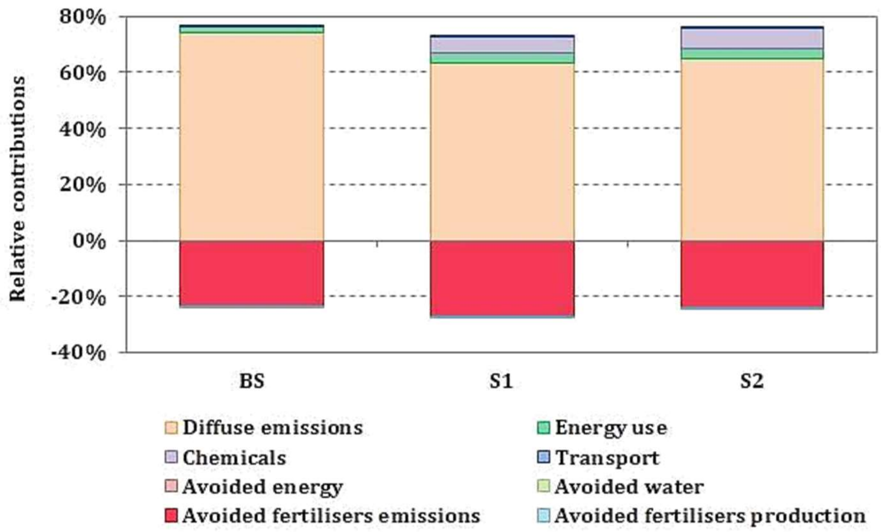

Figure 6.12. Relative contributions to ME from the main processes involved in the different Spanish scenarios.

Finally, paying attention to FE in Figure 6.13 demonstrates the relevance of most contributing factors. Thus, diffuse emissions and energy use have a similar negative influence (about 20%) in all scenarios assessed. In the same line, avoided processes (except for avoided water with a negligible effect) equally contribute to the environmental credits. It should also be noted that, in this category, the absence of chemical requirements favours the performance of the BS compared to the other scenarios, where the use of chemicals is mainly necessary for the struvite precipitation ($Mg(OH)_2$) and BNR (glycerine) stages.

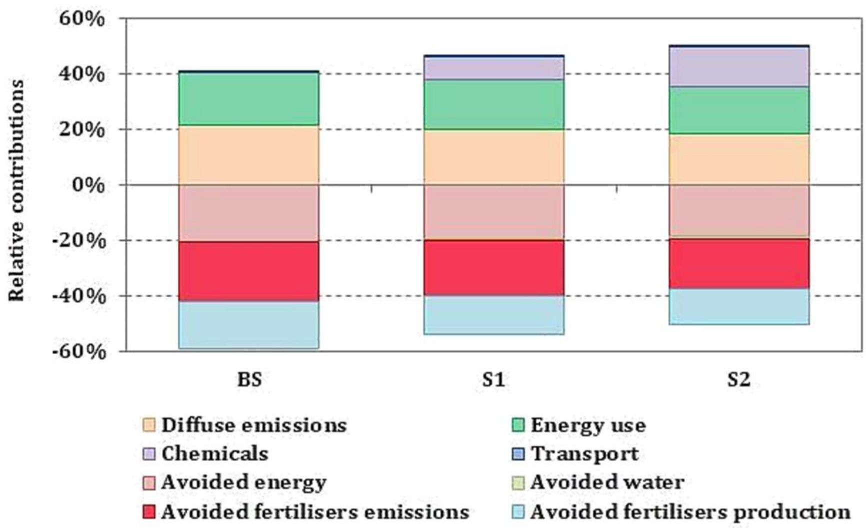

Figure 6.13. Relative contributions to FE from the main processes involved in the different Spanish scenarios.

6.4 CONCLUSIONS

This chapter assessed the potential for improvement of current livestock manure management strategies, following the approach proposed by the MEM project: manure as a source of nutrients. Accordingly, two case studies were evaluated involving pig (The Netherlands) and cow (Spain) manure within European boundaries; similarly, two main scenarios were compared in both cases: conventional practices (BS) vs. MEM prototypes (S1).

The comparative results reported the scenarios proposed within the MEM framework as the most environmentally friendly alternatives, regardless of the impact category and case study. Nitrogen emissions are mainly responsible for the large impacts on acidification and eutrophication potentials (especially ME), while energy use is gaining importance in the other categories. However, the generation of renewable energy from the recovery of biogas in the CHP unit largely offsets energy-based impacts, so that avoided processes and emissions from nutrient recovery are responsible for the highest environmental credits.

Therefore, strategies based on energy recovery and digestate valorisation in terms of high quality recovered fractions could be environmentally justified alternatives to more conventional practices to date. However, in the absence of a clear predominance over the direct use of digestate as a fertiliser, the preference for one or the other scenario still largely depends on the particular situation. In this regard, the integration of social and economic criteria could provide added value to the decision-making process.

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SECTION III

URBAN FRAMEWORK

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CHAPTER 7. ENVIRONMENTAL ANALYSIS OF MSW MANAGEMENT: LCA APPROACH

Summary

The generation of MSW continues to be responsible for several environmental and human health problems, especially in developing regions, but still is relevant for developed countries. This situation poses a potential challenge for society and responsible entities in the search for improved management alternatives that allow progress towards more sustainable scenarios, respecting the particularities and limitations of each specific environment and location.

Accordingly, two case studies were evaluated in this chapter, involving Galicia (Spain) and Astana (Kazakhstan) as representatives of developed and developing regions, respectively. The environmental profiles and potential environmental benefits of current and alternative MSW management schemes, based mainly on low-waste generation, material recovery and energy production, were evaluated on the basis of LCA indicators.

Environmental results showed that GHG emissions are the major contributor to climate change effects, while landfilling disposal of the non-recovered fraction of recyclable materials and energy requirements were responsible for the highest contributions to the other categories. However, the reuse of recycled materials largely offsets the related environmental burdens, along with the generation of renewable energy. In comparative terms, a comprehensive waste treatment system based mainly on aerobic treatment of biowaste in detriment of waste-to-energy systems could significantly reduce the environmental impacts of MSW management in developed regions. Similarly, the proposed treatment scenarios that encourage renewable energy generation and higher recycling rates were responsible for reducing negative environmental burdens compared to current practices in Astana (developing region). The main findings are expected to support stakeholders in improving the environmental context by addressing MSW management concerns in these regions.

CHAPTER 7 - INDEX

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7.1 INTRODUCTION TO MSW MANAGEMENT: DEVELOPED AND DEVELOPING REGIONS

Nowadays, the production of MSW continues to have responsibility for several environmental impacts affecting human health and ecosystems quality, as well as serious economic burdens (Bernstad and la Cour Jansen, 2012; Iriarte et al., 2009). In this context, substantial progress has been made in recent decades in the most economically developed regions of the world (Castaldi, 2014; Commission of the European Communities, 2005). In fact, conventional management alternatives have been improved and/or closed down, while new technologies have been designed and promoted (Achillas et al., 2013; Vergara and Tchobanoglous, 2012). Practices involving the use of biodegradable waste for energy generation linked to growing recovery of recyclable waste are contributing to prevent the related negative impacts (Castaldi, 2014). However, further actions are encouraged to move towards a broader environmentally sustainable context in developed countries (Castaldi, 2014; Vergara and Tchobanoglous, 2012).

On the contrary, despite the positive efforts made in recent times, waste remains a real problem for economically disadvantaged areas (Bezama et al., 2007; Wilson et al., 2015). Policies on MSW management have changed in response to societal and environmental concerns (Antonopoulos et al., 2014; Wilson et al., 2015). However, modern waste management practices have not yet been in line with the continued growth of waste generation in developing countries (Bezama et al., 2007; Wilson et al., 2015). Inadequate collection systems and disposal facilities continues to generate potential risks to public health and environmental pollution (Bezama et al., 2007; Othman et al., 2013). The lack of comprehensive and adequate waste management systems therefore affirms the importance of promoting a thorough evaluation of the state-of-the-art and what needs to be done to address major environmental problems in developing regions (Inglezakis et al., 2017; Othman et al., 2013).

The adoption of the LCA methodology has been recommended with the aim of evaluating alternative technologies to make the most of a comprehensive MSW management (Clearly, 2009; Laurent et al., 2014a,b; Tock and Schummer, 2017). Indeed, LCA has been widely accredited for its ability to assess the potential environmental damages associated with different waste treatment configurations in developed and developing regions (Bernstad and la Cour Jansen, 2012; Clearly, 2009; Laurent et al., 2014a,b; Othman et al., 2013; Tock and Schummer, 2017). However, the lack of consistency in methodological choices, as well as particular local conditions, makes it more difficult to achieve an absolute ranking of alternative schemes for MSW management (Bernstad and la Cour Jansen, 2012; Clearly, 2009). Therefore, it should be desirable to analyse each situation individually pending progress towards a common methodological framework among LCA practitioners (Clearly, 2009).

Accordingly, the main goal of this Chapter 7 was to evaluate the environmental context of available and potential strategies for MSW management under both conductive (developed) and non-conductive (developing) conditions, following the LCA approach. To this aim, two case studies were performed in two specific locations: Galicia (Spain) and Astana (Kazakhstan) as representatives of developing and developed regions, respectively. To the best of my knowledge, no studies focusing on the environmental evaluation of these areas are available in literature to date.

7.2 MSW MANAGEMENT IN DEVELOPED REGIONS – GALICIA (SPAIN)

At present, different MSW management models coexist in Galicia – incineration, composting and anaerobic biodigestion – while landfilling has become an obsolete alternative (PXRUG, 2011). However, Galician legislation has been supporting initiatives aimed at reducing waste generation at source, as well as promoting aerobic treatment (composting) as a priority alternative for the valorisation of organic waste (main fraction) to the detriment of more conventional practices (PXRUG, 2014). Indeed, biodegradable organic waste represents an important fraction in Galician waste, so it is necessary to continue working to ensure the proper management of this fraction and make the most in terms of waste valorisation.

With this objective in mind, this section aims to analise and compare the environmental sustainability of the different strategies currently implemented in Galicia for MSW management from the LCA perspective.

7.2.1 Goal and scope definition

Galicia is a region located in NW Spain with an estimated population of around 3 million inhabitants and a total surface area of about 3 million hectares. It is responsible for a daily generation rate of 1.05 kg of MSW per inhabitant (in 2013), with a predominance of biodegradable organic fraction (42%), followed by paper and cardboard (18%) and packaging waste (15%) (Table 7.1).

Table 7.1. Mean composition of MSW in Galicia (PXRUG, 2014).

Waste Fraction	Percentage (%)
Organics (biodegradable)	42.0
Paper/Cardboard	18.0
Packaging	15.0
Textiles	10.0
Glass	6.00
WEEE[a]	4.00
Others	5.00
Total	100

[a] Waste Electrical and Electronic Equipment.

At present, MSW in Galicia is managed according to three different models, mainly differentiated by the separation at source of the organic fraction and its subsequent valorisation (PXRUG, 2014). Incineration is the major alternative (82%), followed by anaerobic (15%) and aerobic (3%) treatment, focused mainly on energy generation and compost production, respectively. Moreover, for the last several years, some Galician regions were still subject to an alternative management model, based on the delivery of specific fractions of waste to landfill, without any previous treatment (PXRUG, 2011). However, nowadays, the disposal of waste directly to landfill is no longer possible, becoming and obsolete alternative in Galicia, due to the adhesion agreements to the other treatment models and the decommissioning of existing landfills (PXRUG, 2014).

Accordingly, three main waste management systems were considered in this study: (S1) incineration with energy recovery, (S2) aerobic biological treatment to produce compost (composting) and (S3) anaerobic biological treatment to obtain both energy and compost as primary products. The landfill of waste was also evaluated as the reference alternative (S0), although considering the potential valorisation of biogas to generate renewable energy. A cradle-to-grave approach was applied in all cases, taking into account all processes from MSW collection (including transport) to final management of the various fractions, in line with similar studies in literature (Laurent et al., 2014b); emissions and discharges were also taken into consideration, as well as related background processes.

Since several recovered products can be obtained from the different systems evaluated – energy, recycled materials and compost – a system expansion approach was taken among them to avoid allocation issues, in agreement with ISO recommendations (ISO 14040, 2006; ISO 14044, 2006). This means that not only the burdens associated with managing the different waste fractions were taken into account, but also the benefits derived from the production and use of renewable energy (instead of electricity from the grid), the reuse of recovered materials (such as plastic, paper and metals) and the production and subsequent application of compost to soils (accounting for its associated emissions). In this way, the MSW flow to be treated can be considered as a variable common to all management systems for comparative purposes.

Indeed, the main objective of this case study is to evaluate and compare the environmental profile of alternative strategies for MSW management. This has resulted in the definition of a FU of 1 ton of incoming MSW as the basis for calculations. Waste mass-based FUs are frequently used in LCA studies on waste management (Clearly, 2009; Laurent et al., 2014a,b).

- **Systems description: MSW management models in Galicia**

Figures 7.1 – 7.4 show the main stages included for each management alternative. According to the incineration model (Figure 7.1), MSW must be separated (in origin) into four main fractions: organic waste, packaging, glass and paper and cardboard. While the first two fractions are properly managed within the plant facilities, the last two are directly sent to recycling companies. Note that the “reject fraction” (which includes those wastes that are not subjected to separate collection (such as bulbs or plastic toys, among others) is collected together with organic waste. This waste mixture is first submitted to a triage process in which recyclable materials are separated and send to recycling companies, while the remaining waste is treated to produce the refuse derived fuel (RDF) that will be burned in the incineration plant to generate energy. Due to capacity limitations of the plant, some of this mixture must be sent directly to the landfill. However, biogas from the degradation of organic waste is recovered as an energy source. Regarding packaging fraction, it is sent to the sorting plant where the different types of residues (not only plastics), are separated and sent to the recycling companies for reuse.

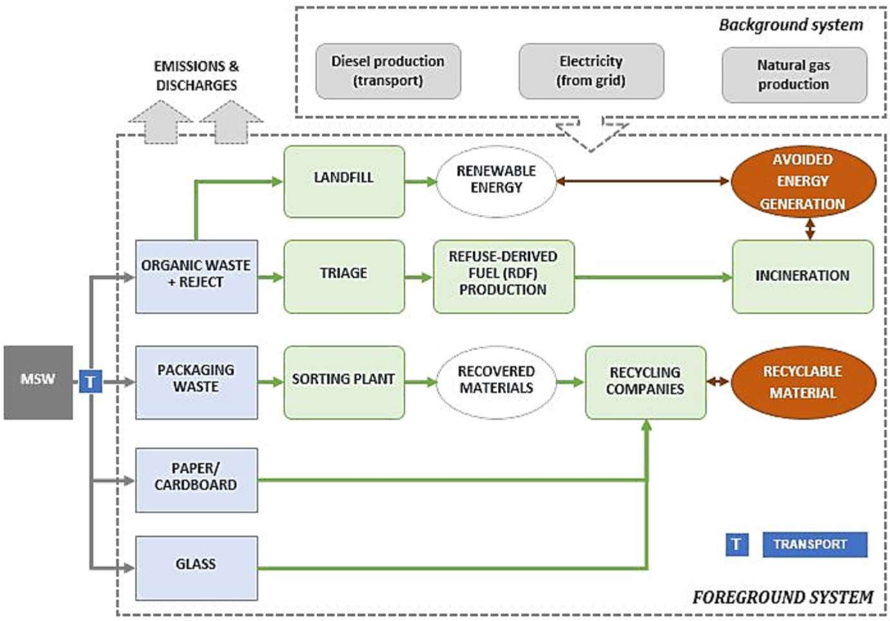

Figure 7.1. System boundaries description of the incineration model. Key: blue boxes represent the different fractions of MSW at source; green boxes make reference to management stages/processes (the CHP unit is included in the landfill facilities); white ellipses represent the different products obtained; brown ellipses include the avoided processes (environmental credits).

In the composting model, a similar treatment strategy is followed for the packaging fraction (Figure 7.2); however, since separation at source is different, the "reject fraction" must be treated together with this packaging waste (instead of organic waste). There are large discrepancies in the management of the organic fraction, which is sent to the composting plant, after the triage stage, to generate compost. Therefore, in this case, not only renewable energy and recovered materials are obtained, but also compost that can be used as fertiliser and/or soil amendment for agricultural production. Nevertheless, the incineration model achieves significantly higher energy efficiency.

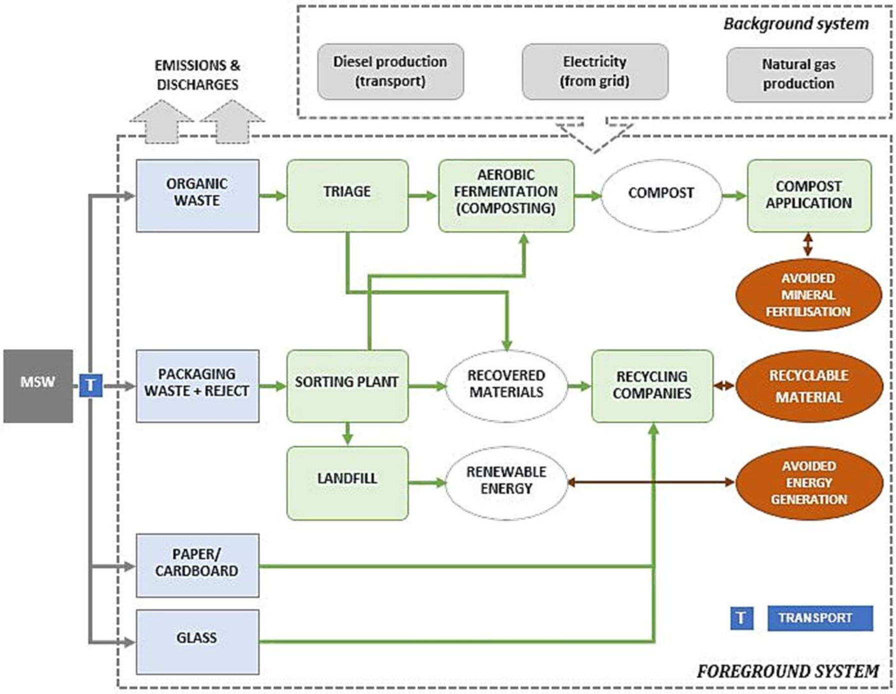

Figure 7.2. System boundaries description of the composting model. Key: blue boxes represent the different fractions of MSW at source; green boxes make reference to management stages/processes (the CHP unit is included in the landfill facilities); white ellipses represent the different products obtained; brown ellipses include the avoided processes (environmental credits).

The scheme for the model involving anaerobic biodigestion technologies (Figure 7.3) is very similar. Again, the "reject fraction" is collected together with the packaging waste and sent to the sorting plant to recover the recyclable material. However, in this case, the organic fraction is first subjected to the anaerobic digestion process to produce both biogas and digestate. Biogas is burned to generate energy, while the digestate is sent to the composting plant to produce compost.

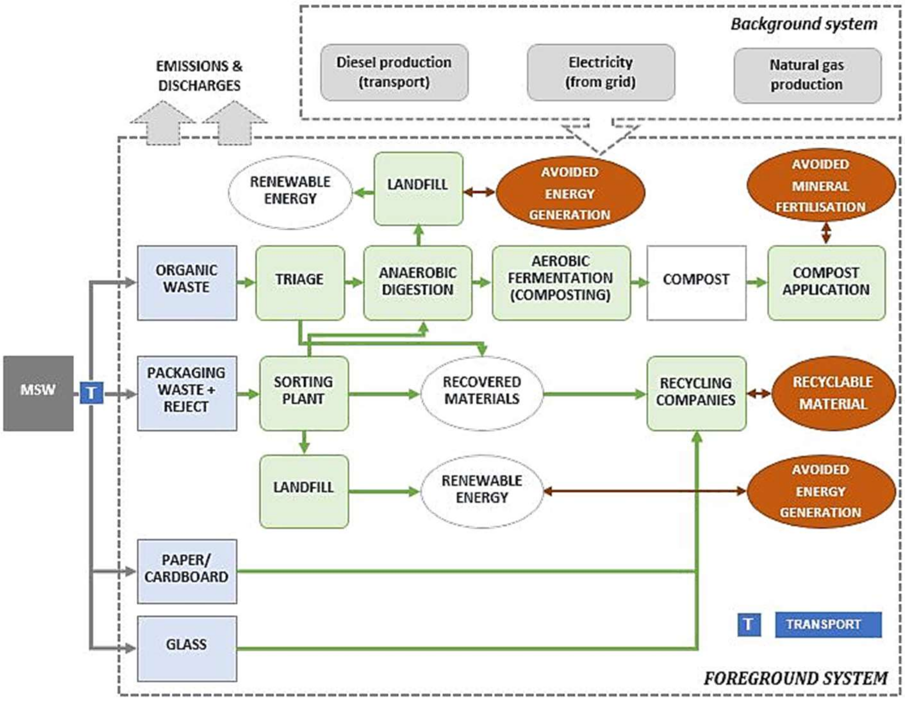

Figure 7.3. System boundaries description of the anaerobic biodigestion model. Key: blue boxes represent the different fractions of MSW at source; green boxes make reference to management stages/processes (the CHP unit is included in the landfill facilities); white ellipses represent the different products obtained; brown ellipses include the avoided processes (environmental credits).

As mentioned above, the landfilling model (Figure 7.4) was also considered as a base case, despite it has become obsolete in Galicia. In this model, it is assumed that renewable energy is generated by the combustion of residual gases in a CHP unit, in a comparable manner to the role of the lanfilling facilities in the other available schemes. It leads to minor diffuse emissions, only due to GHGs losses (mianly in terms of CH_4 and NO_x) intrinsically linked to the operation of this unit. Finally, although electricity is also required by the CHP, it can be assumed that a small fraction of the energy generated is used for its own operation (see epigraph 7.2.2 involving LCI analysis).

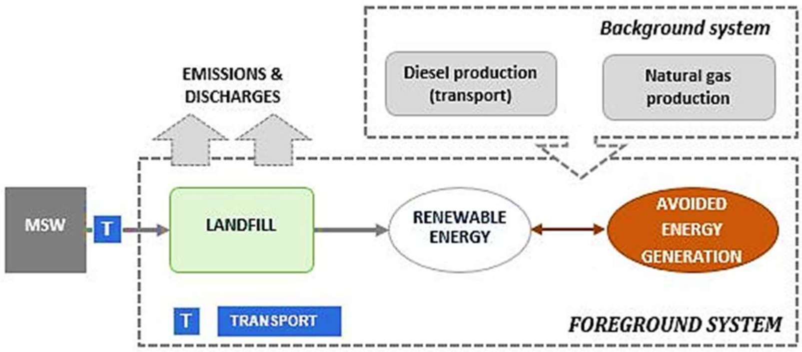

Figure 7.4. System boundaries description of the landfilling model. Key: green boxes make reference to management stages/processes (the CHP unit is included in the landfill facilities); white ellipses represent the different products obtained; brown boxes include the avoided processes (environmental credits).

7.2.2 LCI analysis

Primary data was always prioritised in this study. However, although all available information was compiled from surveys and face-to-face interviews with staff from the management companies, only the inventory data common to all four models were taken into account to ensure the reliability of the benchmarking (Figure 7.5). Consequently, the following inputs were included here: MSW flow and composition, energy requirements, fossil fuel consumption (natural gas and diesel) and transport activities. Two types of outputs were also considered: products and emissions. For the main products, renewable energy and recovered materials were common to all models (except for landfill), while compost was also accounted for in the case of biological treatment of the organic fraction. In this sense, the diffuse emissions of the composting stage (and the global process) and the subsequent application of compost were also estimated, as well as the emissions avoided by the non-use of agrochemicals for fertilisation activities.

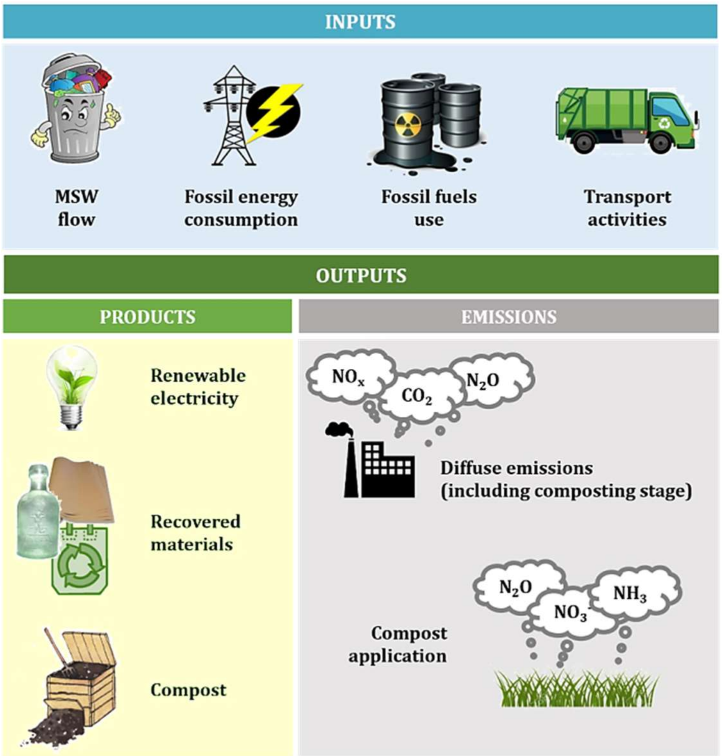

Figure 7.5. Common inventory data considered for the different models in Galicia.

It was also necessary to use relevant literature in the absence of primary information. Thus, mass and energy balances were completed according to the information included in the PXRUG (2014, 2011), while the related diffuse emissions from the composting process (CH_4, N_2O) and the subsequent application of compost to the soil (N_2O, NH_3, NO_3^-) were calculated according to Boldrin et al. (2009) and Bruun et al. (2006), respectively (Table 7.2). The former provided also the factors for estimating the electricity and fossil fuel consumption rates per unit mass of compost generated, in line with Bovea and Powell (2006). According to these authors, carbon from the composting stage is delivered as biogenic CO_2 emissions, which can be accounted for as part of the natural carbon cycle rather than waste emissions. Hence, neutral carbon emissions were assumed in this study. PO_4^{3-} emissions to water were evaluated based on the ratio of 0.01 kg P-PO_4^{3-} per kg of applied P proposed by Rossier (1998). Avoided N- and P-derived emissions due to the use of compost replacing the production and subsequent application of mineral fertilisers were determined according to the guidelines of the Intergovernmental Panel on Climate Change (IPCC, 2006). In the case of N from compost, the MFE was adjusted to 60%, while the uptake rate of P was assumed to be around 90%, according to the literature (Bersntad and la Cour Jansen, 2012; Boldrin et al., 2009).

Table 7.2. Emission coefficients for both N- and P-based compounds from composting process and further compost application.

Compound		Emission Coefficient	Unit	Reference
Composting process	CH_4	0.020-1.800	kg CH_4/t compost[a]	Boldrin et al. (2009)
	N_2O	10.0-120	kg CH_4/t compost[a]	Boldrin et al. (2009)
Compost application	N_2O	0.016	kg N-N_2O/kg N input	Bruun et al. (2006)
	NH_3	0.016	kg N-NH_3/kg N input	Bruun et al. (2006)
	NO_3^-	0.575	kg N-NO_3^-/kg N input	Bruun et al. (2006)
	PO_4^{3-}	0.010	kg P-PO_4^{3-}/kg P input	Rossier (1998)

[a]Wet basis (wb).

Similarly, diffuse emissions from the CHP unit were also assumed in the landfilling model; in this regard, around 0.5% of landfill gases (CH_4, NO_x) were considered to be delivered to air, in accordance with literature (de Vries et al., 2012). Moreover, around 3% of electricity generated was consumed by the CHP unit during its operation (Pöschl et al., 2010).

Finally, secondary data was also taken from the ecoinvent® database to complete the background inventory on electricity generation, fossil fuel production and diesel consumption in transport activities (Althaus et al., 2007; Dones et al., 2007; Spielmann et al., 2007; Wernet et al., 2016). Table 7.3 shows the life cycle inventory data (per FU) of the four models evaluated in the Galician case study.

Table 7.3. LCI data per FU (1 ton of incoming MSW) for the different MSW management models evaluated: S0 – landfilling; S1 – incineration; S2 – composting; S3 – anaerobic biodigestion.

Inputs/Outputs	**S0**	**S1**	**S2**	**S3**	**Units**
Inputs					
Organic line	0.97[a]	0.97[a]	0.34	0.24	t
Packaging line	0.03	0.03	0.66[a]	0.76[a]	t
Electricity	1.75	82.9	37.1	37.1	kWh
Natural gas	11.3	527	-	-	kWh
Diesel	-	-	0.52	7.45	kWh
Transport	51.3	51.3	7.87	6.76	t·km
Outputs – Products					
Renewable energy	56.7	637	10.5	43.3	kWh
Recyclable material	-	34.0	159	56.8	kg
Paper/Cardboard	-	2.55	58.2	14.1	kg
Packaging	-	10.9	73.3	28.9	kg
Glass	-	3.30	7.97	-	kg
Metals	-	17.3	19.9	13.8	kg
Compost	-	-	6.63	94.7	kg
Outputs – Emissions					
Methane (CH_4)	0.23	12.0	0.03	0.43	kg
Nitrogen oxides (NO_x)	0.09	365	-	-	g
Sulphur oxides (SO_x)	-	27.7	-	-	g
Dinitrogen monoxide (N_2O)	-	-	2.17	31.0	kg
Ammonia (NH_3)	-	-	9.00	127	g
Nitrate (NO_3^-)	-	-	1.16	16.6	kg
Phosphate (PO_4^{3-})	-	-	3.30	47.2	g
Outputs – Avoided emissions					
Dinitrogen monoxide (N_2O)	-	-	4.30	61.4	g
Ammonia (NH_3)	-	-	0.03	0.47	kg
Nitrate (NO_3^-)	-	-	0.36	5.19	kg
Phosphate (PO_4^{3-})	-	-	2.97	42.5	g

[a] "Reject fraction" is included in this treatment line.

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7.2.3 Impact assessment

The characterisation factors provided by the ReCiPe Midpoint (H) 1.12 method (Goedkoop et al., 2013) were considered to estimate the potential environmental impacts of the different alternatives in terms of TA, FE, ME and FD. Moreover, the effect on CC was also evaluated according to the IPCC guidelines (2013). This set of categories is in line with related studies in literature, which report the impacts on global warming, acidification and eutrophication potential and resources depletion as key issues in urban waste management systems (Laurent et al., 2014b; Othman et al., 2013).

The contributions of the different processes and activities involved in each management model were also evaluated in relation to the overall environmental results (in percentage). To this aim, they were grouped into seven contributing factors: diffuse emissions (including composting-related emissions, where applicable), energy consumption, fossil fuel use, transport activities, renewable electricity generation, recovered materials (also compost) and avoided emissions (considered as environmental credits).

7.2.4 Results and discussion

Disaggregating environmental results of the different models evaluated are presented below. According to Table 7.4, it can be concluded that the composting model (S2) shows the most environmental-friendly results from a comparative perspective. The lowest energy requirements and the greatest valorisation of their products, especially compost, provide it with the best results in most impact categories, although they depend to a large extent on the category selected. The most notable exception is climate change, where the landfill model (S0) shows the best profile, mainly due to the environmental credits from the valorisation of landfill gases as a renewable energy source. Diffuse emissions in terms of CH_4 and N_2O were found to be responsible for such unfavorable performance of the composting model in terms of the climate change category. In this way, CC was identified to be proportionally affected by GHG emissions from composting practices (especially compost application). It is for this reason that this category was critically penalised when specific attention is paid to anaerobic digestion (S3), since in this model higher compost ratios per ton of treated waste were obtained.The same happens to eutrophication impacts involving NO_3^- release. Again, S0 shows the lowest impacts, although closely followed by the S1 and S2 models. This is due to the fact that nitrogen emissions from compost application on soils are highly offset by the benefits of compost use rather than its artificial analogues on the market.

Table 7.4. Disaggregating environmental results per FU (1 ton of incoming MSW) for the different models evaluated: S0 – landfilling; S1 – incineration; S2 – composting; S3 – anaerobic biodigestion.

Impact category	S0	S1	S2	S3
CC (kg CO_2 eq)	**-1.55**	**254**	**309**	**8047**
Diffuse emissions	6.45	334	575[a]	8214[a]
Electricity consumption	0.43	20.5	10.3	9.19
Fossil fuels use	2.89	144	0.06	0.84
Transport activities	4.98	4.98	1.04	0.13
Electricity generation	-16.3	-184	-3.03	-12.5
Recovered materials	-	-65.6	-270[b]	-101[b]
Avoided emissions	-	0.00	-4.48[c]	-63.9[c]
TA (kg SO_2 eq)	**-0.08**	**-0.70**	**-1.24**	**-1.50**
Diffuse emissions	0.00	0.23	0.02[a]	0.31[a]
Electricity consumption	$2.67 \cdot 10^{-3}$	0.13	0.07	0.06
Fossil fuels use	$7.33 \cdot 10^{-3}$	0.36	0.00	0.00
Transport activities	0.02	0.02	$4.64 \cdot 10^{-3}$	$5.90 \cdot 10^{-4}$
Electricity generation	-0.11	-1.18	-0.02	-0.08
Recovered materials	-	-0.27	-1.21[b]	-0.37[b]
Avoided emissions	-	0.00	-0.10[c]	-1.41[c]
FE (kg P eq)	**$-2.75 \cdot 10^{-3}$**	**-0.05**	**-0.10**	**-0.03**
Diffuse emissions	0.00	0.00	0.00	0.02
Electricity consumption	$2.37 \cdot 10^{-4}$	0.01	$3.25 \cdot 10^{-3}$	0.01
Fossil fuels use	$1.41 \cdot 10^{-4}$	0.01	0.00	0.00
Transport activities	$8.73 \cdot 10^{-4}$	$8.70 \cdot 10^{-4}$	$8.73 \cdot 10^{-5}$	$1.11 \cdot 10^{-5}$
Electricity generation	$-4.00 \cdot 10^{-3}$	-0.04	$-7.20 \cdot 10^{-4}$	$-2.97 \cdot 10^{-3}$
Recovered materials	-	-0.03	-0.10[b]	-0.02[b]
Avoided emissions	-	0.00	$-1.88 \cdot 10^{-3}$[c]	-0.03[c]

[a] Compost emissions; [b] Including compost; [c] Avoided emissions due to compost application.

Table 7.4 (cont.). Disaggregating environmental results per FU (1 ton of incoming MSW) for the different models evaluated: S0 – landfilling; S1 – incineration; S2 – composting; S3 – anaerobic biodigestion.

Impact category	S0	S1	S2	S3
ME (kg N eq)	**$2.79 \cdot 10^{-4}$**	**0.06**	**0.12**	**2.59**
Diffuse emissions	0.00	0.01	0.27	3.83
Electricity consumption	$1.66 \cdot 10^{-3}$	0.08	0.01	0.04
Fossil fuels use	$9.89 \cdot 10^{-5}$	0.01	0.00	0.00
Transport activities	$1.29 \cdot 10^{-3}$	$1.29 \cdot 10^{-3}$	$2.37 \cdot 10^{-4}$	$3.01 \cdot 10^{-5}$
Electricity generation	$-2.77 \cdot 10^{-3}$	-0.03	$-5.14 \cdot 10^{-4}$	$-2.12 \cdot 10^{-3}$
Recovered materials	-	-0.01	-0.07	-0.02
Avoided emissions	-	0.00	-0.09	-1.25
FD (kg oil eq)	**-1.37**	**-20.4**	**-130**	**-64.7**
Diffuse emissions	0.00	0.00	0.00	0.00
Electricity consumption	0.09	4.11	2.48	1.84
Fossil fuels use	0.99	46.9	0.00	0.27
Transport activities	1.73	1.73	0.38	0.05
Electricity generation	-4.18	-47.0	-0.78	-3.19
Recovered materials	-	-26.1	-131[b]	-56.1[b]
Avoided emissions	-	0.00	-0.53[c]	-7.62[c]

[a] Compost emissions; [b] Including compost; [c] Avoided emissions due to compost application on agricultural soils.

Focusing on the environmental performance of each model individually, Figure 7.6 shows that the use of both electricity and fossil fuels is responsible for the highest impacts when MSW is managed according to the incineration model. These consumptions are mainly linked to the requirements of natural gas to dry the organic waste prior to the RDF production. On the contrary, renewable energy generation would report the most important credits, partially offseting the environmental impacts. As all organic waste is sent to the incineration plant, this model generates high levels of electricity.

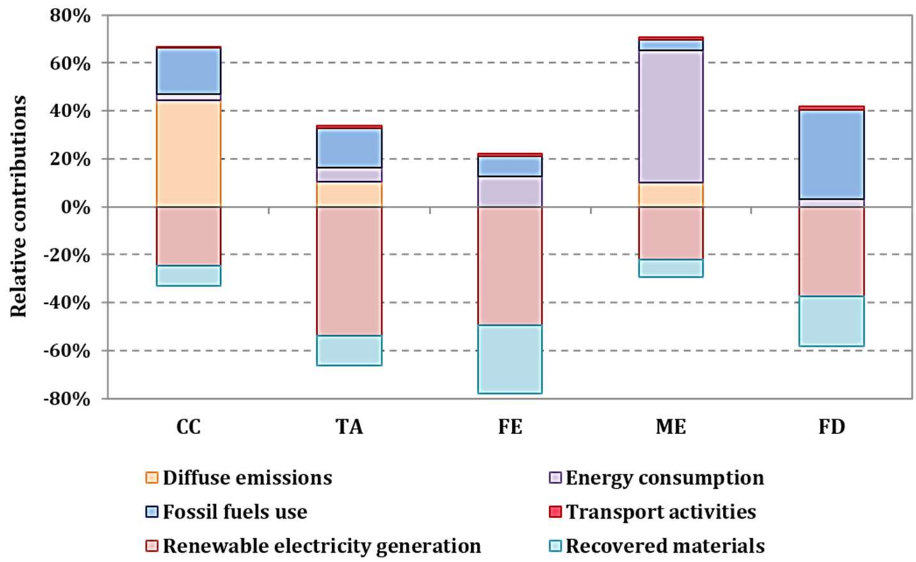

Figure 7.6. Relative contributions (in percentage) from the different processes involved in the incineration model (S1).

The main outcomes are different in the case of the composting model –S2 (Figure 7.7); while composting emissions are responsible for the greatest impacts, instead of energy use, the recovery of recycable materials is the most important factor in offsetting these impacts. In this case, organic waste is mainly used as raw material for the production of compost and not for energy purposes, so that electricity generation rates are lower compared to the incineration strategy, as it comes only from the valorisation of biogas from landfill sites.

Similar results can be found in the anaerobic digestion model –S3 (Figure 7.8) and again composting-related emissions are the main impact responsible. In relative terms, however, avoided emissions from compost application and energy generation play a more important role as environmental credits in this case, together with the recovery of materials.

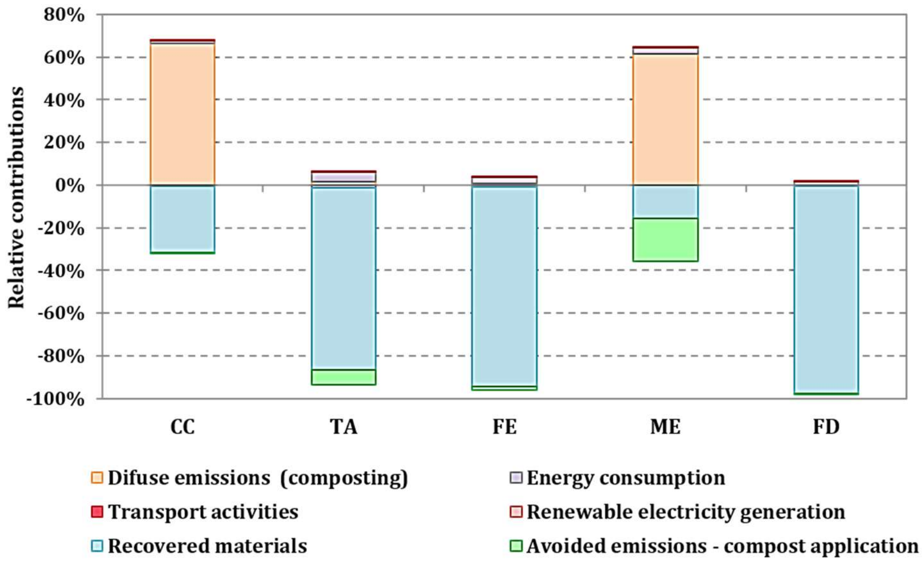

Figure 7.7. Relative contributions (in percentage) from the different processes involved in the composting model (S2). Fossil fuels use was not included in the figure due to its minor effect in relation to the other contributing activities (far lower 1%).

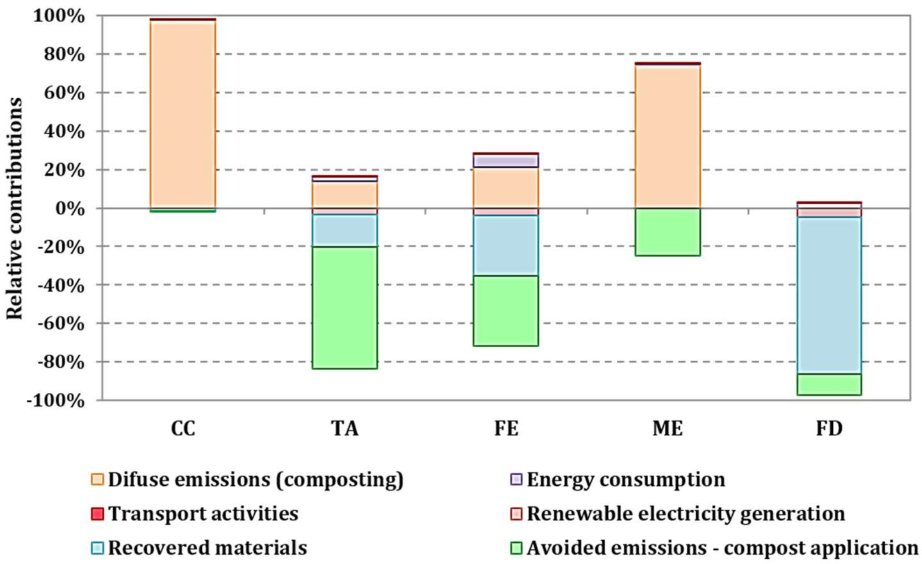

Figure 7.8. Relative contributions (in percentage) from the different processes involved in the anaerobic biodigestion model (S3). Fossil fuels use was not included in the figure due to its minor effect in relation to the other contributing activities (far lower 1%).

7.3 MSW MANAGEMENT IN DEVELOPING REGIONS – ASTANA (KAZAKHSTAN)

Landfilling remains the dominant alternative for MSW management in the Republic of Kazakhstan. However, it seems that increasing environmental concerns have forced authorities to make progress towards more environmental-friendly strategies for MSW treatment (Inglezakis et al., 2017). In this context, the Program of Modernisation of Municipal Solid Waste Management for 2014-2050 has recently been developed as a legal tool to move towards a green economy by increasing the efficiency and environmental reliability of MSW management services in Kazakhstan (Ministry of Environment and Water Resources, 2014). It proposes the gradual implementation of measures to promote higher recycling rates and updated management technologies such as anaerobic digestion, composting and biogas valorisation (Ministry of Environment and Water Resources, 2014).

Consequently, in recent years mechanical-biological treatment plants (MBTs) have been introduced for the recovery of recyclable materials, although Kazakhstan has not yet established the recovery of the organic fraction of waste through the generation of renewable energy (Inglezakis et al., 2017). This approach is in line with other developing regions on waste management that apply similar technologies as a first step towards environmental sustainability (Abeliotis et al., 2012; Bezama et al., 2007; Othman et al., 2013; Yay, 2015). Some authors went beyond conventional practices and also referred to waste-to-energy plants, where priority is given to thermal treatment (incineration, gasification) or anaerobic digestion systems for generating energy and organic fertilisers as by-products (Chaya and Gheewala, 2007; Othman et al., 2013; Panepinto et al., 2015; Song et al., 2017). In most cases, they were found as potential environmental-friendly technologies compared to conventional ones. However, most of them are still in their early stages, and preference for one or the other option may depend to a large extent on MSW composition, as well as the local income level (Chaya and Gheewala, 2007; Othman et al., 2013; Tock and Schummer, 2017).

In this context, the present section focuses on the evaluation of the predominant MSW management technologies in Kazakhstan, as well as the potential improvement alternatives to achieve a more sustainable framework in the country, on the basis of the LCA guidelines (ISO 14040, 2006).

7.3.1 Goal and scope definition

The current situation of the capital city of the country (Astana) was analised as representative of the MSW management scheme in Kazakhstan. Astana has an estimated population of 872,619 inhabitants and an MSW generation rate of approximately 1,118 t/day in 2013 (Inglezakis et al., 2015; NSC, 2016). However, due to the lack of an adequate waste collection system, only around 800 t/day (72% of the total MSW generated) are subjected to further treatment (Inglezakis et al., 2014, 2015), while the rest is managed uncontrolled at source. The average composition of municipal waste in Astana is led by organic waste (27.6%) – mostly food waste – followed by plastics (15.5%), glass (14.9%) and paper and cardboard (11.2%) (Table 7.5).

Table 7.5. Mean composition of MSW in Astana (Inglezakis et al., 2017).

Waste Fraction	Percentage (%)
Organics (food waste)	27.6
Plastics	15.5
Glass	14.9
Paper/Cardboard	11.2
Garden	2.80
Metals	0.95
Wood	0.55
Others	26.5
Total	100

Currently, the waste management system in Astana is mainly based on the mechanical separation of only a small fraction of recyclable materials and organic waste (before landfill disposal), at the expense of landfilling practices applied in other regions of the country (Inglezakis et al., 2014). However, despite the effort, this waste management strategy is far from ideal, and further studies are being developed for the implementation of more sustainable treatment schemes in the city (Inglezakis et al., 2017).

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In this section, a cradle-to-grave analysis was conducted (in line with previous one) covering all stages, from collection and transport of waste to final management and/or disposal, considering current and potential scenarios. Again, since both recycled materials and renewable energy can be sourced as main outputs of the system, the environmental benefits of their further use as substitutes for non-recycled materials and fossil energy, respectively, were also considered to address the allocation requirements (system expansion approach). However, related burdens must also be accounted for, since the subsequent usage stage can entail consequences on the environment.

Finally, analogous to the previous case study in Galicia, a FU of 1 ton of incoming MSW was considered the best choice as a common basis for comparison, in agreement with the literature (Clearly, 2009; Laurent et al., 2014a; Othman et al., 2013).

- **Systems description: MSW management models in Astana**

A total of four alternative scenarios for MSW management were considered: mechanical treatment (MT) without landfill gases valorisation and minor material recycling (current scenario – S0), landfilling without landfill gases valorisation and material recycling (S1), MT with landfill gases valorisation and minor material recycling (S2) and MT with landfill gases valorisation and major material recycling (S3). The main stages of each management scheme are shown in Figures 7.9 – 7.12. It should be noted that, since selective collection of MSW at source is not still available in Astana to date, MT facilities were designed to be able to manage mixed incoming waste, although integrating a modern sorting system in order to separate some fractions of recyclable materials. Moreover, a cogeneration (CHP) unit was assumed to be used for energy generation from the combustion of landfill gases in hypothetical scenarios involving their valorisation (S2 and S3).

The same input flow and MSW composition (see Table 7.5) was considered for all the scenarios. In this sense, it is important to underline the limitations on the waste acceptance capacity of the MT plant, below 50% of the collection rates (≈340 t/day vs. 800 t/day) in 2013; consequently, around 340 tons were assumed to be daily treated within the MT plant in all the scenarios, while the remaining waste must be briquetted to be sent to landfill (Inglezakis et al., 2017).

However, different recycling rates were evaluated. Thus, a greater (20%) recyclability of potential recovered materials in S3 was assumed, avoiding the production of their analogues in the market, in accordance with the recovery capacity initially projected for the MT plant. By contrast, only a small fraction (6%) was considered recyclable for further use in S0 and S2 according to current practices. Finally, recovery of materials does not exist in S1 (landfilling). The following recovered materials were included in the calculations: paper/cardboard (26.6%), plastics (35.7%), glass (35.4%) and metals (2.3%).

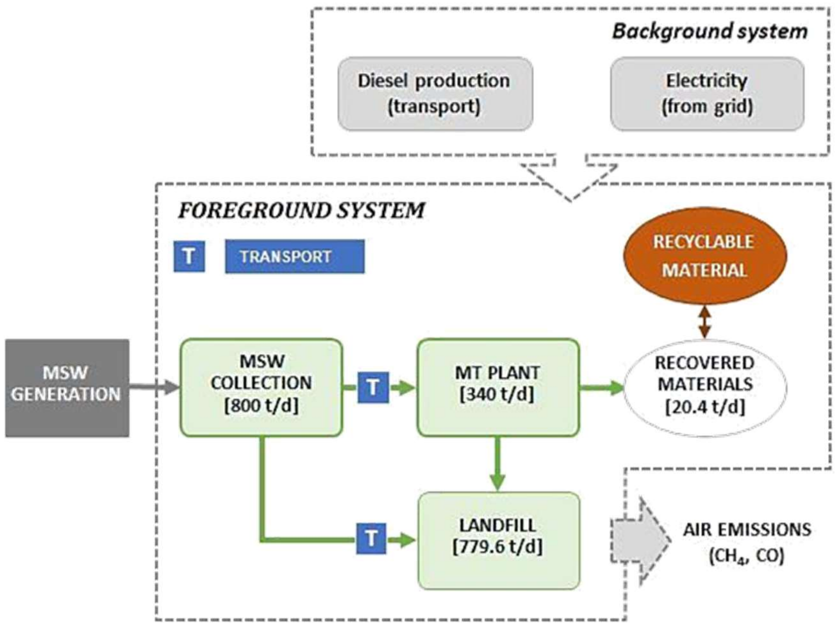

Figure 7.9. System boundaries description of the MT plant scenario (S0). Key: grey boxes represent the different management stages/processes; white ellipses make reference to the different products obtained; brown ellipses include the avoided processes (environmental credits).

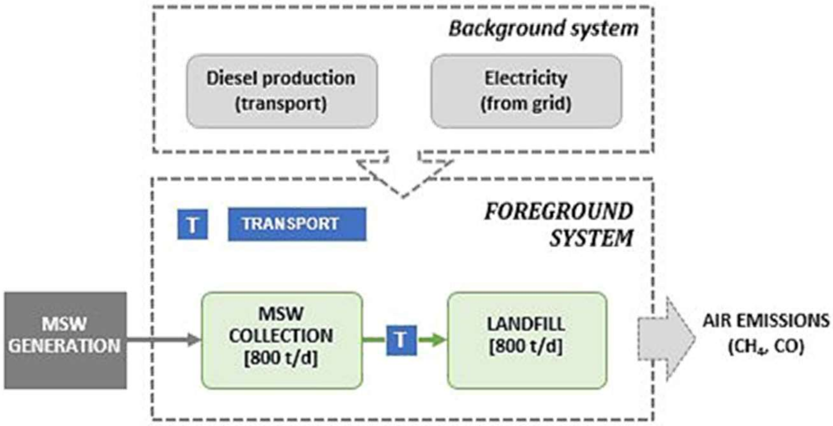

Figure 7.10. System boundaries description of the landfilling scenario (S1).

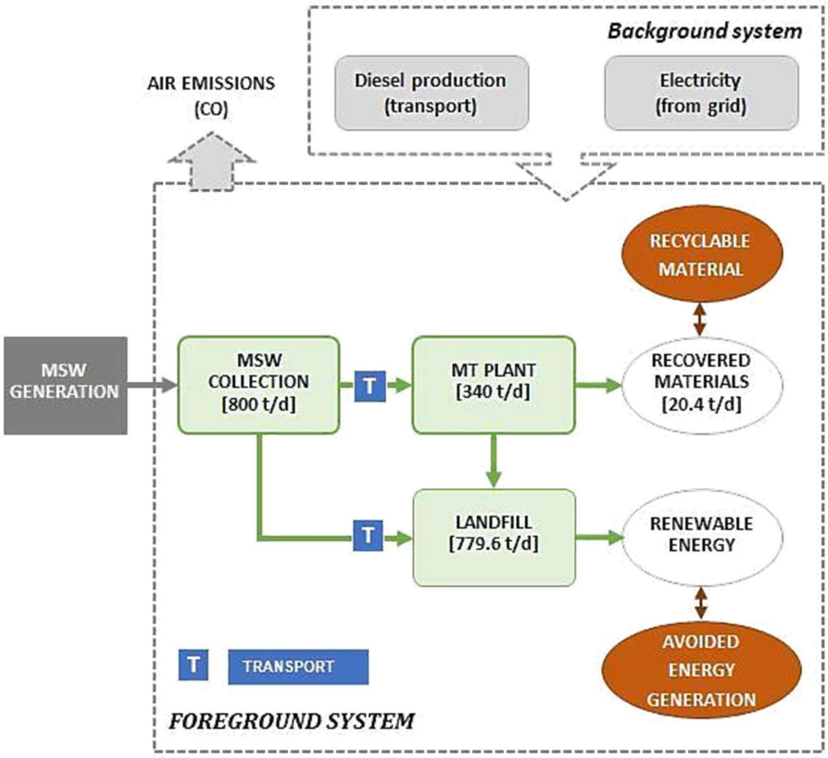

Figure 7.11. System boundaries description of the MT plant + 100% biogas valorisation scenario (S2). Key: green boxes represent the different management stages/processes (the CHP unit is included in the landfill facilities); white ellipses make reference to the different products obtained; brown ellipses include the avoided processes (environmental credits).

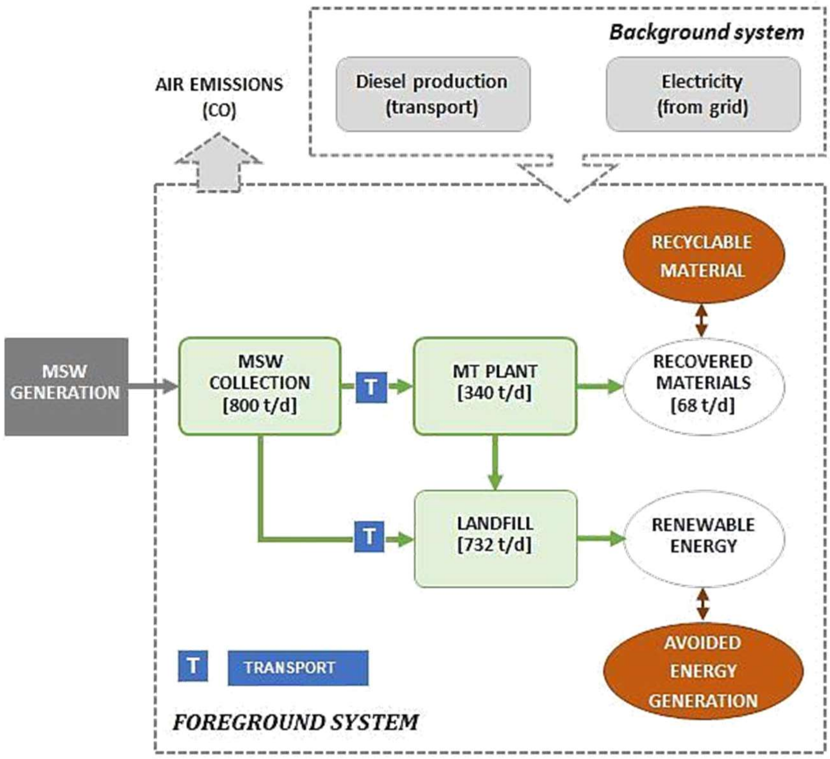

Figure 7.12. System boundaries description of the MT plant + 100% biogas valorisation + major material recycling scenario (S3). Key: green boxes represent the different management stages/processes (the CHP unit is included in the landfill facilities); white ellipses make reference to the different products obtained; brown ellipses include the avoided processes (environmental credits).

7.3.2 LCI analysis

Data collection to quantify input and output flows for the different scenarios under assessment (S0 – S3) was conducted on an annual basis (2013-2014). A standard approach to data collection was followed to ensure the reliability of comparative results. The following inventory information, available for the different alternative scenarios, was considered (Figure 7.13): MSW input flow and composition, transport activities, electricity requirements and renewable energy generation (only in case of landfill gases valorisation), recovered materials (except for landfilling), land use and diffuse emissions in terms of CH_4 and CO; biogenic CO_2 was assumed to be discharged into the atmosphere but without related environmental impacts.

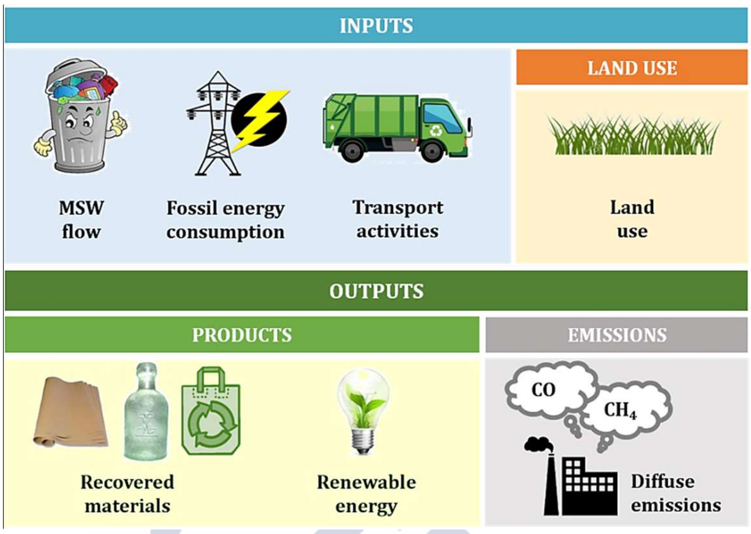

Figure 7.13. Common inventory data considered for the different models in Astana.

Primary inventory data regarding MSW flow and composition, transport distances and collection system (transport fleet, collection frequency) as well as recovered rates for recyclable materials (only current state) and final land use requirements was prioritised and collected on the basis of personal communications from facilities managers (Table 7.6). However, it was also necessary to collect secondary data from the literature to complete the inventory of the different systems in the absence of primary information. Landfill GHG emissions (landfill gases) were estimated based on the emission factors provided by Abeliotis et al. (2012) and Bernstad and la Cour Jansen (2012) for CH_4 and Cherubini et al. (2009) for CO, respectively.

Moreover, the energy potential of such landfill gases was also taken into consideration (only in case of their valorisation). Thus, a calorific value of 9.45 kWh/m^3 CH_4 was assumed, resulting in a ratio of 5.67 kWh/m^3 landfill gas with a composition of 60% CH_4 (40% biogenic CO_2) (IDAE 2014); note that total valorisation was assumed in absence of landfill gas losses on relevant scenarios. Average rates of 40% and 48% were considered for electrical and thermal efficiencies, respectively (Pöschl et al., 2010); however, 4.5% of electricity produced was assumed to be consumed as an input for CHP running (Pöschl et al. 2010). Similarly, the rate of electricity consumption (≈43 kWh t-1 MSW recovered) from MT facilities as a whole was also estimated in line with available studies in literature (Abeliotis et al. 2012, Bernstad and la Cour Jansen 2012); no energy requirements were assumed in landfill facilities (Abeliotis et al. 2012).

Table 7.6. LCI data per FU (1 ton of incoming MSW) for the different scenarios evaluated: S0 – MT plant; S1 – landfilling; S2 – MT + 100% biogas valorisation; S3 – MT + 100% biogas valorisation + major material recycling.

Inputs/Outputs	S0	S1	S2	S3	Units
Inputs					
Electricity	1.10	0.00	1.10	3.66	kWh
Transportation	5.50	5.50	5.50	5.50	t·km
Land use	$4.18 \cdot 10^{-2}$	$4.28 \cdot 10^{-2}$	$4.18 \cdot 10^{-2}$	$3.94 \cdot 10^{-2}$	m^2
Emissions					
CH_4 (air)	56.3	56.3	0.00	0.00	kg
CO (air)	1.79	1.79	1.79	1.79	g
Materials disposal					
Paper/Cardboard	98.8	112	98.8	64.8	kg
Plastics	147	155	147	127	kg
Glass	146	149	146	141	kg
Metals	13.7	9.50	13.7	5.46	kg
Organics	276	276	276	276	kg
Others	293	299	293	302	kg
Total	975	1000	975	915	kg
Renewable energy					
Electricity	-	-	299	299	kWh
Heat	-	-	376	376	MWh
Materials recycling					
Paper/Cardboard	13.2	-	13.2	43.8	kg
Plastics	8.50	-	8.50	28.5	kg
Glass	2.55	-	2.55	8.50	kg
Metals	1.27	-	1.27	4.25	kg
Total	25.5	-	25.5	85.0	kg

Finally, ecoinvent® database (Classen et al., 2009; Dones et al., 2007; Hischier et al., 2007; Spielmann et al., 2007; Wernet et al., 2016) was used for the background inventory regarding energy (electricity and heat) generation and diesel production (for transport activities). Avoided manufacture of recovered materials (paper/cardboard, plastics, glass, metals) due to recycling activities, together with avoided fossil energy generation, were also taken into consideration (avoided processes), accounted as potential environmental credits.

7.3.3 Impact assessment

Analougous to the Galician case study in the previous section, the following impact categories were selected in line with related studies in the literature (Clearly, 2009; Laurent et al., 2014b; Othman et al., 2013): CC, TA, FE, ME and FD. Moreover, since large surfaces are occupied by management facilities in most scenarios (mainly due to landfill sites), impact from land use (LU) was also considered in this section. Again, the IPCC (2013) report was used to evaluate the impacts on CC, while the characterisation factors reported by the ReCiPe Midpoint (H) 1.12 method (Goedkoop et al., 2013) were considered for all the other categories.

Six main contributing factors were defined to facilitate the interpretation of the results: diffuse emissions, disposal of materials (to landfill), recovered glass, recovered metals, recovered plastics and recovered paper/cardboard. Diffuse emissions make reference to GHGs directly emitted to the atmosphere while material disposal includes the fraction of MSW discharged to landfill. The energy requirements for the operation of the MT plant and transport activities from waste collection to the location of the plant had minor influence on the overall results, so that both factors were not included in the analysis of the environmental results.

7.3.4 Results and discussion

- **Environmental results of the current scenario (S0)**

Figure 7.14 shows the environmental impacts associated with the current MSW management strategy in Astana (S0). According to the results, the landfill of about 94% of the recyclable material (material disposal) was the main contributor to the environmental burdens in most impact categories, especially in ME due to the effect of related nitrogen emissions. Similarly, GHG (CH_4 and CO) emissions from the degradation of organic waste in the landfill was responsible for the greatest impacts (around 73%) in CC.

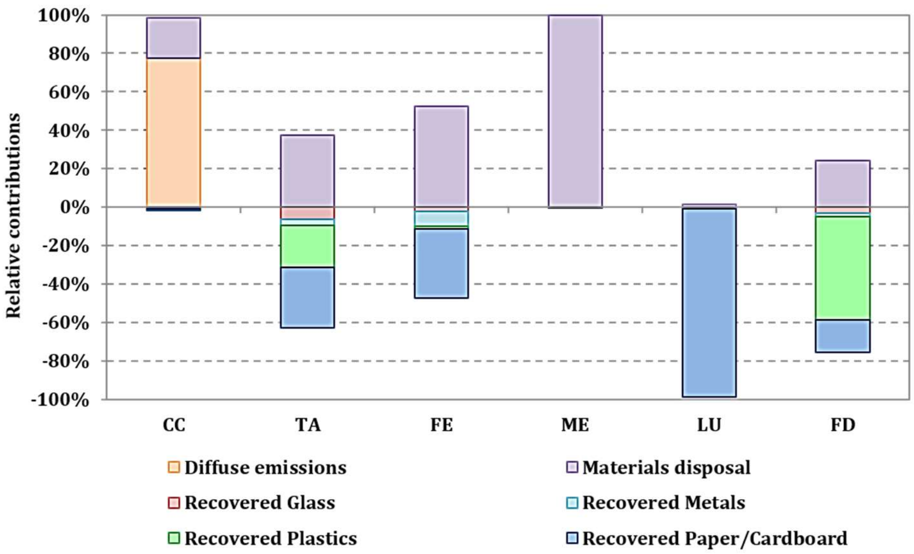

Figure 7.14. Environmental results (in %) associated with the current MSW management scenario (S0) in Astana. Note: positive values (above x-axis) represent environmental impacts while negative results (below x-axis) make reference to environmental credits.

In contrast, the recovery and subsequent reuse of the recyclable material had a significant beneficial effect (below the x-axis), largely offseting the environmental impacts in terms of TA, FE, LU and FD. This could mainly be attributed to recovered paper and cardboard (from 17% in FD to 98% in LU) followed by recovered plastics (up to 53% in FD). Particular attention should be paid to the environmental-friendly contribution of paper/cardboard recycling to LU; it was directly linked to the arable land required for the cultivation of raw materials for the industrial production of both goods (paper and cardboard). On the contrary, this effect is practically negligible (below 3%) in those categories strongly influenced by diffuse emissions (CC) and discharges (ME) to the environment.

Finally, no environmental credits have been registered for power generation, as there is currently no valorisation of gases from landfilling waste at the MT plant.

- **Comparative environmental results (S0 – S3)**

Table 7.7 presents the characterisation results of the different scenarios and impact categories and compares them with the current situation (S0). In view of the results, the scenarios in which landfill gases valorisation could potentially take place – i.e. S2 and S3 – showed the best environmental performances. The production of renewable energy from the use of the CH_4 generated in the landfill was the main responsible for such desirable results in terms of CC mitigation. CH_4 is not released to the atmosphere, so that GHG emissions are deducted at source; this is the rationale behind the significantly lower balance of S2 and S3 in CC. Moreover, the largest fraction of recovered materials made the difference between S2 and S3; while major rates (20%) of recyclable materials were assumed to be recovered in S3, only 6% was considered in S2. The increase in the fraction of recovered materials resulted in lower production of its analogues in the market and, therefore, higher environmental credits due to avoided processes. As a result, the environmental impacts resulting from the increased energy requirements of S3 were fully offset by the favourable contribution of its greater recycling capacity.

When land use (LU) is evaluated individually, environmental-friendly (negative) results can be found in all scenarios except S1. Renewable energy generation did not make a relevant contribution in this category, while recovered materials exerted the greatest influence, according to previous results (S0 – Figure 6.13). Thus, while S0 and S2 shared close results based on the same fraction recovered, again S3 proved to be the best scenario with greater recyclability.

Table 7.7. Global environmental results per FU (1 ton of incoming MSW) for the different scenarios evaluated: S0 – MT plant; S1 – landfilling; S2 – MT + 100% landfill gases valorisation; S3 – MT + 100% landfill gases valorisation + major material recycling.

Impact category	S0	S1	S2	S3	Units
CC	2283	2338	430	123	kg CO_2 eq
TA	-0.07	0.12	-0.89	-1.43	kg SO_2 eq
FE	1.50	14.0	-36.1	-56.4	g P eq
ME	2.73	2.81	2.70	0.20	kg N eq
LU	-91.3	1.25	-102	-318	m^2
FD	-12.6	6.58	-50.1	-125	kg oil eq

According to the above, the environmental results would be in line with some innovative proposals promoted by authorities in Astana. Thus, modern technologies have been recently projected in the city involving the collection and valorisation of CH_4 as an energy source, seeking the avoidance of related climate change impacts (Inglezakis et al., 2017). However, these measures would be insufficient to amend the problems arising from land use: large volumes of waste are directly linked to large landfill sites and, consequently, to extensive land requirements. Moreover, it was demonstrated that the direct disposal of certain waste fractions also indirectly contributes to the greater impact on terrestrial resources, as well as most impact categories as a whole.

It would be directly linked to the absence of selective MSW collection systems at source, while the implementation of mechanical sorting facilities for the recovery of recyclable materials is still in its infancy. In this sense, it would also be desirable to focus a more targeted effort on the advanced design of the primary treatment stages – collection, conditioning and potential enhancement – also involving the valorisation of organic waste.

7.4 CONCLUSIONS

The principles of the LCA methodology were followed in this chapter to analyse the environmental profiles and potential environmental credits of the implementation of current and alternative strategies for MSW management in a developed (Galicia) and developing (Astana) areas. In both case studies, landfilling of waste was considered as worst-case scenario.

Comparative results revealed those scenarios focussed on the aerobic treatment of biodegradable fractions to produce compost as the most environmental-friendly alternative in Galicia, instead of its valorisation as energy source, in line with local initiatives on waste and environmental management.

Similarly, it was found that waste treatment practices involving higher recycling rates with increased reuse of recovered fractions led to greater environmental sustainability in Astana; however, in this case, the generation of renewable energy from the combustion of landfill gases continues to play a critical role in opposition to the direct release of GHG into the atmosphere according to common practices of the region to date.

The main findings of this chapter are expected to contribute to promoting the development of more sustainable waste management schemes that comply with future environmental regulations in Galicia and Astana, as well as other developed and developing regions in an analogous situation worldwide.

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CHAPTER 8. SUSTAINABILITY ANALYSIS OF MSW MANAGEMENT: AHP METHOD

Summary

Selecting the most efficient scenario among different waste management alternatives can be a complex task based on different criteria. In response, the combination of multi-criteria analysis and the perspective of environmental assessment has emerged as an important support to overcome such decision-making process within the waste sector.

In this context, the present chapter focused on the implementation of the AHP coupled with LCA to compare the sustainability of alternative MSW management models in Galicia: landfilling, incineration (with energy recovery) and composting. To this aim, both economic and social indicators were integrated with the environmental impacts previously estimated in Chapter 7. The overall results identified the composting scheme with the most sustainable behaviour, in line with the regulatory measures recently proposed by the Galician authorities on waste management. The favourable environmental component compared to the other alternatives was responsible for its promising results when a balanced weight of criteria is assumed; only economic results could act as penalising dimension. A sensitivity analysis reinforced composting as the most environmental-friendly model, as well as an economically and socially rational alternative. However, it also revealed how incineration or landfilling had better profiles when particular attention was paid to the social and economic dimensions, respectively. This underlines that, although the composting model is the one that currently leads the ranking of sustainable priorities for alternative waste management in Galicia, the outcomes may partly depend on the weight of the criteria scores. Therefore, additional efforts could be made to improve the socio-economic profile of the composting model in accordance with regional regulations that promote this alternative to move towards a more sustainable waste management framework in Galicia.

CHAPTER 8 - INDEX

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8.1 AHP METHOD FOR ASSESSING MSW SUSTAINABILITY

As noted in Chapter 7, the selection of an appropriate treatment process is an important issue to be evaluated before designing and implementing any waste management strategy. However, sometimes there is no single solution to the problems of MSW management, as each situation has specific challenges to address (Antonopoulos et al., 2014; Contreras et al., 2008; Fragkou et al., 2010). In this context, the MCDA have emerged as an interesting support decision tool in the waste sector (Achillas et al., 2013). Among them, the AHP method has been especially preferred when the optimal alternative of waste management schemes has to be identified (Achillas et al., 2013; Soltani et al., 2015); although it is one of the early MCDA methods (Saaty, 1980), it is still widely used today.

Moreover, numerous research works on waste management have addressed the AHP methodology linked to environmental analysis through the LCA methodology (Contreras et al., 2008; Dong et al., 2014; Manfredi and Cristobal, 2016; Soltani et al., 2015; Yap and Nixon, 2015). This offers the possibility of integrating additional indicators (such as economic and social criteria) into the purely environmental perspective. However, there are not yet publications that exploit the potential added value of the combined AHP approach in the Spanish regions.

In this context, the main goal of Chapter 8 was to apply the AHP method, together with the LCA perspective, to assess and compare the sustainability of MSW management in Galicia on the basis of environmental, economic and social indicators. In this way, the most sustainable alternative (integrative approach) was identified along with the optimal one for each of the three pillars of sustainability (individually), in a complementary way to the environmental analysis developed in Chapter 7.

8.2 MSW MANAGEMENT IN GALICIA: SUSTAINABILITY ASSESSMENT

8.2.1 Goal and scope definition

Therefore, this chapter focuses on comparing the sustainable performance of alternative strategies for MSW management in Galicia. To this purpose, environmental impacts (Chapter 7) and economic and social indicators were analysed together following the AHP principles. The most sustainable alternative was identified and a sensitivity analysis was also conducted to validate the robustness of the comparative results based on a variable weighting covering all three dimensions of sustainability.

As explained in Chapter 7, three main MSW models coexist in Galicia: incineration with energy recovery, aerobic biological treatment for compost production (composting) and anaerobic biodigestion; landfilling disposal is already an obsolete alternative (PXRUG, 2014; 2011). However, recent studies reveal the saturation of the incineration plant, as well as the present challenges of existing anaerobic biodigestion facilities to properly manage incoming organic fraction (PXRUG, 2014). Indeed, intrinsic properties of biodegradable waste (such as moisture and fermentability) give it high recycling possibilities; this results in a greater attention on getting the largest possible yields by specific initiatives involving a sustainable management of this fraction (PXRUG, 2014). Accordingly, alternatives for improvement have been proposed in Galicia based on increasing the current incineration capacity and providing a greater relevance for those infrastructures dedicated to composting practices, following the requirements and recommendations of the current regulations in this region (PXRUG, 2014).

Bearing this is mind, two main waste management systems were considered in this chapter for comparison: incineration (S1) and composting (S2). Landfilling (S0) was also evaluated, although only as a reference of the most precarious alternative (analogous to Chapter 7); anaerobic biodigestion was excluded from the study in agreement with the last action lines posed by Galician authorities (PXRUG, 2014). A detailed description of the system boundaries and the main stages included in each model is provided in Chapter 7 (see epigraph 7.2.1). Also in accordance with this chapter, a cradle-to-gate assessment was performed and a system expansion approach was applied to address allocation challenges. A FU of 1 ton of incoming MSW was selected as the basis for the calculations, in line with similar studies in literature focusing on waste management (Clearly, 2009; Laurent et al., 2014a,b).

8.2.2 Criteria selection

Figure 8.1 shows the hierarchical structure defined in this study, which involves the main goal, criteria and management alternatives. A detailed description of the criteria and sub-criteria selected for the evaluation is included below.

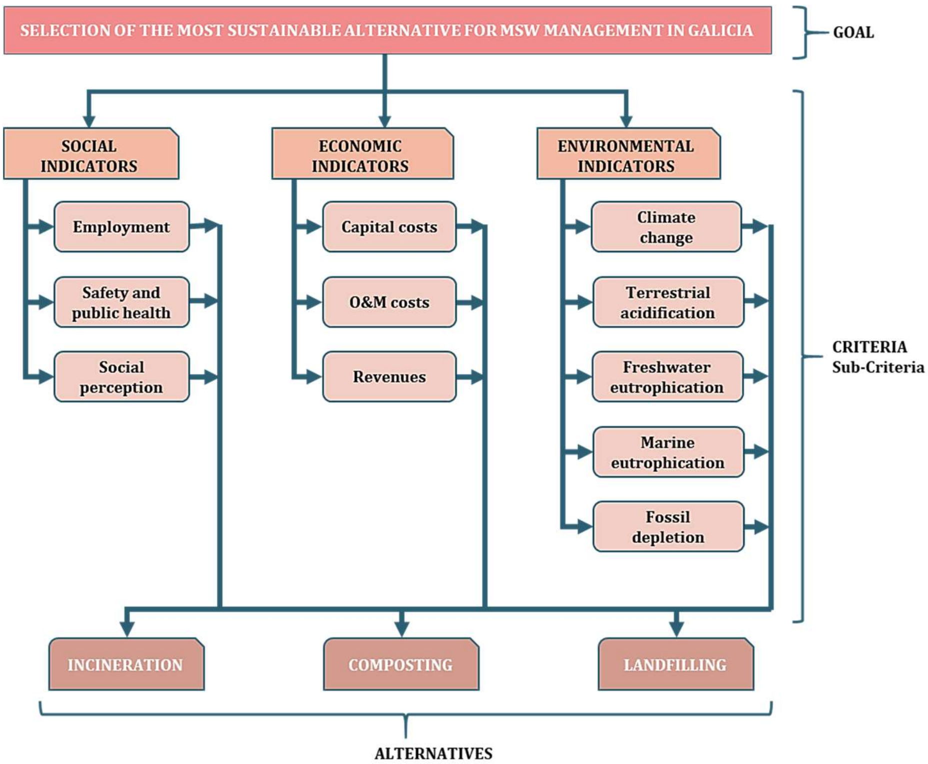

Figure 8.1. Hierarchical structure for the comparative evaluation of the alternative models for MSW management in Galicia.

8.2.2.1 Economic criteria

The economic performance of the different alternatives was evaluated based on three main criteria: capital cost, operational and maintenance (O&M) costs and revenues. The capital cost is related to the construction of the waste management facilities (including land and transport fleet costs), while O&M costs are primarily considered in the operation of the waste management plant (Antonopoulos et al., 2014; Economopoulos et al., 2010;

Perkoulidis et al., 2010). Both capital and O&M costs were estimated by applying the cost functions reported in the literature and average values were considered for assessment (Antonopoulos et al., 2014; Economopoulos et al., 2010; Perkoulidis et al., 2010). Tables 8.1 and 8.2 summarise the equations used. Note that the costs were updated to 2013 (reference year), assuming an annual interest rate of about 6%; similarly, the annual amortised capital costs were also estimated taking into account the same interest rate (6%) and a lifespan of the facilities of 20 years, according to related studies in the literature (Antonopoulos et al., 2014; Perkoulidis et al., 2010).

Table 8.1. Equations applied to estimate the capital costs for the different MSW management models evaluated: S0 – landfilling; S1 – incineration; S2 – composting.

Management model	Capital costs	Units	Data sources
Landfilling	$0.0057x^{0.61}$	10^6 €	Tsilemou and Panagiotakopoulos (2006)
Incineration	$0.0049x^{0.80}$	10^6 €	Tsilemou and Panagiotakopoulos (2006)
	$0.0035x^{0.83}$	10^6 €	Murphy and McKeogh (2006)
Composting	$0.0021x^{0.76}$	10^6 €	Tsilemou and Panagiotakopoulos (2005)

x: design capacity (10^3 t/year)

Table 8.2. Equations applied to estimate the O&M costs for the different MSW management models evaluated: S0 – landfilling; S1 – incineration; S2 – composting.

Management model	O&M costs	Units	Data sources
Landfilling	$103.86x^{-0.30}$	€/t	Tsilemou and Panagiotakopoulos (2006)
Incineration	$726.37x^{-0.29}$	€/t	Tsilemou and Panagiotakopoulos (2006)
	$755.97x^{-0.29}$	€/t	Murphy and McKeogh (2006)
Composting	$1624x^{-0.48}$	€/t	Tsilemou and Panagiotakopoulos (2005)

x: design capacity (10^3 t/year)

Revenues were assumed to be derived from the products sold, including energy and recovered materials (also compost), which can be estimated taking into account energy generation and recovery rates, along with the price of each product. The market prices considered for the assessment are summarised in Table 8.3.

Table 8.3. Market prices considered for recovered materials and surplus energy.

Output (material/energy)	Market price	Units	Data sources
Energy	0.105	€/kWh	BOE (2009, 2007)
Compost	27.9	€/t	COGERSA[a]
Paper/Cardboard	83.0	€/t	ASPAPEL[b]
PEAD/PEBD	895	€/t	ANARPLA[c]
PET	720	€/t	ANARPLA[c]
Steel	303	€/t	MetalRadar
Aluminium	1785	€/t	MetalRadar

[a] Consortium for the Management of Solid Waste of Asturias; [b] Spanish Association of Pulp, Paper and Cardboard Manufacturers; [c] National Association of Plastic Recyclers.

8.2.2.2 Social criteria

Three criteria – employment, social perception and public health and safety (public H&S) – were also selected to assess the social dimension of the different management alternatives, such as the most common social indicators used in similar studies in the literature (Achillas et al., 2013; Antonopoulos et al., 2014; Khan and Faisal, 2008; Su et al., 2007). Employment refers to the number of employees associated with each waste management scheme, so it is preferable to create a large number of jobs. Public H&S was addressed on the basis of both waste valorisation and waste sent to landfill, in percentage values. Finally, social perception was evaluated taking into account not only the level of satisfaction of the population but also their active participation in waste management tasks; however, qualitative ratios were defined for this indicator. The social outcomes were estimated on the basis of public surveys and personal communications.

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8.2.2.3 Environmental criteria

In accordance with Chapter 7, the LCA guidelines were followed (ISO 14040, 2006) and the characterisation factors provided by the ReCiPe Midpoint (H) 1.12 method (Goedkoop et al., 2013) were considered to estimate the potential environmental impacts of the different alternatives. The following indicators were selected: CC, TA, FE, ME and FD. Detailed information on environmental data collection and estimations are provided in the previous Chapter 7 (see epigraph 7.2.2 LCI Analysis).

8.2.3 Results

8.2.3.1 AHP results

The environmental, economic and social results for the different sub-criteria and management alternatives are compiled in Table 8.4 per ton of incoming MSW (FU). Minimum scores are preferable for both environmental and economic indicators, except for revenues, which should be as high as possible. Conversely, maximum ratios are desirable for social indicators, although the fraction of waste sent to landfill should be reduced.

Table 8.4. Environmental, economic and social results for the different management models per FU (1 ton of incoming MSW): S0 – landfilling; S1 – incineration; S2 – composting.

Criteria	**Sub-criteria**	**S0**	**S1**	**S2**	**Units**
Environmental	CC	-8.96	254	309	kg CO_2 eq
	TA	-0.08	-0.70	-1.24	kg SO_2 eq
	FE	$-2.98 \cdot 10^{-3}$	-0.05	-0.10	kg P eq
	ME	$-1.47 \cdot 10^{-3}$	0.06	0.12	kg N eq
	FD	-1.59	-20.4	-130	kg oil eq
Economic	Capital Costs	46.4	441	403	€
	O&M Costs	3.15	23.3	25.3	€
	Revenues	4.97	144	46.0	€
Social	Employment	$1.99 \cdot 10^{-4}$	$4.40 \cdot 10^{-4}$	$5.29 \cdot 10^{-3}$	No. employees
	Social perception	Low	High	Medium-Low	
	Public H&S	100	51.3	59.1	% to landfill
		0.00	8.10	23.1	% valorised

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According to Table 8.4, the alternative focused on the disposal of MSW to landfill (S0) would be the worst option from environmental and social perspectives, with the exception of CC mitigation, due to the environmental credits derived from the valorisation of landfill gases as a renewable energy source. Only the economic component can partially offset the unfavourable results of landfilling, with much lower costs than incineration and composting.

8.2.3.2 Base case: equal criteria weighting

Tables 8.5 and 8.6 show the pair-wise comparison matrix and priority vectors associated with each management model when equal weight (33.3%) is assumed for the environmental, economic and social criteria (Base Scenario – BS). A more detailed explanation of the numerical processing and intermediate calculation stages is included in Annex II (Supplementary material).

Table 8.5. Pair-wise comparison matrix in Base Scenario (BS): equal weight (33.3%) for environmental, economic and social criteria.

Scenario	Criteria	Economic	Social	Environmental
BS	Economic	1	1	1
	Social	1	1	1
	Environmental	1	1	1

Table 8.6. Overall priority vectors for each management model in Base Scenario (BS): equal weight (33.3%) for environmental, economic and social criteria.

Management model	Overall Priority Vector
Landfilling	0.28
Incineration	0.34
Composting	0.38

Additionally, Figure 8.2 shows the AHP results for the three models evaluated, both individual and integrated scores, in terms of the weighting attributed to each criterion (economic, social and environmental) in BS.

Overall values show that the management system based on the composting of MSW ranks first in the priority classification (37%), followed closely by incineration (35%) and, finally, landfilling (29%). The environmental-friendly results associated with this management alternative would be responsible for its highest sustainable profile, except in the case of CC, where diffuse emissions from the composting stage (and subsequent application of compost) cause an unfavourable influence. However, the impacts from the composting practices are mainly offset by the emissions avoided by the use of compost as an organic fertiliser (as a substitute for mineral ones) and by the reuse of recyclable materials, which positively affects the other impact categories, especially in terms of FD. Finally, the generation of renewable energy would not have a significant favourable effect on the composting model, as much attention is paid to the production of compost from organic waste.

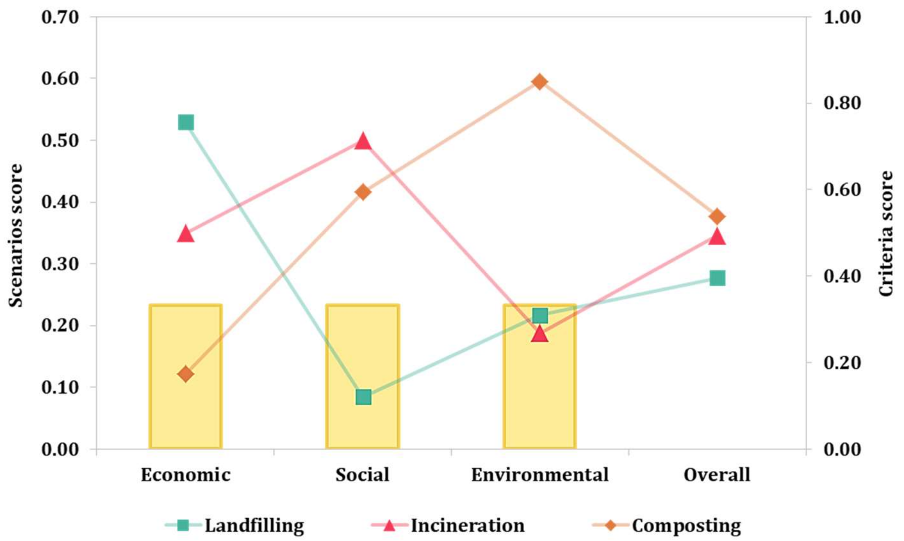

Figure 8.2. AHP results when equal criteria (33.3%) are assigned to the three MSW management models evaluated (Base Scenario – BS). Left axis makes reference to the weighting score of the different models (landfilling, incineration and composting) while left axis represent the weight of the different criteria (economic, social and environmental).

In contrast, the net energy balance (generation of surplus energy) represents a significant environmental credit when MSW is managed according to the incineration model. However, it is not sufficient to offset the high energy demand, mainly from the drying process prior to the production stage of RDF, which causes greater environmental impacts in most impact categories. The environmental disaggregation results for the different models were evaluated in detail in Chapter 7 (epigraph 7.2.4 Results and discussion).

However, different conclusions can be drawn when each criterion is analysed separately. Thus, while incineration would be the best option focusing only on the social dimension, landfilling would result in the best economic profile. The rationale behind these results lies mainly on the high social acceptance of incineration as a suitable option for MSW management, as well as the lower capital and operational costs related to landfill facilities, respectively.

8.2.4 Discussion: sensitivity analysis

Accordingly, a sensitivity analysis was conducted in this chapter to evaluate how the dominance of one criterion or another could affect the final conclusions of the benchmarking. The three sets of criteria – economic, social and environmental – were assumed to share the same relevance in BS and the composting model was found to be the most sustainable alternative for MSW management in Galicia. However, the above findings revealed also how variations in the relative relevance of the different criteria can lead to totally divergent results. This is in line with similar studies in the literature, which reported that variations in the relative weights attributed to each element of the hierarchical structure can be potentially responsible for totally divergent outcomes (Antonopoulos et al., 2014; Contreras et al., 2008; Dong et al., 2014; Karimi et al., 2011; Yap and Nixon, 2015), in accordance with AHP principles.

Therefore, an increased relevance (75% corresponding to level 6 of the Saaty's Fundamental Scale - see Chapter 2) was assigned individually to the economic, social and environmental criteria to determine which management model reports the most sustainable performance for each scenario. In this way, three alternative scenarios were proposed: economic (Scenario A – SA), social (Scenario B – SB) and environmental (Scenario C – SC) weighting schemes (75%). In addition, an economic-environmental perspective was additionally considered based on a weighting factor of 45% for both dimensions to the detriment of social criteria (Scenario D – SD). A detailed criteria weighting is given in Table 8.7.

Table 8.7. Detailed criteria weighting factors according to the four scenarios included in the sensitivity analysis (pair-wise comparison matrix): economic dimension priority (Scenario A – SA); social dimension priority (Scenario B – SB); environmental dimension priority (Scenario C – SC); economic-environmental priority approach (Scenario D – SD).

Scenario	Criteria	Economic	Social	Environmental
SA	Economic	1	1/6	1/6
	Social	6	1	1
	Environmental	6	1	1
SB	Economic	1	6	1
	Social	1/6	1	1/6
	Environmental	1	6	1
SC	Economic	1	1	6
	Social	1	1	6
	Environmental	1/6	1/6	1
SD	Economic	1	1/5	1
	Social	5	1	5
	Environmental	1	1/5	1

Figure 8.3 displays the sustainability results for the different scenarios, while Table 8.8 shows the priority ranking associated with each case in accordance with the overall results obtained by applying the AHP principles.

Table 8.8. Overall priority vectors for each management model for the different alternative scenarios proposed based on alternative criteria weighting: 75% weighting factor for the economic perspective (SA); 75% weighting factor for the social perspective (SB); 75% weighting factor for the environmental perspective (SC); 45% weighting factor for both economic and environmental perspectives (SD).

	Overall priority vectors (Priority Ranking)			
Management model	**SA**	**SB**	**SC**	**SD**
Landfilling	0.43	0.16	0.24	0.35
Incineration	0.35	0.44	0.25	0.29
Composting	0.22	0.40	0.51	0.36

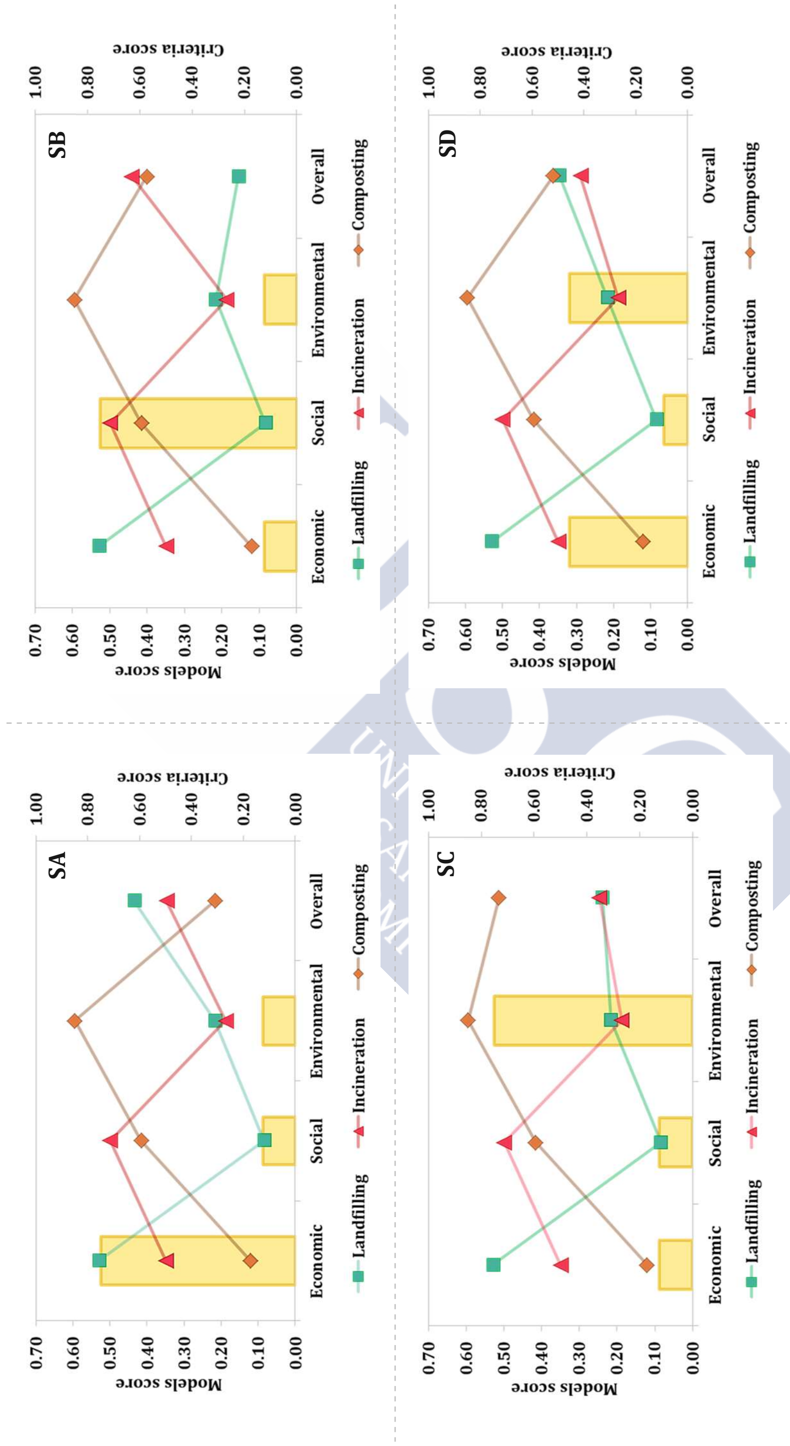

Figure 8.3. Sensitivity analysis results based on alternative criteria weighting: 75% weighting factor for the economic perspective (SA); 75% weighting factor for the social perspective (SB); 75% weighting factor for the environmental perspective (SC); 45% weighting factor for both economic and environmental perspectives (SD).

As expected, the composting model would be again the most desirable alternative when the environmental perspective is predominant (Figure 8.3 – SC), according to the previous results in the base scenario (BS), although with greater supremacy (49%) over the other alternatives. However, priorities may change depending on the weighting of criteria. The lower associated costs give landfill sites a much more favourable economic profile compared to other models (Figure 8.3 – SA), but its environmental performance and social perception would relegate it to the last position. Only the incineration model could represent a more competitive management alternative, especially for public opinion (Figure 8.3 – SB), although its economic-environmental profile should be subject to significant improvements. In contrast, composting would share a social ratio close to incineration (40% vs. 44%, respectively) and its unfavourable performance in the economic area would be partially offset from an economic-environmental approach (Figure 8.3 – SD).

8.3 Conclusions

MCDA techniques were shown to be useful tools for decision-making in the case of waste management systems, where several criteria have to be evaluated together. Among them, the AHP method has been widely applied to analyse the sustainability of different MSW management schemes. Moreover, the LCA methodology has also been followed in studies related to the environmental assessment of alternative waste treatment configurations; however, no similar studies are available for NW Spanish region. In this study, both methodologies (AHP linked to LCA) were applied to assess the sustainability of three alternatives for MSW management in Galicia: landfilling, incineration and aerobic fermentation (composting). According to the results, composting would be the best option when the three pillars of sustainability are weighted similarly, mainly due to its environmental-friendly performance, followed by incineration and landfilling. However, priorities may change when each criterion is analysed individually, so incineration would be the preferred option based on popular opinion. In this regard, further awareness-raising actions should be undertaken to lead the popular preferences towards more sustainable alternatives, in line with the European guidelines on strategies for sustainable waste management.

Moreover, comparative results reveal how the main outcomes can depend to a large extent on the environmental, economic and social dimensions. Accordingly, a sensitivity analysis was performed to determine the extent to which these variations in the weight of the criteria may affect the order of priority between the three management models. Comparative results showed that the composting model would remain the best environmental option and even share ratios close to the socially more favourable alternative (incineration). It would only have the last position in terms of economic criteria, but partially offset when it is extended to an economic-environmental perspective. In this way, the robustness of the main outcomes was demonstrated, so that they could make a useful contribution to advancing the development of MSW management systems committed to sustainability in Galicia.

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SECTION IV

CONCLUSIONS

CHAPTER 9. GENERAL FINDINGS AND CONCLUSIONS OF THE THESIS

The main goal of this doctoral thesis was to analyse and compare the environmental sustainability of conventional practices against advanced technologies on waste management, involving both agricultural and urban sources, at European level. This theme is in line with the growing awareness of the related effects of inadequate management of solid wastes, as well as the changing view of their role in society as potential added-value products. Accordingly, the European authorities have promoted a progressive migration towards good practices in integrative waste management strategies, ensuring economic and social development in harmony with environmental protection. Methodologies on environmental impact analysis applied in this thesis were proved as useful tools for this purpose, also in combination with other support instruments that integrate the socio-economic perspective. The main findings and conclusions drawn from both Section II and Section III are set out below.

- **Section II. Agricultural framework**

The aim of this section was to focus on the environmental performance and potential for improvement of advanced strategies for waste management of agricultural systems, with particular attention to the livestock sector. To this end, different stages of the agricultural value chain were environmentally evaluated, from animal feed production to final product manufacture and waste valorisation.

The LCA assessment of several varieties of winter and summer cereals cultivated for **animal feed** in the Po Valley (leading agricultural area in Europe in the cultivation of cereal crops) reported the following:

↳ Field emissions and agrochemical production were found to be the most relevant contributors to environmental damages during cereal cultivation, regardless of the variety and regime evaluated. It was mainly related to the manufacture of mineral fertilisers and their subsequent application to soils, which leads to the discharge of N-based compounds to the environment. Agricultural activities also had a decisive influence, directly related to diesel use and related combustion emissions during the operation of agricultural machinery.

↳ In comparative terms for cropping systems, rye and maize classes 600-700 (average ratios) were considered as the most environmental-friendly alternatives for winter and summer cereals, respectively, based on the crude protein content as the basis for comparison (FU).

In this regard, a sensitivity analysis was conducted to evaluate the potential influence of alternative FUs on the comparative results, and it was found that:

↳ Similar results can be obtained when alternative cropping systems are compared according to mass-based criteria, involving biomass production either in wet or dry basis. This could be closely attributed to the proportionality between biomass yield and protein content in all the cereals assessed.

↳ However, the relative performances were not the same when comparing systems on the basis of land use instead of mass criteria. In this case, the impacts of higher requirements of agrochemicals per hectare on agricultural activities gained relevance to the detriment of greater biomass yields. This highlights how the choice of the best and/or worst alternative can depend to a large extent on the FU considered for the calculations.

Moving forward to the **farm level**, some common conclusions emerged when three different Spanish systems involving cow dairy (Catalonia) and pork (Galicia/Catalonia) sectors were environmentally evaluated up to the farm gate:

↳ Animal feed production (especially fodder) was identified as one of the major contributors to the environmental impacts of the farms evaluated, along with diffuse emissions from both animal husbandry at farm and manure management. Carbon emissions from enteric fermentation played a critical role in the dairy system due to the highest production of CH_4 from ruminant livestock. However, their relative influence was reduced in pork farms by the contribution of nitrogen-based emissions (N_2O, NH_3, NO_3^-).

↳ The supremacy of feed production and on-farm diffuse emissions, in relative terms, were also shared by similar studies involving pork and dairy systems, either within or outside Spanish boundaries. In addition, the environmental results (absolute values) were also in line with the main results reported by previous studies carried out in the field.

On the other hand, interesting outcomes were also obtained when the related environmental impacts were assessed individually for each particular case study:

↳ Among impacts associated with feed production at the dairy farm, alfalfa was identified as responsible for the major contributions. Therefore, the potential for improvement of using maize silage and grass silage as alternative sources of protein was environmentally assessed; however, the comparative results did not demonstrate any significant advantages of substitutes in the target system.

↳ Similarly, promising areas for engagement with regard to pork production systems were also proposed. Among them, alternative proportions between the ingredients of fodder formulations were encouraged, together with more efficient consumption patterns, due to their potential environmental benefits.

↳ When the entire pork chain was evaluated, further stages of slaughterhouse and pork processing (pork cutting) were identified as minor contributors to the global environmental results, in comparison with related impacts up to the farm gate.

Other environmental indicators in terms of energy efficiency and water use in different agricultural systems were also assessed, resulting in the following outcomes:

- Feed production was also found as the largest contributor when particular attention was paid to water use (WF results) in the dairy system. Rationale behind this lies in the water indirectly used and polluted during the cultivation activities (green and grey WFs), rather than direct consumption rates (blue WF).

- In this regard, it was highlighted how methodological limitations on the WD scope can lead to potentially biased outcomes. Compared to the integrated approach in the WF values, WD is no longer able to integrate impacts beyond the abstraction of water from surface and groundwater courses.

- The results of ep-EROI revealed energy efficiencies of pork products from Galician pig farming close to the estimated values for pork production in other countries. Similar scores were also recorded for other products used as protein sources in the Spanish human diets (such as milk and mussels, among others).

Finally, two integrative technological schemes were evaluated, seeking to meet demand for more sustainable and responsible **manure management** practices. They were proposed in the framework of the MEM project, based on the principles of energy recovery and agri-food waste valorisation. From the results obtained, it could be concluded that:

- Nitrogen-based emissions, in terms of NH_3 and NO_3^-, were responsible for significant contributions to acidification and eutrophication, respectively; energy requirements had the greatest adverse impacts on the other environmental indicators. On the contrary, the credits derived from energy generation (biogas combustion) and organic fertilisation (recovered nutrients) were capable to offset the impacts to a large extent.

- Accordingly, the comparative analysis showed that MEM prototypes as more environmentally sustainable choices in both case studies (The Netherland and Spain). Conventional management practices were highly penalised by the effect of diffuse emissions from direct application of digestate to soils.

➯ However, since the environmental supremacy of MEM prototypes could not be guaranteed in all circumstances, socio-economic criteria should also be integrated into decision-making (along with environmental indicators) for each specific situation.

- **Section III. Urban framework**

In this section, attention has been focused on the analysis of the potential for improvement of alternative management schemes for MSW from urban areas, in comparison with conventional practices. To this end, two case studies in Galicia (Spain) and Astana (Kazakhstan) were selected on behalf of developed and developing regions, respectively. Moreover, not only have the environmental indicators been evaluated jointly, but also the social and economic variables, from a sustainability approach in developing scenarios.

The environmental assessment performed to analyse the management of MSW in **Astana (Kazakhstan)** led to the following conclusions:

➯ Despite efforts to make progress in the implementation of advanced waste management technologies, energy recovery through the combustion of landfill gas has established itself as the most widespread alternative to date, without further treatment.

➯ However, waste treatment practices based on increasing recycling rates to obtain also larger volumes of recovered fractions, apart from energy recovery, led to a more favourable environmental profile.

Similarly, the comparative results related to the strategies evaluated in the **Galician (Spain)** case study revealed that:

➯ Alternatives based mainly on the aerobic treatment (composting) of the waste were responsible for the most environmental-friendly performance. In this case, priority is given to the generation of compost from the organic matter of MSW, so its use as an organic fertiliser reported the greatest environmental credits.

- In contrast, other strategies focusing on the generation of renewable energy, either through anaerobic biodigestion or thermal processes (incineration or combustion of landfill gas), did not provide such environmentally friendly behaviour. These results were in line with existing European legislation and recommendations on waste management and environmental protection.

Finally, when the **AHP method** was applied to the study, and both social and economic indicators were integrated with previous LCA results, it was also confirmed that:

- The priority ranking remained unchanged when equal weights were attributed to the three pillars of sustainability, led by composting and followed by incineration and landfilling, respectively.

- In addition, a sensitivity analysis showed that these priorities were slightly dependent on the influence of each criterion individually analysed. The composting model remains an environmentally and socially sound alternative, while its choice would only be economically unwise. However, in view of the results, Galician authorities could encourage their promotion, trying to comply with European guidelines.

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SUPPLEMENTARY MATERIAL – ANEXX I

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Table A1.1. Main inventory data (flow and composition) for S3 (MEM Acidification) in Spain.

Composition	Units	Feeding mixtue	Digestate	Permeate	Centrate	Struvite effluent	Struvite	BNR effluent (water)
Flow	kg/d	125	122	108	92.1	135	1.69	136
DM	g/kg	73.9	49.9	30.9	15.4	10.5	150	10.0
COD	g/kg	79.3	42.4	23.6	5.19	3.54	-	3.00
TN	g/kg	4.36	4.45	3.91	3.45	2.23	-	0.67
$N-NH_4+$	g/kg	1.99	2.69	2.67	2.47	1.56	-	0.00
TP	g/kg	0.79	0.75	0.54	0.47	0.05	-	0.05
$P-PO_4^{3-}$	g/kg	0.60	0.60	0.42	0.40	0.04	-	0.04

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SUPPLEMENTARY MATERIAL – ANNEX II

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The AHP method was developed in Chapter 8 according to the following steps:

1. Definition of the problem and the hierarchical structure.

The first step of the AHP method is to subdivide the problem of decision making into several levels, including the goal of the analysis, the criteria and sub-criteria selected and the alternatives to be evaluated. Figure 8.1 displays the hierarchical structure associated with the comparative assessment developed in Chapter 8 to determine the most sustainable alternative for MSW management in Galicia.

2. Pair-wise comparison matrices.

In accordance with the AHP principles, a total of 14 pair-wise comparison matrices should be defined (see Figure 8.1): one for defining the relationship between the different criteria and the goal (Table A2.1), three for the sub-criteria with respect to the upper-level criteria (Table A2.2 – equal weight) and, finally, ten for comparing the three models with each other in relation to all criteria and sub-criteria (Table A2.3). The Saaty's Fundamental Scale should be used in this step (see Chapter 2).

3. Normalised matrices and criteria priority vectors.

Once the pair-wise matrices are created, the normalised matrices must be estimated, as well as the priority vector associated with each criterion in relation to the different models.

4. Overall priority vector.

Finally, the priority vector makes it possible to determine the most favourable alternative (model) in accordance with the goal of the study and the selected criteria. It can be estimated by aggregating the weights of the different criteria for each alternative across the hierarchical structure (Table A2.4).

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Table A2.1. Pair-wise comparison matrix, normalised matrix and priority vector in Base Scenario (BS): criteria.

Step	Criteria	Economic	Social	Environmental
Pair-wise comparison matrix	Economic	1	1	1
	Social	1	1	1
	Environmental	1	1	1
Normalised matrix	Economic	0.33	0.33	0.33
	Social	0.33	0.33	0.33
	Environmental	0.33	0.33	0.33
Priority vector	Economic		0.33	
	Social		0.33	
	Environmental		0.33	

Table A2.2. Pair-wise comparison matrix, normalised matrix and priority vector in Base Scenario (BS): sub-criteria.

		Economic criteria		
Step	**Sub-criteria**	**Capital Costs**	**O&M Costs**	**Revenues**
Pair-wise comparison matrix	Capital costs	1	1	1
	O&M costs	1	1	1
	Revenues	1	1	1
Normalised matrix	Capital costs	0.33	0.33	0.33
	O&M costs	0.33	0.33	0.33
	Revenues	0.33	0.33	0.33
Priority vector	Capital costs		0.33	
	O&M costs		0.33	
	Revenues		0.33	

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Table A2.2 (cont.). Pair-wise comparison matrix, normalised matrix and priority vector in Base Scenario (BS): sub-criteria.

Social criteria				
Step	**Sub-criteria**	**Employment**	**Social perception**	**Public H&S**
Pair-wise comparison matrix	Employment	1	1	1
	Social perception	1	1	1
	Public H&S	1	1	1
Normalised matrix	Employment	0.33	0.33	0.33
	Social perception	0.33	0.33	0.33
	Public H&S	0.33	0.33	0.33
Priority vector	Employment		0.33	
	Social perception		0.33	
	Public H&S		0.33	

Environmental criteria						
Step	**Sub-criteria**	**CC**	**TA**	**FE**	**ME**	**FD**
Pair-wise comparison matrix	CC	1	1	1	1	1
	TA	1	1	1	1	1
	FE	1	1	1	1	1
	ME	1	1	1	1	1
	FD	1	1	1	1	1
Normalised matrix	CC	0.20	0.20	0.20	0.20	0.20
	TA	0.20	0.20	0.20	0.20	0.20
	FE	0.20	0.20	0.20	0.20	0.20
	ME	0.20	0.20	0.20	0.20	0.20
	FD	0.20	0.20	0.20	0.20	0.20
Priority vector	CC			0.20		
	TA			0.20		
	FE			0.20		
	ME			0.20		
	FD			0.20		

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Table A2.3. Pair-wise comparison matrix, normalised matrix and priority vector in Base Scenario (BS): models.

		Capital Costs		
Step	**Models**	**Landfilling**	**Incineration**	**Composting**
Pair-wise comparison matrix	Landfilling	1	7	9
	Incineration	1/7	1	4
	Composting	1/9	1/4	1
Normalised matrix	Landfilling	0.80	0.85	0.64
	Incineration	0.11	0.12	0.29
	Composting	0.09	0.03	0.07
Priority vector	Landfilling		0.76	
	Incineration		0.17	
	Composting		0.06	
		O&M Costs		
Step	**Models**	**Landfilling**	**Incineration**	**Composting**
Pair-wise comparison matrix	Landfilling	1	7	9
	Incineration	1/7	1	4
	Composting	1/9	1/4	1
Normalised matrix	Landfilling	0.80	0.85	0.64
	Incineration	0.11	0.12	0.29
	Composting	0.09	0.03	0.07
Priority vector	Landfilling		0.76	
	Incineration		0.17	
	Composting		0.06	
		Revenues		
Step	**Models**	**Landfilling**	**Incineration**	**Composting**
Pair-wise comparison matrix	Landfilling	1	1/9	1/5
	Incineration	9	1	4
	Composting	5	1/4	1
Normalised matrix	Landfilling	0.07	0.08	0.04
	Incineration	0.60	0.74	0.77
	Composting	0.33	0.18	0.19
Priority vector	Landfilling		0.06	
	Incineration		0.70	
	Composting		0.24	

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Table A2.3 (cont.). Pair-wise comparison matrix, normalised matrix and priority vector in Base Scenario (BS): models.

		Employment		
Step	**Models**	**Landfilling**	**Incineration**	**Composting**
Pair-wise comparison matrix	Landfilling	1	1	1/9
	Incineration	1	1	1/9
	Composting	9	9	1
Normalised matrix	Landfilling	0.09	0.09	0.09
	Incineration	0.09	0.09	0.09
	Composting	0.82	0.82	0.82
Priority vector	Landfilling		0.09	
	Incineration		0.09	
	Composting		0.82	
		Social perception		
Step	**Models**	**Landfilling**	**Incineration**	**Composting**
Pair-wise comparison matrix	Landfilling	1	1/8	1/3
	Incineration	8	1	5
	Composting	3	1/5	1
Normalised matrix	Landfilling	0.08	0.09	0.05
	Incineration	0.67	0.75	0.79
	Composting	0.25	0.15	0.16
Priority vector	Landfilling		0.08	
	Incineration		0.74	
	Composting		0.18	
		Public H&S		
Step	**Models**	**Landfilling**	**Incineration**	**Composting**
Pair-wise comparison matrix	Landfilling	1	1/6	1/4
	Incineration	6	1	4
	Composting	4	1/4	1
Normalised matrix	Landfilling	0.09	0.12	0.05
	Incineration	0.55	0.70	0.76
	Composting	0.36	0.18	0.19
Priority vector	Landfilling		0.09	
	Incineration		0.67	
	Composting		0.24	

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Table A2.3 (cont.). Pair-wise comparison matrix, normalised matrix and priority vector in Base Scenario (BS): models.

		CC		
Step	**Models**	**Landfilling**	**Incineration**	**Composting**
Pair-wise comparison matrix	Landfilling	1	9	8
	Incineration	1/9	1	1/2
	Composting	1/8	2	1
Normalised matrix	Landfilling	0.81	0.75	0.84
	Incineration	0.09	0.08	0.05
	Composting	0.10	0.17	0.11
Priority vector	Landfilling		0.80	
	Incineration		0.08	
	Composting		0.12	
		TA		
Step	**Models**	**Landfilling**	**Incineration**	**Composting**
Pair-wise comparison matrix	Landfilling	1	1/5	1/7
	Incineration	5	1	1/2
	Composting	7	2	1
Normalised matrix	Landfilling	0.08	0.06	0.09
	Incineration	0.38	0.31	0.30
	Composting	0.54	0.63	0.61
Priority vector	Landfilling		0.08	
	Incineration		0.33	
	Composting		0.59	
		FE		
Step	**Models**	**Landfilling**	**Incineration**	**Composting**
Pair-wise comparison matrix	Landfilling	1	1/4	1/9
	Incineration	4	1	1/5
	Composting	9	5	1
Normalised matrix	Landfilling	0.07	0.04	0.09
	Incineration	0.29	0.16	0.15
	Composting	0.64	0.80	0.76
Priority vector	Landfilling		0.07	
	Incineration		0.20	
	Composting		0.73	

Table A2.3 (cont.). Pair-wise comparison matrix, normalised matrix and priority vector in Base Scenario (BS): models.

		ME		
Step	**Models**	**Landfilling**	**Incineration**	**Composting**
Pair-wise comparison matrix	Landfilling	1	1/4	1/9
	Incineration	4	1	1/5
	Composting	9	5	1
Normalised matrix	Landfilling	0.07	0.04	0.09
	Incineration	0.29	0.16	0.15
	Composting	0.64	0.80	0.76
Priority vector	Landfilling		0.07	
	Incineration		0.20	
	Composting		0.73	
		FD		
Step	**Models**	**Landfilling**	**Incineration**	**Composting**
Pair-wise comparison matrix	Landfilling	1	1/2	1/9
	Incineration	2	1	1/7
	Composting	9	7	1
Normalised matrix	Landfilling	0.08	0.06	0.09
	Incineration	0.17	0.12	0.11
	Composting	0.75	0.82	0.80
Priority vector	Landfilling		0.08	
	Incineration		0.13	
	Composting		0.79	

Table A2.4. Overall priority vector for each model in Base Scenario (BS).

Management model	**Overall Priority Vector**
Landfilling	0.28
Incineration	0.34
Composting	0.38

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NOMENCLATURE

AD	Anaerobic Digestion
AcoD	Anaerobic co-Digestion
AHP	Analytical Hierarchy Process
ALO	Agricultural Land Occupation
BNR	Biological Nitrogen Removal
CC	Climate Change
CED	Cumulative Energy Demand
CF	Carbon Footprint
CFC	Chlorofluorocarbon
CH_4	Methane
CHP	Cogeneration unit
CO_2	Carbon dioxide
COD	Chemical Oxygen Demand
Cr	Chrome
Cu	Cooper
DB	Dichlorobenzene
ELCD	European Reference Life Cycle Database
ep-EROI	Edible Protein Energy Return On Investment
FAO	Food and Agriculture Organization
FD	Fossil Depletion
FE	Freshwater Eutrophication
FET	Freshwater Ecotoxicity
FPCM	Fat and Protein Corrected Milk
FU	Functional Unit
FT	Freshwater Toxicity

GHG	Greenhouse Gas
GPR	Gas Production Rate
H_2	Hydrogen
H&S	Human and Safety
HFC	Hydrofluorocarbon
HT	Human Toxicity
IDF	International Dairy Federation
ILCD	International Reference Life Cycle Data System
IPCC	Intergovernmental Panel on Climate Change
IR	Ionising radiaton
ISO	International Organization of Standardization
JRC	Join Research Centre
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LCT	Life Cycle Thinking
LU	Land Use
MBT	Mechanical-Biological Treatment
MCDA	Multi-Criteria Decision Analysis
MD	Metal Depletion
ME	Marine Eutrophication
MEM	ManureEcoMine (project)
MET	Marine Ecotoxicity
MFE	Mineral Fertiliser Equivalent
Mg^{2+}	Magnesium
MSW	Municipal Solid Waste
MT	Mechanical Treatment
N	Nitrogen
N_2O	Dinitrogen monoxide
NE	Northeast
NH_3	Ammonia
NH_4^+	Ammonium

NLO	Natural Land Occupation
NMVOC	Non-Methane Volatile Organic Compound
NO_2^-	Nitrite
NO_3^-	Nitrate
NO_x	Nitrogen oxides
NVZ	Nitrate Vulnerable Zones
NW	Northwest
O&M	Operational and Maintenance
OD	Ozone Depletion
OECD	Organisation for Economic Co-operation and Development
OLR	Organic Loading Rate
P	Phosphorus
PMF	Particulate matter formation
PNA	Patial Nitritation/Anammox
PO_4^{3-}	Phosphate
POF	Photochemical Oxidant Formation
RDF	Refuse Derived Fuel
SETAC	The Society of Environmental Toxicology and Chemistry
S/L	Solid/Liquid
SO_2	Sulphur dioxide
TA	Terrestrial Acidification
TET	Terrestrial Ecotoxicity
UF	Ultrafiltration
ULO	Urban Land Occupation
UNEP	United Nations Environment Program
VFA	Volatile Fatty Acid
VS	Volatile Solids
WFD	Water Framework Directive
WBCSD	World Business Council for Sustainable Development
WCED	World Commission on Environment and Development
WD	Water Depletion
WEEE	Waste Electrical and Electronic Equipment

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WF	Water Footprint
WFN	Water Footprint Network
WRI	World Resources Institute
WTE	Waste-to-Energy
Zn	Zinc

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RESUMEN (SPANISH)

Con una población mundial en constante crecimiento, la demanda de productos alimentarios ha aumentado de forma significativa en los últimos tiempos y, con ella, la intensificación de los regímenes de producción en el sector de la alimentación. Esta situación ha provocado importantes problemas de seguridad alimentaria, así como innumerables daños ambientales. En este contexto, la producción agrícola, y más particularmente el sector ganadero, ha adquirido un papel protagonista debido a su vinculación directa con la industria alimentaria. En efecto, se ha demostrado la importante contribución de este sector al cambio climático y a las emisiones de gases de efecto invernadero liberados a la atmósfera, responsable también del alto consumo y de la contaminación de los recursos hídricos, entre muchos otros impactos ambientales. Además, las prácticas de agricultura intensiva, en su afán por satisfacer las necesidades de la población, han desarrollado una fuerte dependencia del uso de agroquímicos, especialmente fertilizantes, cuya producción genera una alta huella de carbono; la aplicación a largo plazo de estos fertilizantes puede contribuir además al deterioro productivo de los suelos, causando otros problemas ambientales como la acidificación y la eutrofización. Sin embargo, en Europa se generan grandes volúmenes de estiércol animal, residuo que representa una importante fuente de nutrientes todavía sin explotar.

Por otro lado, el desarrollo socio-económico sumado al crecimiento urbano de la población y la intensificación de sus actividades, ha dado lugar a la generación de volúmenes cada vez mayores de residuos sólidos en zonas urbanas. Por lo tanto, la gestión de estos residuos se ha convertido en una cuestión clave en todo el mundo, con el objetivo de mitigar adecuadamente sus efectos adversos sobre la salud humana y la calidad de los ecosistemas. Esta problemática es especialmente preocupante en las regiones menos desarrolladas, donde el tratamiento de estos residuos se encuentra todavía en sus primeras fases.

De acuerdo con lo expuesto, es evidente la relevancia de los avances en la gestión de los residuos generados, tanto en el ámbito agrícola como en el urbano, y en las estrategias de mitigación pertinentes. En este contexto, el desarrollo eficiente de tecnologías para el tratamiento del estiércol generado es cada vez más importante, postulándose como un factor clave para contrarrestar las cargas ambientales derivadas de las prácticas agrícolas. Sin embargo, a pesar de los esfuerzos para diseñar y desarrollar varias estrategias de gestión, solo algunas de ellas han podido ser implementadas con éxito a gran escala. De forma similar, la gestión de residuos sólidos urbanos (RSU) también se ha investigado con mayor intensidad en los últimos tiempos, principalmente en las zonas más desarrolladas. En este sentido, y de acuerdo con diversos estudios publicados hasta la fecha, las prácticas de gestión convencionales han ido mejorando gradualmente, en la búsqueda de un tratamiento más eficiente de los residuos biodegradables y aumentando las tasas de reciclaje. No obstante, los resultados más recientes siguen alentando a seguir trabajando en el desarrollo de soluciones respetuosas con el medio ambiente, en consonancia con las iniciativas legales en el ámbito de la protección del medio ambiente y en apoyo de la evolución socio-económica de la población.

En este contexto, se han desarrollado también diversas metodologías de análisis ambiental aplicadas al sector de la gestión de residuos, consideradas como herramientas útiles en el progreso hacia un entorno sostenible. Su función consiste principalmente en determinar en qué medida los avances en materia de gestión de residuos representan realmente una alternativa mejorada desde el punto de vista medioambiental en comparación con el perfil de las prácticas más convencionales. Entre ellas, las metodologías basadas en la perspectiva de ciclo de vida, con el Análisis de Ciclo de Vida (ACV) a la cabeza, han adquirido mayor relevancia. De hecho, se pueden encontrar muchos estudios en la literatura, basados en la aplicación de esta perspectiva a la gestión de residuos y su sostenibilidad, ya sea de origen agrícola o urbano. Asimismo, se espera que la integración del enfoque socio-económico en los estudios ambientales pueda apoyar el desarrollo de prácticas ambientalmente sostenibles sin comprometer la prosperidad económica de las regiones. Esta es la razón por la que otras metodologías complementarias también se han vuelto más relevantes en los últimos tiempos, en combinación con el enfoque de ACV, destacando la contribución de las herramientas de toma de decisiones bajo una perspectiva multi-criterio. Estas herramientas permiten identificar la alternativa óptima entre las diferentes opciones consideradas, pudiendo integrar indicadores cuantitativos y cualitativos de las diferentes áreas de interés en cada caso particular. Entre ellas, el Proceso Analítico Jerárquico (AHP – "Analytical Hierarchy Process") ha sido aceptado internacionalmente por la comunidad científica como una herramienta fiable para tratar problemas complejos de toma de decisiones en diversas áreas del conocimiento, incluida la gestión de residuos.

De acuerdo con lo anterior, el objetivo principal de esta tesis doctoral fue evaluar la sostenibilidad ambiental de diferentes alternativas de gestión de residuos de los sectores agrícola y urbano, incluyendo prácticas más convencionales y/o tecnologías avanzadas. Con este fin, los principios de la metodología de ACV se aplicaron a diferentes situaciones particulares de gestión de residuos (casos de estudio), en combinación con otras herramientas de evaluación como AHP y huella hídrica (HH)(WF – "Water Footprint"). Por lo que se refiere a los entornos agrícolas, se consideró apropiado centrar la atención no sólo en los procesos responsables de la generación de residuos (principalmente estiércol animal), sino también en las fases previas y su posible vinculación con las estrategias de gestión posteriores. Por último, la participación social y la viabilidad económica se integraron con los resultados ambientales para abordar las limitaciones de la toma de decisiones en los entornos urbanos.

De acuerdo a los objetivos propuestos, la presente tesis se ha estructurado en cuatro secciones principales: (I) Contextualización, (II) Entorno agrícola, (III) Entorno urbano y (IV) Conclusiones.

- **Sección I – Contextualización**

El objetivo fundamental de esta primera sección radica en proporcionar una visión global de la situación actual y las perspectivas futuras en la gestión de residuos agrícolas y urbanos, así como de las herramientas de evaluación de sostenibilidad ambiental disponibles en ese ámbito.

De forma más precisa, en el **Capítulo 1** se presenta una revisión bibliográfica sobre las principales fuentes de generación de residuos, las tecnologías actuales de gestión y las estrategias potenciales de valorización, diferenciando entre los sectores agrícola y urbano. Asimismo, al final de este capítulo, se presenta también un breve resumen del marco legal Europeo vigente, así como sus propuestas sobre el progreso hacia economías circulares sostenibles, también en el ámbito de gestión de residuos.

En el **Capítulo 2** se presentan, en primer lugar, los conceptos de desarrollo sostenible y ciclo de vida junto con las diferentes herramientas metodológicas disponibles y, a continuación, se profundiza en los principios fundamentales de las metodologías aplicadas en los capítulos siguientes de la tesis. Finalmente, en este capítulo se presentan también los objetivos principales y estructura de la tesis en sus diferentes secciones y capítulos.

- **Sección II – Entorno agrícola**

El fin último de esta segunda sección se centra en aplicar los principios de ACV para evaluar el perfil ambiental de diferentes alternativas actuales y avanzadas en la gestión de residuos ganaderos, procedentes de los sistemas de producción lechera y porcina. Con este objetivo, se realizó un análisis integrado, teniendo en cuenta, además, los posibles impactos ambientales de las etapas anteriores que pueden contribuir de alguna manera a la generación de residuos en la granja, así como a su valorización posterior como fuente de nutrientes.

En el **Capítulo 3** se analiza y compara el perfil ambiental de diversas variedades de cereales cultivadas bajo diferentes regímenes de cultivo, incluyendo dos cultivos de verano (maíz y sorgo) y cuatro cultivos de invierno (trigo, triticale, cebada y centeno). Estos cereales se cultivan ampliamente en una de las principales regiones agrícolas de Europa (Valle del Po – Región de Lombardía), utilizados principalmente como fuente de proteínas en la alimentación animal. De ahí la elección de 1 kg de proteína (cruda) como base para el análisis comparativo.

En primer lugar, para cada uno de los sistemas de cultivo evaluados, se identificaron las emisiones de campo, las actividades agrícolas y la producción de agroquímicos (incluyendo pesticidas y fertilizantes) como los factores con mayor impacto en todos los sistemas evaluados, independientemente del tipo de cereal y la variedad estudiada. Sobre la base de los resultados comparativos, el uso de prácticas menos intensivas y mayores rendimientos de biomasa determinó los perfiles más favorables para el medio ambiente del maíz (clase

600-700) y el centeno, entre los cereales de verano e invierno, respectivamente. Sin embargo, los resultados del análisis de sensibilidad demostraron que el ranking de prioridades podía variar al modificar la base de comparación; así, mientras que las conclusiones eran análogas al considerar criterios másicos, los resultados se modifican de forma considerable cuando se realiza la comparación en términos de uso del suelo de cultivo (1 ha). En este caso, las mayores necesidades de agroquímicos por hectárea tienen un mayor efecto en detrimento de los mayores rendimientos de biomasa obtenidos. Esto muestra cómo la elección de la mejor y/o peor alternativa puede depender en gran medida de la base de cálculo en este tipo de sistemas agrícolas.

El **Capítulo 4** se centra en la evaluación del comportamiento medioambiental de un sistema de producción de leche de vaca representativo del sector lácteo en Galicia (Noroeste de España). Para ello, se aplicaron los principios de ACV en combinación con las directrices establecidas por la Federación Internacional de Lechería (IDF – "International Dairy Federation"). De acuerdo con los resultados, las emisiones difusas procedentes de la cría del ganado en la granja (fundamentalmente CH_4 procedente de la fermentación entérica) y la posterior gestión del estiércol generado, fueron identificadas como un factor decisivo en el impacto ambiental global. Asimismo, en línea con el capítulo anterior, la producción de piensos para el ganado (principalmente compuestos de cereales y alfalfa) también contribuyó de forma importante a la mayoría de los indicadores medioambientales analizados.

En este contexto, se llevó a cabo un análisis de sensibilidad para estimar el potencial de mejora del perfil ambiental del sistema mediante la sustitución de la alfalfa en las dietas por ingredientes alternativos como ensilado de hierba y ensilado de maíz. En los resultados se verificaron los créditos ambientales (aunque menores) derivados del uso de ensilado de maíz, definido como una opción ambientalmente sostenible en el capítulo anterior; sin embargo, no se pudo demostrar ninguna ventaja ambiental asociada con el uso de ensilado de hierba con respecto a la situación actual de la alfalfa. Finalmente, se determinó también la relevancia de la producción de alimento como factor clave sobre los impactos sobre el uso del agua, de acuerdo a los resultados del estudio de HH desarrollado de acuerdo con la metodología establecida por la Red de Huella

Hídrica (WFN – "Water Footprint Network"). En este caso, las cargas medioambientales proceden del consumo indirecto de agua durante las actividades de cultivo y de las tasas de contaminación de los recursos hídricos, más que del consumo directo de agua procedente de las actividades agrícolas.

De forma análoga al capítulo anterior, en el **Capítulo 5** se realiza el estudio ambiental de la producción de carne de cerdo en España, tomando como referencia dos casos de estudio ubicados en dos de las áreas españolas de mayor actividad agrícola y ganadera: Galicia y Cataluña. Mientras que en el caso gallego se evaluó la cadena de producción hasta la fase de explotación en la granja, en el caso catalán se amplió el alcance del estudio para integrar también las fases posteriores de matadero y transformación del producto final. Ambos análisis se desarrollaron a partir de una serie de indicadores ambientales comunes; además, se evaluaron la eficiencia energética y el impacto sobre los recursos hídricos para Galicia y Cataluña, respectivamente. A partir de los resultados obtenidos, se pueden extraer las siguientes conclusiones comunes a los dos sistemas estudiados.

De nuevo las emisiones difusas procedentes de la etapa de cría en las granjas destacaron como uno de los mayores responsables de los impactos ambientales; sin embargo, las mayores cargas ambientales en ambos casos se identificaron en la producción de la alimentación animal, fundamentalmente compuesta por piensos en el sector porcino. En este sentido, se ha demostrado que los impactos derivados del cultivo de los cereales que constituyen la mayor parte de la formulación de los piensos tienen una contribución fundamental a los resultados globales, tanto en el caso de los indicadores comunes a ambos casos de estudio, como en relación con los resultados obtenidos sobre el impacto en el uso del agua (HH) en el caso catalán. En consecuencia, se propusieron diversas estrategias para mitigar los impactos asociados a la producción de piensos, con especial atención a la mejora de la eficiencia de los recursos utilizados en formulaciones alternativas. Por último, la carne de cerdo generada en el sistema gallego presentaba un balance energético análogo al de otros productos cárnicos extranjeros, así como al de fuentes alternativas de proteínas en la dieta humana.

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Finalmente, en esta segunda sección, el **Capítulo 6** pretende proponer un enfoque alternativo a la conversión de los residuos ganaderos (estiércol) en productos de valor añadido, minimizando los impactos ambientales de las prácticas de gestión convencionales en las granjas (Capítulos 4 y 5). Este estudio se lleva a cabo en el marco del proyecto europeo ManureEcoMine (MEM), basado en el diseño e implementación (a escala piloto) de diferentes esquemas de tratamiento integral para la gestión sostenible de estiércol de cerdo y de vaca en Holanda y España, respectivamente, en combinación con otros residuos orgánicos procedentes de la industria alimentaria. En ambos casos, los prototipos desarrollados en el proyecto se basan en complementar el proceso de digestión anaeróbica convencional con otras etapas posteriores de recuperación de digestato, incluyendo la unidad de separación sólido-líquido (en una o dos etapas), la precipitación de estruvita y la eliminación biológica de nitrógeno.

Estos prototipos fueron evaluados ambientalmente en comparación con la aplicación directa de digestato como fertilizante orgánico, sin ningún proceso adicional. Los resultados comparativos demostraron los créditos ambientales potenciales de la implementación de los prototipos en ambas ubicaciones, en la mayoría de las categorías de impacto consideradas. Mientras que la recuperación de energía del biogás de la etapa de digestión anaeróbica compensa parcialmente las cargas ambientales en todos los escenarios definidos en el estudio, las emisiones difusas del uso directo del digestato en suelos agrícolas penalizan significativamente los perfiles ambientales de los escenarios convencionales. Por el contrario, las cargas evitadas debido a la recuperación de nutrientes de la corriente de digestato ejercen una influencia favorable en los resultados ambientales de los prototipos, contrarrestando en mayor medida sus impactos. Sin embargo, en ausencia de un claro predominio de estos escenarios sobre las prácticas convencionales, podría concluirse que la preferencia por una u otra alternativa dependerá de cada situación particular, pudiendo integrarse también criterios socioeconómicos en la toma de decisiones.

- **Sección III – Entorno urbano**

De forma análoga al enfoque de la sección anterior, el objetivo de esta sección es analizar el perfil actual y el potencial de mejora ambiental derivado de la implantación de sistemas alternativos de gestión de residuos sólidos en zonas urbanas (RSU), tanto en países desarrollados como en desarrollo.

A tal fin, en el **Capítulo 7** se evaluaron, bajo la perspectiva ACV, dos casos de estudio involucrando la gestión de RSU en Galicia (España) y Astana (Kazajstán), en representación de regiones desarrolladas y en desarrollo, respectivamente. Considerando en primer lugar los resultados del caso gallego, se observó que la alternativa centrada principalmente en el tratamiento aeróbico (compostaje) de la fracción orgánica presentaba el mejor perfil ambiental. Los mayores créditos asociados a la conversión de residuos degradables en compost priorizan la elección de este modelo de gestión frente a otras estrategias más enfocadas a la generación de energía renovable, ya sea mediante procesos de biodigestión anaeróbica o térmicos (incineración o combustión de gases de vertedero).

En contraposición, a pesar de los esfuerzos establecidos recientemente para la implementación de tecnologías de gestión de RSU más avanzadas, la recuperación de energía por combustión de los gases de vertedero se ha afianzado como la alternativa más extendida hasta la fecha en Astana. Sin embargo, se propusieron y evaluaron escenarios de tratamiento alternativos a la situación actual, basados en el aumento de las tasas de reciclaje para obtener mayores volúmenes de materiales recuperados, además de la recuperación de energía en el vertedero; los resultados ambientales demostraron perfiles ambientales más favorables con respecto a las prácticas actuales.

Finalmente, y de forma complementaria al estudio desarrollado en el capítulo anterior para el caso gallego, en el **Capítulo 8** se aplican los principios de la metodología AHP para integrar criterios económicos y sociales junto con los resultados ambientales previamente obtenidos. La integración de los tres grupos de indicadores (que representan los tres pilares de la sostenibilidad) permite comparar la sostenibilidad de diferentes modelos alternativos de gestión de los RSU en Galicia: incineración con recuperación energética, compostaje y vertedero. De acuerdo con los resultados comparativos, el compostaje (tratamiento aeróbico para la producción de compost) se identificó como la alternativa de gestión más sostenible, asumiendo un peso igual para los tres grupos de indicadores. El equilibrio medioambiental más favorable fue decisivo para la elección de la alternativa de compostaje frente a las otras opciones.

Sin embargo, se llevó a cabo un análisis de sensibilidad para determinar en qué medida la ponderación de los indicadores podría afectar a la clasificación de las prioridades. Los resultados consolidaron el compostaje como el modelo de gestión más respetuoso con el medio ambiente, así como una alternativa económicamente viable y socialmente comprometida; sin embargo, los mismos resultados también mostraron que el modelo de incineración y el de vertedero eran líderes en la toma de decisiones al dar prioridad a la dimensión social y económica, respectivamente. En este sentido, podría ser conveniente aunar esfuerzos para mejorar el perfil socio-económico del compostaje de RSU en la comunidad gallega, en línea con las propuestas impulsadas por la Comunidad Europea para avanzar hacia un marco sostenible de gestión de residuos.

- **Sección IV – Conclusiones**

En la última sección se presenta un resumen de los principales resultados y conclusiones obtenidos durante el desarrollo de la tesis. Así, en el **Capítulo 9** se lleva a cabo una revisión de los principales hallazgos identificados en las diferentes secciones y capítulos, su relación con la situación actual de gestión de residuos (en base a diferentes casos de estudio y ubicaciones) y su potencial contribución a las estrategias futuras. De acuerdo a lo establecido en este capítulo, se podría concluir que los estudios desarrollados en la presente tesis han contribuido: (i) a profundizar en el análisis de los impactos ambientales asociados a prácticas convencionales de gestión de residuos agrícolas, en diferentes etapas de la cadena de producción; (ii) a la identificación y evaluación ambiental de los modelos de gestión de RSU que actualmente conviven en diferentes áreas de desarrollo socio-económico; (iii) a la estimación de los potenciales créditos ambientales derivados de las estrategias alternativas de gestión de residuos, basadas en la implementación de tecnologías avanzadas en el marco de la sostenibilidad; (iv) a la incorporación de criterios sociales y económicos, junto con indicadores ambientales, en la toma de decisiones para la gestión integrada y sostenible de residuos.

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