UNIVERSIDADE DE SANTIAGO DE COMPOSTELA

Departamento de Enxeñaría Química

Envisioning the sustainability of bioenergy production from anaerobic digestion

Memoria presentada por

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Informan:

Que a presente memoria titulada "Envisioning the sustainability of bioenergy production from anaerobic digestion" presentada por Dona Lucía Lijó Batalla, para optar ó grao de Doutor en Enxeñaría Química, Programa de Doutoramento en Enxeñaría Química e Ambiental, foi realizada baixo a nosa inmediata dirección no Departamento de Enxeñaría Química da Universidade de Santiago de Compostela.

E para que así conste, firman o presente informe en Santiago de Compostela, maio de 2017.

María Teresa Moreira Vilar   Sara González García

Outline of the thesis

Chapter 5 – Eco-efficiency assessment of agricultural biogas

Chapter 6 – Sustainable management of manure – The LiveWaste project

Abstract

One of the main challenges of modern society is related to the provision of energy systems that cover its needs while preserving the environment. As a consequence, the European Union has set ambitious goals to foster the implementation of renewable energy. Particularly, 20% energy from renewable sources in the final gross consumption of energy and 10% renewable energy in the transportation sector must be accomplished by 2020. In parallel, society has also the challenge to take effective measures to both counteract the drivers of on-going climate change and improve public response to its consequences. Another European target established a reduction target of 20% of greenhouse gas (GHG) emissions by 2020 in comparison with 1990. The 21st Conference of the Parties of the United Nations Framework Convention on Climate Change (UNFCCC) ended with the adoption of the Paris Agreement, which establishes the framework for combating climate change from 2020 onwards. The primary objective set is to avoid that the increase of the average global temperature exceeds 2°C in comparison with the pre-industrial level, which should be attained improving both mitigation and adaptation to climate change. With this scenario in mind, the diversification of technologies and resources for the production of renewable energy and the decarbonisation of the energy system is nowadays mandatory. In this context, biogas is being promoted as a sustainable form of energy as it shares the principles of circular economy by converting these wastes into valuable products such as energy, water and nutrients.

Assessment methodologies are being developed to provide evidence of the benefits and drawbacks of renewable energy production, in terms of its economic, social and environmental sustainability. Among them, Life Cycle Assessment (LCA) is considered an appropriate methodology for the quantification of the environmental consequences of biogas production. Numerous LCA studies are available in the literature concerning biogas production and use. In these studies, biogas production systems from different feedstock (mono- and co-digestion) as well as their possible applications have been assessed from environmental and energy perspectives, with special attention on GHG emissions and fossil fuel depletion. Recently, the combined implementation of Data Envelopment Analysis (DEA) with LCA has also been proposed to analyse the eco-efficiency of multiple similar entities. This novel alternative avoids the use of average inventory data and enriches results interpretation through eco-efficiency verification. This approach has been applied using technical and environmental indicators to different production processes such as wine production and fisheries and even to wastewater treatment plants (WWTPs). The Analytic Hierarchy Process (AHP) is another available methodology for multi-criteria decision making. The AHP can be used for the hierarchical decomposition of a complex problem, helping to identify the best option considering several sets of criteria with different nature. It has been widely applied to analyse the sustainability of energy and waste systems, including for the selection of the best alternative for energy recovery from municipal solid waste, the selection of the best solid waste treatment technology, to rank suitable locations to place a municipal solid waste facility and to analyse the sustainability of cities.

The objective of this doctoral thesis is to examine the environmental sustainability of different biogas systems in the European Union and to evaluate their suitability in accordance with the European energy objectives stated. By applying the LCA methodology and other evaluation tools, such as DEA and AHP, a variety of biogas systems were investigated in the context of different European countries. These biogas systems comprised agricultural and sewage biogas, involving both well-established technologies and more innovative proposals. According to the objectives proposed, the thesis has been divided into four sections, including i) Introduction, ii) Agricultural biogas, iii) Sewage biogas and iv) Conclusions.

Section I: Introduction

This section aims at contextualising the thesis providing general information regarding biogas production and the available tools for the assessment of its environmental sustainability.

✓ Chapter 1 reviews the relation between fossil fuels consumption, energy production and climate change. In addition, it is also discussed the role of renewable energies in the European objectives for combating climate change and assuring energy security, including the objective regarding the limitation in the increase of the average global temperature. Biogas is analysed as a low-carbon energy source that can make a large contribution to the energy supply in the European Union due to the wide range of biomass suitable for anaerobic digestion.

✓ Chapter 2 explores the concept of sustainable development and the available methodologies for the assessment of biogas systems in terms of sustainability, paying special attention to methodologies applied in this thesis, i.e. LCA, DEA in combination with LCA and AHP. The objectives and structure of the thesis are also presented at the end of Chapter 2.

Section II: Agricultural biogas

The environmental sustainability of biogas production from agricultural biomass, mainly animal manure and energy crops, was analysed in detail in Section II.

✓ Chapter 3 assesses, from a life cycle perspective, the environmental sustainability of four different full-scale biogas plants operating in Italy and considered representative of the state-of-the-art. These biogas plants, which digest different agricultural feedstock, were compared with the aim of identifying the most polluting stages in which improvement options could be proposed. In more detail, Plant 1 uses pig slurry as the only substrate; Plant 2 and 3 co-digest animal manure (pig slurry) and energy crops (maize silage and, in the case of Plant 2, triticale silage) and Plant 4 perform the mono-digestion of maize silage. All of them use the biogas in a co-generation heat and power (CHP) engine, the produced electricity is injected in the national grid while heat is not fully exploited. Accordingly, the functional unit (FU) selected was the production of 1 MWh of electricity. The digestate, which is co-produced during anaerobic digestion, is used as an organic fertiliser in the cultivation of energy crops. An avoided product perspective was undertaken for the surplus digestate that is not required for the production of own crops. The results demonstrated that biogas systems based on waste treatment achieved GHG savings when injected in the

Italian electricity mix; however, biogas plants digesting a high ratio of energy crops obtained worse results. Eutrophication and acidification impacts were higher in all cases, mainly associated with the cultivation of energy crops and digestate management. Strategies to mitigate environmental impacts were proposed and analysed. The results proved that the use of the surplus heat in a nearby greenhouse contributed to improve the system profile due to credits derived from avoided production of heat from diesel. The injection of digestate rather than surface spreading influenced the derived emissions, increasing nitrous oxide but lowering ammonia. Factors affecting the performance of the biogas plants were also examined in detail to understand the environmental results obtained. Pig slurry appeared as a poor source of energy, leading to higher digestate production per unit of biogas produced. Related with this, maize silage was identified as the best source of energy due to higher biogas potential compared not only to pig slurry but also to triticale. However, other factors that vary the efficiency of the anaerobic digestion process and the conversion into electricity should be also considered. To conclude, a comparison between the results obtained in the chapter with other studies was carried out in order to identify common outcomes and methodological factors hindering comparison.

✓ Following up this study, Chapter 4 delves into the environmental implications of feedstock selection in biogas systems, not only in terms potential energy production from substrates, but also related to the quantity and quality of the produced digestate. With this regard, two additional biogas plants were modelled and analysed in detail. While Plant 5 performs the digestion of animal waste (pig slurry and chicken manure) and energy crops (maize and triticale silages); Plant 6 apart from maize silage and pig slurry, also includes food waste from supermarkets and households. In terms of GHG emissions, Plants 6 performs better than Plant 5; however, both presented savings compared with the Italian electric profile. In 2014, the European Commission established a benchmark value, defined as 70% savings of GHGs compared to a fossil fuels reference system, as requirement for considering energy production from biomass as sustainable. With this in mind, only Plant 6 attained results comparable to those proposed by the European Commission; on the contrary, GHG emissions produced during the cultivation of energy crops increased the impact produced in Plant 5. Conversely, Plant 5 performs better than Plant 6

regarding acidification and eutrophication impacts, due to higher amount of animal manure digested that ended up in higher amount of digestate managed. Nevertheless, food waste was identified as an interesting source of energy since no environmental burdens from its production are allocated to the biogas system (different from energy crops) and it has high biogas potential (especially compared to animal waste). It has been also assessed the influence of methodological implications, including i) methodologies for the estimation of digestate derived emissions and ii) the selection of the FU. Important differences were identified in the environmental results when different calculation methodologies are applied; however, none of them were selected as the most appropriate one since they consider different factors affecting emissions and are internationally accepted. Among the FU studied, the use of electricity output revealed as an appropriate FU since it takes into consideration the conversion efficiencies of the two most important processes in a biogas system: feedstock into biogas and biogas into bioenergy. Finally, the biogas potentials of substrates used were compared with data published in the literature.

✓ In Chapter 5, the eco-efficiency of 15 real biogas plants is analysed by applying the LCA+DEA approach. The assessment comprised some of the biogas plants already assessed in Chapters 3 and 4 (Plants 2 – 6) and 10 new plants, located in the same area. Most of the biogas plants under study adopt a co-digestion approach treating energy crops (mainly maize silage and triticale, but also ryegrass and sorghum) mixed with animal waste (pig, cow and chicken manure) in different ratios. All of them co-produce heat and electricity in a CHP with the aim of injecting electricity into the national grid. The DEA matrix employed for the study was composed by 3 inputs, including i) feedstock production, ii) consumed electricity and iii) transport; and one single output i.e. produced electricity. The model used to evaluate the matrix was i) the slacks-based measure of efficiency model, since it allows calculating the efficiency scores regardless the units of measure used for the set of inputs and outputs; ii) with variable return to scale because the operational sizes of the plants were very different and iii) considering an input-oriented approach, to minimise the use of resources without a reduction in final electricity production. Among the 15 plants, a total of 9 operated under eco-efficient conditions (efficiency of 100%) and the average efficiency for the sample was 85%, showing a general high operational efficiency among the sample. For the non-efficient plants, the target reduction percentages were also proposed. The comparison of the environmental profile of the original and the virtual plants (after applying theoretical reduction percentages of inputs) showed that the environmental gains differed among impact categories, being GHG and eutrophication emissions, the main reduced environmental impacts. However, regarding the total environmental impact, the reduction targets based on the eco-efficiency principles influenced the plants exhibiting the worst overall environmental impact, expressed as a single score, evidencing the effectiveness of the combined LCA+DEA methodology. On the top of that, an analysis was carried out in order to identify parameters influencing environmental efficiency; however, none has been highlighted as an important driving force of the whole system efficiency, meaning that the eco-efficiency of these complex systems is controlled by a compendium of underlying factors. As a final point, the role of digestate in biogas systems and specifically in LCA studies was discussed in detail.

✓ Chapter 6 includes the assessment of an innovative treatment system for the management of animal manure in Cyprus in the framework of the LiveWaste project (LIFE12 ENV/CY/000544). The pilot plant, designed to perform the anaerobic digestion of animal manure and the recovery and removal of nutrients from the anaerobic effluent, was analysed. Moreover, four possible configurations for the treatment of the produced digestate were proposed. The results showed that the best configuration is the one that maximises phosphorus recovery as struvite, since it is a high quality fertiliser. Additionally, the potential environmental benefits of the implementation of the integrated system at full-scale were evaluated by comparing with the four most typical systems applied for animal waste in Cyprus. The use of anaerobic lagoons was recognised as the worst manure management practice due to on-site emissions. The conventional biogas plant, without digestate processing, presented lower GHG emissions than the plant proposed in the project; however, it produced higher acidification impacts due to ammonia emissions derived from the digestate. To conclude, the AHP was applied to integrate economic, social and environmental indicators in the selection of the most sustainable waste management system in Cyprus. The results showed that, even if the capital and operational costs of the LiveWaste treatment are higher, the environmental, social and economic benefits of this animal waste management option made it the most sustainable.

Section III: Sewage biogas

This section analyses the environmental consequences of anaerobic digestion as a waste valorisation option in the context of wastewater treatment, including different WWTP in terms of treatment capacities, expressed as population equivalent (PE).

✓ Chapter 7 analyses the potential environmental consequences of the co-management of food waste and sewage sludge to provide evidence of the potential benefits of co-digestion. The aim was to address legal barriers and obstacles that hinder the development of integrated circular value chains towards more sustainable waste management options. Different integration rates of food waste were proposed within the facilities of a large WWTP in the United Kingdom serving 150,000 PE: i) Scheme 1 – mono-digestion of sewage sludge as base case; ii) Scheme 2 – the co-digestion of source segregated food waste (SSFW) considering the spare capacity of the existing digester and iii) Scheme 3 – the co-digestion of the total amount of food waste produced by the community in an additional digester of the WWTP. According to the results obtained, the co-digestion of food waste and sewage sludge in a WWTP (Scheme 3) could multiply the electricity from biogas by a factor of 2.7, enhancing the performance of the plant in terms of GHG emissions. Moreover, the additional biogas produced due to the total integration of food waste entails the production of enough electricity to cover the requirements of the whole plant and the surplus can be injected in the national electric grid, delivering economic benefits from its associated incentives. However, higher proportion of biomass digested involves higher production of digestate (increased by 15%), entailing other impacts such as acidification emissions. The main barriers that hinder the application of the proposed schemes in the United Kingdom were also examined in detail to promote the use of food waste in co-digestion with sewage sludge as a sustainable and cross-sectorial solution of urban waste management. The regulatory context of the United Kingdom prevents the co-digestion of these substrate since both digestate and bioenergy from sewage sludge and food waste are controlled under different regulations. The Environmental Permitting scheme requires all biogas plants to obtain a permit or exemption to operate and to spread digestate, entailing different charges. When a feedstock is considered as a waste, the digestate deriving from it is classified as waste until they meet the End of Waste criteria for digestate However, in the United Kingdom, as in other countries (e.g. Sweden, Germany), sewage sludge is excluded from this list. This creates market barriers for the adoption of the integrated management schemes within the water utilities.

✓ Chapter 8 is focused on the assessment of an innovative treatment scheme for the combined management of sewage and domestic organic waste (DOW) at decentralised level in a small community of 2,000 PE. The base case treatment scheme proposed included: i) an upflow anaerobic sludge reactor (UASB) to treat sewage and to produce biogas, ii) a fermentation process to produce volatile fatty acids (VFAs) from DOW, iii) a sequencing batch reactor (SBR) to remove nutrients from the UASB effluent and iv) a composting process to treat the excess sludge and convert it to compost to be applied as soil conditioner. In addition, 12 alternative configurations were compared combining different integration rates of food waste disposers (FWDs), alternative nitrogen removal processes and the removal or not of phosphorus, with the aim of identifying the most suitable one from a technical and environmental point of view. The environmental assessment of base case system showed that the main *hotpots* were direct emissions and electricity production. Dissolved methane is presented in the anaerobic effluent and it is released in further stages of the treatment processes, producing an important source of GHG emissions. Moreover, impacts in eutrophication related categories were derived from the discharge of the treated effluent. The results of the comparative study shown that the use of FWDs increased the environmental impacts compared to the separate collection, while denitrification via nitrite entailed lower impacts in energy-related categories compared to nitrogen removal via nitrate, while achieving good effluent quality. The schemes which perform nitrogen removal and phosphorus uptake removal via nitrite resulted in better environmental profile concerning eutrophication impacts.

Section IV: Conclusions

The last section summarises the findings of the thesis, including the main conclusions and recommendations.

✓ Chapter 9 reviews the work developed in this thesis, identifies the main conclusions and proposes recommendations to improve the sustainability of biogas production. The thesis provides remarkable evidence through extensive application of the LCA methodology in different biogas systems in the European Union. One of the objectives achieved in the thesis was to provide tools that allow the improvement of the environmental performance of conventional biogas production. With this aim, the most polluting processes were identified and measures that reduce derived impacts were proposed and analysed. In addition, this thesis also aimed at promoting the implementation of innovative systems that integrates biogas production with digestate treatment with low eutrophication and acidification impacts. In this sense, LCA proved to be a valuable method to provide evidence of the best treatment configuration from an environmental perspective previous to their implementation at full scale. Moreover, throughout the whole thesis, the methodological barriers in the application of LCA to biogas systems were identified to provide future progress opportunities.

Resumen

Uno de los principales retos que debe afrontar la actividad productiva y económica actual está relacionado con el desarrollo de sistemas energéticos que cubran las necesidades de la sociedad, preservando al mismo tiempo el medio ambiente. Un número creciente de países han puesto en marcha objetivos e iniciativas políticas para incrementar la presencia de energías renovables en sus matrices energéticas. En dicho contexto, la Unión Europea ha promovido diferentes acciones para lograr el objetivo del 20% en términos de consumo bruto de energía procedente de fuentes renovables en 2020. En paralelo, se propone la reducción del 20% de las emisiones de gases de efecto invernadero (GEI) para 2020 en comparación con 1990. En la búsqueda de medidas eficaces para mitigar el cambio climático y mejorar la respuesta a sus consecuencias, La Convención Marco de Naciones Unidas sobre Cambio Climático creó la Conferencia de las Partes en 1995, la cual, a través del Acuerdo de París en 2015, establece el marco de actuación a partir de 2020. El principal objetivo acordado es evitar que el aumento de la temperatura media global sea superior en 2°C al valor registrado en el período preindustrial. Para lograr estos objetivos, se plantea la diversificación de tecnologías y recursos para la producción de energía renovable y la descarbonización del sistema energético. Entre las diferentes alternativas más desarrolladas, la valorización de biomasa para la producción de biogás se considera una fuente sostenible de energía, ya que comparte los principios de economía circular, siendo capaz de convertir residuos en productos de valor añadido como energía, agua de riego y nutrientes.

Son necesarias metodologías de evaluación que permitan evidenciar los beneficios e inconvenientes de la producción de este tipo de energía renovable, en cuanto a su sostenibilidad económica, social y ambiental. Entre ellas, el Análisis de Ciclo de Vida (ACV) se considera una metodología apropiada para la cuantificación de las consecuencias ambientales de la producción de biogás. Existen numerosos

estudios de ACV publicados sobre producción y uso de biogás, incluyendo diferentes sustratos y usos del biogás, prestando especial atención a las emisiones de GEI y al agotamiento de los combustibles fósiles. Recientemente, se ha propuesto la implementación combinada del ACV con el Análisis por Envoltura de Datos (DEA, acrónimo de su nombre en inglés "Data Envelopment Analysis"), con el fin de analizar la eco-eficiencia de múltiples entidades similares. Esta herramienta evita el uso de datos de inventario promedio y enriquece la interpretación de los resultados mediante la verificación de su ecoeficiencia. Este enfoque se ha aplicado utilizando indicadores técnicos y ambientales a diferentes procesos del sector primario tales como vino y pesca, e incluso a sistemas de tratamiento de aguas residuales. El Proceso Analítico Jerárquico (AHP, acrónimo de su nombre en inglés "Analytical Hierarchy Process") puede utilizarse como metodología válida para llevar a cabo la descomposición jerárquica de un problema complejo, ayudando a identificar la mejor opción considerando criterios de diferente naturaleza. Se ha aplicado ampliamente para seleccionar las alternativas más sostenibles en sistemas de energía y de residuos, incluyendo, por ejemplo, la selección de la mejor alternativa para la recuperación de energía a partir de residuos sólidos municipales.

El objetivo de esta tesis doctoral es examinar la sostenibilidad ambiental de diferentes sistemas de biogás en la Unión Europea y evaluar su idoneidad de acuerdo con los objetivos europeos en cuanto a producción de energía. Para ello se aplicó la metodología ACV y otras herramientas de evaluación, como DEA y AHP, en diferentes sistemas de producción de biogás a partir de biomasa agrícola, aguas residuales o lodos de depuradora, abarcando tanto tecnologías convencionales bien establecidas como propuestas más innovadoras. De acuerdo con los objetivos propuestos, la tesis se ha dividido en cuatro secciones, incluyendo i) Introducción, ii) Biogás agrícola, iii) Biogás en plantas de tratamiento de aguas residuales y iv) Conclusiones.

Sección I: Introducción

Esta sección pretende contextualizar la tesis proporcionando información general sobre la producción de biogás y las herramientas disponibles para la evaluación de su sostenibilidad ambiental.

✓ El capítulo 1 aborda la relación entre el consumo de combustibles fósiles, la producción de energía y el cambio climático. Además, se discute el papel de las energías renovables en los objetivos europeos de lucha contra el cambio climático y de seguridad energética. El biogás se analiza como una fuente de energía que puede suponer una gran contribución al suministro de energía y reducción de emisiones GEI en la Unión Europea debido a la amplia gama de biomasa disponible.

✓ El capítulo 2 explora el concepto de desarrollo sostenible y las metodologías disponibles para la evaluación de los sistemas de biogás en términos de sostenibilidad, prestando especial atención a las metodologías aplicadas en esta tesis, es decir, ACV, DEA y AHP. Los objetivos y estructura de la tesis también se presentan al final del capítulo 2.

Sección II: Biogás agrícola

La sostenibilidad ambiental de la producción de biogás a partir de biomasa agrícola, principalmente estiércol y cultivos energéticos, se evalúa en detalle en la Sección II.

✓ En el capítulo 3 se examina y compara la sostenibilidad ambiental de cuatro plantas de biogás reales en Italia con el objetivo de identificar las etapas más contaminantes con potencial de mejora ambiental. La Planta 1 utiliza purín de cerdo como único sustrato, mientras que las plantas 2 y 3 codigieren residuos animales (de gallina y de cerdo) y cultivos energéticos (maíz y triticale) y por último la Planta 4 sólo digiere maíz. Todos los sistemas evaluados incluyen la transformación de biogás en un motor de cogeneración para producir energía eléctrica y térmica; de modo que la electricidad producida se suministra a la red nacional, mientras que el calor se aprovecha para los servicios de aporte de calor de los digestores anaerobios. Por ello la unidad funcional seleccionada fue la producción de 1 MWh de electricidad. El digestato, que se produce durante la digestión anaerobia, se utiliza como fertilizante orgánico en el cultivo de los cereales que usa cada planta. Se empleó una perspectiva de producto evitado para los casos en los que hay un exceso de digestato, considerando que evita el uso de fertilizantes minerales. Los resultados demostraron que los sistemas de biogás pueden suponer reducciones de emisiones de GEI comparado con el perfil energético italiano cuando se opera con un balance adecuado de cultivos energéticos y residuos. A pesar de ello, los impactos en eutrofización y acidificación fueron en todos los casos mayores que el sistema de referencia, asociados no sólo con el cultivo de los cereales, sino también con la gestión del digestato. Se propusieron y analizaron estrategias para mitigar los impactos ambientales. Los resultados demostraron que el aprovechamiento del calor producido en la cogeneración en un invernadero cercano contribuyó a reducir los impactos ambientales debido a los créditos derivados de la producción evitada de calor a partir de diésel. La inyección del digestato en el suelo en lugar de su aplicación en superficie influyó en las emisiones derivadas, con mayor producción de óxido nitroso y menor de amoníaco. Los factores que afectan al rendimiento de las plantas de biogás también fueron examinados en detalle para comprender los resultados ambientales obtenidos. El purín de cerdo se identificó como una fuente limitada de energía, conllevando una mayor producción de digestato por cantidad de biogás producido. Relacionado con esto, el maíz presentó mayor potencial de biogás comparado no sólo comparado con el purín de cerdo, sino también con el triticale. Sin embargo, también se deben considerar otros factores que varían la eficiencia del proceso de digestión anaerobia y la conversión en bioenergía. Para concluir, se realizó una comparación entre los resultados obtenidos con otros estudios con el fin de identificar aspectos comunes y aquellos factores metodológicos que dificultan la comparación.

✓ El capítulo 4 profundiza en las implicaciones ambientales de la selección de materias primas en la producción de biogás, no sólo en términos de potencial energético, sino también en relación con la cantidad y calidad del digestato producido. Para ello se analizaron y modelaron dos plantas de biogás. La Planta 5 utiliza estiércol de gallina y cerdo como sustratos, además de cultivos energéticos como el maíz y el triticale. Sin embargo, la Planta 6, además de purín de cerdo y maíz, también utiliza los residuos de comida procedentes de supermercados y hogares. En términos de emisiones de GEI, la Planta 6 presentó mejores resultados que la Planta 5; no obstante, ambas consiguieron menor huella de carbono comparando con el perfil eléctrico. En 2014, la Comisión Europea estableció un valor de referencia como requisito de cara a considerar la producción de energía de la biomasa como sostenible. De tal forma, se define la producción de energía sostenible como aquella que alcanza el 70% de ahorro de

GEI en comparación con un sistema que utilice combustibles fósiles. En base a eso, sólo la Planta 6 alcanzó resultados comparables a los propuestos por la Comisión Europea, ya que el uso extensivo de cultivos energéticos en la Planta 5 incrementó considerablemente su huella de carbono. Estos resultados también nos indican la necesidad de considerar otros impactos ambientales además de la huella de carbono para realizar un estudio más profundo. La Planta 5 presenta mejores resultados que la Planta 6 en relación a los impactos en acidificación y eutrofización, debido a la mayor cantidad de estiércol utilizado en la Planta 6 que deriva en mayor cantidad de digestato. Sin embargo, los residuos de alimentos se identificaron como una interesante fuente de energía ya que no se asignan cargas ambientales a su producción (al contrario que los cultivos energéticos) y tienen un alto potencial energético (especialmente en comparación con los residuos animales). También se ha evaluado la influencia de supuestos metodológicos, incluyendo i) la metodología seleccionada para la estimación de las emisiones del digestato y ii) la selección de la unidad funcional. La aplicación de diferentes metodologías de cálculo conllevó diferencias importantes en los resultados obtenidos; con todo, ninguna de ellas se identificó como la más adecuada ya que todas ellas consideran diferentes factores que son aceptados internacionalmente. Entre las unidades funcionales estudiadas, el uso de la electricidad producida apareció como una unidad funcional apropiada ya que aborda la eficiencia de los dos procesos más importantes: la conversión de la materia prima en biogás y del biogás en bioenergía. Finalmente, se compararon los potenciales de biogás de los sustratos utilizados con los datos publicados en la literatura.

✓ En el capítulo 5 se analiza la ecoeficiencia de 15 plantas de biogás reales aplicando el enfoque ACV + DEA. La evaluación incluyó algunas de las plantas de biogás ya evaluadas en los capítulos 3 y 4 (Plantas 2 - 6) y 10 plantas nuevas, ubicadas todas ellas en la misma área de estudio. La mayoría de las plantas de biogás adoptan la co-digestión de cultivos energéticos (principalmente maíz y triticale, pero también de centeno y sorgo) mezclados con estiércol (de cerdos, vacas y gallinas). Todos ellos generan electricidad y calor con el objetivo de suministrar electricidad a la red nacional. La matriz de DEA empleada se compuso por 3 entradas, incluyendo i) producción de materia prima, ii) consumo de electricidad y iii) transporte; y una sola salida, la electricidad producida. La matriz fue evaluada mediante el modelo SBM (abreviatura de su nombre en inglés: “slacks-based measure of efficiency model”), ya que permite calcular las puntuaciones de eficiencia independientemente de las unidades de medida utilizadas en las entradas y salidas. Además, las condiciones de medida incluyeron el enfoque de retorno variable a escala, porque los tamaños operativos de las plantas eran muy diferentes, y un enfoque orientado a las entradas, lo cual permite minimizar el uso de recursos (entradas) sin una reducción en la producción final de electricidad (salida). De las 15 plantas, 9 de ellas obtuvieron una ecoeficiencia del 100%; además, la eficiencia media de la muestra fue del 85%, mostrando una alta eficiencia general. Además, para mejorar la eficiencia del resto se calcularon reducciones teóricas de las entradas (producción de biomasa, transporte y consumo eléctrico) que permiten alcanzar altos valores de ecoeficiencia. La comparación del perfil ambiental de las plantas originales y virtuales (después de aplicar porcentajes teóricos de reducción de las entradas) mostró que los beneficios ambientales conseguidos variaban de acuerdo a la categoría de impacto estudiada, siendo las emisiones de GEI y eutrofización los principales impactos ambientales reducidos. En cuanto al impacto ambiental total, los objetivos de reducción basados en los principios de ecoeficiencia influyeron en las plantas que presentaron peor perfil ambiental global, evidenciando la efectividad de la metodología combinada ACV+DEA. Además, se realizó un análisis para identificar los parámetros que influyen en la eficiencia ambiental; sin embargo, ninguno de ellos destacó por su papel en la definición de la eficiencia del sistema, lo que significa que la ecoeficiencia de estos sistemas complejos está controlada por un conjunto de factores. Como punto final, se discutió en detalle el papel del digestato en los sistemas de biogás y específicamente en los estudios de ACV.

✓ En el capítulo 6 se incluye la evaluación de un sistema de tratamiento innovador para la gestión de estiércol en Chipre, en el marco del proyecto LiveWaste (LIFE12 ENV/CY/000544). Se analizó la planta piloto, diseñada para realizar la digestión anaerobia del estiércol y la recuperación y/o eliminación de nutrientes del efluente anaerobio. La planta piloto ofrece flexibilidad en cuanto al sistema de tratamiento del digestato, por lo que cuatro configuraciones diferentes fueron analizadas y comparadas para identificar la mejor desde un punto de vista ambiental. Los resultados mostraron que la mejor configuración es la que maximiza la recuperación de fósforo como estruvita, ya que es un fertilizante de alta calidad. Además, se evaluaron los posibles beneficios ambientales de la aplicación del sistema integrado a escala real comparando con los cuatro sistemas más típicos considerados en la gestión de residuos ganaderos en Chipre. El uso de lagunas anaerobias destacó como la peor práctica de gestión de estiércol debido a las emisiones directas. La planta convencional de biogás presentó menores emisiones de GEI que la planta propuesta en el proyecto; sin embargo, produjo mayores impactos de acidificación debido a las emisiones de amoníaco derivadas del digestato. Para concluir, se aplicó la metodología AHP para integrar indicadores económicos, sociales y ambientales en la selección del sistema de gestión de residuos más sostenible en Chipre. Los resultados mostraron que, aunque el coste de capital y operación del tratamiento propuesto en el proyecto LiveWaste son mayores, los beneficios ambientales, sociales y económicos asociados a este sistema lo convirtieron en el más sostenible.

Sección III: Biogás en plantas de tratamiento de aguas residuales

En esta sección se analizan las consecuencias ambientales de la digestión anaerobia como opción de valorización de residuos en el contexto del tratamiento de aguas residuales, incluyendo diferentes plantas de tratamiento de aguas residuales en términos de capacidad de tratamiento, expresadas como población equivalente (PE).

✓ El capítulo 7 analiza los potenciales beneficios ambientales de la co-digestión de lodos con residuos de comida en los digestores de una planta de tratamiento situada en Reino Unido para evidenciar los potenciales beneficios de la co-digestión y abordar el análisis de las barreras legales que obstaculizan el desarrollo de opciones más sostenibles de tratamiento integral de residuos. Considerando una depuradora de 150.000 PE de capacidad, se propusieron diferentes grados de integración de la gestión de los residuos de comida: i) Esquema 1 – mono-digestión de lodos como caso base; ii) Esquema 2 – co-digestión de residuos de comida considerando la capacidad adicional no aprovechada del digestor y iii) Esquema 3 – co-digestión de la cantidad total de residuos alimentarios producidos por la comunidad, construyendo un digestor adicional. De acuerdo con los resultados obtenidos, la co-digestión de residuos de alimentos y lodos de depuradora en una EDAR podría multiplicar la electricidad del biogás por un factor de 2.7, mejorando el rendimiento de la planta en términos de emisiones de GEI. El biogás adicional producido por la integración total de los residuos de comida supuso la producción de electricidad suficiente para cubrir las necesidades de toda la planta y el excedente puede ser inyectado en la red eléctrica nacional. Con todo, la mayor proporción de biomasa digerida implica la mayor producción de digestato (con un aumento del 15%) y la recirculación de una corriente rica en nitrógeno también afectó a la línea de aguas de la depuradora. Las principales barreras que obstaculizan la aplicación de los esquemas propuestos en Reino Unido se examinaron en detalle. En Reino Unido, el digestato y la bioenergía producida a partir de lodos o de residuos de comida se controlan bajo reglamentos diferentes. El sistema de permisos ambientales requiere que todas las plantas de biogás obtengan un permiso para operar y para aplicar el digestato como fertilizante. Cuando la materia prima utilizada en digestión anaerobia se considera residuo, el digestato derivado también se clasifica como residuo hasta que cumple con unos criterios establecidos. Sin embargo, en el Reino Unido, al igual que en otros países (por ejemplo, Suecia y Alemania), el digestato procedente de lodos de depuradora no puede someterse a este criterio y siempre se considera como residuo, requiriendo diferentes permisos para su uso. Esto crea barreras a los mercados para la adopción de los esquemas de gestión integrada dentro de los servicios de agua.

✓ El capítulo 8 se centra en la evaluación de un esquema de tratamiento innovador para la gestión combinada de aguas residuales y residuos de comida a nivel descentralizado en una pequeña comunidad de 2.000 PE. El esquema de tratamiento base propuesto incluye: i) un reactor anaerobio para tratar aguas residuales y producir biogás, ii) un proceso de fermentación para producir ácidos grasos volátiles a partir de los residuos de comida, iii) un reactor biológico secuencial para eliminar nutrientes del efluente y iv) un proceso de compostaje para tratar el exceso de lodo y convertirlo en compost para aplicarlo como enmienda agrícola. Además, se compararon 12 configuraciones alternativas combinando diferentes niveles de integración de trituradores de residuos alimentarios, procesos alternativos de eliminación de nitrógeno y eliminación o no de fósforo, con el objetivo de identificar el más adecuado desde el punto de vista técnico y ambiental. La evaluación ambiental del sistema base mostró que las principales emisiones directas y la producción de electricidad fueron los principales contribuyentes al perfil ambiental. El efluente anaerobio presenta una concentración importante de metano disuelto, el cual se libera en otras etapas del proceso de tratamiento, suponiendo una importante fuente de emisiones de GEI. Además, la descarga del efluente tratado contribuyó al impacto producido en eutrofización. Los resultados del estudio comparativo demostraron que el uso de trituradoras aumentó los impactos ambientales en comparación con la recogida por separado, mientras que la desnitrificación vía nitrato implicó mayores impactos en las categorías relacionadas con la energía en comparación con la eliminación de nitrógeno vía nitrito. Los esquemas que llevan a cabo la eliminación de nitrógeno y fósforo a través de nitrito produjeron mejores resultados ambientales en términos de eutrofización.

Sección IV: Conclusiones

La última sección resume los resultados de la tesis, incluyendo las principales conclusiones y recomendaciones.

✓ El capítulo 9 revisa el trabajo desarrollado en esta tesis, identifica las principales conclusiones y propone recomendaciones para mejorar la sostenibilidad de la producción de biogás. La tesis proporciona información sobre el desempeño ambiental a través de la aplicación de la metodología ACV en diferentes sistemas de biogás en la Unión Europea. Uno de los objetivos alcanzados en la tesis fue proporcionar herramientas que permitan la mejora ambiental de la producción convencional de biogás. Con este objetivo, se identificaron los procesos más contaminantes y se propusieron y analizaron medidas que reducen los impactos asociados. Además, esta tesis también tenía como objetivo promover la implementación de sistemas innovadores que integran la producción de biogás con tratamiento del digestato, disminuyendo los impactos de eutrofización y acidificación. En este sentido, el ACV demostró ser una metodología valiosa que proporciona información y evidencia de la mejor configuración de tratamiento desde una perspectiva ambiental previa a su implementación a escala completa. Además, a lo largo de toda la tesis, se identificaron las barreras metodológicas en la aplicación de ACV a los sistemas de biogás para proporcionar futuras oportunidades de mejora.

Resumo

Un dos principais retos actuais está relacionado co desenvolvemento de sistemas enerxéticos que cubran as necesidades da sociedade, preservando ao mesmo tempo o medio ambiente. Por iso, a Unión Europea fixou diferentes obxectivos co fin de fomentar o desenvolvemento de enerxías renovables, incluíndo por exemplo que o 20% do consumo bruto de enerxía proveña de fontes renovables en 2020. Outro dos retos que afrontamos é implantar medidas eficaces que consigan contrarrestar os precursores do cambio climático, así como mellorar a resposta ás súas consecuencias. En relación a isto, outro obxectivo europeo é reducir nun 20% as emisións de gases de efecto invernadoiro (GEI) para 2020 en comparación con 1990. A Convención Marco das Nacións Unidas sobre o Cambio Climático creou a Conferencia das Partes en 1995, a cal en 2015 adoptou o Acordo de París, que establece o marco para combater o cambio climático a partir do 2020. O principal obxectivo acordado é evitar que o aumento da temperatura media global supere os 2 °C en comparación co nivel preindustrial, o que debería lograrse mellorando tanto a mitigación como a adaptación ao cambio climático. Con isto en mente, a diversificación de tecnoloxías e recursos para a produción de enerxía renovable e a descarbonización do sistema enerxético debe ser un obxectivo primordial. O biogás esta a ser promovido como unha forma sostible de enerxía xa que comparte os principios da economía circular ao ser capaz de converter refugallos en produtos de valor engadido como enerxía, auga ou nutrientes.

Son necesarias metodoloxías de avaliación que permitan evidenciar os beneficios e inconvenientes da produción deste tipo de enerxía renovable, en canto á súa sustentabilidade económica, social e ambiental. Entre elas, o Análise de Ciclo de Vida (ACV) considérase unha metodoloxía apropiada para a cuantificación das consecuencias ambientais da produción de biogás. Xa hai numerosos estudos de ACV publicados sobre produción e uso de biogás, incluíndo diferentes substratos

e posibles usos do biogás, prestando especial atención a emisións de GEI e ao esgotamento dos combustibles fósiles. Recentemente, propúxose a implementación combinada do ACV co Análise por Envoltura de Datos (DEA, acrónimo do seu nome en inglés "Data Envelopment Analysis"), coa fin de analizar a eco-eficiencia de múltiples entidades similares. Esta nova ferramenta evita o uso de datos de inventario promedio e enriquece a interpretación dos resultados mediante a verificación da ecoeficiencia. Este enfoque aplicouse empregando indicadores técnicos e ambientais a diferentes procesos de produción como o viño ou a pesca e incluso a sistemas de tratamento como depuradoras. O Proceso Analítico Xerárquico (AHP, acrónimo do seu nome en inglés "Analytical Hierarchy Process") é outra metodoloxía dispoñible que pode ser utilizada para a descomposición xerárquica dun problema complexo, axudando a identificar a mellor opción considerando varios conxuntos de criterios de diferente natureza. Aplicouse amplamente para seleccionar as alternativas máis sustentables en sistemas de enerxía e de residuos, incluíndo por exemplo a selección da mellor alternativa para a recuperación de enerxía a partir de residuos sólidos municipais.

O obxectivo desta tese doutoral é examinar a sustentabilidade ambiental de diferentes sistemas de biogás na Unión Europea e avaliar a súa idoneidade de acordo cos obxectivos europeos en canto a enerxía. Aplicando a metodoloxía ACV e outras ferramentas de avaliación, como o DEA e o AHP, investigáronse diferentes sistemas de produción de biogás na Unión Europea, incluíndo biogás producido a partir de biomasa agrícola ou en plantas de tratamento de augas residuais, abarcando tanto tecnoloxías convencionais ben establecidas como propostas máis innovadoras. De acordo cos obxectivos propostos, a tese dividiuse en catro seccións, incluíndo i) Introdución, ii) Biogás agrícola, iii) Biogás en estacións de tratamento de augas residuais e iv) Conclusións.

Sección I: Introdución

Esta sección pretende contextualizar a tese proporcionando información xeral sobre a produción de biogás e as ferramentas dispoñibles para a avaliación da súa sustentabilidade ambiental.

✓ O capítulo 1 examina a relación entre o consumo de combustibles fósiles, a produción de enerxía e o cambio climático. Ademais, discútese o papel das enerxías renovables nos obxectivos europeos de loita contra o cambio climático e de seguridade enerxética. O biogás analízase como unha fonte de enerxía que pode facer unha gran contribución á provisión de enerxía e ás reducións de emisións GEI na Unión Europea debido a ampla gama de biomasa adecuada para a súa produción e ás diferentes formas dispoñibles para convertelo en bioenerxía.

✓ O capítulo 2 explora o concepto de desenvolvemento sustentable e as metodoloxías dispoñibles para a avaliación dos sistemas de biogás en termos de sustentabilidade, prestando especial atención ás metodoloxías aplicadas nesta tese, é dicir, ACV, DEA e AHP. Os obxectivos e estrutura da tese tamén se presentan ao final do capítulo 2.

Sección II: Biogás agrícola

A sustentabilidade ambiental da produción de biogás a partir de biomasa agrícola, principalmente esterco e cultivos enerxéticos, examínase en detalle na Sección II.

✓ No capítulo 3 examínase e compárase dende o punto de vista do ciclo de vida a sustentabilidade ambiental de catro plantas de biogás reais operando en Italia co obxectivo de identificar as etapas máis contaminantes para propoñer e analizar opcións de mellora. A Planta 1 utiliza xurro de porco como o único substrato, mentres que as plantas 2 e 3 codixiren residuos animais (de galiña e de porco) e cultivos enerxéticos (millo, e no caso da Planta 2, triticale) e, por último, a Planta 4 só dixire millo. Todas elas usan o biogás nun motor de coxeración e a electricidade producida é subministrada á rede nacional, mentres que a calor só se aproveita para quentar os reactores. Por iso a unidade funcional seleccionada foi a produción de 1 MWh de electricidade. O dixestato, que se produce durante a dixestión anaerobia, emprégase como fertilizante orgánico no cultivo dos cereais que usa cada planta. Empregouse unha perspectiva de produto evitado para os casos nos que hai un exceso de dixestato, considerando que evita o uso de fertilizantes minerais. Os resultados demostraron que os sistemas de biogás poden supor reducións de emisións GEI comparado co perfil enerxético italiano cando se alcanza un balance no uso de residuos e cultivos enerxéticos. Non obstante, os impactos de eutrofización e acidificación foron maiores que os producidos no perfil enerxético do país en todos os casos, principalmente debido ao cultivo dos cereais e ao manexo do dixestato. Propuxéronse e analizáronse estratexias para mitigar os impactos ambientais. Os resultados demostraron que o aproveitamento da calor producida na coxeración nun invernadoiro próximo contribuíu a reducir os impactos ambientais debido aos créditos derivados da produción evitada de calor a partir de diésel. A inxección do dixestato no terreo en lugar do esparexemento superficial influíu nas emisións derivadas, aumentando o óxido nitroso, pero diminuíndo o amoníaco. Os factores que afectan ao rendemento das plantas de biogás tamén foron examinados en detalle para comprender os resultados ambientais obtidos. O xurro de porco identificouse como unha fonte pobre de enerxía, implicando unha maior produción de dixestato por unidade de biogás producido. Relacionado con isto, o millo presentou o maior potencial de biogás comparado non só co xurro de porco, senón tamén co triticale. Sen embargo, tamén se deben considerar outros factores que varían a eficiencia do proceso de dixestión anaerobia e a conversión do biogás en bioenerxía. Para concluír, realizouse unha comparación entre os resultados obtidos no capítulo con outros estudos coa fin de identificar os resultados comúns e os factores metodolóxicos que obstaculizan a comparación.

✓ O capítulo 4 profunda nas implicacións ambientais da selección de materias primas na produción de biogás, non só en termos de potencial enerxético, senón tamén en relación ca cantidade e calidade do dixestato producido. Para isto analizáronse e modeláronse dúas plantas de biogás. A Planta 5 emprega esterco de galiña e porco como substratos, ademais de cultivos enerxéticos como o millo e o triticale. Sen embargo, a Planta 6, ademais de xurro de porco e millo, tamén utiliza os residuos de comida provenientes de supermercados e fogares. En termos de emisións de GEI, a Plantas 6 obtivo mellores resultados que a Planta 5; non obstante, ambas conseguiron menor pegada de carbono que a do perfil eléctrico. En 2014, a Comisión Europea estableceu un valor de referencia, como requisito para considerar a produción de enerxía da biomasa como sustentable, definido como 70% de aforro de GEI en comparación cun sistema de referencia de combustibles fósiles. En canto a isto, só a Planta 6 alcanzou resultados comparables ós propostos pola Comisión Europea, xa que o uso extensivo de cultivos enerxéticos na Planta 5 incrementou considerablemente as súas emisións de GEI. Outros impactos ambientais deben ser considerados ademais da pegada de carbono para facer un estudo completo. A Planta 5 ten mellores resultados que a Planta 6 con respecto aos impactos en acidificación e eutrofización, debido á maior cantidade de esterco empregado que deriva nunha maior cantidade de dixestato. Porén, os residuos de alimentos identificáronse como unha interesante fonte de enerxía xa que non se asignan cargas ambientais á súa produción (ao contrario que os cultivos enerxéticos) e teñen un alto potencial enerxético (especialmente en comparación cos residuos animais). Tamén se avaliou a influencia de supostos metodolóxicos, incluíndo i) a metodoloxía seleccionada para a estimación das emisións do dixestato e ii) a selección da unidade funcional. A aplicación de diferentes metodoloxías de cálculo trouxo consigo diferencias importantes nos resultados obtidos; non obstante, ningunha delas se identificou como a máis adecuada xa que todas elas consideran diferentes factores e son aceptadas internacionalmente. Entre as unidades funcionais estudiadas, o uso da electricidade producida apareceu como unha unidade funcional apropiada xa que toma en consideración as eficiencias dos dous procesos máis importantes: a conversión da materia prima en biogás e a do biogás en bioenerxía. Finalmente, comparáronse os potenciais de biogás dos substratos utilizados cos datos publicados na literatura.

✓ No capítulo 5 avalíase a ecoeficiencia de 15 plantas de biogás reais aplicando o enfoque ACV+DEA. A avaliación abarcou algunhas das plantas de biogás xa avaliadas nos capítulos 3 e 4 (Plantas 2 - 6) e 10 plantas novas, situadas todas elas na mesma área. A maioría das plantas de biogás adoptan a co-dixestión de cultivos enerxéticos (principalmente millo e triticale, pero tamén de centeo e sorgo) mesturados con esterco (de porcos, vacas e galiñas). Todos eles xeran electricidade e calor co obxectivo de subministrar electricidade á rede nacional. A matriz de DEA empregada estaba composta por 3 entradas, incluíndo i) produción de materia prima, ii) consumo de electricidade e iii) transporte; e unha sóa saída, a electricidade producida. A matriz foi avaliada mediante o modelo SBM (abreviatura de seu nome en inglés: "slacks-based measure of efficiency model"), xa que permite calcular as puntuacións de eficiencia independentemente das unidades de medida utilizadas nas entradas e saídas. Ademais, as condicións de medida incluíron retorno variable a escala, porque os tamaños operativos das plantas eran moi diferentes; e un enfoque orientado ás entradas, o que permite minimizar o uso de recursos (entradas) sen unha redución na produción final de electricidade (saída). Das 15 plantas, 9 delas obtiveron unha ecoeficiencia do 100%; ademais, a eficiencia media da mostra foi do 85%, mostrando unha alta eficiencia xeral. Ademais, para mellorar a eficiencia do resto calculáronse reducións teóricas das entradas (produción de biomasa, transporte e consumo eléctrico) que permiten alcanzar a ecoeficiencia. A comparación do perfil ambiental das plantas orixinais e virtuais (despois de aplicar porcentaxes teóricos de redución das entradas) mostrou que os beneficios ambientais conseguidos cambiaban segundo a categoría de impacto estudiada, sendo as emisións de GEI e eutrofización os principais impactos ambientais reducidos. En canto ao impacto ambiental total, os obxectivos de redución baseados nos principios de ecoeficiencia influíron nas plantas que presentaron o peor perfil ambiental xeral, expresado nunha única unidade de medida, evidenciando a efectividade da metodoloxía combinada ACV+DEA. Así mesmo, realizouse unha análise para identificar os parámetros que inflúen na eficiencia ambiental; con todo, ningún deles destacou polo seu papel na definición da eficiencia do sistema, o que significa que a ecoeficiencia destes sistemas complexos está controlada por un compendio de factores. Como punto final, discutiuse en detalle o papel do dixestato nos sistemas de biogás e especificamente nos estudios de ACV.

✓ No capítulo 6 incluíuse a avaliación dun sistema de tratamento innovador para a xestión de esterco en Chipre, no marco do proxecto LiveWaste (LIFE12 ENV/CY/000544). Analizouse a planta piloto, deseñada para realizar a dixestión anaerobia do esterco e a recuperación e/ou eliminación de nutrientes do efluente anaerobio. A planta piloto ofrece flexibilidade en canto ao sistema de tratamento do dixestato, polo que catro configuracións diferentes foron analizadas e comparadas para identificar a mellor dende un punto de vista ambiental. Os resultados mostraron que a mellor configuración é a que maximiza a recuperación de fósforo como estruvita, xa que é un fertilizante de alta calidade. Ademais, avaliáronse os posibles beneficios ambientais da aplicación do sistema integrado a escala real comparando cos catro sistemas máis típicos aplicados aos refugallos animais en Chipre. O uso de lagoas anaerobias destacaron como a peor práctica de manexo de esterco debido ás emisións directas. A planta convencional de biogás presentou menores emisións de GEI que a planta proposta no proxecto; non obstante, produciu maiores impactos de acidificación debido ás emisións de amoníaco derivadas do dixestato. Para concluír, aplicouse a metodoloxía AHP para integrar indicadores económicos, sociais e ambientais na selección do sistema de xestión de residuos máis sustentable en Chipre. Os resultados mostraron que, aínda que o coste de capital e operación do tratamento proposto no LiveWaste son maiores, os beneficios ambientais, sociais e económicos desta opción convertérono no máis sustentable.

Sección III: Biogás en depuradoras

Nesta sección analízanse as consecuencias ambientais da dixestión anaerobia como opción de valorización de residuos no contexto do tratamento de augas residuais, incluíndo as diferentes estacións depuradoras en termos de capacidade de tratamento, expresadas como poboación equivalente (PE).

✓ No capítulo 7 analízanse os potenciais beneficios ambientais da co-dixestión de lodos con residuos de comida nos dixestores dunha depuradora situada no Reino Unido para evidenciar os potenciais beneficios da co-dixestión para abordar as barreras legais que obstaculizan o desenvolvemento de opcións máis sustentables de tratamento integral de residuos nese país. Considerando unha depuradora de 150.000 PE de capacidade, propuxéronse diferentes grados de integración da xestión dos residuos de comida: i) Esquema 1 – mono-dixestión de lodos como caso base; ii) Esquema 2 – co-dixestión de residuos de comida considerando a capacidade adicional non aproveitada do dixestor e iii) Esquema 3 – co-dixestión da cantidade total de refugallos alimentarios producidos pola comunidade, construíndo un dixestor adicional. De acordo cos resultados obtidos, a co-dixestión de residuos de alimentos e lodos nunha planta de tratamento de augas residuais (Esquema 3) podería multiplicar a electricidade producida a partir do biogás por un factor de 2.7. O biogás adicional producido pola integración total dos residuos de comida supuxo a produción de electricidade suficiente para cubrir as necesidades de toda a planta e o excedente pode ser inxectado na rede eléctrica nacional. Sen embargo, a maior proporción de biomasa dixerida implica a maior produción de dixestato (aumentado nun 15%) e a recirculación dunha corrente rica en nitróxeno tamén afectou á liña de augas da planta. As principais barreiras que obstaculizan a aplicación dos esquemas propostos no Reino Unido examináronse en detalle. No Reino Unido, o dixestato e a bioenerxía producidos a partir de lodos ou de residuos de comida contrólanse baixo regulamentos diferentes. O sistema de permisos ambientais require que todas as plantas de biogás obteñan un permiso para operar e para aplicar o dixestato como fertilizante. Cando a materia prima utilizada en dixestión anaerobia se considera residuo, o dixestato derivado tamén se clasifica como un residuo ata que cumpre cuns criterios establecidos. Sen embargo, no Reino Unido, ao igual que en outros países (por exemplo, Suecia ou Alemaña), o dixestato proveniente de lodos de depuradora non pode someterse a este criterio e sempre e considera como residuo, requirindo diferentes permisos para o seu uso. Isto crea barreiras nos mercados para a adopción dos esquemas de xestión integrada dentro dos servicios da auga.

✓ O capítulo 8 céntrase na avaliación dun esquema de tratamento innovador para a xestión combinada de augas residuais e residuos de comida a nivel descentralizado nunha pequena comunidade de 2.000 PE. O esquema de tratamento base proposto inclúe: i) un reactor anaerobio para tratar augas residuais e producir biogás, ii) un proceso de fermentación para producir ácidos graxos volátiles a partir dos residuos de comida, iii) un reactor biolóxico secuencial para eliminar nutrientes do efluente e iv) un proceso de compostaxe para tratar o exceso de lodo e convertelo en compost para aplicalo como emenda agrícola. Ademais, comparáronse 12 configuracións alternativas combinando diferentes niveis de integración de trituradores de residuos alimentarios, procesos alternativos de eliminación de nitróxeno e eliminación ou non de fósforo, co obxectivo de identificar o máis adecuado dende o punto de vista técnico e ambiental. A avaliación ambiental do sistema base mostrou que as principais emisións directas e a produción de electricidade foron os principais contribuíntes ao perfil ambiental. O efluente anaerobio presenta unha concentración importante de metano disolto, o cal se libera noutras etapas do proceso de tratamento, supoñendo unha importante fonte de emisións de GEI. Ademais, a descarga do efluente tratado contribuíu ao impacto producido en eutrofización. Os resultados do estudio comparativo demostraron que o uso de trituradoras aumentou os impactos ambientais en comparación ca recollida por separado, mentres que a desnitrificación vía nitrato implicou maiores impactos nas categorías relacionadas ca enerxía en comparación ca eliminación de nitróxeno vía nitrito. Os esquemas que levan a cabo a eliminación de nitróxeno e de fósforo a través de nitrito produciron mellores resultados ambientais en canto a eutrofización.

Sección IV: Conclusións

A última sección resume os resultados da tese, incluíndo as principais conclusións e recomendacións.

✓ O capítulo 9 revisa o traballo desenvolvido nesta tese, identifica as principais conclusións e propón recomendacións para mellorar a sustentabilidade da produción de biogás. A tese proporciona información sobre o desempeño ambiental a través da aplicación extensiva da metodoloxía ACV en diferentes sistemas de biogás na Unión Europea. Un dos obxectivos alcanzados na tese foi proporcionar ferramentas que permitan a mellora ambiental da produción convencional de biogás. Con este obxectivo, identificáronse os procesos máis contaminantes e propuxéronse e analizáronse medidas que reducen os impactos derivados. Ademais, esta tese tamén tiña como obxectivo promover a implementación de sistemas innovadores que integran a produción de biogás co tratamento do dixestato, diminuíndo os impactos de eutrofización e acidificación. Neste sentido, o ACV demostrou ser un método valioso para proporcionar evidencia da mellor configuración de tratamento dende unha perspectiva ambiental previa á súa implementación a escala completa. Ademais, ao longo de toda a tese, identificáronse as barreiras metodolóxicas na aplicación de ACV ós sistemas de biogás para proporcionar futuras oportunidades de progreso.

Section I:

Introduction

Chapter 1: The role of biogas in sustainable development

Summary

The society demands the provision of energy sources that cover the needs of the economies and preserves the environment. The diversification of technologies and resources for the production of renewable energy creates many opportunities to improve the environmental profile of energy generation, but the increased complexity also leads to increased challenges.

Renewable energy can be produced from biomass, hydropower, geothermal, solar wind and marine sources. Biogas is a versatile renewable source of energy produced from the anaerobic digestion of different types of biomass. Due to the wide range of biomass suitable for anaerobic digestion, biogas can make a large contribution to the energy supply in the European Union. The objective of Chapter 1 is to provide background about biogas development in the European Union and to provide information about the diversity of biogas systems. It can be produced from nearly all biomass sources, including dedicated energy crops, agricultural and livestock waste, industrial and domestic organic waste and sewage. Moreover, biogas can be used for heat production, co-generation of heat and electricity, or converted into biomethane to be used as vehicle fuel or distributed in the natural gas network. The produced digestate can be used as an organic fertiliser due to its content in nutrients. However, depending on the particular case, digestate must be processed prior spreading on agricultural land with the purpose of removal of excess nutrients or their recovery to improve the fertiliser potential of the digestate.

Outline of Chapter 1

1.1. Energy and climate change

The access to affordable and reliable energy has played a major role in human and economic development and, in general, in society's prosperity and well-being. Since the Industrial Revolution took place during the 18th century, unprecedented quantities of fossil fuels (i.e. oil, coal, and natural gas) have been used (Chu and Majumdar, 2012). As shown in Figure 1.1, ever since, modern society uses more and more energy produced from fossil resources for industry, services, households and transport; particularly intensive in the case of crude oil, which has led to a correlation between economic growth and its market price (IEA, 2004). The increase in population since then has also played an important role in the consumption of fossil fuels. At the beginning of the industrial revolution, the world population was 700 million, while today it is over 7 billion; moreover, it is projected to grow up to 9 billion by 2050 (Lee, 2011; U.S. Census, 2016). The International Energy Agency (IEA) has estimated that the world's energy demand will increase 48% by 2040, which has been related to the increase of the global population as well as the improvements in the life-style and industrialisation of developing nations (Panwar et al., 2011).

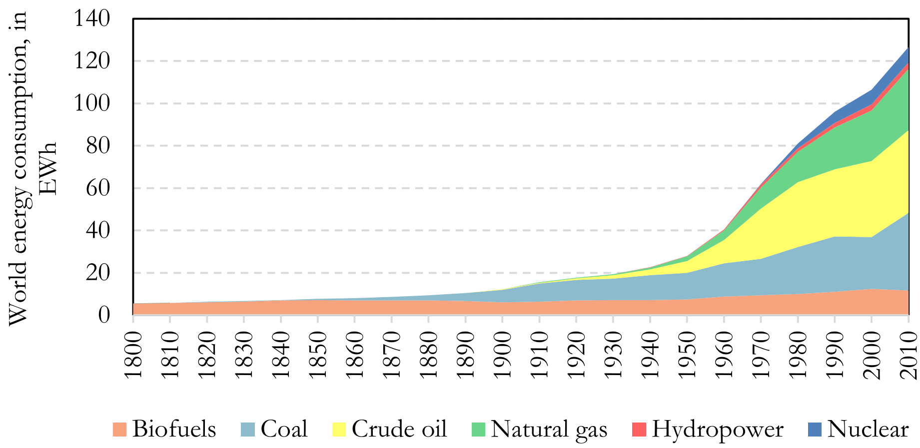

Figure 1.1. World energy consumption in exawatt-hour (EWh) by source, adapted from Roser (2016).

This tendency has led to unprecedented levels of carbon dioxide in the atmosphere. According to Hodkinson et al. (2001), three quarters of the carbon dioxide present in the atmosphere are the result of the combustion of fossil fuels. In more detail, before the Industrial Revolution, the concentration of atmospheric carbon dioxide was maintained around 280 ppm for several thousand years; thereafter, it rose continuously, reaching 367 ppm in 1999 (Hodkinson et al., 2001). Indeed, the present atmospheric level has not been exceeded for the last 420,000 years and the increasing rate over the past century is unprecedented, at least for the past 20,000 years. As a result, two key climate change indicators (i.e. global surface temperature and Arctic sea ice area) have broken numerous records converting 2016 in the warmest year of the modern temperature record, which dates to 1880 (NASA, 2016). In more detail, emissions of anthropogenic carbon dioxide have increased by more than 50% over the past 25 years (van der Hoeven, 2015). While emissions increased by 1.2% per year in the last decade of the 20th century, the average rate between 2000 and 2014 accelerated to 2.3%, particularly driven by a rapid rise in power generation in countries outside the OECD[1] (van der Hoeven, 2015). Moreover, emissions from energy generation in emerging and developing countries have doubled since the start of the 21st century, being China the highest contributor. On the other hand, emissions from the industrial sector in OECD countries have been reduced by a quarter; nonetheless, these countries still lead emissions from transport and building sector (van der Hoeven, 2015). It is important to highlight that this fact was the first sign of a decoupling between energy-related emissions and economic growth (van der Hoeven, 2015). The carbon emissions related to energy production for the year 2014 in selected areas of the world are presented in Figure 1.2. According to Chu and Majumdar (2012), the energy sector needs a change to provide affordable, accessible and sustainable energy. Energy efficiency and conservation are essential together with decarbonising energy sources. Moreover, with the aim of reducing carbon emissions on the timescale needed to mitigate climate, the development of cost-effective alternative sources of energy is necessary.

[1] Organisation for Economic Cooperation and Development (OECD) Organisation that includes 35 countries: Australia, Austria, Belgium, Canada, Chile, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Iceland, Ireland, Israel, Italy, Japan, Korea, Latvia, Luxembourg, Mexico, Netherlands, New Zealand, Norway, Poland, Portugal, Slovak Republic, Slovenia, Spain, Sweden, Switzerland, Turkey, United Kingdom, United States.

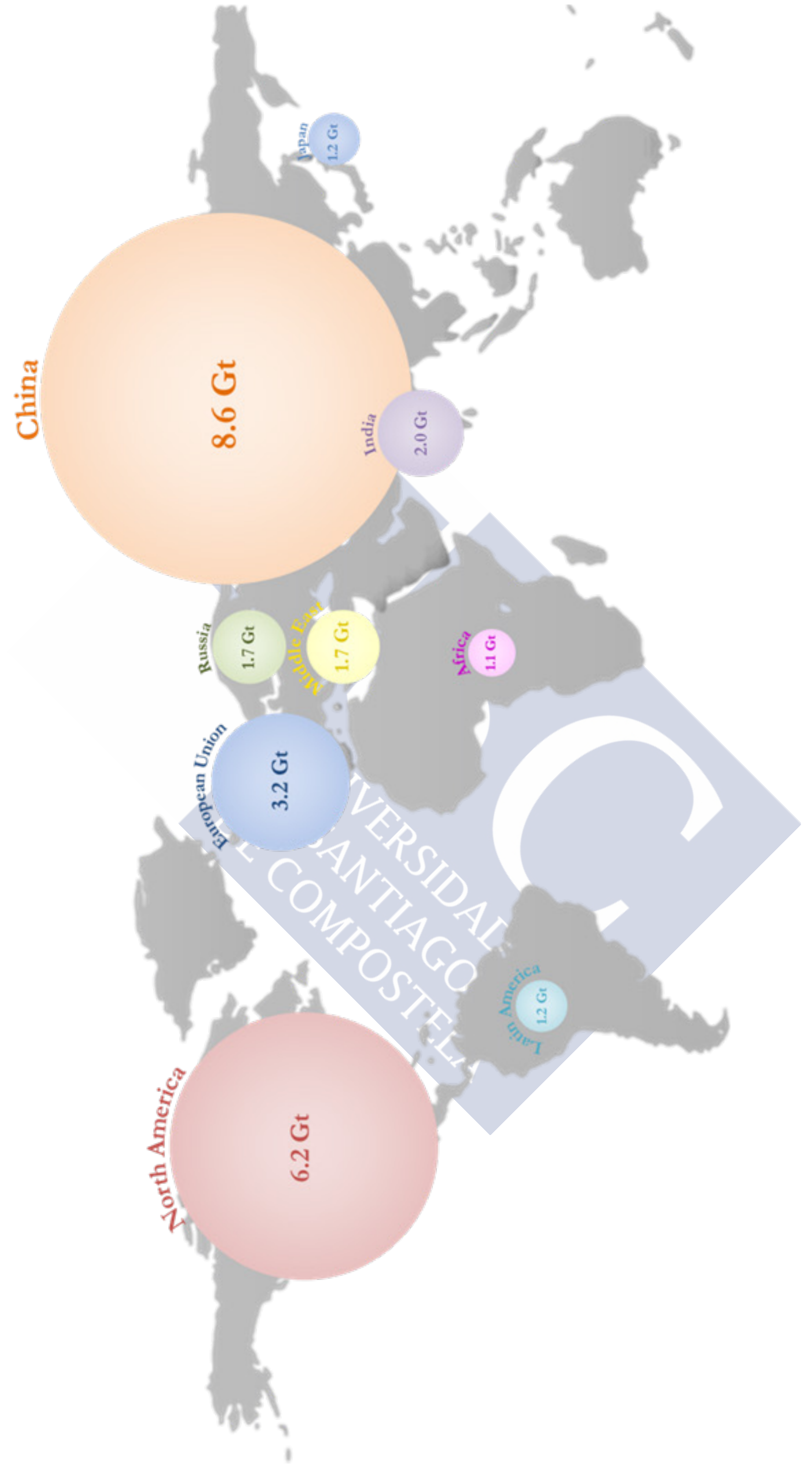

Figure 1.2. Carbon dioxide emissions from fossil-fuel combustion in 2014 in gigatonnes (Gt), adapted from van der Hoeven (2015)

Concerns over global climate change increase on a daily basis as extreme weather events multiply over the world and scientific evidence of anthropogenic changes in climate accumulates. More into detail, climate change increase frequency and intensity of heat waves, intensify floods and droughts and alter the distribution of vector-borne diseases, which influence risk of disasters and malnutrition (Panwar et al., 2011). Humankind has the challenge to take effective measures to both counteract the drivers of on-going climate change and improving public response to its consequences (Adamo, 2015). With this regard, the Conference of the Parties (COP) has been created, which is the most relevant decision-making body of the United Nations Framework Convention on Climate Change (UNFCCC). Until now, there have been 21 Conferences between 1995 and 2015. The last one, COP21 was held in Paris in 2015, and ended with the adoption of the Paris Agreement, establishing the global framework for combating climate change from 2020 onwards. It is a historic agreement to tackle climate change by promoting a transition towards a low emission and resilient economy. The primary objective adopted in the COP21 was to avoid that the increase of the average global temperature exceeds 2°C in comparison with the pre-industrial level. To do so, COP21 considers that mitigation and adaptation have equivalent importance. Mitigation is the number of actions needed for stabilising concentrations of greenhouse gases (GHGs) in the atmosphere, either by reducing sources of emissions and/or enhancing sinks and reservoirs. Adaptation is the adjustments in ecological, social or economic systems in response to the effects of climate change. An analysis performed by IEA has shown that limiting temperature rise to 2°C will require an important decrease of near-term global energy-related emissions and a marked decline thereafter, which is in line with the targets of the Paris Agreement. For this reason, the IEA has proposed a Bridge Scenario, encouraging the use of existing technologies that could deliver that peak in global energy-related emissions by 2020 at no cost to global economic activity (IEA, 2016). In more detail, the five energy policy measures set as a bridge to further action includes: i) increasing energy efficiency, ii) limiting least-efficient coal power, iii) raising renewable investments, iv) reforming fossil-fuel subsidies and v) reducing upstream methane. Adoption of these measures can lock-in the recently observed decoupling of emissions increase and economic growth, an important first step to move the energy world towards the ambitious 2°C climate target. Therefore, actions regarding energy efficiency and renewable energy are vital to deliver the Nationally Determined Contributions (NDCs) defined in the Paris agreement (IEA, 2016). However, an analysis of the IEA proved that the aforementioned NDCs are consistent with an average global temperature increase of around 2.7°C; therefore, for achieving the objective of 2°C it is required to surpass the NDCs (IEA, 2016). Therefore, the Paris Agreement is providing a significant push for further investment and deployment of renewables. However, investments remain below levels consistent with long-term climate goals. Enhanced policy measures could accelerate renewables deployment and maintain consistency with the early emissions peak and subsequent downward trajectory required to stay below 2°C.

1.2. Renewable energy in Europe

As shown, one of the key pillars of the energy strategy in the European Union to mitigate climate change is based on the promotion of low-carbon renewable energy sources. Renewable energy is produced from biomass, hydropower, geothermal, solar wind and marine sources (Panwar et al., 2011). The foremost benefits of renewable energy production are summarised in Figure 1.3.

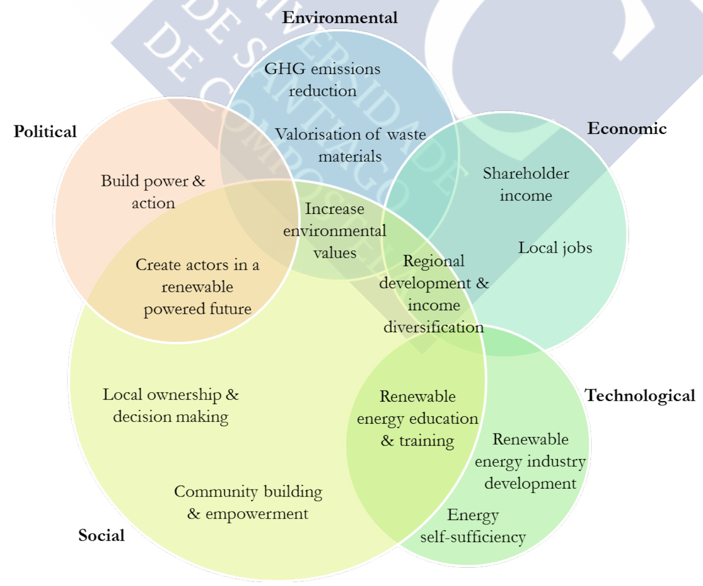

Figure 1.3. Benefits of renewable energy production, adapted from Ellabban et al. (2014)

The development of renewables can contribute to solve the presently most relevant objectives such as improving energy supply reliability, solving problems of local energy supply, creating job opportunities in rural areas and ensuring sustainable development of the remote regions (Panwar et al., 2011). Moreover, renewable energy production in decentralised way is one of the options to meet the rural and small scale energy needs in a reliable, affordable and environmentally sustainable way. In 1997 the European Commission published "Energy for the future: Renewable sources of energy", a White Paper for a Community Strategy and Action Plan setting the basis for the European Union policy on renewable energy (European Commission, 1997), which proposed increasing the share of renewable energy consumption in the European Union to 12% in 2010. At the beginning of the 21st century, the European Union promoted the production of electricity from renewable energies with Directive 2001/77/EC (European Parliament, 2001). In 2007, the European Commission proposed an integrated energy and climate change package, which included the commitment to achieve at least a 20% reduction of GHG emissions by 2020 compared to 1990 levels. Thereafter, the EU Directive 2009/28/EC set the target of a 20% share of energy from renewable sources in the gross final consumption of energy and 10% renewable energy in transport by 2020 (European Parliament, 2009). Each Member State has its own target for the share of energy from renewable sources; accordingly, each one have implemented various policies to increase the production of renewable electricity. The Member States had to prepare National Renewable Energy Action Plans with detailed roadmaps and measures taken to reach the 2020 renewable energy targets (Scarlat et al., 2015).

There are two major policy support systems commonly adopted by governments in the European Union: i) price-based feed-in systems that include feed-in tariff (FIT) and feed-in premium (FIP), and ii) quota systems e.g. green certificates or renewable portfolio standards (RPS) that are quantity-based systems (Schallenberg-Rodriguez, 2017). The origins of these support systems are quite diverse and are motivated by different political and economic needs. On one side, FIT, which is the most common system in the European Union, offer long-term (about 15–20 years) stable and guaranteed purchase agreements with green power producers to sell their electricity into the grid (Alizamir et al., 2016; Nicolini et al., 2017). The tariff rate is usually differentiated by the source and the size of the project. The advantage of this system is its effectiveness in promoting technology development and in achieving higher production of electricity from renewable sources. On the other side, FIP offers a premium (an additional payment) on top of the electricity market price (Schallenberg-Rodriguez, 2017), which implies that the money received per kilowatt-hour by the producer is less predictable in the scheme as it depends on the electricity price. Finally, quota obligations are used to impose a minimum production or consumption of electricity from renewable sources (Nicolini et al., 2017). These systems include two different concepts: the quota that is the percentage of renewable power to be supplied or consumed and the physical certificate generated that guarantees that the electricity comes from renewable sources. Therefore, the generators are obliged to provide a required number of certificates to demonstrate the compliance with a certain percentage of renewable electricity. They may obtain these documents from their own electricity generation, by purchasing renewable electricity or certificates without purchasing the actual power from a generator (Nicolini et al., 2017).

• **Germany** – This country is the leader in the European Union in the successful development of renewable energy as a result of combined efforts between government agencies and private-sector (Izadian et al., 2013). Germany was the first country in Europe that attempted to develop a FIT system in 1979. Electricity distribution companies were obliged to purchase renewable electricity based on avoided costs. However, this system had no significant impact due to the low avoided costs estimated. In 1990, the country approved the first feed-in tariff law (named as *Stromeinspeisegesetz*, StrEG). The law required electric utilities to connect renewable electricity generators to the grid and buy electricity at rates of 65–90% of the average tariff for final customers. In 2000, this law was replaced a new one (named as *Erneuerbare Energien Gesetz*, EEG). The new law set FIT values for 20 years and differentiate the tariff per technology. The most recent renewable energy law (EEG 2014) aims at reducing the financial cost of energy transition by slowing the growth of the most expensive electricity generating sectors. The country's goal is that 80% of its energy will be from renewable sources by 2050 (Izadian et al., 2013).

• **United Kingdom** – In 2000, the United Kingdom government announced that 10% of the energy produced in the country would come from renewable energy by 2010. As a consequence, the country put forth several policies to achieve this goal, including the first European quota system in Europe, which has been implemented in 2002. The Renewable Obligation (RO) is the primary mechanism to support the deployment of renewable electricity generation and it allowed ambitious growth targets for renewable electricity production (Cherrington et al., 2013). The RO imposed an obligation on all electricity suppliers to supply their customers with specified amounts of renewable energy. Suppliers could comply with these obligations by either presenting Renewable Obligation Certificates (ROCs) or by making a buy-out payment (Izadian et al., 2013). The ROCs were allocated by technology banding: emerging technologies were awarded more certificates than mature technologies. As a result, renewable electricity generation increased from 1.8% in 2002 to 6.8% in 2010 (Schallenberg-Rodriguez, 2017). Since 2010, FIT works alongside the RO to promote the deployment of small-scale renewable and low-carbon electricity generation technologies (Cherrington et al., 2013).

• **Italy** – The first attempt to pluralise electricity production by promoting renewable energies was in 1992 through the Law 6/92, a FIT scheme (Benedetti, 2014). In 2002 it was replaced by a green certificate scheme that lasted until 2012. In the beginning, green certificates were given to renewable energy producers for a period of 12 years, regardless the type of energy produced (Mela and Canali, 2014). Beginning in 2008, the duration of green certificates was extended to 15 years, and the number of them given to producers was linked to the type of renewable source. In 2005, the Italian government also introduced the first FIT incentives specifically for electricity generated by photovoltaic solar systems (named as *Conto Energia*). These payments were designed to last 20 years and to encourage both small and large producers to invest in the installation of photovoltaic systems. Between 2005 and 2013 five *Conto Energia* schemes were introduced by ministerial decrees. From 2008, small producing facilities could opt for an alternative incentive system in which green certificates were substituted by a FIT scheme (Benedetti, 2014). Producers could benefit for 15 years, after which they would have to sell energy at market prices (Mela and Canali, 2014). The FIT scheme was actualised in 2013, when the Decree 6 July 2012 shifted the Italian renewable energy policy promoting the development of smaller plants.

• **Spain** – In 1994 the Spanish sector attempted to implement a FIT that would force utilities to purchase renewable energy from wind, solar and hydroelectric power at rates above the market value (IER, 2014). Moreover, Spain increased government-funded renewable development during the first decade of 21st century, adding bonus to the FIT system, adopting aggressive RPSs and subsidising renewable energy. However, these renewable energy policies resulted in an electricity rate deficit that raised electricity prices and taxes. As recognised by the government, the costs of the final support for the electricity from renewable energy sources were significantly higher than they had anticipated (IER, 2014). That situation, together with the economic crisis, forced the Spanish government to announce the end of the program with the Royal Decree Law 1/2012 and replaced it with a less lucrative subsidy in 2013.

1.3. Biogas as an energy option

Biogas production has several advantages, even when compared with other renewable energy alternatives, since it can be produced when needed and easily stored (Panwar et al., 2011). Anaerobic digestion is performed by microorganisms that degrade biomass in an oxygen-free environment, producing biogas that can be used for bioenergy purposes and digestate. Biogas is the main product of the anaerobic digestion process and consists of a mixture of gases, mainly composed of methane (50%–75%), carbon dioxide (25%–50%) and other gases (2%–8%) such as water, nitrogen, oxygen, ammonia and hydrogen sulphide (Da Costa-Gomez, 2013). There is a great variety in biogas systems, due to different feedstock biomass, digestion technologies, bioenergy pathways as well as digestate management options (Poeschl et al., 2012). Biogas is a very versatile form of energy since it can be produced from nearly all kind of biomass and it can be used for heat production, co-generation, as vehicle fuel or distributed in the natural gas grid after being upgraded (Poeschl et al., 2012). The produced digestate can be used as an organic fertiliser due to its content in nutrients. It can be treated before spreading on agricultural land to reduce transport costs due to its high water content.

1.3.1. Biogas and circular economy

Biogas as an energy source can provide a significant contribution to the European efforts to develop a more circular economy, as the way to achieve sustainable, low carbon, resource efficient and competitive economy (European Commission, 2015). The concept of circular economy has emerged in response to drawbacks of the conventional linear economy based on 'take, make, consume and dispose' model of growth, in favour of an economy where the value of products, materials and resources is maintained for as long as possible, and the generation of waste minimised. Therefore, nowadays there is an increasing need to develop and apply treatment processes that convert waste into resources through the recovery of valuable products. In this sense, the anaerobic digestion of biomass, especially waste streams, shares the principles of the circular economy by converting these wastes into several valuable products such as energy, water and nutrients, as presented in Figure 1.4. In more detail, recovering bioenergy from waste helps to achieve the European goals regarding renewable energy production and climate change reduction (European Parliament, 2009). In addition, the reuse of digestate in agriculture in safe conditions contributes to nutrients recycling, decreasing the need of mineral fertilisers and also provides water, alleviating pressure on limited water resources (Norton-Brandão et al., 2013). The recovery of biofertilisers including nitrogen and phosphorus from the digestate is especially interesting since the production of mineral fertilisers entails the depletion of non-renewable natural resources such as phosphate rock, oil and natural gas and the production of considerable environmental impacts derived from the extraction, manufacture and use of these fertilisers (Ten Hoeve et al., 2014). The materials that are recovered from waste can be injected back into the economy and used as primary raw materials are named as "secondary raw materials" (European Commission, 2015). However, there is still uncertainty regarding their quality, and it is one of the most important obstacle that prevent the spread the use of secondary raw materials, including digestate and compost. In this context, the development of standards that guarantee their safety use is essential. The Waste Framework Directive introduced the procedure for defining end-of-waste (EoW) criteria, which are criteria that a given waste stream has to fulfil not to be considered as a waste (European Union, 2008). The EoW criteria promote high compost and digestate quality standards by including certain product quality requirements.

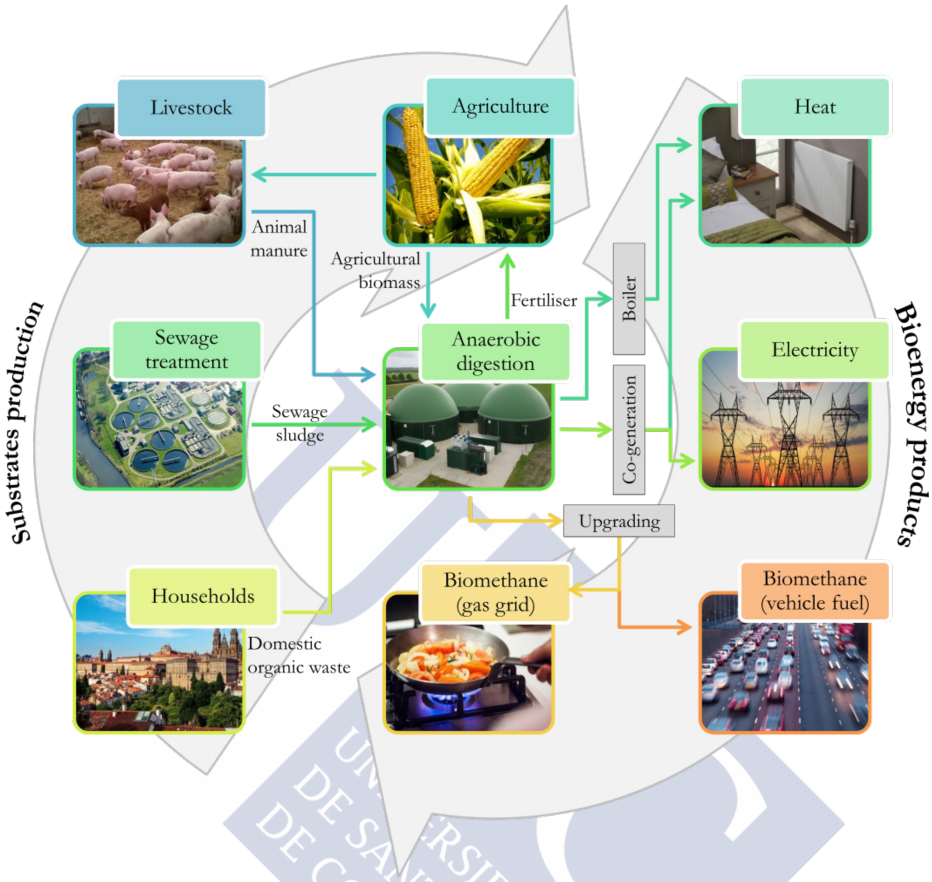

Figure 1.4. Schematic representation of the sustainable biogas cycle

1.3.2. Feedstocks for biogas production

The large amounts of animal manure and slurries produced by the animal breeding sector as well as other organic waste streams from industries and municipalities represent a constant pollution risk with a potential negative impact on the environment, if not managed properly (Holm-Nielsen et al., 2009; Panwar et al., 2011). During the 1970s, anaerobic digestion was applied for the stabilisation of animal waste and the sewage sludge produced in wastewater treatment plants (WWTPs) (Al Seadi et al., 2013). Anaerobic digestion of organic wastes offers several benefits by reducing odours and pathogens (Holm-Nielsen et al., 2009) while producing a renewable fuel (biogas) and an organic fertiliser (digestate). On the contrary, the cultivation of dedicated crops for bioenergy purposes such as cereals was developed in the 1990s in countries like Germany and Austria (Al Seadi et al., 2013). The most widespread feedstocks used for anaerobic digestion are:

• **Energy crops** – Energy crops are common substrates for bioenergy production due to their high biogas potential. Their cultivation requires a high input of fertilisers, pesticides and energy for agricultural activities and transport, entailing substantial environmental impacts due to emissions to air, water and soil (European Commission, 2014). They can be immediately fed to the digester after harvesting or stored as silage for year-round availability. The used energy crops for energy purposes are various types of grass, cereals, beet, potato and sunflower. Among them, maize is the most widely used in Europe (European Commission, 2014). The rising demand for maize can entail a change in the use of soil, increasing the pressure to convert grass- and peat-lands in areas for maize cultivation (European Commmission, 2014). Regarding this issue, alternative crops such as sugar beet have been recently proposed as an alternative to maize for bioenergy production (Jacobs et al., 2017). However, the concerns about the use of cereals for energy purposes are not only related to the environmental impacts of its production. According to Mela and Canali (2014), more than 10% of the available agricultural area in the Po Valley (Italy) was occupied by energy crops, especially maize, maybe displacing the cultivation of food crops. Since the different energy crops render into diverse energy yields per hectare, it is essential to increase the efficiency of agricultural land use (Gissén et al., 2014). It is even possible that in the future, the agricultural land used for energy purposes may be limited by European regulations (Gissén et al., 2014).

• **Livestock waste** – Livestock waste from a variety of animals (pigs, cattle, poultry, horses and many others) can be used as feedstock for biogas production. More specifically, manure processing to biogas through anaerobic digestion recovers the energy contained in the substrate and reduces the risk from pathogens during land spreading (Akbulut, 2012). They offer an adequate carbon to nitrogen (C/N) ratio (25:1) and they are rich in various nutrients, which are necessary for the growth of anaerobic endogenous microorganisms (Al Seadi et al., 2013). In addition, they have a high buffer capacity that can help to stabilise the process in case of pH decrease and they have also a low dry matter content (manure around 10-30% and liquid slurry less than 10%), which gives a low methane yield per unit volume of digested feedstock and makes biomass transport costs high (Al Seadi et al., 2013). The treatment of manure through anaerobic digestion depends on the economic viability of biogas plants installed in areas of livestock production, in which incentive policies play a major role. Smaller and dispersed installations allow a reduction of emissions associated with both manure transport and digestate management, while better supporting local farmer's income (Negri et al., 2016).

• **Source-segregated food waste (SSFW)** – It refers to the organic fraction of household waste that is separately collected, providing a clean and high-quality material for use for anaerobic digestion, while reducing the organic material going to landfills or incineration and increasing recycling and nutrients recovery (Al Seadi et al., 2013). The rationale behind the use of SSFW as a promising organic substrate for anaerobic digestion is its high methane potential and it does not compete for land use. When intending the anaerobic digestion of organic household wastes, high purity must be ensured since low-purity waste may cause technical malfunctions of the biogas plant due to the presence of foreign materials that are a source of pollutants and can have a negative impact on the utilisation of digestate as a fertiliser (Al Seadi et al., 2013). Therefore, it is important to remark that the use of SSFW for anaerobic digestion is limited by the optimisation of separate collection scheme (Cavinato et al., 2011). The content of impurities depends to a large extent on the human factor, that is, the awareness and motivation of the population involved in collection systems. In most wet digestion processes, these compounds are removed by complex pre-treatments. The grade of contamination of the SSFW varies in different regions and depends on the degree of maturity of the collection scheme (Zhang et al., 2013). Moreover, the composition of SSFW varies among regions and seasons as well as with different collection schemes (Zhang et al., 2013), which may affect the stability of the operation of biogas plants using SSFW as the only substrate. In addition, the anaerobic mono-digestion of SSFW can also lead to inhibition in the long-term operation due to nutrients imbalance (insufficient trace elements and excess of macronutrients) as well as due to C/N ratio higher than the optimal reported (20-30:1) (Zhang et al., 2011).

• **Sewage** – Energy and nutrients recovery from wastewater, together with reduced energy requirements, are essential factors to make conventional WWTPs more environmentally sustainable (Campos et al., 2016). This recovery of energy in WWTPs has been mainly conducted using anaerobic digestion with the main driver of energy recovery and sludge stabilisation (Mills et al., 2014). Anaerobic digestion of primary and secondary sludge is a standard technology around the world (Al Seadi et al., 2013; Appels et al., 2008). Since sewage sludge has a methane potential similar to animal slurries, it is commonly co-digested with other substrates or pre-treated including mechanical disintegration, chemical hydrolysis, thermal hydrolysis and enzymatic degradation. Moreover, the primary factor preventing its further application is that sewage sludge, due to its nature, has a high content in organic and chemical pollutants, resulting in high risks related to the use of the derived digestate as an organic fertiliser. Therefore, while the use of sewage sludge in anaerobic digestion is regulated by national legislation, the use of the digestate is controlled by quality standards. There are countries in which the use of digested sewage sludge for agricultural purposes is banned, while in others its utilisation as a fertiliser is governed by strict requirements concerning the limit values of concentrations of heavy metals and persistent organic pollutants as well as the sanitation conditions for inactivation of pathogens and other biologic vectors. Moreover, anaerobic digestion can be an alternative to the aerobic activated sludge system to treat urban wastewater since it is a net energy producing process with ten times lower sludge production.

1.3.3. Anaerobic digestion process

Anaerobic digestion implies complex microbial processes that take place in the absence of oxygen. The microbial population mainly corresponds to diverse genera of obligate anaerobic bacteria and facultative anaerobic bacteria. The anaerobic digestion process includes four main steps: i) hydrolysis, ii) acidification, iii) acetogenesis and iv) methanogenesis (Da Costa-Gomez, 2013). Complex polymers are converted into monomers by extra-cellular enzymes during hydrolysis, while these monomers are transformed into volatile fatty acids (VFAs) and hydrogen during acidogenesis. Acetate, carbon dioxide and hydrogen are produced from VFA in the acetogenesis phase and finally converted into methane during methanogenesis. Some of the important parameters that influence the effectiveness of the anaerobic digestion process are:

• **Temperature** – It controls the extent of the growth rate and metabolism of microorganisms and hence, the enzymatic activity of the microbial population. There are three common temperature ranges at which anaerobic treatment can be achieved: i) psychrophilic (< 20ºC), ii) mesophilic (20–45ºC) and iii) thermophilic (45–60ºC) (Appels et al., 2008). Although production has been documented under psychrophilic temperatures, anaerobic treatment is usually carried out at an either mesophilic or thermophilic range of temperatures.

• **pH** – Each group of micro-organisms has a different optimum pH range, however, in general, should lie between 6.6 and 7.6 (Appels et al., 2008). Values outside this range can be detrimental to process stability since methanogenic bacteria are extremely sensitive to pH changes (the range should be between 6.5 and 7.2), compared to the other groups of bacteria such as acidogenic (that stand variable pH, between 4.0 and 8.5). Deviations from the optimum pH range are usually a result of the increased accumulation of acidic or alkaline products such as VFAs and ammonia, respectively.

• **Hydraulic retention time (HRT)** – This parameter indicates the average period that the feedstock remains in the digester for treatment. The HRT can be defined as the rate of the reactor volume and the volumetric daily flow rate of the organic substrate. The HRT must be long enough to allow anaerobic bacteria to complete their metabolism and proliferation. HRTs vary depending on the type of reactor used. Short HRTs are insufficient for a stable treatment due to a washout of methanogenic bacteria, while long HRTs result in high operating costs.

• **Organic loading rate (OLR)** – This factor indicates the amount of volatile solids to be fed into the digester each day (Babaee and Shayegan, 2011). Both the loading rate and the level of biochemical activity depend on the type of waste fed into the digester (Babaee and Shayegan, 2011).

1.3.4. Biogas cleaning

The required quality of the biogas in terms of its composition depends on how the biogas is going to be utilised. In this sense, the content of certain gases might affect the equipment for biogas utilisation causing corrosion and mechanical wear (Abatzoglou and Boivin, 2009). The most common impurities in raw biogas are water, hydrogen sulphide, ammonia and oxygen. Additionally, impurities can also lead to unwanted emissions when biogas is combusted during utilisation.

• **Water** – It is present in the biogas leaving the digester due to partial evaporation of the moisture present in the substrates (Petersson, 2013) and it should be removed to avoid corrosion problems associated to carbonic acid. The parameters that affect the solubility of water in the gas must be evaluated (Petersson, 2013). In more detail, water condenses if pressure increases or if temperature decreases; therefore, compression and/or cooling are possible technologies to remove water from biogas.

• **Hydrogen sulphide** – It is formed by reducing-sulphate bacteria and the digestion of proteins containing sulphur. The presence of hydrogen sulphide during the use of biogas can lead to corrosion since it forms sulphuric acid in combination with water. The maximum hydrogen sulphide concentrations in relation with different technologies for the use of biogas are presented in Table 1.1. The combustion of biogas with hydrogen sulphide will also lead to emissions of sulphur oxides produced during the combustion. This compound can be removed by supplying a small flow of air or oxygen into the digester or into a biological filter. Both compounds react producing elementary sulphur through biological oxidation, catalysed by *Thiobacillus* bacteria usually present in the digester (Petersson, 2013). The presence of hydrogen sulphide can also be prevented by introducing iron chloride or iron sulphate to produce insoluble iron sulphide that will precipitate in the digester.

Table 1.1. Maximum hydrogen sulphide concentrations (in parts per million by volume, ppmv) in biogas depending on the technology used, adapted from Llaneza et al. (2010)

Energy production system	**Maximum H_2S (ppmv)**
Co-generation	<1,000
Micro-gas turbines	<70,000
Fuel cells	<50
Grid Injection	<1
Vehicle fuel	<1

• **Carbon dioxide** – The use of biogas as a substitute for natural gas requires the removal of carbon dioxide, converting biogas into biomethane with methane content up to 95%. It can be done through absorption, adsorption, permeation or cryogenic upgrading (Petersson, 2013). Two possibilities are water scrubbing and pressure swing adsorption (PSA) (Weiland, 2010). Water scrubbing is an absorptive method using only water as an inorganic solvent. It implies the dissolution of gas in the absorption agent. Within PSA, carbon dioxide is retained on the surface of solids such as activated carbon, zeolites or carbon molecular sieves (Beil and Beyrich, 2013). Low temperatures and high pressures increase the adsorption rate. Besides carbon dioxide, other impurities can be also removed such as hydrogen sulphide, ammonia or water; however, in practice, they are removed before the biogas is injected into the adsorption columns (Petersson, 2013).

1.3.5. Biogas use alternatives

Over the years, biogas utilisation technologies have improved and nowadays, several technologies convert biogas into more useful forms of energy (Kaparaju and Rintala, 2013). The selected pathway among all available options depends on the specific case as well as on the biogas properties. The energy potential of the biogas depends on its methane and, in some cases, its hydrogen content. With this respect, some energetic equivalence between biogas and other energy sources are presented in Figure 1.5.

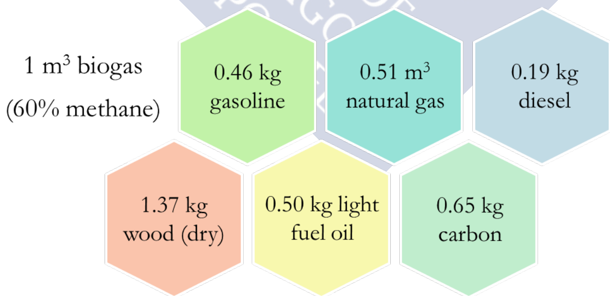

Figure 1.5. Energy equivalences of biogas with other fuels

• **Heat and electricity production** – The production of heat in boilers is the simplest way of using biogas. The conversion efficiencies for heat production range from 75% to 90%. However, biogas is usually used in co-generation heat and power (CHP) engines (Weiland, 2010); however, alternatives to the standard motor CHP are micro-gas turbines and fuel cells (Kaparaju and Rintala, 2013). A comparison among the three options is presented in Table 1.2. As shown, CHP engines can achieve electricity efficiencies between 30-42% while thermal ones are around 40%-50% (Kaparaju and Rintala, 2013). The economics of on-site CHP applications are enhanced by effective use of recovered heat generated by the engine jacket and exhaust gas. Micro-gas turbines result in a lower electric efficiency (25%–30%) but have long maintenance intervals. Fuel cells provide higher electric efficiency (40%-45%) but need previous gas cleaning, because it is sensitive to impurities (Weiland, 2010).

Table 1.2. Energy from biogas, adapted from Kaparaju and Rintala (2013)

Parameter	**Units**	**CHP engine**	**Micro-gas turbine**	**Fuel cell**
Unit capacity	kW_e	110-3000	30-300	300-1500
Plant size		Small/ medium	Small	Small
Electrical efficiency	%	30-42	25-30	40-45
Thermal efficiency	%	40-50	30-35	30-40
Biogas purification	%	Medium	Medium	High
Investment costs	€/kW_e	400-1100	600-1200	3000-4000
O&M costs	€/kW_e	0.01-0.02	0.008-0.015	0.003-0.010

• **Biomethane for natural gas grid** – The upgrading of the biogas produces biomethane and it can be used similarly to natural gas without the need to change any settings on equipment (Urban, 2013). Countries like Germany, Sweden and Switzerland have defined quality standards for biomethane injection into the natural gas grid (Weiland, 2010). The biogas injection equipment depends on the operating conditions of the natural gas grid in terms of pressure, gas composition, length of the pipeline as well as the type of upgrading technology (Urban, 2013).

• **Biomethane for vehicle fuel** – The produced biomethane after upgrading can also be used as vehicle fuel. Utilisation of methane in the transport sector is widely distributed in Sweden and Switzerland (Weiland, 2010). The upgraded biogas is stored at 200 to 250 bar in gas bottles.

1.3.6. Digestate management

The composition and quality of the digestate produced after anaerobic digestion highly depend on the composition and quality of the feedstock digested (Evangelisti et al., 2014). As in anaerobic digestion, nutrients loss is not significant, it can be considered that the total nutrients content in the feedstock and the digestate is similar (Evangelisti et al., 2014; Møller et al., 2009). However, the total nitrogen entering the reactor is mainly in its organic form and then undergoes to hydrolysis and ammonification processes during digestion (Fantin et al., 2015). This can be considered as one of the main drawbacks of the anaerobic digestion process since it enhances the potential for ammonia emission during the digestate storage if compared to the untreated liquid slurry.

The use of digestate as fertiliser is by far its main utilisation and it is considered to be the most sustainable option, as it allows the recycling of nutrients, helping to preserve the limited natural resources such as fossil resources of mineral phosphorus (Al Seadi et al., 2013). However, the application of organic fertilisers to agricultural land is regulated by the European Nitrate Directive 91/676/EEC (EEC, 1991). During many years organic substrates such as animal manure and slurries have been applied to land to recycle their nutrients. However, this practice entails important impacts to air, water and soil as well as to biodiversity (FAO, 2006). Moreover, these impacts are concentrated in specific locations as a result of the current tendencies in livestock production related to intensification and specialisation (Oenema et al., 2007). As an example, European pig production is mainly developed in eight zones: Denmark, Belgium, Netherlands, northern Germany, Brittany (France), Catalonia and Aragon (Spain) and Po Valley (Italy) (Bernet and Béline, 2009). Within these areas, the use of manure as fertiliser leads to an over-application of nutrients, mainly nitrogen and phosphorus, on agricultural soils resulting in water and soil pollutions, resulting in the designation of nitrate vulnerable zones (NVZs), as shown in Figure 1.6.

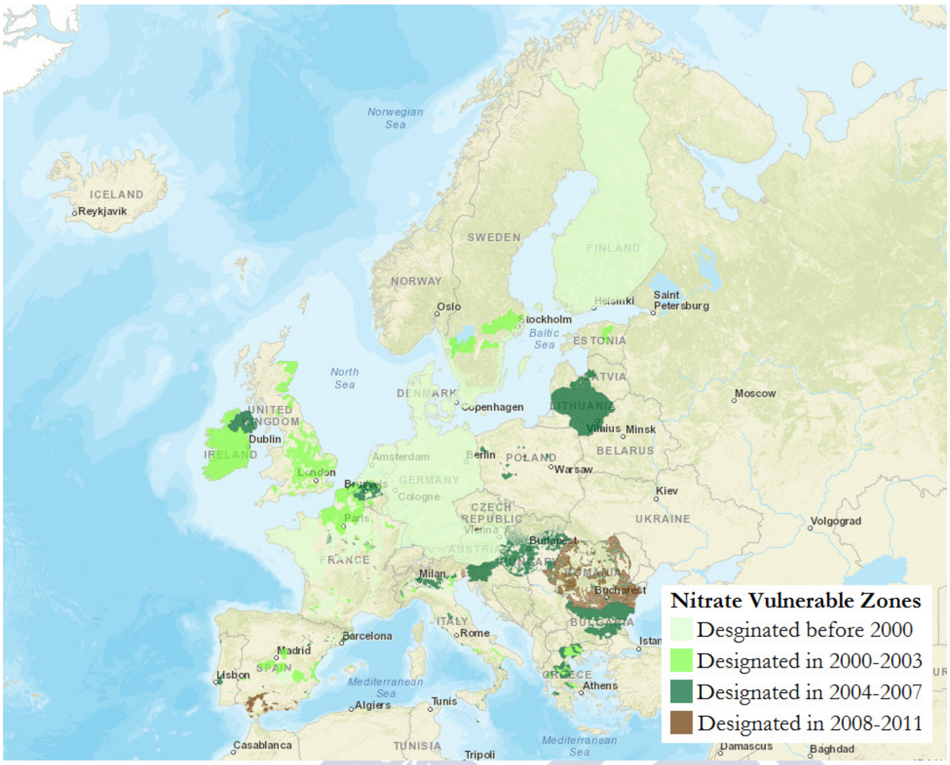

Figure 1.6. Designated Nitrate Vulnerable Zones in the European Union[2]

These NVZs include surface freshwater and ground water containing or that could contain eutrophic if no action is taken to reverse the trend, a concentration of more than 50 mg/L of nitrates as well as freshwater bodies, estuaries, coastal waters and marine waters found to be eutrophic or that could become eutrophic if no action is taken to reverse the trend (EEC, 1991). Moreover, Member States can also choose to apply measures to the whole territory, instead of designating NVZs. In these areas, mandatory actions established to be implemented by farmers, including Codes of Good Agricultural Practices and the limitation of its application up to 170 kg of nitrogen per hectare and year. Therefore, sometimes it is necessary to transport the digestate to regions with nutrients deficits to redistribute away from intensive areas (Rehl and Müller, 2011). However, digestate transportation causes logistical problems since it consists of 95% water on average. Various treatment options are available for reducing the amount of

[2] http://fate-gis.jrc.ec.europa.eu/geohub/MapViewer.asp

water or for the separation or removal of nutrients. Digestate processing can be approached in two ways: i) digestate conditioning for improved quality of digestate as fertiliser and ii) digestate treatment aiming the reduction of impacts related to nutrients discharge.

• **Solid/liquid separation** – It consists of separating the solid phase from the liquid. Both fractions can be used without further treatment in agricultural land (Al Seadi et al., 2013). Solid/liquid separation results in most of the phosphorus with the solid fraction and most of the nitrogen with the liquid fraction (Bauer et al., 2009), which facilitates nutrients management. A variety of solid-liquid separation technologies is available, including centrifuges, screw press, bow sieves, double circle bow sieves, sieve belt presses and sieve drum presses (Al Seadi et al., 2013). The most relevant ones are the decanter centrifuge and the screw press.

• **Struvite precipitation** – The recovery of phosphorus from digestate is especially interesting since actually phosphorus is taken from the phosphate rock, a non-renewable geological reserve (Campos et al., 2016; Tao et al., 2016). When the digestate contains magnesium ions, ammonium and orthophosphates, struvite ($MgNH_4PO_4 \cdot 6H_2O$) may be formed. Struvite typically precipitates in the form of stable white crystals having an orthorhombic pyramidal crystal lattice. Struvite is an appropriate fertiliser as it get dissolved slowly in the environment due to its low solubility, providing nutrients at a rate suitable for crop uptake (Tao et al., 2016). One of the main disadvantages of the process is related to chemicals consumption. In some cases, the addition of magnesium compounds and sodium hydroxide may be necessary to provide enough magnesium and to rise pH up to 9, as required for the precipitation of struvite (Tao et al., 2016).

• **Ammonia stripping** – Ammonia stripping by air or steam is a relatively simple gas–liquid mass transfer process that can be controlled for efficient ammonia removal and recovery (Zeng et al., 2006). The reduction of the ammonium concentration in the digestate is based on the low solubility of ammonia. Since equilibrium between ammonia and ammonium is mainly related to temperature and pH, when increasing temperature or pH, equilibrium is shifted towards ammonia. Therefore, by injecting air or steam, ammonia is easily transferred towards the gas phase (Zeng et al., 2006). This process has the advantage that it does not require the addition of expensive chemicals and that it recovers a standardised nitrogen fertiliser.

• **Biological nitrogen removal (BNR)** – This process is usually based on conventional nitrification/denitrification techniques. Nitrification is the aerobic oxidation of ammonia to nitrite and nitrate by autotrophic nitrifying bacteria, while denitrification is the anoxic reduction of nitrate to nitrite and nitrogen gas by heterotrophic bacteria (Malamis et al., 2013). For this process, it may be necessary the addition of an external carbon source since most of the organic carbon was removed in anaerobic digestion (Frison et al., 2013). Because nitrite is an intermediate of both reactions, intensive research has been carried out these last years to develop new processes based on nitrogen removal over nitrite named as short-cut nitrogen removal (Malamis et al., 2013). This process has several advantages compared with conventional nitrification/denitrification, including lower requirements of oxygen (25% less) and organic carbon (40%). It is also possible to use nitrite as an electron acceptor for anaerobic ammonium oxidation to nitrogen gas, known as the anammox process, according to the autotrophic nitrogen removal process.

• **Enhanced biological phosphorus removal (EBPR)** – This process enables the removal of phosphorus by alternating anaerobic and aerobic phases that promote the growth of phosphorus accumulating organisms (PAOs). In these conditions, PAOs can store phosphate as intracellular polyphosphate, leading to phosphorus removal via biomass removal. In addition, phosphorus removal is also possible under anoxic conditions, enabling denitrifying phosphorus removal via nitrite (Peng et al., 2011). This results from the ability of certain denitrifying bacteria to store significant amounts of polyphosphate as they have a metabolism that is very similar to PAOs growing in conventional EBPR processes. Consequently, nitrogen and phosphorus removal can both be accomplished in the same reactor under anoxic conditions, increasing the process attractiveness (Malamis et al., 2013).

1.4. List of acronyms

BNR	Biological nitrogen removal
C/N	Carbon to nitrogen ratio
CHP	Co-generation heat and power
COP	Conference of the Parties
EBPR	Enhanced biological phosphorus removal
EoW	End-of-Waste
FIP	Feed-in premium
FIT	Feed-in tariff
GHG	Greenhouse gas
HRT	Hydraulic retention time
IEA	International Energy Agency
NDC	Nationally Determined Contributions
NVZ	Nitrate vulnerable zone
OECD	Organisation for Economic Cooperation and Development
OLR	Organic loading rate
PAO	Phosphorus accumulating organism
PSA	Pressure swing adsorption
RO	Renewable Obligation
ROC	Renewable Obligation Certificate
RPS	Renewable portfolio standards
SSFW	Source segregated food waste
UNFCCC	United Nations Framework Convention on Climate Change
VFA	Volatile fatty acid
WWTP	Wastewater treatment plant

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Chapter 2: Sustainability assessment of biogas systems

Summary

The need to work towards sustainable development is well recognised by international organisations and it is considered a priority in the current agenda of governments. In this context, ensuring the sustainability of alternative and renewable energy sources is fundamental to achieve the decarbonisation of the energy sector. As a consequence, different measurement tools that allow monitoring the economic, social and environmental dimensions of sustainability have been developed, enabling to achieve the balance between economic and social progress while providing environmental protection and preserving the resources of the planet. The objective of Chapter 2 is to present a brief description of the available methods for measuring sustainability in biogas production systems, paying special attention to Life Cycle Assessment (LCA), a methodology able to quantify the environmental benefits and impacts of products and process as well as its combination with Data Envelopment Analysis (DEA) which is a mathematical model that measures the relative eco-efficiency of multiple homogeneous units and the Analytical Hierarchy Process (AHP), a multi-criteria analysis to integrate economic, social and environmental indicators in decision-making processes. In more detail, LCA is a worldwide accepted and standardised methodology for assessing the environmental consequences of production processes and activities through their entire life cycle. In this sense, LCA has been widely applied to analyse the environmental sustainability of biogas production systems, as shown in the literature review. Finally, the goal and structure of this thesis are summarised at the end of the chapter.

Outline of Chapter 2

2.1. Roots of sustainable development

The public awareness regarding the effect that human activity has on the surrounding environment started in the second half of 19th century (Sikdar, 2003). In 1962, Rachel Carson published the environmental science book entitled "Silent Spring", pointing out the devastating effects of the indiscriminate use of pesticides on ecosystems. The awareness of people and authorities regarding the dependency on fossil fuels started due to the oil crisis occurred during the decade of 1970, which caused the increase of crude oil price, having an enormous impact on global economy. In addition, during the same period, it was discovered that chlorofluorocarbons (CFCs), which were widely used as refrigerants and solvents, were mainly responsible for the depletion of the ozone layer (Noakes, 1995). However, CFCs were not completely forbidden until 1989, when the Montreal Protocol was ratified. In 1983, the World Commission of Environment and Development (WCED) was established by the United Nations with the aim of developing proposals and solutions to deal with the degradation of the environment and natural resources. The term sustainable development came into use in policy circles after the publication in 1987 of the book "Our Common Future", also known as "The Brundtland Report", written by the WCED headed by Gro Harlem Brundtland. In this document, the term sustainable development was firstly defined as "the development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED, 1987). This concept of sustainable development is simple from a qualitative point of view (Sikdar, 2003): the Earth's natural resources are limited and they are increasingly used; moreover, they are mainly consumed by a minority of people living in the wealthy nations, which creates intra and inter-generational inequity. Therefore, according to the report, the present inefficient lifestyle of the developed nations is not sustainable in the long term due to the disproportionately large resource consumption per capita in developed countries that results in environmental degradation and societal inequity (Sikdar, 2003; WCED, 1987). Sustainable development was recently re-defined by Griggs et al. (2013) as "the development that meets the needs of the present while safeguarding Earth's life-support system, on which the welfare of current and future generations depends". Taken this definition into account, the protection of Earth's life-support system and poverty reduction should be the main two priorities when stating the goals of sustainable development (Griggs et al., 2013), as shown in Figure 2.1.

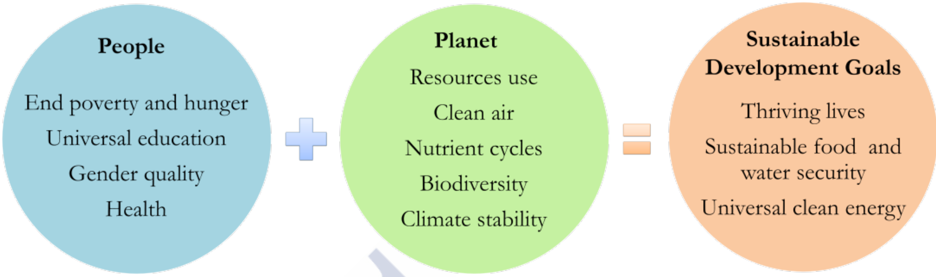

Figure 2.1. Definition of sustainable development goals, adapted from Griggs et al. (2013)

Moreover, there is an international consensus that sustainable development should integrate three main pillars (Griggs et al., 2013; Lozano, 2008). These three pillars should be addressed not only from an individual perspective but also including the dynamic inter-relations among them (Lozano, 2008). As depicted in Figure 2.2, the integration of each pair of dimension is considered a partial state of sustainability, whereas sustainability is located in the centre of the diagram, where the three dimensions meet (Lozano, 2008).

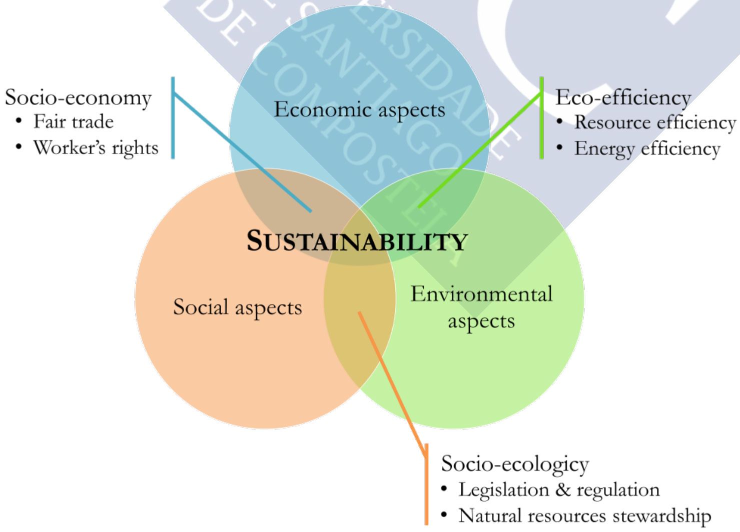

Figure 2.2. Three dimensions of sustainability and their interconnections, adapted from Sikdar (2003)

Conventionally, researchers have searched for improvements in one dimension of sustainability; therefore, only focusing on economic, social or environmental aspects (Sikdar, 2003). Other studies have been published regarding the measure of two dimensions, corresponding to the interactions of any two aspects of sustainability, including eco-efficiency, socio-ecology and socio-economy. However, sustainability is measured when the three dimensions are analysed.

2.2. Methodologies for sustainability assessment

The need of methodologies for measure the different pillars of sustainability is becoming more important to provide solid facts and data as the basis for strategic decision making by governments, international organisations and companies. With this respect, several methodologies and tools have been proposed and developed to measure different dimensions of processes sustainability, including:

• **Life Cycle Assessment (LCA)** – LCA is an internationally accepted methodology for the assessment of the environmental burdens associated with all inputs and outputs related to the entire life cycle of a product or process, from the extraction of raw materials up to the disposal of wastes (ISO 14040, 2006). LCA is probably the most widespread methodology for the evaluation of the environmental profile products or processes. Numerous LCA studies are available in the literature concerning biogas production and use (De Vries et al., 2012; Fantin et al., 2015; Poeschl et al., 2012a, 2012b). In these studies, biogas production systems from different feedstock as well as their possible applications have been assessed from environmental and energy perspectives, with special attention on GHG emissions and fossil fuel depletion (Börjesson and Berglund, 2007, 2006).

• **Life Cycle Assessment (LCA) + Data Envelopment Analysis (DEA) –** The implementation of DEA in combination with LCA has been proposed to analyse the eco-efficiency, that is the operational and environmental performances of multiple similar entities (Iribarren et al., 2010). This novel alternative avoids the use of average inventory data and enriches results interpretation through eco-efficiency verification (Iribarren et al., 2010). This approach has been applied using technical and environmental indicators to different production processes such as wine production (Vázquez-Rowe et al., 2012) and fisheries

(Vázquez-Rowe et al., 2010) and even to WWTPs (Lorenzo-Toja et al., 2015). Moreover, Iribarren et al. (2016) applied this methodology also integrating socio-economic indicators for sustainability assessment.

• **Multi-criteria Analysis (MCA)** – This approach includes a set of decision-making methods for addressing complex problems characterised by high uncertainty, opposite objectives and different sources of data and perspectives (Milutinović et al., 2014). Due to the inherent characteristics, the proposed options can be positive for some criteria but negative for others. Therefore, MCA does not provide a unique solution optimising all the criteria but a set of compromise solutions among which the decision-maker has to choose. Among the different available MCA methodologies, Saaty (1980) developed the Analytic Hierarchy Process (AHP) as one of the available alternatives for multi-criteria decision making and as a tool for analysing the decision-making process. The AHP is can be used for the hierarchical decomposition of a complex problem, helping to identify the best option considering several sets of criteria with different nature (Chatzimouratidis and Pilavachi, 2009). This methodology has been widely applied to analyse the sustainability of energy and waste systems, including for the selection of the best alternative for energy recovery from municipal solid waste (Nixon et al., 2013), the selection of the best solid waste treatment technology (Samah et al., 2010), to rank suitable locations to place a municipal solid waste facility (De Feo and De Gisi, 2010) and to analyse the sustainability of cities (González et al., 2013).

2.3. Life Cycle Assessment

The LCA methodology involves the holistic assessment of a product or process from an environmental perspective over its entire life, from raw material production, manufacture, distribution, use and disposal (ISO 14040, 2006). This methodology identifies the most polluting processes in the life-cycle of a product or process. Moreover, it allows the implementation of improvements options, considering upstream and downstream environmental consequences of these decisions, allowing the avoidance of shifting environmental burdens from one environmental concern to another, from one country to another or from one stage to another in a product's life cycle (Hauschild et al., 2011). This assessment method can also be used to analyse the environmental performance of different processes or products with the same function to identify the most sustainable from an environmental point of view. It is also possible to implement this methodology to optimise the environmental profile of a single product during its design stage, known as eco-design (Wolf et al., 2010).

The first studies similar to LCAs date from the early 1970s, as a consequence of the growing concern about environmental issues such as energy efficiency, pollution control and waste management. One of the first studies was accomplished by the Midwest Research Institute in 1969 for the Coca-Cola Company (Guinée et al., 2011). They performed a "Resource and Environmental Profile Analysis" that compared in quantitative terms the resource requirements, emission loading and waste flows of different beverage containers. Afterwards, LCA was slowly developed from 1970 to 1990 with widely diverging approaches, terminologies and results, without a common theoretical framework due to a lack of international scientific discussion and exchange platforms (Guinée et al., 2011). Standardisation of the methodology happened between 1990 and 2000, when a number of workshops were organised and LCA guides and handbooks were published (Guinée et al., 2011). The Society of Environmental Toxicology and Chemistry (SETAC) started playing a leading and coordinating role in bring LCA practitioners together to collaborate on the continuous improvement and harmonisation of LCA framework, terminology and methodology. In addition, they provided one of the first accepted definitions of LCA, published in Consoli et al. (1993): "an objective process to evaluate the environmental burdens associated with a product, process, or activity by identifying and quantifying energy and materials used and wastes released to the environment, and to evaluate and implement opportunities to affect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing extracting and processing raw materials; manufacturing, transportation and distribution; use, re-use, maintenance; recycling and final disposal". Next to SETAC, the International Organisation for Standardisation (ISO) has been involved since 1994 with the task of standardisation of LCA methods and procedures. There are currently two international standards, the ISO 14040 and 14044 which facilitate the consolidation of procedures and methods of LCA, helping to contribute to the general acceptance of LCA by all stakeholders and the international community. In the first place, the ISO 14040

(2006) provides an overview of the methodology, including applications and limitations of LCA studies. Accordingly, LCA can contribute in identifying opportunities to improve the environmental performance of products and processes, informing decision-makers in industry and governments and marketing, including eco-labelling and environmental product declaration). Moreover, the ISO 14044 (2006) provides guidelines for data collection and validation as well as for the impact assessment phase. ISO 14040 and 14044 established four general phases that are required for the completion of an LCA study. The stages of an LCA study are schematically represented in Figure 2.3, and include i) Goal and scope definition, ii) Life cycle inventory (LCI), iii) Life cycle impact assessment (LCIA) and iv) Interpretation.

2.1.1. Goal and scope definition

The goal of an LCA study states the intended application, the reasons for carrying out the study, the intended audience and the decision-context (e.g. support decision on governmental recommendations). The decision-context is one key criterion for determining the most appropriate method for modelling the analysed process or product, which can be attributional or consequential (Wolf et al., 2010).

• **Attributional life-cycle model** – It includes all the processes that are identified to contribute to the supply-chain of the system relevantly. Therefore, it represents the actual data of the system under study, supposing that it is embedded into a static technosphere. Consequently, it depicts the potential environmental impacts that can be attributed to a process or product over its life cycle.

• **Consequential life-cycle mode** – It integrates the supply-chain as it is theoretically expected as a consequence of the analysed decision, including the changes derived from the interaction between the system and the markets. The objective is to identify the consequences that a decision has for other processes. Hence, this model is not reflecting the actual or estimated supply-chain, but a hypothetic generic supply-chain is modelled along market-mechanisms and potentially including political interactions and consumer behaviour changes.

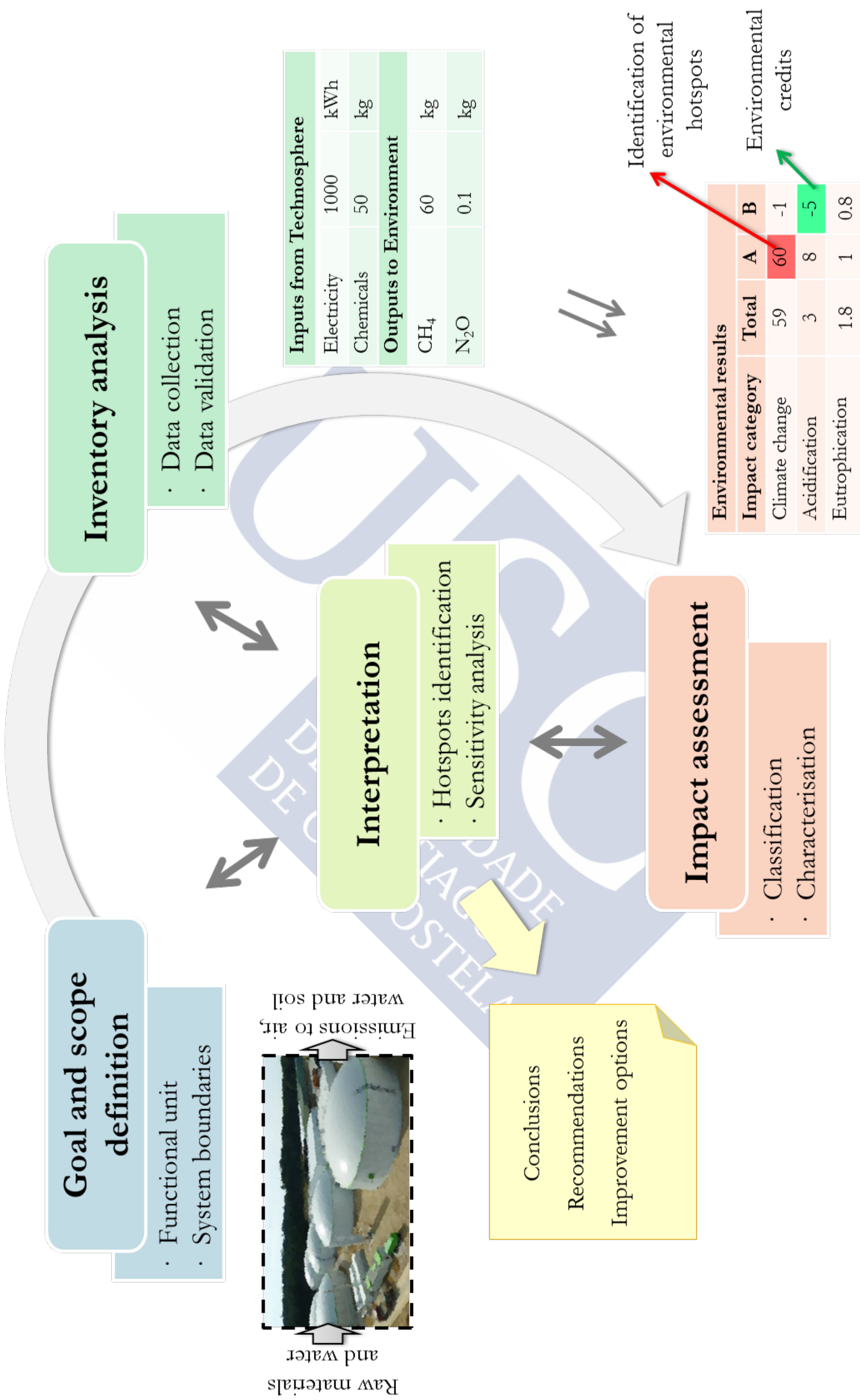

Figure 2.3. Different phases of LCA and key elements

Regarding the scope of the study, it should be sufficiently well defined to ensure that the depth and detail of the study are compatible and sufficient to address the established goal. The scope includes the identification and selection of several relevant aspects involved in the analysis, including the product system to be studied, the functions, the functional unit (FU), the system boundary, allocation procedures, methodology of impact assessment and impact categories selected, data requirements, assumptions, limitations and data quality requirements. It should also be considered that LCA is an iterative technique, and while the following steps are being performed, the scope may require modification to meet the original goal of the study.

Function provided by the system and functional unit

The selection of the function of the system is one of the main methodological issues in LCA studies as it is related to the definition of the FU and the system boundaries. A system may have a number of possible functions and the one selected depends on the goal and scope of the study. The FU defines the quantification of the identified functions of the system, providing a reference to which the inputs and outputs are related. As mentioned, some systems have more than one function since they provide more than one product or service. Different approaches are used for solving multi-functionality as presented in Figure 2.4. The choice of the most appropriate one depends among others on the goal situation of the study, available data and information, and the characteristics of the multifunctional process or product (Wolf et al., 2010).

• **Subdivision** – Subdivision of multifunctional processes refers to the collection of data individually for several mono-functional processes that are constituents of the multifunctional process and render into the production of the product under study. This can be performed unless any of the included single-operation unit processes is still multifunctional.

• **System expansion** – Within this approach two options are possible: i) to add another function to make the system comparable (i.e. system expansion in the stricter sense) or ii) to subtract not required functions substituting them by the ones that are replaced (i.e. substitution by system expansion).

• **Allocation** – This approach solves the multi-functionality by partitioning the individual inputs and outputs flows between the co-products according to certain criteria. According to ISO 14044, allocation should be avoided whenever possible by applying subdivision or system expansion. When it cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects a relationship between them, which can be physical, economic, energetic or exergetic.

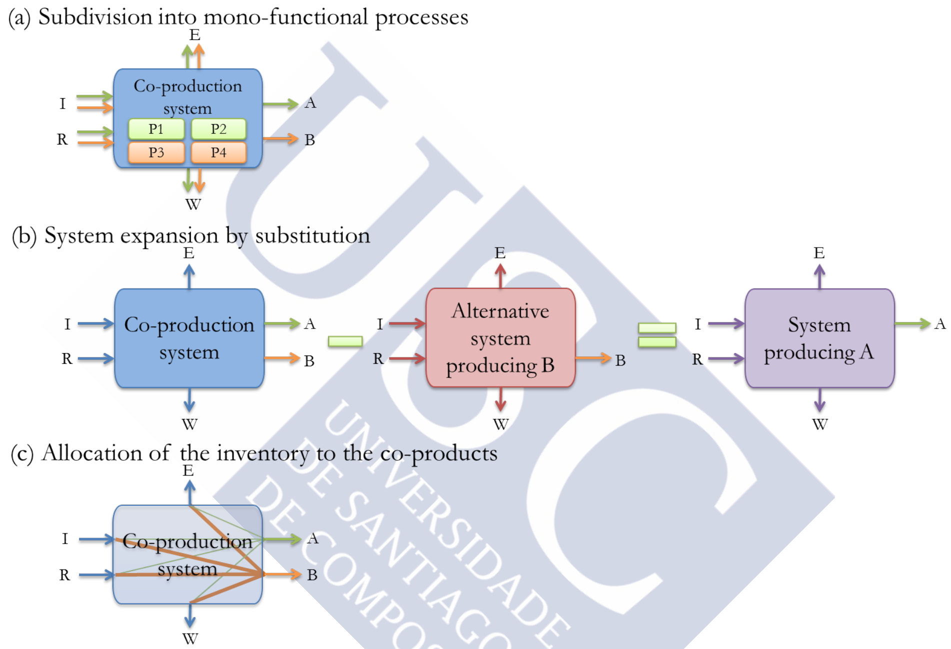

Figure 2.4. Different approaches for solving the multi-functionality problem including (a) subdivision, (b) system expansion and (c) allocation, adapted from Wolf et al. (2010). Acronyms: I – inputs, R – resources, E – emissions, W – wastes, A – product A, B – product B

It is important to highlight at this point that these methodological issues must be addressed when performing the LCA of biogas production, since these systems involve the co-production of different products, from both the anaerobic digestion process and the final use of the produced biogas. Moreover, this is especially relevant when anaerobic digestion is selected as the treatment option for organic waste management since resource recovery from waste results in increasingly multi-functional systems (Heimersson et al., 2017). In more detail, in these waste management systems the main function is the treatment of organic waste, while other secondary functions can be identified, including: i) the production of biogas, ii) the production of electricity, heat and/or biomethane from biogas and iii) the production of digestate since it is a suitable organic fertiliser. Therefore, how these multiple products are considered in LCA studies is becoming increasingly important (Heimersson et al., 2017). As shown before, this can be handled in different ways. In this complex systems, subdivision cannot be conducted since it is not possible to inventory the system in such detail that allows linking each flow to each product (Heimersson et al., 2017). Using substitution to handle multi-functionality to avoid allocation is consistent with the recommendations in the ISO 14044 and the International Reference Life Cycle Data (ILCD) Handbook (ISO 14044, 2006; Wolf et al., 2010). This can be done by given the system a credit for the secondary functions by awarding the system with the avoided negative impacts corresponding to the avoided product or service that the secondary functions replace.

System boundary

The system boundary defines which unit processes belong to the analysed system (ISO 14040, 2006). These processes are recognised because they are required for providing the function to the system as defined by the FU. Therefore, the system boundary separates the analysed system with the rest of the technosphere, defining the boundary where the system exchanges elementary flows with nature (Wolf et al., 2010). The choice of unit processes to be included in the study depends on the goal and scope definition of the study, its intended application and audience, the assumptions made, data and cost constraints and cut-off criteria (ISO 14040, 2006). It can be distinguished two types of processes: i) foreground processes that refer to the process required to produce the product under study and ii) background processes that include the processes required to produce generic materials, energy, transport and waste management. The ILCD Handbook suggest that the system boundary shall be represented in a "system boundary diagram", that is a semi-schematic diagram that explicitly shows which parts and life cycle stages of the system are initially intended to be included and excluded (Wolf et al., 2010).

Specifically applied to biogas systems, the definition of the system boundaries of some systems digesting organic waste streams may be conflictive. In more detail, the distribution of the burdens related to the production of the waste between the producer system and the treatment system can be problematic (Doka, 2007). The question is if the waste can be regarded as a valuable product since it can produce biogas or as a waste material that needs to be managed. Organic wastes are considered as a zero value products since the biogas plants do not usually have to pay to receive them. In these cases, it is a common practice in LCA studies to consider that the production of organic wastes such as manure, food and industrial wastes are excluded from the system boundaries of the biogas system since they are considered waste streams from other production systems (i.e. livestock and food sectors). Broadly speaking, although their use in biogas plants is an option for their management, the production of such wastes would not be influenced by a change in the biogas management scheme. A similar analysis can be applied to the digestate fraction from anaerobic digestion. Farmers that use the produced digestate as an organic fertiliser usually do not have to pay for it; therefore, it can be considered as a waste and the environmental impacts derived to its handling should be allocated to the anaerobic digestion process. This brings an important problem when applying the digestate on agricultural land; while the emissions of mineral fertilisers or animal manure are fully attributed to the agricultural production, the emissions from digestate would be attributed to the biogas system. This would lead to questionable conclusions from LCA studies. According to Doka (2007), there are good reasons to include the digestate application as a waste in the LCA of biogas systems, but equally justifiable reasons can be found to set the cut-off boundary to include digestate application in agriculture as a recycled material.

Following the subject of system boundary, special mention also deserves the consideration (or not) of biogenic carbon within the system boundaries of LCA studies. This consideration is especially important when dealing with biogas systems since they digest biomass, either energy crops or organic waste, which can be considered a temporary storage of carbon. Biogenic carbon is defined as carbon contained or derived from biomass that was accumulated during plant growth as a result of photosynthesis (Wiloso et al., 2012). Conventionally, LCA studies do not assign any environmental burden to carbon dioxide emissions from biogenic sources (Brandão et al., 2013). In these cases, it is considered carbon neutrality on the basis that the expected carbon dioxide uptake from biomass growth is the same than the expected carbon emitted over the full life cycle in the same amount, either naturally decomposed or burned (Wiloso et al., 2012). Therefore, it is considered that there is not a net increase in the atmospheric content of carbon dioxide and the benefits of temporally removing it from the atmosphere and the impacts related to its latter emission are excluded from many LCA studies (Brandão et al., 2013; Wiloso et al., 2012). However, with the aim of validating this assumption, the biomass that was previously harvested should be replaced by new growing one in relative short term. In this sense, the use of annual crops may not increase the amount of atmospheric carbon due to compensation by the relatively un-delayed photosynthesis (Wiloso et al., 2012). There are many authors that disagree with this statement. Carbon sequestration during biomass growth can be accounted as a negative emission in LCA. The argument to support this approach is that during the time between the harvesting of biomass and its decomposition or burnt, the concentration of carbon dioxide in the atmosphere is temporarily decreased and some radiative forcing is avoided. Other authors support the idea that temporary storage of biogenic carbon may have a negative effect due to the change of concentration gradient between the atmosphere and the oceans, making that the oceans absorb less carbon dioxide (Wiloso et al., 2012). The consideration of temporary carbon storage and delayed emissions within the system boundaries of LCA studies is discouraged by the ILCD Handbook, unless the goal of the study clearly includes it (Wolf et al., 2010). In any case, the consideration of biogenic carbon within the system boundary in LCA studies is still under discussion.

Data quality requirements

According to ISO 14044, data quality requirements shall be specified to guarantee that the goal and scope of the LCA are met. It should include time-related coverage (data age and the minimum period of time over which data should be collected), geographical coverage (area from which data for each unit process should be collected), technology coverage, precision (measure of the variability of the data values for each process), completeness (percentage of flow that is measured or estimated), consistency (assessment of whether the methodology for data collection is applied uniformly along the analysis), reproducibility (assessment of the extent to which information about the methodology and data values would allow an independent practitioner to reproduce the results reported in the study), sources of the data and uncertainty of the information.

2.1.2. Life cycle inventory

This stage involves data collection as well as the calculation of the remaining data required to complete all relevant inputs and outputs of each unit process defined within the system boundary. The LCI data can be divided into primary and secondary data; while the former is provided by the producers of goods and operators of processes and services as well as their associations; the latter is provided by databases and represent generic data. The process of conducting an inventory analysis is iterative; that is, if the knowledge of the system increases, new data requirements may be identified. Data for each unit process defined in the system boundary should be collected from the system under study and expressed on the basis of the FU selected. They would include energy inputs, raw material inputs, ancillary inputs, other physical inputs, products, co-products and waste, emissions to air, discharges to water and soil, and other environmental aspects.

Specifically regarding LCA studies dealing with biogas systems, the calculation of different LCI data is required in different unit processes and at different stages of the life cycle, especially data related to direct emissions such as from digestate storage or application since they are not usually measured due to its difficulty. The estimation of this kind of data represents a crucial issue in biogas LCA studies because they play an important role in the environmental results. Therefore, in order to consider these emissions within the system boundary, they should be estimated through available methodologies in the literature. However, there are several different methodologies and there is not a general consensus on which one should be selected.

2.1.3. Life cycle impact assessment

In the LCIA phase, the LCI data from the previous stage is translated into different impact categories and category indicators related to human health, natural environment and resource depletion in support of interpretation (Wolf et al., 2010). Therefore, the LCIA is aimed at evaluating the significance of potential environmental impacts produced by the system under study (ISO 14040, 2006).

The LCIA analyses the potential environmental impacts that are caused by interventions that cross the boundary between the system under study and the nature. The impact assessment may include the iterative process of reviewing the goal and scope of the LCA study to determine if the objectives of the study have been met, or to modify the goal and scope if the assessment indicates that they cannot be achieved.

LCIA stage has different steps with elements both mandatory and optional as described in ISO 14044 (2006). The mandatory elements are: i) impact categories, indicators and characterisation models selection, ii) classification and iii) characterisation; while the optional components are: i) normalisation, ii) grouping and iii) weighting.

• **Impact categories, indicators and characterisation models selection** – Impact categories are a comprehensive set of environmental issues and their selection should be connected to specific environmental issues related to the product system being studied, also considering the specific goal and scope of the study. Each impact category is quantitatively represented by category indicators expressed each one in a specific unit of measurement calculated according to a selected characterisation model. The appropriateness of the characterisation model used to obtain the indicators in the context of the goal and scope of the study should also be described in LCA studies.

• **Classification** – In this step the LCI results are assigned to the different impact categories selected taking into account the effect that the substances have on the environment. Therefore, a cause-effect pathway is used to identify the relationship between the environmental intervention (for instance, the emission of a certain chemical) and its potential effects on the environment.

• **Characterisation** – The LCI results assigned to each impact category are converted to common units using the characterisation factors of the model selected. The quantitative results obtained are aggregated within the same impact category, resulting in a numerical indicator result. The usefulness of the indicator results for a given goal and scope depends on the accuracy, validity and characteristics of the LCI results and the characterisation models.

LCA professionals can choose impact indicators among the two main types: midpoint and endpoint categories, which differ in the stages along the cause-effect chain where they calculate the impact. Midpoint categories reflect the environmental impacts produced at some point between the environmental stressors (origin of the impact) and the final of the cause-effect chain. Examples of midpoint categories are climate change, terrestrial acidification or freshwater ecotoxicity. Endpoint categories reflect the environmental impact produced at the end of the cause-effect chain. There are three main endpoint categories including damage to human health, damage to ecosystems, damage to resources. The required steps to obtain the endpoint indicators from LCI results is summarised in Figure 2.5.

• **Normalisation** – This step estimates the magnitude of the category indicator results relative to some reference information. The aim of the normalisation is to add information about the relative significance of the results. It helps in checking for inconsistencies and adds information on the relative significance of the results.

• **Grouping** – It involves the aggregation of impact categories into one or several sets. In this way, they are listed on specific characteristics or to rank their priority according to a hierarchy.

• **Weighting** – It converts indicator results of different impact categories by using numerical factors by using numerical factors based on value-choices related to priority criteria. It can provide a final single impact score, although it is based on subjective value judgments rather than on scientific criteria. Thus, the data prior to weighting should be available to avoid loss of information.

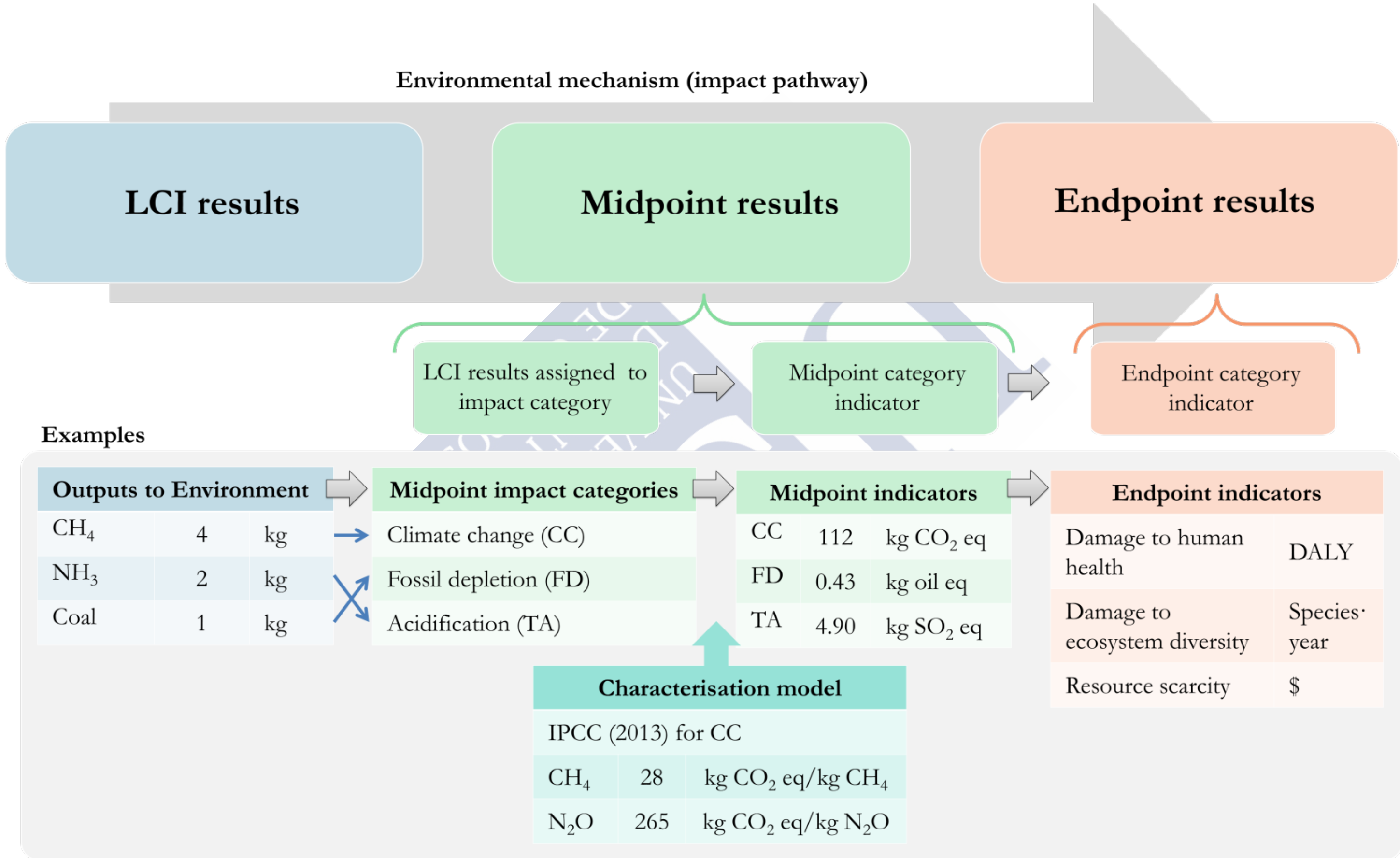

Figure 2.5. Steps from inventory to category endpoints within the LCIA, adapted from ISO 14044 (2006) and Wolf et al. (2010)

Relevant impact categories and characterisation models

In accordance with the ILCD Handbook, the selection of the impact categories and characterisation models should enjoy international acceptance (Hauschild et al., 2011). In addition, the category indicators shall include those that are relevant for the specific study performed according to the goal and scope as well as to the LCI results. The characterisation model for each category indicator shall be scientifically and technically valid. Moreover, the totality of characterisation factors should have no relevant gaps in coverage of the impact category they relate to. ReCiPe is the most recent and harmonized methodology available in LCA. In addition, it includes the characterisation models recommended in ILCD handbook (Hauschild et al., 2011). This methodology includes eighteen midpoint and three endpoint indicators, as presented in Figure 2.6. As it can be seen, marine eutrophication is not included at the endpoint level, not because it does not have an impact on ecosystems, but because it has not been modelled yet (Huijbregts et al., 2016). Moreover, each midpoint indicator is expressed for three different perspectives.

- The individualistic perspective (I) is based on the short-term interest, impact types that are undisputed, and technological optimism with regards to human adaptation.

- The hierarchist perspective (H) is based on scientific consensus with regards to time-frame and plausibility of impact mechanisms.

- The egalitarian perspective (E) is the most precautionary perspective, taking into account the longest time-frame, and all impact pathways for which data is available.

Midpoint		
Impact category	**Acronym**	**Unit of measure**
Climate change	CC	kg CO_2 to air
Ozone depletion	OD	kg CFC-11 to air
Terrestrial acidification	TA	kg SO_2 to air
Freshwater eutrophication	FE	kg P to freshwater
Marine eutrophication	ME	kg N to marine water
Human toxicity	HT	kg 14DCB to urban air
Photochemical oxidant formation	POF	kg NMVOC to urban air
Particulate matter formation	PMF	kg PM10 to air
Terrestrial ecotoxicity	TET	kg 14DCB to soil
Freshwater ecotoxicity	FET	kg 14DCB to freshwater
Marine ecotoxicity	MET	kg 14DCB to marine water
Ionising radiation	IR	kg U235 to air
Agriculturla land occupation	ALO	$m^2 \cdot yr$ agricultural land
Urban land occupation	ULO	$m^2 \cdot yr$ urban land
Natural land transformation	NLT	m^2 natural land
Water depletion	WD	m^3 water
Mineral resource depletion	MD	kg FE
Fossil fuel depletion	FD	kg oil

Endpoint	
Impact category	**Unit of measure**
Human health	DALY
Ecosystems	Species·year
Resources	$

Figure 2.6. Midpoint and endpoint categories of the ReCiPe methodology and interconnections

For the selection of relevant impact categories, initial knowledge based on experience gained from studies about similar systems may help to identify which impact categories have important overall relevance and which may appear as irrelevant for a specific system (Hauschild et al., 2011). Among the available impact categories in ReCiPe methodology, the following ones presented importance in biogas systems:

• **Climate change (CC)** – It is defined as the impact of human GHG emissions on the radiative forcing (i.e. heat radiation absorption) of the atmosphere (Guinée et al., 2002). Greenhouse gases accumulate in the atmosphere and alter Earth's energy balance. Therefore, this environmental indicator takes into account the amount of GHGs released or associated to the biogas system under study with the aim of measuring the potential of climate change. Typically, biogas systems result in emissions of three main GHG that are carbon dioxide, methane and nitrous oxide, they raise from storage of organic waste or digestate, from leakages of biogas, from biogas burnt or from the application of organic and mineral fertilisers. Carbon dioxide has been set up as a reference gas to measure the emissions of different greenhouse gases based on their climate change potential.

• **Ozone depletion (OD)** – Ozone is continuously formed and destroyed by the action of sunlight and chemical reactions in the stratosphere (Goedkoop et al., 2009). Ozone depletion occurs if the rate of ozone destruction is increased due to emissions of anthropogenic substances which persist in the atmosphere. Stratospheric ozone is vital for life because it hinders from harmful solar ultraviolet B (UV-B) radiation. If not absorbed, UV-B radiation below 300 nm will reach the troposphere and the earth surface. This radiation can have harmful effects on human health, animal health, terrestrial and aquatic ecosystems, biochemical cycles and on materials. Within biogas production, there are no significant direct emissions of ozone depleting substances, but they are emitted in background processes such as infrastructure, electricity, chemicals and fossil fuels production. This environmental indicator uses trichlorofluoromethane (CFC-11) as the reference gas to which all emissions are related.

• **Terrestrial acidification (TA)** – Atmospheric deposition of acidifying inorganic substances can cause a change in acidity in the soil. For almost all plant species, there is a clearly defined optimum of acidity and a serious deviation from this optimum is harmful to that specific kind of species (Goedkoop et al., 2009). Acidifying pollutants have a wide variety of impacts on soil, groundwater, surface waters, biological organisms, ecosystems and materials. The major acidifying pollutant related to biogas production is ammonia, since it is produced as a by-product of the microbial decomposition of the organic nitrogen compounds in organic substrates. In addition, ammonia emissions derived also from the application of organic and mineral fertilisers. Moreover, nitrogen oxides and sulphur dioxide are important contributing substances. This environmental indicator is measured in terms of mass of sulphur dioxide equivalent emitted. Herein, this environmental indicator measures the emissions of acidifying pollutants to the atmosphere that arise from the livestock waste management system.

• **Freshwater and marine eutrophication (FE and ME)** – Repeated over-applications of organic and mineral fertilisers to soil, above crop requirements, have led to the accumulation of macro-nutrients, such as nitrogen, phosphorus and potassium. At saturation, nutrients are lost to either surface or ground waters. Nutrient enrichment may cause an undesirable shift in species composition and elevated biomass production in aquatic ecosystems. In addition, high nutrient concentrations may also render surface waters unacceptable as a source of drinking water. In aquatic ecosystems, increased biomass production may lead to depressed oxygen levels. Biomass growth in different aquatic ecosystems may be limited by different nutrients. Freshwaters are typically limited by phosphorus, whereas nitrogen usually is the limiting nutrient of biomass yield in marine waters (Goedkoop et al., 2009). Therefore, marine and inland waters are treated as two different environmental indicators of aquatic eutrophication: i) freshwater eutrophication is measured in mass of phosphorus equivalent and the main emissions which affect this environmental indicator are phosphate, phosphoric acid and phosphorus and ii) marine eutrophication is measured in mass of nitrogen equivalent and the most important related emissions are ammonia, total nitrogen, nitrate, nitrite and nitrogen oxides. In a typical biogas system, these impacts derived from the application of the produced digestate as a fertiliser.

• **Photochemical oxidant formation (POF)** – This environmental indicator comprises the formation of reactive chemical compounds such as ozone by the action of sunlight on certain primary air pollutants, mainly nitrogen oxides and non-methane volatile organic compounds. Photo-oxidants may be formed in the troposphere under the influence of ultraviolet light through photochemical oxidation of non-methane volatile organic compounds (NMVOC) and carbon monoxide in the presence of nitrogen oxides (Guinée et al., 2002). Ozone is considered the most important of these oxidising compounds, along with peroxyacetylnitrate. Within biogas systems, there are direct emissions of substances which affect the formation of photochemical oxidants such as methane and nitrogen oxides. In addition, there are other indirect emissions of these substances in background processes such as infrastructure, electricity, chemicals and fossil fuels production. This environmental indicator is measured in terms of NMVOC equivalent.

• **Fossil depletion (FD)** – This environmental indicator measures the consumption of fossil fuels along the livestock waste management. This indicator refers to a group of non-renewable resources that contain hydrocarbons. The group ranges from volatile materials (like methane), crude oil and non-volatile materials (like coal) (Goedkoop et al., 2009). It is directly linked with the degree of mechanization of the system under study. This indicator is measured in terms of mass of oil equivalent, based on the energy content of different fossil fuels.

LCIA methodologies are a collection of individual characterisation models for a specific set of impact categories. The use of several different LCIA methods makes it difficult to compare LCA results and interpret them. To some extent, the presence of different LCIA methodologies respond to the request of representing different environmental approaches that may be of interest in certain applications; but in any case a default or baseline method is needed (Hauschild et al., 2011). Regarding standardisation, ISO 14044 addresses this issue in general terms and most existing LCIA methodologies can be applied.

2.4. Life Cycle Assessment + Data Envelopment Analysis

Eco-efficiency analysis has emerged as a valuable tool towards the target of sustainable development since it connects business and environmental goals, engaging companies in the agenda of sustainable development (Syrrakou et al., 2006). This approach provides a practical tool for economic prosperity involving more efficient use of resources and lower emissions (NRTEE, 2001). In this matter, the World Business Council for Sustainable Development defines eco-efficiency as "the attainment of delivering competitively priced goods and services that satisfy human needs and bring quality of life, while reducing environmental impacts and resource intensity throughout the life cycle" (NRTEE, 2001).

In this context, DEA also arose as a methodology able to measure the relative efficiency of multiple homogeneous entities when the productive process shows a structure composed of multiple inputs and outputs. DEA is a linear programming methodology used to quantify in an empirical manner the comparative productive efficiency of multiple similar entities named Decision Making Units (DMU) (Cooper et al., 2007). A DMU is each one of the homogenous units whose input/output conversion undergoes assessment. Given a certain number of inputs and outputs, DEA identifies the efficient DMUs within a certain sample and computes an efficiency score and target efficient values for those DMUs considered inefficient. In order to evaluate eco-efficiency, the LCA methodology can be used to determine the indicators from an environmental perspective, including the use of primary energy use, raw materials utilisation and emissions to the environment (Syrrakou et al., 2006). In this sense, the combination of LCA and DEA can provide quantitative life-cycle-based benchmarks that orientate the performance towards environmental sustainability (Vázquez-Rowe and Iribarren, 2015). The innovative nature of LCA+DEA methodology may develop into a challenging identification of potential uses. Even though LCA+DEA studies to date have mainly assessed agrifood systems, the LCA+DEA methodology can be applied to any sector. For instance, Iribarren et al. (2013) have recently carried out the LCA + DEA study of wind farms, showing that this methodology can be useful for the benchmarking of energy conversion systems.

To date, LCA+DEA methodology has been used applying two different methods. One the one hand, "five step LCA+DEA method" is the recommended approach to undertake eco-efficiency verification analyses through the quantification of the environmental consequences of operational inefficiencies. On the other hand, the "three step LCA+DEA method" is a preliminary approach directed toward the estimation of environmental impact efficiency and the simultaneous benchmarking of operational and environmental parameters.

• **The five step LCA+DEA method** – Vazquez-Rowe et al. (2010) established the five steps required for the combined operational and environmental assessment of multiple similar units. These steps are:

- Development of the LCI for each DMUs. This stage involves input and output data collection for the assessed systems.

- Performance of the LCIA for every DMU. This second step involves the characterisation of the environmental profile of the current DMUs from the LCI developed in the first step.

- Determination of the operational efficiency for each DMU by running the matrix that includes relevant input and output data with the selected DEA model. In this way, target DMUs are calculated by applying the reduction targets proposed by the methodology. Target DMUs refer to virtual units that consume less input and/or produce more output. Thus, operational benchmarking is attained.

- LCIA of the target DMUs from the new LCI data arising from the third step. Consequently, the potential environmental impacts associated with the virtual DMUs are determined.

- Quantification of the environmental consequences of operational inefficiencies (eco-efficiency verification). The comparison between the potential environmental impacts for the virtual DMUs and those for the current DMUs quantifies the environmental impacts generated by inadequate operational practices.

• **The three step LCA+DEA method** – The first two steps of this method are coincident with those for the five-step method. However, the third phase comprises a DEA matrix with a higher number of inputs due to the inclusion of the potential environmental impacts determined in the second step as additional inputs. In this sense, the benchmarking results directly estimate targets for both LCI inputs/outputs and the potential environmental impacts. Therefore, unlike the five-step method, this option avoids the environmental characterisation of the target DMUs by implementing environmental impacts as inputs when performing DEA in the third step.

A range of different models is available to run the matrix resulting from the selection of relevant inputs and outputs of the DMUs under study, including variants of the Charnes-Cooper-Rhodes (CCR), the Banker-Charnes-Cooper (BCC) and the assurance region (AR) models. However, the most common method for analysing the eco-efficiency of systems using the LCA+DEA method is the slacks-based measure of efficiency (SBM), a number of examples are available in literature (Avadí et al., 2014; Iribarren et al., 2015, 2013, 2010; Lorenzo-Toja et al., 2015; Vázquez-Rowe et al., 2012). The SBM model is elastic regarding the calculation of the inefficiencies for the different DMUs since it performs the computation regardless the units of measure used for the different inputs and outputs. In a similar way, unlike the CCR and BCC models, the SBM model considers non-radial characteristics of inputs and outputs, which makes it more appropriate for monitoring inputs with vague interconnections. In addition, the SBM model accounts for all inefficiencies, whereas other models such as the CCR only take into consideration purely technical efficiency. Finally, the SBM model provides a series of target values for the minimised input and/or output that deliver appropriate benchmarks to calculate the target theoretical environmental profile of inefficient DMUs (Lorenzo-Toja et al., 2015). With this regard, the input-oriented approach allows to minimise the use of resources and, therefore, an optimisation of operational inputs, while maintaining the number of outputs; on the contrary, the output-oriented approach is focused on the maximisation of outputs (Cooper et al., 2007). Finally, this method also can be performed according to constant or variable returns-to-scale; the former limits the effect of different scales within the eco-efficiency results; although the latter is more suitable for DMUs of the same scale (Cooper et al., 2007).

2.5. Analytical Hierarchy Process

The AHP methodology is a robust and flexible multi-criteria decision-making tool for the hierarchical decomposition of complex problems (Chatzimouratidis and Pilavachi, 2009). The AHP is designed to structure a decision process in a scenario affected by multiple independent factors (Bottero et al., 2011). The AHP hierarchical structure allows decision makers to easily address problems in terms of relevant criteria and sub-criteria. In the analysis, a complex problem can be divided into several sub-problems that are organised according to hierarchical levels, where each level denotes a set of criteria or attributes related to each sub-problem. Therefore, the analysis is based on three fundamental principles: i) breaking down the problem; ii) pairwise comparison of the various alternatives; iii) synthesis of the preferences (Bottero et al., 2011). The model for the assessment of sustainability though the AHP includes several methodological steps as shown in Figure 2.7.

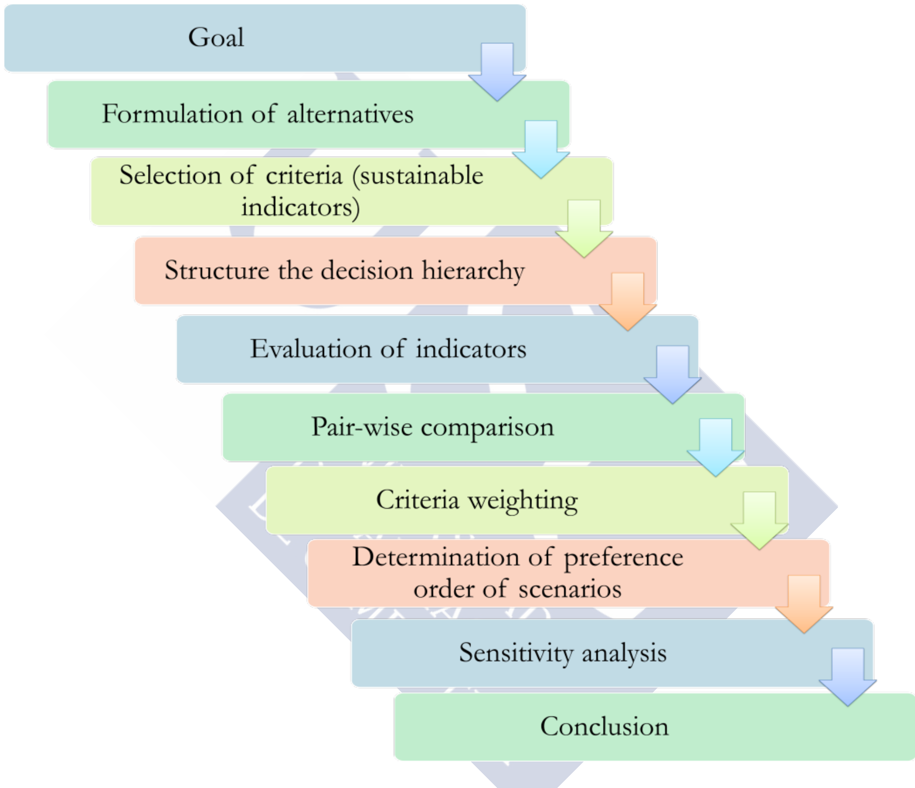

Figure 2.7. Different steps defined within the AHP process

The decision procedure using the AHP is made up of four main steps (Saaty, 2008):

- To define the problem and the goal of the study.

- To identify the criteria to be considered for the decision-making, which are the sustainable indicators including environmental, economic and social aspects. When the criteria are identified, the set of specific indicators should be selected;

this is done considering which of the indicators better translates a comprehensive and meaningful assessment of sustainability for each specific case, according to the objective of the study. Next, the decision hierarchy done is from the top with the goal of the decision to the bottom with the possible alternatives, going through the criteria and sub-criteria that would be used to determine which alternative fits best the goal of the decision, as presented in Figure 2.8.

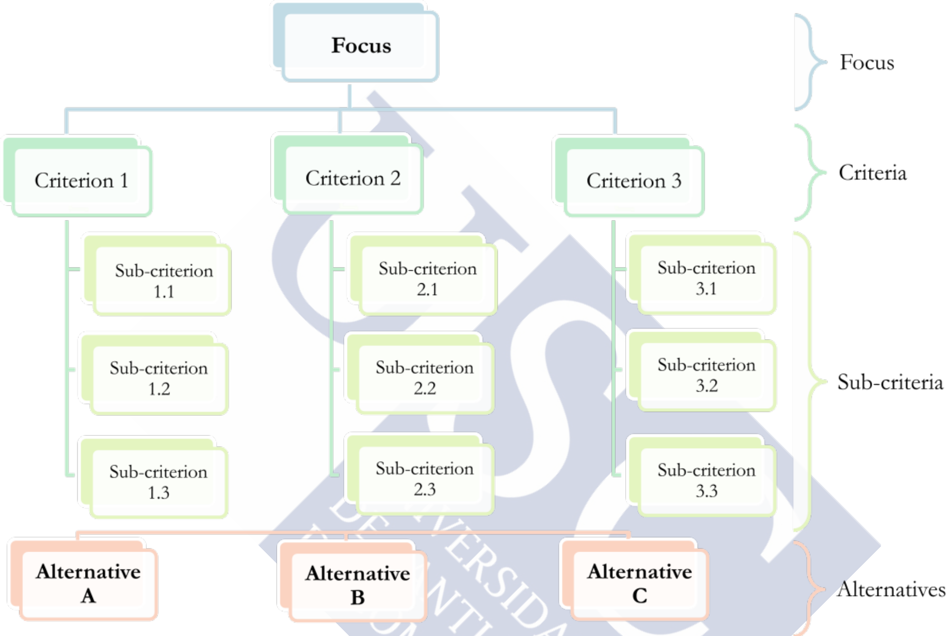

Figure 2.8. Structure of the decision hierarchy

- To build the pair-wise comparison matrix. The weighing is done after consultation with experts working in the field of study, who should rank the importance of the criteria with respect to the goal. Each element of the matrix in the upper level should be used to compare elements in the level immediately below. This step is performed to determine the relative importance of each alternative in terms of each criterion. In order to make the pair-wise comparisons, a scale indicating how many times more important preferred or dominant one element is with respect to the parent element is required. The measurement scale is provided by the method and goes from 1 to 9; meaning 1 (equal importance) that two criteria contribute equally to the objective and 9 (extreme importance) that the experience and judgment favour one criterion over another to the greatest extent possible.

- To use the priorities obtained from the comparisons to weigh the priorities in the level immediately below. When the weighted values are added, the overall global priority is obtained and the preferred alternative is identified.

2.6. Literature review

Focusing on the specific topic of biogas production, LCA is a useful tool for decision-makers involved in the evaluation of new biogas projects such as governments. It can also enhance the overall environmental performance and boosting eco-efficiency related to the operation of the biogas production chain (Huttunen et al., 2014). As a result, several LCA studies have been published in the last years. According to Bacenetti et al. (2016), around 105 studies have been identified performing the environmental assessment of anaerobic digestion processes through the LCA methodology. Even though environmental policies encourage biogas production and utilisation in order to assure GHG savings, these LCA studies showed that environmental impacts and benefits of existing biogas plants are influenced by many factors including existing practices for biogas production, the renewable energy policies and local contexts. In this sense, Whiting and Azapagic (2014) demonstrated that that biogas production could lead to significant reductions in most impacts compared to fossil-fuel alternatives, including climate change, which can be reduced by up to 50%. However, they also identified increased environmental impacts in acidification and eutrophication potentials in comparison with electricity produced from natural gas. Nevertheless, the results obtained in different LCA studies of biogas production are highly dependent on methodological choices about the system boundaries, assumptions made in the calculations, characterisation models and the impact categories selected. As general conclusions, these studies have reported that the environmental impact and biogas yield considerably depend on factors such as the selection of raw materials for digestion, the final use of the biogas and the management of the resulting digestate (Boulamanti et al., 2013).

• **Feedstock selection** – The options usually considered for biogas production comprise agricultural residues including animal waste manure and crop residues, specifically cultivated energy crops, SSFW and different industrial organic waste. In addition, sewage sludge is commonly managed through anaerobic digestion in the facilities of WWTPs. The composition of the feedstock used has an impact on the potential methane yield; for example, high fat content renders in biogas with high content in methane, while feedstock with high water content such as animal manure typically has a low methane production (Bacenetti et al., 2013). Moreover, the composition of a specific raw material can vary noticeably between sites and time, not only regarding feedstock such as food waste, but also among different crop species (Dressler et al., 2012; González-García et al., 2013). Besides the variation in methane production potential, the different raw materials considered vary significantly in harmful environmental impacts related to their production (De Vries et al., 2012; Huttunen et al., 2014).

Embracing all these issues, Poeschl et al. (2012a, 2012b) performed a complete LCA study with the aim of comparing different biogas systems to establish the basis to improve the environmental sustainability of biogas production. Among other factors, different scenarios were analysed by considering different feedstock, mono- and co-digestion schemes. With regard to single feedstock digestion, they highlighted the high energy density of straw and maize silage; while results in co-digestion pointed out that environmental sustainability is achievable with mixtures that includes high proportion of agricultural residues and food waste, providing a basis for feedstock flexibility. Several studies have proved that the cultivation of dedicated energy crops can increase climate change, eutrophication and acidification due to the agricultural machine and fertilisers required (Bacenetti et al., 2016). Moreover, their cultivation involve a significant amount of land occupation for, which can lead to direct and indirect emissions related to land use change (Boulamanti et al., 2013). In particular, biogas plants in countries where public subsidies are granted for electricity and/or biomethane production from biogas (e.g. Germany, Italy and UK) are fed mainly with energy crops (Bacenetti et al., 2016; Huttunen et al., 2014). Among them, the most used energy crops is maize silage due to its high specific biogas production, storability and biomass yield (Poeschl et al., 2012a, 2012b). However, several studies highlighted that biogas plants fed with animal waste achieve better environmental performance compared to the ones fed with energy crops, mainly due to the fact that no environmental loads are associated with their production, but credited for the avoided impacts of their traditional management. In this context, transport of feedstock with low biogas potential may be a key factor (Bacenetti et al., 2015). Transport may be an important factor for other substrates if enough long transport distances are encountered, which is especially important in the collection of SSFW in areas with low population density, discouraging centralised treatment facilities.

• **Biogas use** – The produced biogas can be used for different bioenergy purposes, including heat production, co-generation of heat and electricity and biomethane production that can be used as a substitute for natural gas or vehicle fuel. Excess electricity is usually sold to the electricity grid, while extra heat can be used for heating; however, it is often underutilised, especially during the summer months. Sometimes, the difficulties of finding profitable use for all of the biogas results in situations where, from the biogas producer's perspective, it is more feasible to burn the biogas in a torch. Regarding biogas use, Poeschl et al. (2012a, 2012b) concluded that the most viable pathway for sustainable biogas use was co-generation and tri-generation, due to enhanced combined electricity and thermal conversion efficiency.

• **Digestate management** – The management of the produced digestate is often identified as an important source of environmental impacts. Indeed, Boulamanti et al. (2013) identified digestate management as the second most important factor affecting environmental sustainability of biogas systems, just after feedstock selection. In general, biogas plants aim at utilising the digestate as an organic fertiliser in agriculture. With this regard, Fantin et al. (2015) pointed out that the choice of methods for calculation of emissions from digestate storage and application is a critical point for LCA practitioners, significantly affecting the environmental results in impact categories related to acidification and eutrophication. Moreover, the increasing number of biogas plants results in larger transportation distances for the spread of digestate on land to avoid oversupply of nutrients. The transport of digestate may cause environmental impacts since it is composed of 95% of water in average. To overcome this problem, Rehl and Müller (2011) performed a LCA study to compare the environmental impacts and the energy efficiency of seven treatment options of biogas digestate. The treatment options included i) the conventional management of digestate (storage and application on agricultural land), ii) composting (stabilisation), iii) belt dryer, iv) rum dryer, v) solar dryer, vi) thermal vaporisation (concentration), vii) separation, ultra-filtration and reverse osmosis and ionic exchanger. According to the results of the study, belt drying was identified as the most polluting option in terms of climate change and acidification, while solar drying of separated digestate was the best option. In this study, digestate treatment options were investigated exclusively from an environmental point of view. However, not only ecological aspects are relevant for decision making but also socio-political aspects, economic aspects, legal aspects, regional aspects such as biogas plants density as well as technical aspects. The predominant criterion for choosing a treatment option is the profitability, which arise from a reduction of the transportation costs, higher revenues due to an increase of the product value and an extension of the market by novel fertiliser products. With the same purpose, Vázquez-Rowe et al. (2015) compared the environmental impacts of spreading digestate directly and four different treatment technologies, such as i) drying and pelletizing, ii) composting, iii) biological treatment, reverse osmosis and drying and iv) ammonia stripping and drying. In the results, the authors identified relevant environmental gains when the digestate is treated using the examined conversion technologies prior to spreading, although important trade-offs between impact categories were observed and discussed. Finally, the results obtained by Poeschl et al. (2012a, 2012b) indicated that the recovery of residual biogas from the digestate storage was a significant factor of the performance of digestate processing and handling due to the reduced biogas loss to the atmosphere and the replacement of fossil fuel used in energy generation.

2.7. Objectives and structure of the thesis

The objective of this doctoral thesis is to quantify the environmental benefits and drawbacks related to the production of bioenergy through the anaerobic digestion process, according to a life cycle perspective. The document is structured in four sections, each of them sub-divided into different chapters, as shown in Figure 2.9.

• **Section I: Introduction** – This section aims at contextualising the thesis providing general information regarding biogas production and the available tools for the assessment of its environmental sustainability. In more detail, Chapter 1 focuses on the relation between energy production and climate change, renewable energy production in Europe and biogas production as an energy option or as a waste valorisation opportunity. Finally, Chapter 2 explores the available methodologies for the assessment of biogas systems in terms of sustainability, paying special attention to the methodologies of LCA, LCA+DEA and AHP.

• **Section II: Agricultural biogas** – This section analyses the environmental implications of bioenergy production through the anaerobic digestion of biomass available in an agricultural context. Therefore, Chapter 3 assesses the environmental sustainability of four different full-scale biogas plants operating in Italy and using different feedstock including energy crops and animal manure. In line with this, Chapter 4 delves into the environmental implications of feedstock selection in biogas systems, including not only the environmental burdens of its production but also the derived biogas and digestate production, in terms of their amount and quality by analysing two more biogas plants. In Chapter 5, the eco-efficiency of 15 real biogas plants is analysed by applying the LCA+DEA approach. Finally, an innovative technology proposed for the management of livestock waste in Cyprus is analysed in Chapter 6, especially in comparison with the conventional management options in the area of study. In addition, the AHP methodology is applied to analyse the three dimensions of sustainability and to select the best waste management practice.

• **Section III: Sewage biogas** – This section analyses the environmental consequences of anaerobic digestion as a waste valorisation option in the context of wastewater treatment. In the first place, Chapter 7 analyses the potentially benefits of the co-management of sewage sludge and food waste in the United Kingdom, since co-digestion of these waste streams are prevented by the environmental policies. Finally, Chapter 8 is focused on the assessment of an innovative treatment scheme for the combined management of sewage and food waste at decentralised level by proposing and analysing several alternative schemes to identify the most suitable from a technical and environmental point of view.

• **Section IV: Conclusions** – The last section summarises the findings found, including the main conclusions and recommendations in Chapter 9.

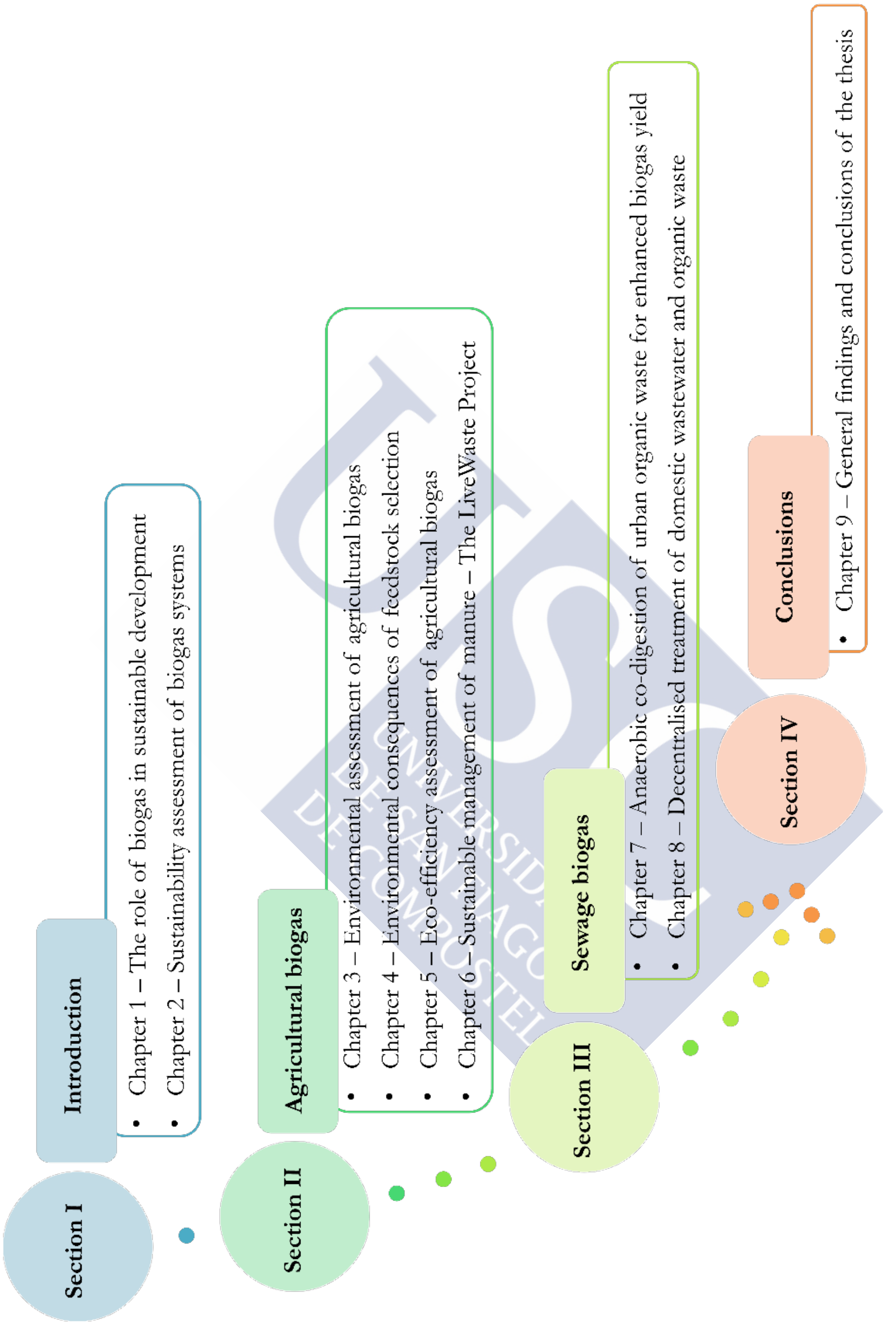

Figure 2.9. Outline of the doctoral thesis

2.8. List of acronyms

AHP	Analytical hierarchy process
AR	Assurance region
BCC	Banker-Charnes-Cooper
CC	Climate change
CCR	Charnes-Cooper-Rhodes
CFC	Chlorofluorocarbons
DEA	Data envelopment analysis
DMU	Decision making unit
FD	Fossil depletion
FE	Freshwater eutrophication
FU	Functional unit
GHG	Greenhouse gas
ILCD	International Reference Life Cycle Data
ISO	International Organisation for Standardisation
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
MCA	Multi-criteria analysis
ME	Marine eutrophication
NMVOC	Non- methane volatile organic compounds
OD	Ozone depletion
POF	Photochemical oxidant formation
SBM	Slacks-based measure of efficiency
SETAC	Society of Environmental Toxicology and Chemistry
SSFW	Source segregated food waste
TA	Terrestrial acidification
WCED	World Commission of Environment and Development
WWTP	Wastewater treatment plant

2.9. References

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Section II:

Agricultural biogas

Chapter 3: Environmental assessment of agricultural biogas

Summary

Through substantial incentives, biogas has become an important source of renewable energy in Europe. Specifically, agricultural biogas can play a significant role in addressing international targets regarding climate change and security of energy supply. However, the adoption of anaerobic digestion may not necessarily lead to sustainable practices, especially when energy crops are widely used as a substrate for biogas production.

The environmental sustainability of electricity production from agricultural biogas was evaluated in Chapter 3. With this purpose, four different full-scale biogas plants operating in Northern Italy managing animal waste (pig slurry) and energy crops (maize and triticale) as substrates following mono- or co-digestion schemes were analysed. Environmental results identified the cultivation of energy crops and the management of digestate as the *hotspots,* producing between 37% and 55% of the greenhouse gas (GHG) emissions (63-68 kg CO_2 eq/t straw). The main responsible factors of these results were the consumption of diesel in the agricultural activities regarding energy related impact categories. Moreover, emissions of ammonia, nitrate and phosphate to air and water derived from the application of digestate and other mineral fertilisers were the primary source of impacts, entailing between 77% and 97% of the impacts in eutrophication and acidification categories. In a sensitivity analysis, actions for the improvement of the environmental profile of the biogas plants were evaluated. The results obtained in this chapter were compared with other studies in the literature.

Outline of Chapter 3

3.1. Introduction to agricultural biogas

According to the Biogas Barometer, biogas can be classified according to the conditions or circumstances where it is produced, including landfill biogas, sewage biogas and “other biogas” that includes multiple feedstocks such as energy crops, agricultural waste, animal manure, green waste, SSFW and food and industrial waste (EurObserv’ER, 2014). The agricultural biomass for anaerobic digestion represents a significant source of biogas, as shown by the two most important European biogas producer countries: Germany and Italy. In 2013, around 13,379 thousand tonnes of oil equivalent (ktoe) of biogas were produced in the European Union, of which 50% was produced in Germany and 16% in Italy. It is estimated that around 82% of the biogas plants in these two countries are of agricultural type, performing the digestion of energy crops, agricultural waste and animal manure (Jacobs et al., 2016). The environmental sustainability of bioenergy systems is usually analysed in terms of GHG savings (Vázquez-Rowe and Iribarren, 2015), calculated as the difference of the carbon footprint between the assessed biogas system and the conventional fossil equivalent (European Parliament, 2009). However, other main environmental concerns arise from this kind of energy systems, including eutrophication and acidification of aquatic and terrestrial ecosystems. In this sense, LCA allows covering a wider range of environmental impacts (Vázquez-Rowe and Iribarren, 2015).

The objective of Chapter 3 was to analyse, from a life cycle perspective, the environmental profile of four real Italian biogas plants, considered representative of agricultural biogas production in Europe. These plants perform the anaerobic digestion of animal manure and energy crops in different ratios according to mono-digestion or co-digestion schemes. Specific objectives include the identification of the most relevant elements that contribute to the environmental impact of the four biogas plants. Moreover, differences in the environmental performance of the four plants were highlighted to assess the sustainability of biogas production.

3.2. Goal and scope definition

As aforementioned, four real biogas plants located in the Po Valley (Northern Italy) and considered representative of the state-of-the-art of agricultural biogas production, were evaluated from a cradle-to-gate perspective (ISO 14040, 2006). All energy and material inputs and outputs as well as emissions associated were quantified in detail. Moreover, the most critical stages from an environmental point of view, named as *hotspots*, were identified and alternatives for some processes included were proposed to improve the environmental profiles. The biogas plants under study perform the anaerobic digestion of animal manure (i.e. pig slurry) and energy crops (i.e. maize and triticale silages) as the only substrates or following a co-digestion scheme. The primary characteristics of the four biogas plants are shown in Table 3.1. The interest of biogas plants using pig slurry as substrate is the wide availability of this substrate in the area of study since Northern Italy is one of the most important European regions for livestock production (Eurostat, 2010; Holm-Nielsen et al., 2009). On the other hand, maize and triticale are the most common energy crops used for biogas production in Italy due to its high yield of dry matter per hectare and high potential of methane production (De Vries et al., 2012b; Dressler et al., 2012).

• **Plant 1** – The first plant under study is a small biogas plant located in Lodi with an electrical power of 250 kW. This small plant digests on average 76,650 t/year of pig slurry as single substrate, which produces a yearly average of 1,025,650 m^3 of biogas, accounting for an electricity production of 2,200 MWh/year. The produced digestate is fully applied as an organic fertiliser in a surrounding agricultural system with no further treatment than its storage in an open tank.

• **Plant 2** – The second plant is located in San Giorgio di Lomellina and it has higher capacity than Plant 1, with an electrical power of 500 kW. It follows a co-digestion scheme using animal manure and energy crops. In more detail, the average inputs are 4,750 t/year of triticale silage, 7,000 t/year of maize silage and 10,200 t/year of pig slurry. As a result, 2,205,700 m^3 of biogas are generated every year, equivalent to the production of 4,400 MWh of electricity. Regarding the digestate, 60% is recirculated into the digester to decrease the solids content while the remaining 40% can be used for the cultivation of maize, triticale as well as in other agricultural systems.

• **Plant 3** – The third plant, located in Vercelli, is the largest plant in terms of electrical power (999 kW). It also performs the co-digestion of pig slurry and maize silage (19,700 and 16,500 t/year, respectively). It produces 8,100 MWh of electricity each year from 4,174,140 m^3 of biogas. In this plant, digestate is separated into liquid and solid fractions. Part of the liquid fraction is also recirculated to the digester (68%) while the remaining is stored with the solid fraction and applied in the cultivation of the maize digested in the plant. In this case, more digestate than the produced is required for the cultivation of maize; therefore, extra digestate is supplied from a nearby biogas plant.

• **Plant 4** – The last plant has an electrical power capacity of 520 kW and is located in Pavía. Maize silage is the only substrate used for anaerobic digestion (12,100 t/year). As a result, 4,150 kWh of electricity are cogenerated each year from 2,372,500 m^3 of biogas. The liquid fraction produced after the solid/liquid separation is entirely recirculated to the digester, while the solid part is stored and applied as fertiliser for the maize used in the plant. As in Plant 3, more digestate than produced is required and it comes from a nearby biogas plant.

Table 3.1. Main parameters of the biogas plants under study

			Plant 1	**Plant 2**	**Plant 3**	**Plant 4**
District			Lodi	San Giorgio di Lomellina	Vercelli	Pavía
AD	Feedstock		Pig slurry	Pig slurry, maize and triticale	Pig slurry and maize	Maize
	Digester		1	1	1	1
	Temperature	(°C)	40	40	40	40
	HRT	(days)	30-40	35-45	35-50	25-35
	OLR	(kg TVS/m^3·d)	0.74	2.07	1.60	3.34
CHP	Electrical power	(kW)	250	999	500	520
	Electrical efficiency	(%)	35.7	40.7	38.5	37.0
	Thermal efficiency	(%)	51.0	44.0	48.0	47.1

AD – anaerobic digestion; CHP – cogeneration heat and power; HRT – hydraulic retention time; OLR – organic loading rate; TVS – total volatile solids

3.2.1. Function and functional unit

As discussed in Chapter 2, in some cases in agricultural biogas production, especially for these plants digesting only organic residues, the function of the system could be considered "the treatment of waste". However, in this case "the production of electricity" was considered as the function in common for all the plants under study due to the economic incentives and the outstanding use of energy crops. Therefore, the function of the systems under study was the supply of electricity to the medium voltage Italian national grid. Consequently, the FU selected was the provision of 1 MWh of electricity to the Italian electricity grid from biogas valorisation.

3.2.2. Description of the system boundaries

The environmental study of each plant included the production and supply of substrates used in the anaerobic digestion process, the biogas use for the generation of electricity and heat as well as the management of the produced digestate. Figure 3.1 presents a general flowchart of the main unit processes considered within the four biogas systems under study. For all biogas plants, the systems under study were divided into five main subsystems: SS1: feedstock production; SS2: feedstock supply; SS3: bioenergy production, SS4: digestate management and SS5: surplus digestate use.

As previously discussed in Chapter 2, the consideration of emissions of biogenic carbon is a controversial issue in LCA studies. In this study, carbon dioxide emissions from biogenic sources were considered as carbon neutral as commonly performed in LCA studies of biogas systems; therefore, no environmental impacts were allocated to the emissions of biogenic carbon dioxide.

SS1: Feedstock production

As shown in Figure 3.1, pig slurry was excluded from the system boundaries because it was considered the main waste stream of pig breeding farms and its production is unaffected by its valorisation in anaerobic digestion (see Chapter 2). On the contrary, the cultivation of energy crops was included in the study since cereals used in these plants are exclusively cultivated for bioenergy purposes.

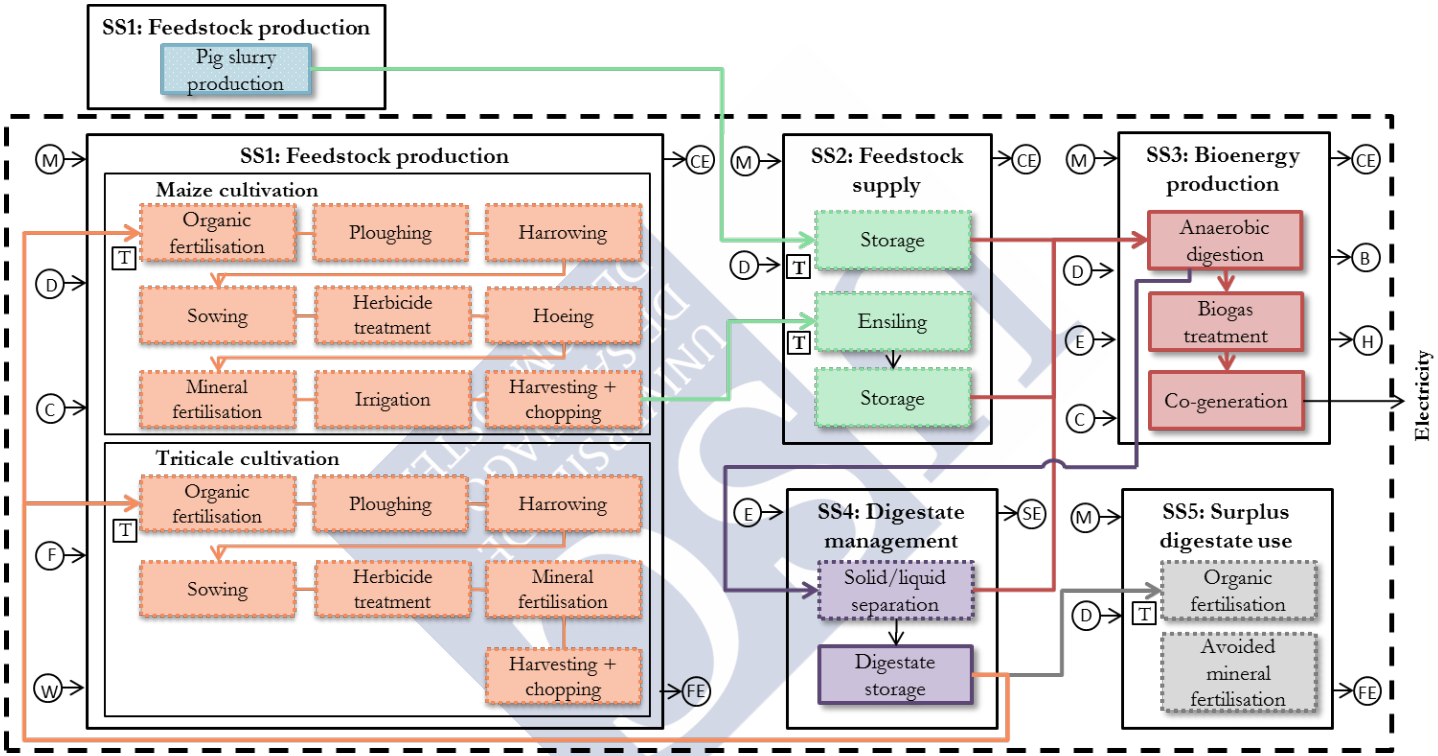

Figure 3.1. Process chain and system boundaries of the biogas plants under study. Dotted boxes indicate processes not performed in all biogas plants. Acronyms: T – transport; M – machinery; D – diesel; C – chemicals; F – fertilisers; W – water; E – electricity; B – biogas losses, H – heat dissipated; CE – combustion emissions; FE – fertilisation emissions; SE – storage emissions

Therefore, this subsystem included all the agricultural field operations involved in the maize and triticale cultivation that consists in a double-crop system where triticale is firstly grown during winter (from September to May), and maize is cultivated in the same agricultural land in summer (from May to September) (maize class 500). However, regarding Plants 3 and 4, the agricultural scheme is a single crop system, where only maize is cultivated during summer (maize class 700). Field activities performed in both crops are ploughing, harrowing, fertilising, pesticides application, sowing (shown in Figure 3.2), harvesting and chopping. Contrarily to triticale, maize cultivation also requires irrigation and hoeing operations. As abovementioned, the digestate used in the organic fertilisation comes from the same biogas plant where the energy crops are going to be used. Moreover, in the case of Plants 3 and 4, additional digestate from another biogas plant is required to fulfil the fertilisation requirements.

Figure 3.2. Sowing performed in the cultivation of energy crops

Table 3.2 displays the main agricultural operations performed during triticale and maize cultivation. More detailed information concerning the cultivation of these crops can be found in published literature (Bacenetti et al., 2014; González-García et al., 2013).

Table 3.2. Field operations per hectare of (a) triticale and (b) maize cultivation

(a)

Operation	Time	Tractor		Inputs/outputs
		Work capacity (ha/h)	Diesel (kg/ha)	
Organic fertilisation	September	2.5	4.71	Digestate 160 kg N/ha
Ploughing	September	0.8	22.64	-
Harrowing	September	0.5	24.24	-
Sowing	October	0.7	8.53	350 seeds/m^2
Herbicide control	October	3.0	3.32	Terbuthylazine + Alachlor: 5 kg/ha
Mineral fertilisation	November	3.0	3.17	Ammonium nitrate: 60 kg/ha
Mineral fertilisation	February	3.0	3.17	Urea: 60 kg/ha
Harvesting	May	2.5	27.17	37 t triticale/ha

(b)

Operation	Time	Tractor		Inputs/outputs
		Work capacity (ha/h)	Diesel (kg/ha)	
Organic fertilisation	May	0.3	27.2	Digestate 340 kg N/ha[a] 180 kg N/ha[b]
Ploughing	May	0.9	22.6	-
Harrowing	May	0.8	23.7	-
Sowing	May	1.0	13.4	20 kg/ha
Herbicide control	June	3.0	3.5	Lumax 5 kg/ha
Mineral fertilisation	June	8.0	6.1	Urea 60 kg
Irrigation (x5)	July-August	0.8	22.4	3,600 m^3/ha
Harvesting	September	1.0	76.0	75 t maize/ha[a] 48.8 t maize/ha[b]

[a] Single-crop system (maize class 700); [b] double-crop system (maize class 500)

Therefore, the subsystem boundaries included all inputs such as the production of agricultural machinery (tractors and implements), mineral fertilisers (urea, potassium- and phosphorus-based fertilisers for maize and urea and ammonium nitrate for triticale), herbicides (Lumax and S-metolachlor in case of maize and Terbuthylazine and Alachlor for triticale), seeds and fossil fuel for the operation of the agricultural machinery. Outputs such as emissions derived from fuel use and fertilisers application (i.e. ammonia, nitrous oxide, nitrogen gas, nitrate and phosphate) were also taken into account. However, emissions from carbon stock change linked to land use (positive or negative) were not taken into account because cereals have been cultivated in this area for more than 20 years (Boulamanti et al., 2013).

SS2: Feedstock supply

In this subsystem, the environmental burdens considered involved all inputs and outputs required for the delivery of the biomass up to the gate of the bioenergy plant and the ensiling operations of the maize and triticale straw. The average transport distances of each plant are summarised in Table 3.3. The supply of pig slurry comprised both the collection of pig slurry in the breeding farm and its delivery up to the biogas plant using a tractor. Pig slurry is collected and delivered on a daily basis; therefore, its storage inside the plant does not exceed more than three days. Concerning maize, chopped maize is transported to the biogas plant by lorry and it is subsequently ensiled and stored. It was considered that 10% of the mass of the energy crops ensiled is lost during storage.

Table 3.3. Average transport distances in the delivery of the feedstock

	Plant 1	Plant 2	Plant 3	Plant 4
Pig slurry	4 km	50% 2.5 km 50% pumped	2 km	-
Maize straw	-	1.5 km	5 km	3 km
Triticale straw	-	1.5 km	-	-

In both systems, the production of the machinery and diesel fuel required for the operations were taken into account within the boundaries of the systems as well as combustion emissions derived from diesel fuel use.

SS3: Bioenergy production

This subsystem encompassed all inputs and outputs required for the production of biogas by anaerobic digestion (such as the loading of feedstock to the digester, the anaerobic digestion process itself and the biogas treatment) as well as its conversion into bioenergy in a CHP engine.

Pig slurry is loaded by a lobe pump from the storage tank; while crops silages are placed in a feeding tank (as in Figure 3.3) and then introduced in the digester through a screw auger (shown in Figure 3.4). The feeding operation is repeated every hour.

Figure 3.3. Loading operations in the biogas plant

Figure 3.4. Anaerobic digester and silage loading system

In all plants under study, anaerobic digestion takes place as a single-stage digestion process in a continuous stirred tank reactor (CSTR), operated at 40ºC by circulating hot water. The HRT and OLR are different in each plant (see Table 3.1). In order to keep the solids content around 6–10% inside the digester, a dilution of feedstock with the raw digestate produced (Plant 2) or its liquid fraction (Plant 3 and 4) is required.

As a result of the anaerobic digestion, biogas and digestate are produced. Biogas is stored in a gasholder dome placed at the top of the digesters. In all plants under study, the biogas produced is filtered, dehumidified and desulphurised. Dehumidification is carried out by a traditional refrigeration unit that cools down the biogas and removes water vapour. In addition, a wet scrubber with a washing water solution with sodium hydroxide (8%) assures the removal of sulphur compounds from the biogas stream before being burned.

Bioenergy is produced from the biogas in an internal combustion engine CHP (shown in Figure 3.5). The specific power capacities as well as the electrical and thermal efficiencies are specified in Table 3.1. In all biogas plants, the electricity produced is totally fed into the Italian national grid. In contrast, thermal energy is used to heat up the biomass inside the digester, by recirculating hot water coming from the engine jacket. Surplus heat is dissipated by dry-coolers as a waste to the atmosphere. Finally, the electricity required in these processes, consumed mainly in the loading operations and in the digester mixer, is directly taken from the grid.

Figure 3.5. Cogeneration engine

Inputs of diesel, water, sodium hydroxide, lubricant oil and electricity are required in this subsystem and, consequently, their production (background processes) was included within the system boundaries. In addition, the production of infrastructure of the plants was taken into account. The derived emissions from the combustion of biogas in the CHP were accounted, including carbon monoxide, carbon dioxide, methane, non-methane volatile organic compounds (NMVOC), nitrogen oxides and sulphur dioxide. Moreover, biogas losses also occur caused by leakages in valves and pipe connections; thus, it has been considered that around 1.5% of the total volume of produced biogas is emitted into the atmosphere (1% from the digestion plant and 0.5% from the gas engine), according to Poeschl et al. (2012a).

SS4: Digestate management

Digestate is produced as a result of the anaerobic digestion process. In Plants 3 and 4, it is firstly separated into its liquid and solid fractions by a screw press (Figure 3.6). Then, the digestate that is not recirculated into the digester is stored. Raw and liquid digestate are stored in open tanks; while the solid digestate is stored in piles. In more detail, they are stored for an average period of 150 days until they can be applied on land. Residual biogas as well as nitrogen-based compounds, including ammonia, nitrous oxide and nitrogen, are emitted during the digestate storage; and therefore, were also taken into account within the system boundaries. Leaching during storage was considered negligible, assuming that the tank is fully sealed.

Figure 3.6. Solid/liquid separation of the digestate

SS5: Surplus digestate use

The digestate can be used as a potential organic fertiliser because it contains active fertiliser ingredients such as ammonium nitrate, triple superphosphate and potassium sulphate (Börjesson and Berglund, 2006; Poeschl et al., 2012a). In fact, the produced digestate is used as an organic fertiliser in the cultivation of the energy crops used in the same plant (Plant 2, 3 and 4). Therefore, the required machinery and derived emissions of the application of digestate were included in the subsystem "Feedstock production". However, whenever the production of digestate was higher than the required for the growing of the energy crops, the digestate was used as fertiliser source of nitrogen in another crop system, reducing the consumption of nitrogen-based mineral fertilisers. Therefore, an avoided product perspective was considered. Thus, a system expansion strategy was performed between the digestate, electricity and/or biogas, which prevented from applying allocation procedures. Several studies have also included environmental credits from the phosphorus and potassium content present in the digestate (Poeschl et al., 2012a, 2012b); though, this perspective was not taken into account in this study since accumulation of these nutrients has been previously pointed out due to over-application of organic fertilisers such as manure and digestate (Nkoa, 2014). In particular, in Northern Italy, where widespread livestock activities are performed, the soil presents high contents of phosphorus and potassium due to the repetitive application of pig and cow slurry (Bacenetti et al., 2016a).

The requirements of agricultural machinery and the corresponding emissions from diesel use as well as the emissions derived from digestate and mineral fertilisers application on field (ammonia, nitrous oxide, nitrogen, nitrate and phosphate) were included. However, it was considered that the occupation of agricultural land was totally allocated to the crop cultivated.

3.3. Life cycle inventory

In order to make a reliable evaluation, site-specific data should be used, when available, to better reflect the real situation of the biogas plants evaluated. For this reason, primary specific data for “feedstock production”, “feedstock supply” and “bioenergy production” concerning inputs (feedstock, electricity, heat, sodium hydroxide and lubricant oil) and outputs (biogas, digestate, heat and electricity) were directly obtained by questionnaires made to farmers and plant workers (Table 3.4). The daily composition of the feedstock depends on its seasonal availability. In this study, average data from the year 2012 was managed.

Regarding “feedstock production”, inputs (machinery, diesel, digestate, fertilisers, pesticides and water) and outputs (crop productivity) linked to all agricultural activities for maize and triticale cultivation were taken from studies where real data coming from the plantations under study were used (Bacenetti et al., 2014; González-García et al., 2013).

With the aim of validating the primary data collected as well as to calculate other inventory data required in the study, mass balances were developed for each biogas plant regarding total solids (TS), total volatile solids (TVS), total nitrogen (TN), total ammonia nitrogen (TAN) and total phosphorus (TP).

Table 3.4. Summary of the main primary data per FU corresponding to the foreground system

			Plant 1	**Plant 2**	**Plant 3**	**Plant 4**
Inputs	Pig slurry	(t/FU)	35.0	2.34	2.55	-
	Maize silage	(t/FU)	-	1.59	3.06	2.92
	Triticale	(t/FU)	-	1.42	-	-
	Digestate	(t/FU)	-	6.69	6.81	7.93
	Electricity	(kWh/FU)	97	55	105	104
	NaOH	(kg/FU)	0.21	0.08	0.07	0.14
	Lubricant oil	(kg/FU)	0.65	0.30	0.27	0.37
Outputs	Biogas	(m^3/FU)	468	497	649	573
	Raw digestate	(t/FU)	34.5	11.1	11.7	10.2
	Electricity	(MWh/FU)	1.00	1.00	1.00	1.00
	Heat	(MWh/FU)	1.43	14.92	1.08	1.27

Cereal silages are characterised by a high TS content (~33%), with a volatile fraction around 87-95%. This high biodegradability of the solid materials results in high biogas yields (Negri et al., 2016). On the contrary, pig slurry shows a lower solid content, around 3.5%, with 85% of TVS. Regarding nutrients, energy crops show a higher concentration of TN (3.75-4.48 kg TN/t), than pig slurry (2.43 kg TN/t). However, the total content of TAN that enters in the digester, which may cause inhibition of the process, comes from the addition of pig slurry (1.83 kg TAN/t). Therefore, the characteristics of the digestate in each plant were calculated from mass balances considering the composition of the feedstock used in the digester and the composition of the produced biogas. The separation of the digestate in a screw press was simulated according to Bauer et al. (2009).

During the storage of the digestate, either as raw digestate or as liquid and solid fractions, residual biogas and other emissions of nitrogen-based compounds such as ammonia, nitrous oxide, nitrogen oxides and nitrogen gas are produced. These emissions were estimated using the emission factors provided by De Vries et al. (2012b). As shown in Table 3.5, these authors differentiated the emissions resulting from the storage of the raw digestate and the solid fraction, indicating that the liquid storage of digestate entails higher emissions of methane than the solid storage; while the former involves larger emissions of nitrous oxide and nitric oxide.

Table 3.5. Emission factors for the storage of the digestate, according to De Vries et al. (2012b)

		Raw digestate	**Solid digestate**
Ammonia	(kg NH_3-N/kg $TAN_{digestate}$)	0.04	0.04
Nitrous oxide	(kg N_2O-N/kg $N_{digestate}$)	0.001	0.02
Nitrogen gas	(kg N_2-N/kg $N_{digestate}$)	0.01	0.1
Nitric oxide	(kg NO-N/kg $N_{digestate}$)	0.001	0.02
Methane	(kg CH_4/t digestate)	0.17	0.004

The estimation of emissions from the application of digestate, ammonium nitrate and urea as fertilisers was required. There are available different emission models for the calculation of direct field emissions from the application of fertilisers. The emission models take into account different types of fertilisers, application methods as well as both soil and climate conditions, among others (Brockmann et al., 2014).

In this study, emissions of ammonia, nitrous oxide, nitrogen and nitrate were calculated using the methodology proposed by Brentrup et al. (2000). This study provides emission factors for the volatilisation of ammonia from organic fertilisers depending on average air temperature, infiltration rate, time between application and rainfall or between application and incorporation into the soil. In this sense, the climate in this area of study is a transition between the Mediterranean climate and the Central European climate, with rainfall mainly concentrated in fall and spring (with an average annual precipitation around 745 mm) and a medium annual temperature of 12°C. Regarding the soil characteristics, it is mainly 52% sand, 30% silt and 17% clay (González-García et al., 2013). On the contrary, ammonia volatilisation due to the application of mineral fertilisers depends on the country and the type of fertiliser. For both organic and mineral fertiliser, nitrogen lost as nitrous oxide is estimated as 1.25% of the TN applied on land, while 9% is released as nitrogen gas. Leachates of nitrate were estimated as the balance of nitrogen inputs (i.e. amount nitrogen supplied by fertilisers and the atmospheric nitrogen) and outputs (i.e. nitrogen content in the crop as well as the remaining emissions previously estimated) (Brentrup et al., 2000). Phosphate leaching due to the application of digestate was considered as 1% of the TP applied, according to Rossier et al. (1998).

When more digestate is produced than the required for the cultivation of the crops used in the biogas plant (Plant 1 and 2), surplus digestate is used in a different agricultural system, replacing the use of ammonium nitrate. Regarding this matter, a ratio of 65% was considered as the nitrogen fertilising replacement value for digestate (De Vries et al., 2012). It was used to calculate the avoided emissions attributed to nitrogen mineral fertilisers.

The Italian electricity mix used for the study was modelled based on Dones et al. (2007) and updated using the data for the average electricity production and import/export data for Italy (Terna Rete Italia, 2015). Finally, background data regarding the production of all required inputs were taken from ecoinvent® version 3.2 database (Wernet et al., 2016). Specifically, data regarding the production of chemicals was taken from Althaus et al. (2007), data for the production bioenergy infrastructure from Jungbluth et al. (2007) and data concerning transport activities, agricultural machinery and agrochemicals productions from Spiermann et al. (2007) and Nemecek and Käggi (2007).

A detailed description of inventory data corresponding to each subsystem included in the system boundaries are displayed in Tables 3.6, 3.7, 3.8, 3.9 and 3.10.

Table 3.6. Global inventory data regarding feedstock production (SS1) per FU

	Plant 2				**Plant 3**		**Plant 4**	
	Maize cultivation		Triticale cultivation		Maize cultivation		Maize cultivation	
Materials and fuels								
Digestate	2.56	t	1.45	t	4.28	t	4.53	t
Seeds	0.51	kg	0.05	kg	0.78	kg	0.94	kg
Pesticides	0.16	kg	0.16	kg	0.25	kg	0.30	kg
Urea	1.61	kg	1.94	kg	2.46	kg	2.96	kg
Ammonium nitrate			1.94	kg				
Diesel	4.89	kg	5.02	kg	34.2	kg	8.84	kg
Tractor	0.44	kg	0.47	kg	3.93	kg	0.76	kg
Agricultural tillage	0.52	kg	0.45	kg	0.80	kg	0.84	kg
Transport								
Tractor and trailer	3.85	t·km	2.18	t·km	22.2	t·km	18.1	t·km
Resources								
Water	96.6	m^3			148	m^3	178	m^3
Products								
Straw	1.75	t	1.20	t	2.67	t	3.22	t
Emissions to air								
Ammonia	2.10	kg	1.29	kg	3.27	kg	3.85	kg
Nitrous oxide	0.16	kg	0.11	kg	0.25	kg	0.29	kg
Nitrogen	0.77	kg	0.51	kg	1.14	kg	1.34	kg
Emissions to water								
Nitrate	3.95	kg			6.16	kg	7.23	kg
Phosphate	0.12	kg	0.07	kg	0.22	kg	0.15	kg

Table 3.7. Global inventory data regarding feedstock supply (SS2) per FU

	Plant 1		**Plant 2**		**Plant 3**		**Plant 4**	
Materials and fuels								
Straw			2.94	t	2.67	t	3.22	t
Animal waste	34.9	t	2.34	t	2.02	t		
Diesel			1.29	kg	1.18	kg	1.42	kg
Tractor			0.08	kg	0.07	kg	0.09	kg
Agricultural tillage			0.07	kg	0.06	kg	0.08	kg
Transport								
Tractor and trailer	140	t·km	7.34	t·km	17.4	t·km	9.65	t·km
Products								
Silage			2.67	t	2.43	t	2.92	t
Animal waste	34.9	t	2.34	t	2.02	t		
Emissions to air								
Ammonia	2.04	kg	0.14	kg	0.12	kg		
Methane	3.37	g	0.23	g	0.22	kg		

Table 3.8. Global inventory data regarding bioenergy production (SS3) per FU

	Plant 1		Plant 2		Plant 3		Plant 4	
Materials and fuels								
Silage			2.67	t	2.43	t	2.92	t
Animal waste	34.9	t	2.34	t	2.02	t		
Digestate (recirculated)			6.69	t	5.4	t	7.93	t
Diesel			0.42	kg	1.18	kg	0.45	kg
Tractor			0.05	kg	0.07	kg	0.06	kg
Agricultural tillage			0.04	kg	0.06	kg	0.04	kg
Chemicals (NaOH)	0.21	kg	0.08	kg	0.06	kg	0.14	kg
Lubricant oil	0.60	kg	0.30	kg	0.14	kg	0.63	kg
Biogas plant	$1.4 \cdot 10^{-4}$	p	$1.4 \cdot 10^{-4}$	p	$1.5 \cdot 10^{-4}$	p	$1.6 \cdot 10^{-4}$	p
Cogeneration unit	$4.0 \cdot 10^{-5}$	p	$4.0 \cdot 10^{-5}$	p	$4.0 \cdot 10^{-5}$	p	$4.0 \cdot 10^{-5}$	p
Energy								
Electricity			55.1	kWh	83.4	kWh	103	kWh
Products								
Electricity	1,000	kWh	1,000	kWh	1,000	kWh	1,000	kWh
Heat	1,428	kWh	1,246	kWh	793	kWh	1,552	kWh
Digestate	34.5	t	11.1	t	9.28	t	10.2	t
Emissions to air								
Carbon dioxide	98.4	kg	53.3	kg	31.8	kg	56.8	kg
Methane	5.34	kg	2.68	kg	2.73	kg	3.04	kg
Carbon monoxide	53.8	g	27.0	g	14.5	g	28.5	g
Nitrogen oxides	16.8	g	8.44	g	4.54	g	8.89	g
NMVOC	2.24	g	1.13	g	0.61	g	1.19	g
Nitrous oxide	2.80	g	1.41	g	0.76	g	1.48	g
Sulphur dioxide	23.5	g	11.8	g	6.36	g	12.5	g

Table 3.9. Global inventory data regarding digestate management (SS4) per FU

	Plant 1		Plant 2		Plant 3		Plant 4	
Materials and fuels								
Digestate	34.5	t	11.14	t	9.28	t	10.2	t
Products								
Digestate (own crops)			3.27	t	3.88	t	2.29	t
Digestate (recirculation)			6.69	t	5.40	t	7.93	
Digestate (surplus)	34.5	t	1.19	t				
Emissions to air due to digestate storage								
Ammonia	3.56	kg	0.57	kg	0.56	kg	0.26	kg
Nitrous oxide	0.27	kg	0.05	kg	0.53	kg	0.59	kg
Nitrogen	0.85	kg	0.17	kg	0.87	kg	0.93	kg
Nitrogen oxide	0.18	kg	0.04	kg	0.36	kg	0.40	kg
Methane	5.86	kg	0.76	kg	0.66	kg	$8.9 \cdot 10^{-3}$	kg
Carbon dioxide	12.3	kg	1.91	kg	2.26	kg	0.03	kg

Table 3.10. Global inventory data regarding surplus digestate use (SS5) per FU

	Plant 1		Plant 2		Plant 3	Plant 4
Materials and fuels						
Digestate	34.5	t	1.19	t	-	-
Diesel	7.43	kg	0.08	kg	-	-
Tractor	0.95	kg	0.01	kg	-	-
Agricultural tillage	1.94	kg	0.02	kg	-	-
Transport						
Tractor and trailer	103	t·km	1.53	t·km	-	-
Ammonium nitrate	52.6	kg	1.01	kg	-	-
Emissions to air						
Ammonia	17.4	kg	0.34	kg	-	-
Nitrous oxide	1.31	kg	0.03	kg	-	-
Nitrogen	5.99	kg	0.12	kg	-	-
Emissions to water						
Nitrate	32.2	kg	0.62	kg	-	-
Phosphate	2.25	kg	0.02	kg	-	-
Avoided emissions to air						
Ammonia	1.28	kg	0.02	kg	-	-
Nitrous oxide	1.01	kg	0.02	kg	-	-
Nitrogen	4.64	kg	0.09	kg	-	-

3.4. Life cycle impact assessment

The potential environmental impacts produced by these four biogas plants were reported in terms of several impact categories of different characterisation methodologies (see Chapter 2). Firstly, CC was selected to measure the contribution to the greenhouse effect. It was determined by considering the characterisation factors provided by the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2013). Moreover, the ReCiPe Midpoint H methodology (Goedkoop et al., 2009) has also been applied to identify other environmental impacts produced in several impact categories, including: OD as an indicator of the contribution to the ozone hole; POF as a measure of the formation of reactive chemical compounds, such as tropospheric ozone; FD for the reduction of fossil resources; TA as indicative of the influence of the acid rain phenomenon; FE to quantify the potential enrichment of nutrients in surface water and ME to analyse marine water enrichment in nutrients. An additional impact category, agricultural land occupation (ALO), was considered to address the area required due to the relevant use of energy crops in these plants.

It is important to note that within the results, positive values presented in tables and figures report environmental burdens, whereas negative values account for environmental credits derived from avoided mineral fertilisation.

3.4.1. General results

The characterisation results corresponding to the FU chosen, that is 1 MWh of electricity fed into the Italian national grid, for the four biogas systems under assessment are presented in Table 3.11. In addition, the environmental impacts produced for the production of 1 MWh of electricity according to the Italian electricity mix is also given as a reference system.

Table 3.11. Characterisation results corresponding to Plant 1, 2, 3 and 4 and the reference system per FU.

		Plant 1	Plant 2	Plant 3	Plant 4	Italian electricity
CC	(kg CO_2 eq/FU)	100	346	478	522	474
OD	(kg CFC-11 eq/FU)	$-7.6\cdot10^{-6}$	$2.0\cdot10^{-5}$	$2.1\cdot10^{-5}$	$2.6\cdot10^{-5}$	$5.8\cdot10^{-5}$
TA	(kg SO_2 eq/FU)	47.5	11.8	10.5	11.6	1.54
FE	(kg P eq/FU)	0.75	0.10	0.11	0.09	0.12
ME	(kg N eq/FU)	9.16	1.51	1.82	2.15	0.06
POF	(kg NMVOC/FU)	0.52	1.32	1.36	1.66	0.98
ALO	(m^2a/FU)	25.0	357	212	256	109
FD	(kg oil eq/FU)	-14.3	45.9	48.9	60.7	137

In Table 3.11 remarkable differences can be observed not only regarding the impacts attributed to each biogas plant, but also among the impact categories selected since different emissions of hazardous substances and extractions of natural resources has different influence on the impact categories at the midpoint level.

Climate change

As shown, not all biogas plants achieved reductions of GHG emissions compared with the Italian electric profile. Plant 1 and 2, which digest higher ratio of animal waste, entailed lower impact in CC compared with the reference system (between 27% and 79% lower emissions); however, the extensive use of energy crops worse the environmental profile, increasing the environmental impacts over the produced in the reference system.

The environmental impacts of each biogas plant for CC split up per subsystem can be found in Figure 3.7. It can be noticed the remarkable contribution of the cultivation step, resulting in a direct link between the ratio of energy crops digested and the environmental impacts produced in CC. Therefore, the largest difference can be found between Plant 1, which only digests pig slurry (100 kg CO_2 eq/FU), and Plant 4, which uses maize silage as the only feedstock (522 kg CO_2 eq/FU).

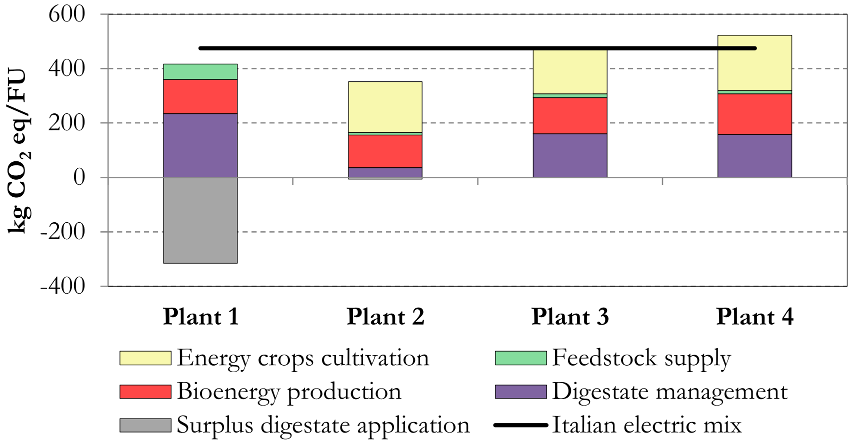

Figure 3.7. Relative contributions to CC of the process involved in the biogas systems

More in detail, the feedstock production (SS1) contributed with ratios between 37% and 55% in CC in Plants 2, 3 and 4. A detailed assessment of this subsystem pointed out the significance of agricultural activities, producing more than 50% of the impacts in this category. It is mainly due to derived carbon dioxide emissions from the consumption of diesel in agricultural machinery. Within the different activities involved in the cultivation of maize, the irrigation process presented the highest contribution ratio since it is performed several times. Considering that triticale cultivation does not need irrigation, the harvesting process was identified as the main environmental *hotspot* due to the large requirement of diesel in this process. Finally, the application of mineral and organic fertilisers was also designated as a source of GHG due to direct field emissions of nitrous oxide, producing 35% of the impacts.

As shown in Figure 3.7, feedstock supply (SS2) only had a notable influence in Plant 1, being responsible of 8% of the impacts due to carbon dioxide emissions from the combustion of diesel required for the transport of pig slurry to the biogas plant. These impacts are connected with the high quantity of feedstock that is needed per FU (34.9 t pig slurry/FU), due to the low biogas potential yield of pig slurry, as discussed in Section 3.5.1. Bioenergy production (SS3) also had an important and constant role in this impact category, producing between 12% and 28% of the impacts in CC. Biogas losses that derive from different types of leakage in pipes and valves connections produced around half of the impacts related to this subsystem, due to direct emission of methane. In addition, the electricity from the grid required to run the biogas plant provides from 25% to 35% of the contributions to this subsystem. It is because the emissions derived from the combustion of non-renewable energy sources within the Italian energy profile.

Digestate management (SS4) includes the separation into liquid and solid fractions, if performed, and the storage of the digestate for the time required. Emissions derived from the storage of the digestate produced significant environmental impacts in CC (10-36%), especially due to derived nitrous oxide emissions. Regarding Plant 1, the environmental impacts in this subsystem produced are related to the higher amount of digestate produced per FU (34.5 t digestate/FU). Differences identified between Plant 2 and Plants 3 and 4 are attributed to the separation of the digestate. In more detail, Plants 3 and 4 perform the separation of the liquid and solid fraction of the digestate, followed by the separated storage of both substrates. According to De Vries et al., (2012a), while the storage of liquid organic substrates such as digestate entails higher emissions of methane compared with the storage of the solid fraction, the emissions of nitrous oxide are larger for the former (see Table 3.6). Surplus digestate use (SS5) includes the application of remaining digestate depending on the digestate required for the first subsystem. Special mention deserves at this point the positive effect of this subsystem produced in Plant 1. More deeply, despite the fact that the management of the digestate produces significant environmental impacts due to emissions derived from its application, the environmental credits associated to the avoided production of ammonium nitrate and avoided emissions from the application of this mineral fertiliser helped to counteract the environmental impacts produced in terms of CC. Conversely, it has slight or no influence on the environmental profile of Plants 2, 3 and 4 due to the requirements of digestate in the feedstock production subsystem.

Ozone depletion, photochemical oxidant formation and fossil depletion

With regard to other energy-related categories, the agricultural activities (SS1) performed in Plants 2, 3 and 4 also played a key role in the environmental performance of OD, POF and FD, as shown in Figure 3.8 for the case of POF. More specifically, regarding these three biogas plants, these impacts meant 66-71% in OD, 73-76% in POF and 66-70% in FD, and they are mainly attributable to the consumption of diesel by the agricultural machinery. In this case, impacts produced in OD were associated to diesel production. In the same way, the environmental impacts in POF were produced as a consequence of diesel use during the operation of the agricultural machinery. Finally, diesel consumption also produced an impact in FD.

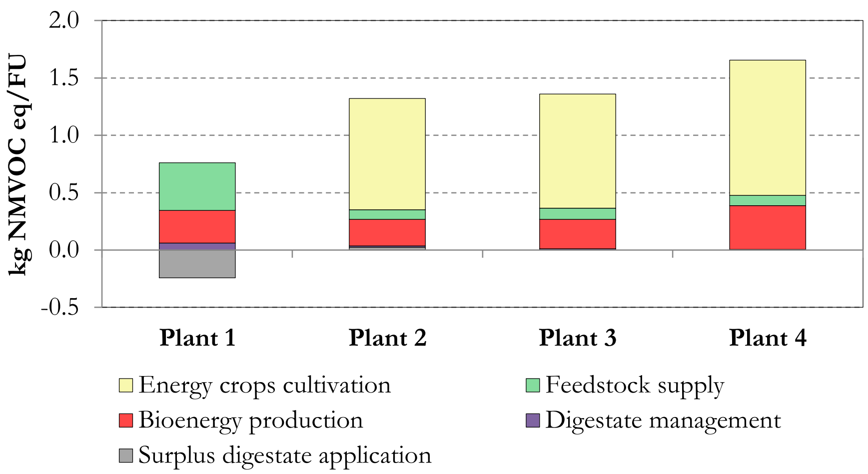

Figure 3.8. Relative contributions to POF of the process involved in the biogas systems

Once again, feedstock supply (SS2) has not a remarkable contribution. However, the behaviour is different in Plant 1, since the production and consumption of the diesel required for transport of pig slurry produced 16%, 32% and 18% of the environmental impacts in OD, POF and FD, respectively. The reason is once again the high amount of pig slurry that needs to be digested to achieve the same electricity production (34.9 t pig slurry/FU) (see Section 3.5.1). Bioenergy production (SS3) represented between 16-24% of the environmental impacts in

OD, 18-22% in POF and 17-25% in FD. In more detail, the impacts in OD derived from the production of electricity consumed in the biogas plant. Regarding POF, the main reason was the emission of nitrogen oxides derived from the combustion of the biogas in the CHP. Finally, fossil fuels consumed in the production of the electricity from the grid produced the contributions to FD.

Digestate management (SS4) entailed slight environmental impacts in these categories since this subsystem comprised emissions from energy requirements for separation, if performed, and emissions from the storage of digestate. However, as in the case of CC, surplus digestate use (SS5) produced a positive effect in Plant 1 regarding these impact categories. The use of digestate in a different agricultural system had a positive effect due to avoided mineral fertilisation; in fact, the avoided impacts derived from the production of ammonium nitrate ended up in environmental benefits in OD; POF and FD.

Terrestrial acidification, freshwater and marine eutrophication

The environmental impacts produced in TA, FE and ME were related to the cultivation of energy crops (SS1) regarding Plants 2, 3 and 4 and the use of surplus digestate (SS5) concerning Plant 1. More deeply, these impacts were associated to direct emissions that arise from the application of mineral fertilisers and digestate on land.

In Plants 2, 3 and 3, field emissions of ammonia arose from the application of fertilisers in the production of energy crops (SS1) and produced between 77% and 91% of the impacts in TA. Of these emissions, around 86-94% derived from the digestate rather than mineral fertilisers (urea and ammonium nitrate). Concerning Plant 1, ammonia emissions from the use of surplus digestate produced 80% of the impacts.

FE and ME measure eutrophication of aquatic bodies produced by a nutrient enrichment of the aquatic environment, especially due to emissions of nitrogen and phosphorus compounds (Goedkoop et al., 2009). Phosphate and nitrate leaching derived from the application of digestate and contributed with 80-88% and 84-97% to the environmental profile of FE and ME, respectively. As mentioned, nitrate leaching was calculated as a balance of inputs and outputs of nitrogen in the agricultural system where fertilisers are applied. Therefore, the amount of digestate applied and the yield and composition of the crop has an important influence.

In addition, the contributions from SS4 in Plant 1 were much higher in these three impact categories than the impacts produced in SS1 for the remaining biogas plants, as shown in Table 3.12 and Figure 3.9 for TA. The reason behind these results is directly linked with the amount of digestate produced per FU; that is much higher in Plant 1 due to the low biogas potential of the mono-digestion of pig slurry. These aspects are further discussed in Section 3.5.1.

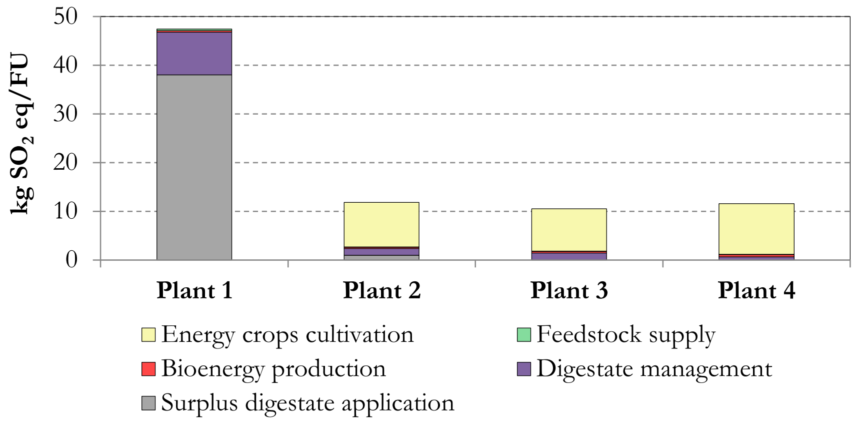

Figure 3.9. Relative contributions to TA of the process involved in the biogas systems

Agricultural land occupation

Finally, the environmental impacts produced in ALO are directly linked with the use of agricultural land, in terms of surface and time. These impacts depend directly on the type of cultivation and the yield of the crop. In more detail, triticale, although it is a less energy-intensive crop, is a winter crop that is cultivated during 8 months per year. On the contrary, maize is a summer crop only cultivated during 4 months per year. Moreover, maize yields a higher amount of straw compared with triticale.

3.4.2. Strategies to mitigate environmental impacts

The current environmental performance of each biogas plant was assessed. However, it is possible to suggest possible improvement actions that can enhance the environmental profile of agricultural biogas production. In order to determine how potential improvement actions could affect the environmental profile of these biogas plants, Plant 3 was selected as a representative example of agricultural biogas production since maize silage and pig slurry are the most commonly co-digested substrates in the Po Valley (Carrosio, 2013). Therefore, in this section the comparison of the base scenario (BS) previously analysed and different alternative scenarios (AS) was performed.

• **Heat valorisation (AS1) –** Heat is produced together with electricity in the CHP. While heat is only partially recycled in heating the anaerobic reactors, electricity is entirely sold to the Italian grid. The surplus heat represents a significant share of the total amount produced by the CHP. This heat might be used to heat nearby greenhouses during the coldest months, to dry forage, to drive a small Organic Rankine cycle (ORC) turbine or an absorption chiller. In this alternative scenario (AS1), surplus heat was considered to be used in two greenhouses from October to March, resulting in environmental credits due to avoided heat produced in a diesel boiler. Each greenhouse has a global area of 1 ha and it is used to grow tomato.

• **Covering the liquid digestate storage tank (AS2)** – Each fraction of the digestate (solid and liquid) is separately stored. Biogas as well as nitrogen-based emissions were identified since storage was considered in open tank for the liquid fraction and in pile for the solid fraction. In this alternative scenario (AS2), covering the storage tank of the digestate liquid fraction was proposed in order to reduce these emissions (Wulf et al., 2006).

• **Digestate land application technique (AS3)** – Emissions during and after digestate application on arable land are highly influenced by the application technique used (Wulf et al., 2006). As base case, surface spreading was considered as application technique of digestate because it is the most widespread procedure for organic fertilisers in the Po Valley (78%) (ISPRA, 2008). However, injection is also carried out but to a lesser extent. According to Wulf et al. (2006), when injection of the digestate is performed, ammonia emissions are nearly half of these from surface spreading with a splash plate, because of the decrease in ammonia volatilisation (Wulf et al., 2002). However, nitrous oxide emissions are around 2.7 times higher due to the promotion of anaerobic sites (Wulf et al., 2002). In addition, the consumption of diesel fuel is higher when injection is applied compared with surface spreading because the slurry tank is equipped with a couple of anchors working at 20-25 cm depth. Environmental consequences of injection were analysed as an alternative scenario (AS3).

The comparison among BS and each alternative scenario under study (AS1, AS2 and AS3) is shown in Figure 3.10.

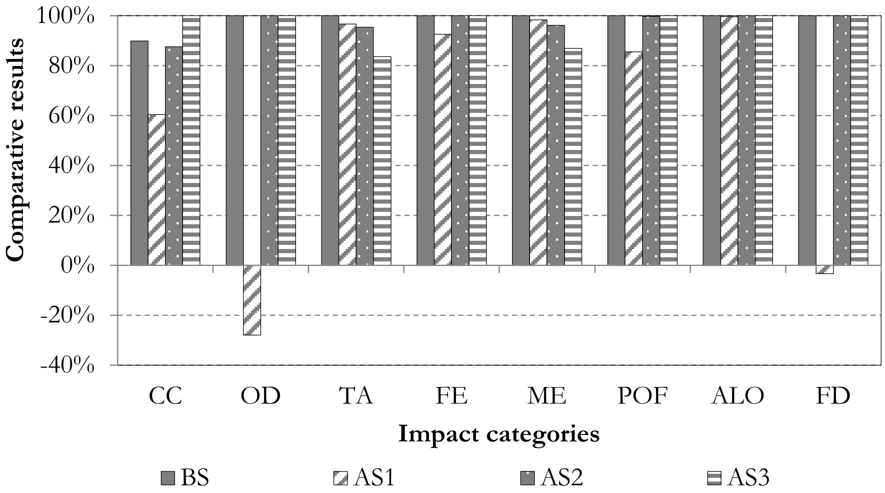

Figure 3.10. Comparative results of the base case (BS) and alternative scenarios (AS)

As can be seen, environmental credits derived from surplus heat valorisation (AS1) significantly improved the system profile; especially regarding energy related categories such as CC, OD, POF and FD. More deeply, AS1 reduced 33% the impacts produced in CC and 14% in POF; regarding OD and FD, the environmental enhancement ended up in environmental benefits. Covering the storage tank (AS2) reduced emissions of residual biogas, nitrous oxide and ammonia, influencing mainly CC and TA. However, the overall effect in the profile of the plant had a slight positive effect (<5%) due to important emissions of these compounds produced in the agricultural step. Finally, the increase of nitrous oxide emissions connected with liquid fraction of digestate injection in arable land (AS3) increased the environmental impacts of CC by 11%. However, the results for TA were 16% lower compared with BS due to the reduction of ammonia emissions.

3.5. Discussion

3.5.1. Performance of the biogas plants under study

In order to understand the environmental results obtained, the several factors that influence their environmental profile should be considered, such as the type of feedstock digested (if it is considered as a waste or not), the combination of feedstocks and the biogas yield of the mixture, the efficiency of the CHP, the amount of digestate produced per FU and its management and use. The average feedstock digested for the production of 1 m^3 of biogas in each plant is presented in Table 3.12.

Table 3.12. Average feedstock input per unit of volume of biogas produced

		Plant 1	Plant 2	Plant 3	Plant 4
Maize	(kg/m^3 biogas)	-	3.19	4.72	5.62
Triticale	(kg/m^3 biogas)	-	2.18	-	-
Pig slurry	(kg/m^3 biogas)	74.7	4.70	3.93	-

It can be observed that the higher the ratio of pig slurry digested, the higher the total amount of feedstock required for the production of 1 m^3 of biogas. In this sense, Plant 1 needs the largest proportion of feedstock (74.7 kg pig slurry/m^3 biogas); according to the composition of pig slurry, this amount only provides 2.62 kg TS/m^3 biogas (2.26 kg TVS/m^3 biogas). This is because pig slurry is a highly diluted substrate (3.5%TS), explaining why Plant 1 does not need the recirculation of the digestate in order to dilute the feedstock inside the digester. Plant 2 and 3, which perform the co-digestion of energy crops and pig slurry, uses 10.1 and 8.66 kg of feedstock/m^3 biogas, respectively; which, in terms of TS, accounts for 1.92 and 1.70 kg TS/m^3 biogas (1.75 and 1.60 t TVS/FU). Finally, Plant 4 that only uses maize silage as feedstock requires the lowest amount of feedstock for the production of 1 m^3 biogas, which is 5.62 kg. Despite this fact, the ratio is similar to the previous one in terms of TS and TVS (1.85 kg TS and 1.76 kg TVS/m^3 biogas). This is the rationale behind the higher environmental impacts per FU in TA and ME in Plant 1, since it results in much higher requirements for feedstock transportation as well as derived digestate production. As shown, even in terms of TVS, Plant 1 needs the higher amount due to the moderate the specific gas potential (SGP) of pig slurry (450 m^3 $biogas_N$/kg TVS), especially when compared with the one from maize (650 m^3

$biogas_N$/kg TVS). This may also help to explain why Plant 2 needs more feedstock than Plant 3, since the SGP of triticale is also lower than the one for maize (580 m^3 $biogas_N$/kg TVS). Nevertheless, the composition of the feedstock is not the only parameter affecting the production of biogas; for example, the amount of TVS digested in Plant 4 is larger than in Plant 3. Therefore, it is important to remind that there are many other factors that vary the efficiency of the anaerobic digestion process such as the OLR and the HRT of the reactors (see Table 3.1). Finally, the efficiency of the CHP converting the biogas produced in energy also affects the environmental results and varies among biogas plants.

3.5.2. Sustainable biogas production in Europe

As previously presented in Chapter 2, numerous LCA studies have analysed the environmental benefits and weaknesses of biogas production from different feedstocks (Boulamanti et al., 2013; De Vries et al., 2012a; Fantin et al., 2015; Fusi et al., 2016; Lansche and Müller, 2012; Poeschl et al., 2012b). An analysis was performed in this section in order to compare the results obtained in this chapter with other published studies in the literature.

Poeschl et al. (2012a, 2012b) investigated a wide variety of biogas scenarios including mono- and co-digestion with different feedstocks, biogas uses and digestate management processes. The FU selected was 1 tonne of feedstock digested. Differently from this study, environmental credits were accrued from the substitution of nitrogen, phosphorus and potassium-based fertilisers (however, they did not consider the whole fertilisation process) as well as fossil energy carrier within the system boundaries. Otherwise, De Vries et al. (2012b) assessed the environmental consequences of the anaerobic mono-digestion of pig slurry and its co-digestion with different substrates including maize silage, crude glycerine, beet tails, wheat yeast concentrate and roadside grass. In this case, the authors followed a consequential approach by including all processes affected by the anaerobic digestion systems. Therefore, within the system boundaries they considered avoided management of pig slurry, land use change emissions and the production of the substitutes for the initial use of the co-substrates. In both cases, these important differences in system boundaries made the numerical comparison of the results obtained in these studies unable. Nevertheless, key elements were identified in both studies that are in agreement with this one. More deeply, in the study performed by Poeschl et al. (2012a, 2012b) the best results were obtained when straw (considered as an agricultural waste) and cattle manure were used as feedstock for anaerobic mono-digestion. As in this study, the environmental burdens related to their production were not included in the system boundaries. Moreover, the largest environmental impacts were reached by the use of energy crops due to impacts linked to the agricultural step. In order to maximise the environmental impacts mitigation, the authors proposed the use of higher proportion of agricultural waste and animal waste. De Vries et al. (2012b) determined that the anaerobic mono-digestion of pig slurry attained better environmental results compared to conventional manure management; however, it represents a limited source of energy compared with its co-digestion with another feedstock.

Boulamanti et al. (2013) evaluated different biogas systems using manure and maize as single substrates in co-digestion; while Fantin et al. (2015) analysed a biogas plant located in Italy co-digesting energy crops (maize, triticale and sorghum), animal waste (cow slurry) and agricultural waste (pressed sugar beet pulps and winery waste). In both studies, the produced biogas was used to co-produce electricity and heat; while the former is entirely fed into the national grid, the latter is partially used in the system and surplus is wasted. In addition, the FU selected was the electricity output (1 MJ) and the impact assessment methodology selected was the one proposed by the ILCD Handbook (Hauschild et al., 2011) in both cases. Fusi et al. (2016) also assessed five Italian full-scale biogas plants performing mono- and co-digestion of energy crops (maize), animal manure (pig and cow slurry) and waste from the food industry (tomato peel and seeds). The FU selected in this study was 1 MWh of electricity generated, the system boundaries included the replacement of animal manure as fertiliser by the digestate and the impact methodology used was the CML 2001 (Guinée et al., 2002). Table 3.13 presents the comparative results obtained by these studies presented per megawatt-hour of electricity produced (MWh_e).

Table 3.13. Comparison between the obtained results with other studies

Study	Feedstock	CC ($kg\ CO_2\ eq/MWh_e$)	OD (kg CFC-11 eq/ MWh_e)	TA ($kg\ SO_2\ eq/MWh_e$)	FE (kg P eq/ MWh_e)	ME (kg N eq/ MWh_e)	POF (kg NMVOC/ MWh_e)
Present study	Pig slurry	77.5	$-1.2\cdot10^{-5}$	47.2	0.74	9.15	0.22
	Pig slurry, maize and triticale silages	343	$2.0\cdot10^{-5}$	11.9	0.10	1.52	1.31
	Pig slurry and maize silage	485	$2.1\cdot10^{-5}$	10.5	0.11	1.82	1.36
	Maize silage	538	$2.4\cdot10^{-5}$	11.5	0.09	2.15	1.55
Fantin et al. (2015)	Cow slurry, maize, triticale and sorghum silages and sugar beet pulps and winery waste,	288	$1.4\cdot10^{-5}$	-[b]	0.04	4.43	1.62
Boulamanti et al. (2013)	Manure	-589		4.82	0.01	2.61	6.14
	Manure and maize silage	286	-[a]	6.14	0.04	4.50	6.61
	Maize silage	500		6.61	0.05	5.07	6.79
Fusi et al. (2016)	Pig slurry, maize silage and tomato peel and seeds.	227	-[a]	2.58			
	Pig slurry and maize silage	37		4.65	-[c]		-[d]
	Pig slurry, maize silage and maize ear silage	291		5.45			
	Maize silage	408		4.91			
	Cow slurry	-395		0.92			

[a]OD in "kg R11 eq"; [b]TA in "mol H+ eq"; [c]eutrophication as a single category in "$kg\ PO_4\ eq$"; [d]POF in "$kg\ C_2H_2\ eq$"

As shown, the environmental results of agricultural biogas differ among LCA studies mainly due to different system boundaries, inventory data calculation and impact assessment method. It should be noticed than even if the same impact category is studied in two LCA studies, different characterisation factors may be used due to different methodology updates. Moreover, uncertainties and use of specific local factors for indirect effects may give rise to wide variability of the final results (Dressler et al., 2012).

Regarding CC, Fantin et al. (2015) estimated lower impacts compared with the co-digestion scenarios inn this study. The main reasons for this difference can be i) the use of agro-residues (sugar beet pulps and winery waste) which may increase the biogas yield without including the environmental impacts of their production and ii) the calculation of emissions derived from digestate storage and application on land. In this case, they considered the IPCC (2006), which considers lower nitrous oxide emissions from the storage of the solid digestate (0.005 kg N_2O-N/kg N) than the considered in the study of De Vries et al. (2012a) (0.02 kg N_2O-N/kg N). Boulamanti et al. (2013) achieved important GHGs savings when animal manure is used for mono-digestion, mainly due to the credits provided by the use of digestate; however, how these environmental credits were considered it is not explained in detail (whether they included avoided use of mineral fertilisers or avoided conventional management of manure). Regarding maize mono-digestion, it achieved almost the same emissions as the reference system (production of electricity from the grid), phenomenon that was also identified in our study and was attributed to the intensive cultivation of the crops, as well as the emissions due to the open storage of the digestate. In this study, the authors used the results obtained in Amon et al. (2006) for the calculation of derived emissions from digestate. Fusi et al. (2016) also attained environmental savings for the mono-digestion of animal waste; in this case because they considered that the use of digestate replaces the use of manure as an organic fertiliser, avoiding emissions from the storage and application of manure on land. The environmental results obtained in this study regarding OD and POF are in the same range than those obtained by Fantin et al. (2015), and lower than those obtained by Boulamanti et al. (2013) concerning POF. Boulamanti et al. (2013) and Fusi et al. (2016) also obtained lower results in TA due to the higher amount of environmental credits accounted that helped to counteract these environmental impacts. Regarding eutrophication, Fantin et al. (2015) and Boulamanti et al. (2013) calculated lower impacts in FE than this study; however the methodology employed to calculate phosphate leaching is not explained. On the contrary, environmental impacts in ME in both studies were higher (ME). For the calculation of nitrate leaching, the IPCC, (2006) used by Fantin et al. (2015) considered an emission factor of 0.30 kg NO_3^--N/kg $N_{applied}$ while Brentrup et al. (2000) suggested a nitrogen balance which highly depend on the crop system (biomass yield, biomass composition, digestate application, digestate composition, etc.).

3.6. Conclusions

The wide spread of anaerobic digestion for renewable energy production requires a deep and detailed analysis of the environmental benefits and negative effects of this technology. This chapter presents an LCA study of four real biogas plants for electricity production located in Italy which uses energy crops and animal waste as input materials. The results demonstrated that bioenergy systems that not perform the extensive use of energy crops can achieve GHG emission savings when compared to conventional fossil reference systems. However, for other impact categories such as TA, FE and ME, most bioenergy systems lead to increased impacts when compared to the reference, especially due to the management of the produced digestate. Nevertheless, the environmental results presented are strongly dependent of the specific substrate selected and the digestate management scheme. Regarding the plants that include energy crops as feedstock, biomass production identified as the main environmental *hotspot* due to impacts related to the cultivation of the biomass. Similarly, concerning the plant that performs the only digestion of pig slurry, the spread of the produced digestate in agricultural land also produced important environmental impacts. Moreover, this chapter suggested some options for the improvement of the environmental profile of biogas plants, which could be useful for future planning of other anaerobic digestion plants. The comparison with other available studies revealed that the key factor explaining the variability of the results were the methodological assumptions made, especially with regard to the system boundaries and the replaced processes selected.

3.7. List of acronyms

ALO	Agricultural land occupation
AS	Alternative scenario
BS	Base scenario
CC	Climate change
CHP	Cogeneration heat and power
CSRT	Continuous stirred tank reactor
FD	Fossil depletion
FE	Freshwater eutrophication
FU	Functional unit
GHG	Greenhouse gas
HRT	Hydraulic retention time
ILCD	International Reference Life Cycle Data
IPCC	Intergovernmental Panel on Climate Change
LCA	Life cycle assessment
ME	Marine eutrophication
NMVOC	Non-methane volatile organic compounds
OD	Ozone depletion
OLR	organic loading rate
ORC	Organic Rankine cycle
POF	Photochemical oxidant formation
SGP	Specific gas production
SS	Subsystem
SSFW	Source segregated food waste
TA	Terrestrial acidification
TAN	Total ammonia nitrogen
TN	Total nitrogen
TP	Total phosphorus
TS	Total solids
TVS	Total volatile solids

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Chapter 4: Environmental consequences of feedstock selection

Summary

The feedstock selection has a crucial role in the definition of the environmental profile of biogas production systems, not only due to its potential biogas production but also the composition and amount of the produced digestate depends directly on it. The objective of Chapter 4 was to analyse the link between substrate selection and the environmental profile of biogas systems. Therefore, the environmental performance of two biogas plants that perform the co-digestion of different feedstocks (energy crops and organic waste) were assessed and compared in detail from a life cycle perspective. The first plant performs the co-digestion of energy crops (78%) and animal waste (22%); while the second one consumes a low ratio of energy crops (4%), food waste (29%) and an important share of animal waste (67%). The use of crops implies higher energy potential in terms of total volatile solids (TVS) digested due to the biogas production rate associated to this type of feedstock (580-650 Nm^3 biogas/t TVS). Despite this fact, the environmental impact in climate change per unit of electricity produced is higher in the first plant (334 kg CO_2 eq/MWh) than the second one (197 kg CO_2 eq/MWh) due to the remarkable environmental burdens derived from the cultivation of energy crops.

Beyond international regulations on climate change, the inclusion of other impact categories such as acidification and eutrophication provides a more exhaustive assessment. In fact, the management and composition of the digestate had a crucial role in the definition of the environmental profile. In this sense, it has been demonstrated that the derived emissions from digestate storage and its application vary on the methodology selected. Additionally, the influence of different functional units on the environmental results was also evaluated.

Outline of Chapter 4

4.1. Contextualisation of the study

As discussed in Chapter 3, energy crops are common substrates for bioenergy production due to their high biogas potential. Among them, maize is the most widely used in the European Union (European Commission, 2014). Nevertheless, the rising demand for maize can entail a change in the use of soil, increasing the pressure to convert grass- and peatlands in areas for maize cultivation (European Commission, 2014). Pardo et al. (2017) indicated that the use of maize improved the performance of the anaerobic digestion process, but its cultivation involved important environmental burdens not only related to energy and fertilisers consumption, but also to changes in indirect land use changes (iLUC). In line with the results obtained in Chapter 3, Whiting and Azapagic (2014) reported that electricity from the anaerobic digestion of agricultural biomass can lead to significant reduction in GHG emissions compared to fossil-fuel alternatives. However, these authors found that the acidification and eutrophication potentials were 25 and 12 times higher, respectively.

In this context, alternative feedstocks that are both economic and environmental sustainable are beginning to be considered. The use of local available organic waste, such as agricultural and food waste, would not only help to improve energy scarcity and security, but also fulfils other desirable objectives such as sustainable management of waste streams (Venkatesh and Elmi, 2013). Agri-food waste is an interesting alternative co-substrate for biogas production, due to high SGP and the lack of burdens usually assigned to the production of waste streams, but special attention should be paid on possible environmental impacts linked to the shifting from its current use as animal feed. Evangelisti et al. (2014) and Walker et al. (2017) reported satisfactory technical and environmental results regarding the anaerobic digestion of food waste from different sources. The aim of Chapter 4 was to assess and compare, from a life cycle perspective, the environmental performance of two biogas plants that process different ratios of energy crops and animal waste. In addition, one of the plants also uses food waste from households and retailers as co-substrate. The main objective was to analyse in detail the influence of substrate selection in biogas production, not only regarding its potential biogas, but also the amount and quality of the produced digestate.

4.2. Goal and scope definition

As explained, the aim of this study was to analyse the environmental implications of substrate selection in biogas production. With this regard, two Italian biogas plants (named as Plants 5 and 6, following the reference numbers used in Chapter 3) located in the Po Valley (Northern Italy) were selected. While Plant 5 treats energy crops and animal manure, Plant 6 also processes food waste. Mass balances were developed for each plant to validate operation data and to calculate other inventory data such as derived emissions. The environmental results were connected to the mass balances to identify the influence of substrate selection on the environmental performance.

• **Plant 5** – The first plant, located in Alzate di Momo, performs the two-stage anaerobic digestion process in three digesters (2+1) with an average temperature of 44°C. It uses different ratios of energy crops (maize and triticale silages) and animal waste (pig slurry and chicken manure) as shown in Table 4.1. The produced biogas is converted into electricity and heat in a CHP engine with an electrical power of 999 kW. Before storage, the produced digestate is separated into its liquid and solid fraction. The solid digestate is stored in piles; while the liquid digestate is stored in a covered tank. Both fractions are used as an organic fertiliser in the cultivation of maize and triticale used in the same plant.

• **Plant 6** – The second plant is situated in Castelleone and consists of a single-stage anaerobic digestion process with four digesters, operating under mesophilic conditions (40°C). As indicated in Table 4.1, it digests energy crops (maize silage), animal manure (pig slurry) and also SSFW as well as food waste from retailers and supermarkets. The biogas is burnt in two CHP units with an electrical power of 832 kW (equivalent to a total electrical power of 1664 kW). The produced digestate is directly stored in a covered tank until it can be applied in the cultivation of energy crops. Nevertheless, taking into account the requirements of nitrogen, surplus digestate is produced and therefore, it is available for its use in another crop system.

Table 4.1. Main parameters of the biogas plants under study

			Plant 5	Plant 6
District			Alzate di Momo	Castelleone
AD	Feedstock		Maize, triticale, pig slurry and chicken manure	Maize, pig slurry, SSFW and food waste
	Digesters		2+1	4
	Temperature	°C	44	40
	HRT	days	40-55	20-30
	OLR	kg TVS/m^3·d	2.48	4.50
CHP	Electrical power	kW	999	1664
	Electrical efficiency	%	41	39
	Thermal efficiency	%	44	45

AD – anaerobic digestion; CHP – co-generation heat and power; HRT – hydraulic retention time; OLR – organic loading rate; TVS – total volatile solids

4.2.1. Function and functional unit

In both biogas plants, the total amount of electricity produced is injected into the national grid whereas the extra heat that is not used in the digester is wasted to the atmosphere. Therefore, following the perspective adopted in Chapter 3, it was considered that the main function of both biogas systems is "the supply of electricity to the national grid" and the FU selected was 1 MWh of electricity produced.

4.2.2. Description of the system boundaries

The system boundaries of these two systems follow the same criteria established in Chapter 3. Therefore, they comprised all inputs and outputs required for all the life-cycle stages, from the production of energy crops to the injection of electricity into the national grid. As displayed in Figure 4.1, the different stages were divided in five subsystems as done in Chapter 3. Within feedstock production, only the production of energy crops is included within the system boundaries; it has been considered that the production of waste streams (pig slurry, chicken manure, SSFW and food waste) is unaffected by its further valorisation in these biogas plants.

(a)

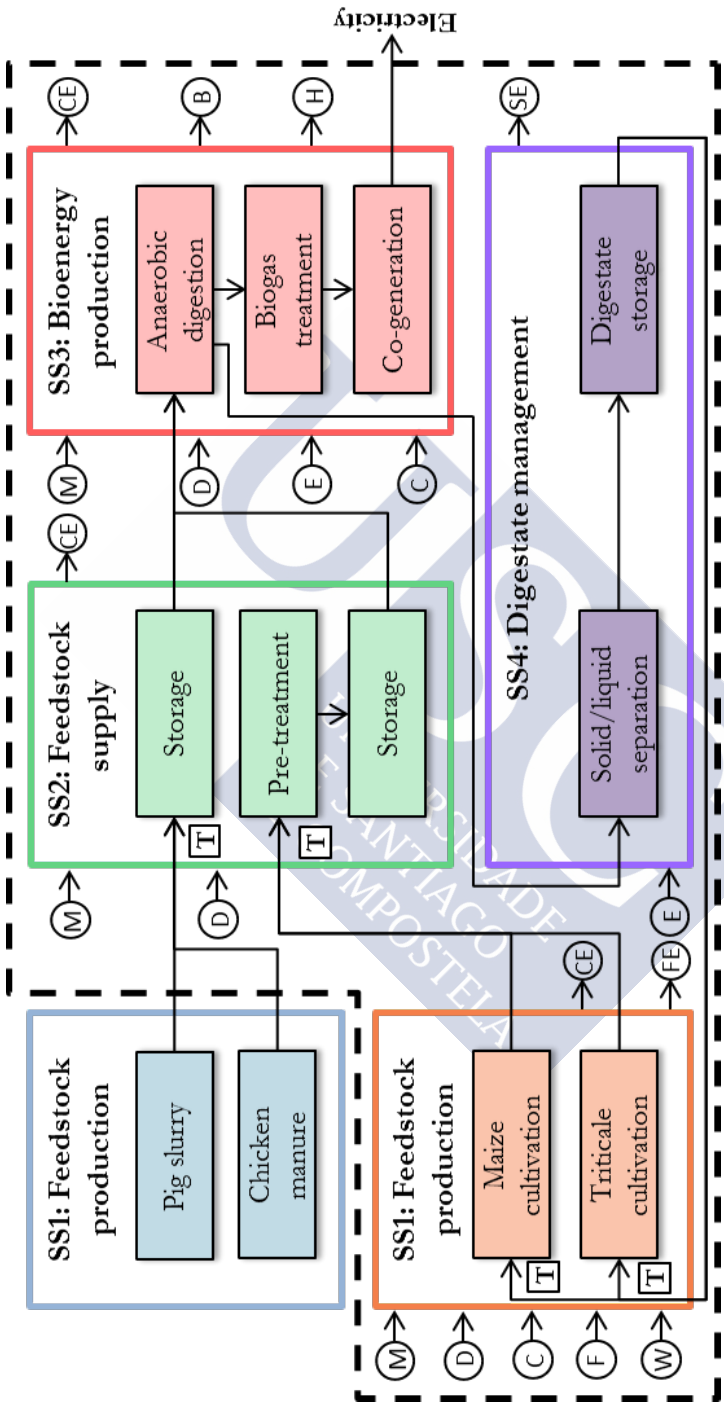

Figure 4.1. Process chain and system boundaries of (a) Plant 5 and (b) Plant 6. ◆ – environmental credits included. Acronyms: T: Transport, M – Machinery; D – Diesel; C – Chemicals; F – Fertilisers; W – Water; E – Electricity; B – Biogas losses; H – Heat dissipated; CE – Combustion Emissions; FE – Fertilisation Emissions; SE – Storage Emissions; L – Landfill

(b)

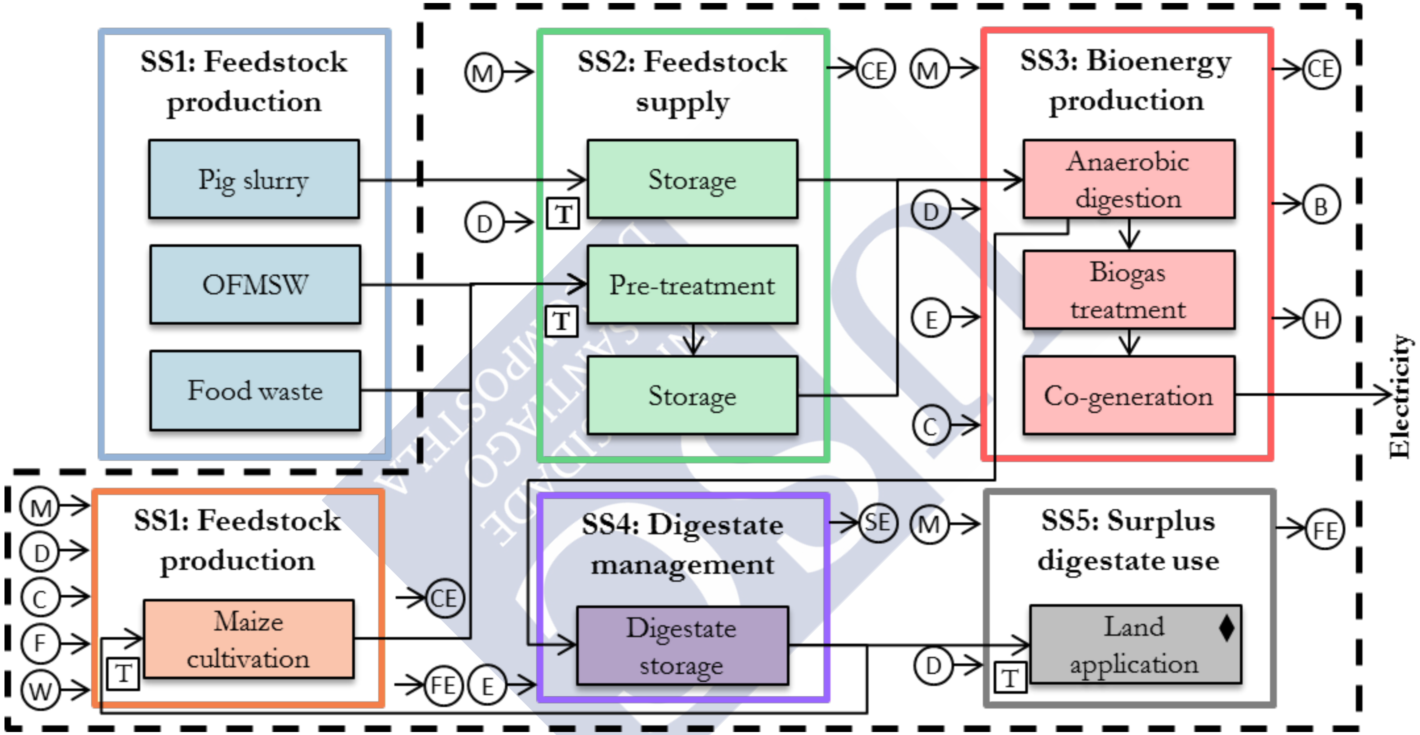

Figure 4.1. Process chain and system boundaries of (a) Plant 5 and (b) Plant 6. ♦ – environmental credits included. Acronyms: T: Transport, M – Machinery; D – Diesel; C – Chemicals; F – Fertilisers; W – Water; E – Electricity; B – Biogas losses; H – Heat dissipated; CE – Combustion Emissions; FE – Fertilisation Emissions; SE – Storage Emissions; L – Landfill

4.3. Life cycle inventory

The cultivation of maize and triticale was computed with the data presented in Chapter 3. The required data of each biogas plant was collected by questionnaires fulfilled by plant workers. The main primary data regarding the most important inputs and outputs of each biogas plant are presented in Table 4.2.

Table 4.2. Summary of the main primary data per day regarding the foreground system

			Plant 5	Plant 6
Inputs	Maize silage	(t/d)	43.9	9.86
		(km)	2	6
	Triticale silage	(t/d)	4.98	
		(km)	3	
	Pig slurry	(t/d)	12.6	178
		(km)	1.5	-[a]
	Chicken manure	(t/d)	1.35	
		(km)	3	
	SSFW	(t/d)		65.7
		(km)		20
	Food waste	(t/d)		11.0
		(km)		20-70
	Iron chloride	(L)	72	
	Lubricant oil	(kg)	3.74	0.6
Outputs	Biogas	(m^3/d)	12,746	17,423
		(% CH_4)	44.6	60
	Raw digestate	(t/d)	48.0	247
	Electricity	(kWh/d)	22,759	39,820
	Heat	(kWh/d)	22,579	42,475

[a] In Plant 6 pig slurry is pumped to the plant instead of transported by road

As in Chapter 3, complete mass balances were performed for each biogas system in terms of TS, TVS, TN, TAN and TP. The average composition of the feedstock input in the biogas plants used for the development of the mass balances can be found in Table 4.3.

Table 4.3. Physico-chemical characterisation of the feedstock used in Plant 5 and 6

		Maize silage	**Triticale silage**	**Pig slurry**	**Chicken manure**	SSFW	**Food waste**
TS	(%)	33	32	3.50	48	58	24
TVS	(% TS)	95	87	85	84	60	92
SGP	(Nm^3/t TVS)	650	580	450	250	375	660
TN	(kg N/t)	3.75	4.48	2.43	1.34	1.6	2.8
TAN	(kg NH_4-N/t)	0	0	1.83	1	0	0
TP	(kg P/t)	1.06	0.06	2.1	0.87	2.2	1.8

TS – total solids, TVS – total volatile solids; SGP – specific gas production; TN – total nitrogen; TAN – total ammonia nitrogen; TP – total phosphorus; SSFW – source-segregated food waste

The quality of digestate is strongly related to the characteristics of the organic substrates used to produce it. Within the calculations performed, it was assumed that there is no loss of nutrients during the anaerobic digestion process, being the total nutrient content in the digestate similar to that of the waste (Evangelisti et al., 2014). The contents of water and ammonia in the produced biogas were also considered negligible (Fantin et al., 2015). Regarding the solid/liquid separation, when performed, it was modelled according to the separation efficiencies presented in Bauer et al. (2009). In more detail, it was considered that 15% of mass, 55% of TS, 60% of TVS, 30% of TN, 30% of TAN and 65% of TP were transferred to the solid stream and the remaining fractions were present in the liquid stream. The mass balances calculated per functional unit (i.e. 1 MWh electricity produced) regarding the main inputs and outputs of both biogas plants are presented in Figure 4.2.

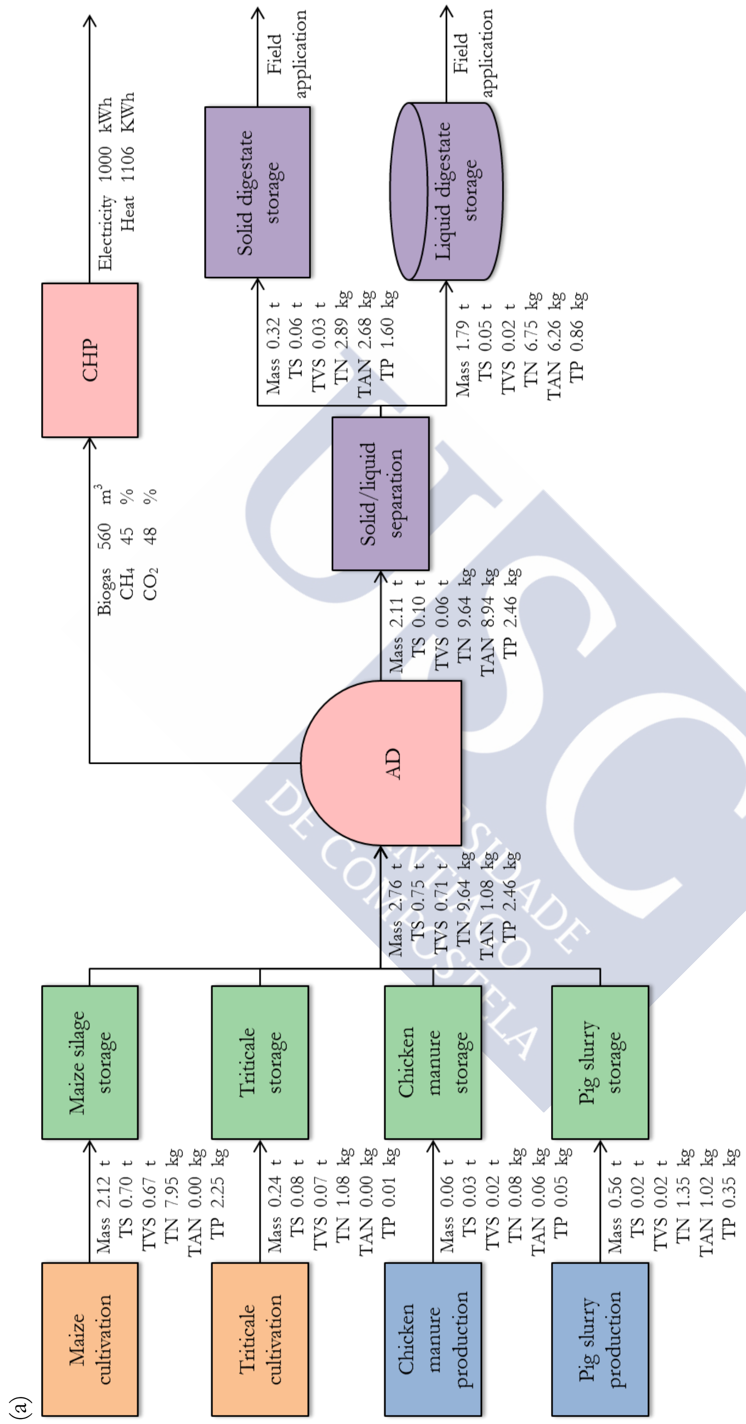

Figure 4.2. Main mass balances of (a) Plant 5 and (b) Plant 6 per FU (i.e. 1 MWh electricity produced). Acronyms: TS – total solids; TVS – total volatile solids; TN – total nitrogen, TAN – total ammonia nitrogen; TP – total phosphorus

(b)

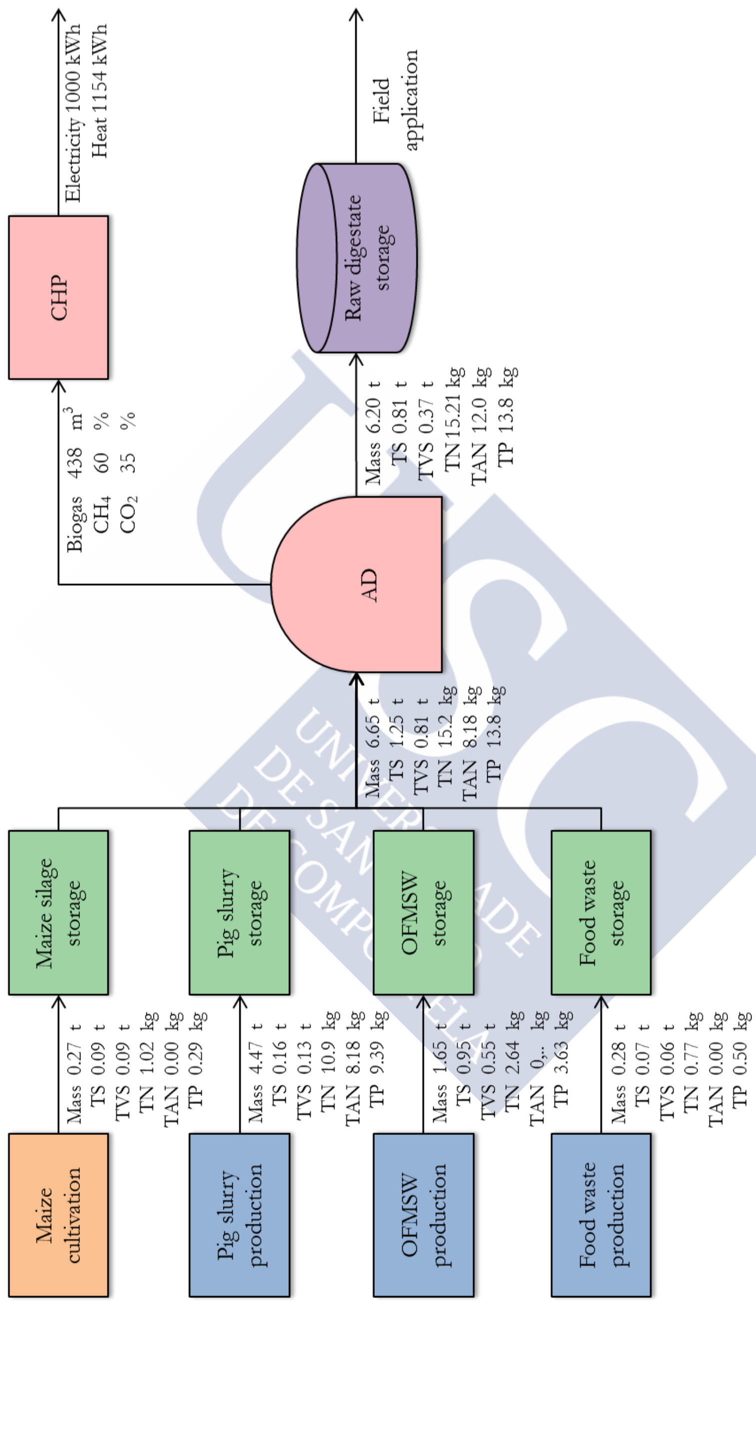

Figure 4.2. Main mass balances of (a) Plant 5 and (b) Plant 6 per FU (i.e. 1 MWh electricity produced). Acronyms: TS – total solids; TVS – total volatile solids; TN – total nitrogen, TAN – total ammonia nitrogen; TP – total phosphorus

Remarkable differences in the behaviour of both plants can be observed in Figure 4.2, not only in terms of mass inputs and outputs, but also regarding the composition of the streams. More deeply, for the production of 1 MWh of electricity, Plant 6 requires the digestion of more feedstock (6.65 t/FU) than Plant 5 (2.76 t/FU); while the values of TVS are not so different (0.71 t TVS/FU in Plant 5 and 0.81 t TVS/FU in Plant 6). Moreover, the specific biogas production of Plant 5 (352 m^3 biogas/t TVS_{fed}) is also higher than the one of Plant 6 (322 m^3 biogas/TVS_{fed}). These differences can be explained taking into account the variability of the feedstock digested. Plant 5 digests a high ratio of energy crops which have high ratio of TVS; while Plant 6 uses a high amount of pig slurry, which provides low amount of TVS. In addition, this substrate also delivers important content of nutrients (2.43 kg TN/t pig slurry and 2.1 kg TP/t pig slurry), but nitrogen is mostly in the form of TAN (1.83 kg TAN/t pig slurry). As a result of this, Plant 6 ends up with the production of a higher amount of digestate (6.20 t digestate/FU) than Plant 5 (2.11 t digestate/FU). The content of nutrients present in the digestate requires a management scheme to be implemented.

Regarding the calculation of the remaining LCI data, emissions derived from digestate storage and application were estimated as explained in Chapter 3 using emission factors provided in the literature (Brentrup et al., 2000; De Vries et al., 2012b). In addition, it has been considered that the closed storage of digestate emitted 80% lower emissions that the open one (Whiting and Azapagic, 2014). In the same way, the Italian electric profile provided in the ecoinvent® database was updated as explained in Chapter 3. Finally, background data regarding the production of all the required inputs such as chemicals, bioenergy infrastructure, transport activities, agricultural machinery and agrochemicals were taken from ecoinvent® database version 3.2 (Wernet et al., 2016). The summary of the main inventory data collected regarding Plant 5 and 6 per subsystems can be found in Tables 4.4, 4.5, 4.6, 4.7 and 4.8.

Table 4.4. Global inventory data regarding feedstock production (SS1) per FU

	Plant 5				Plant 6	
	Maize cultivation		Triticale cultivation		Maize cultivation	
Materials and fuels						
Digestate	6.04	t	0.26	t	0.59	t
Seeds	0.62	kg	0.01	kg	0.08	kg
Pesticides	0.20	kg	0.03	kg	0.03	kg
Urea	1.96	kg	0.39	kg	0.25	kg
Ammonium nitrate			0.39	kg		
Diesel	46.1	kg	1.01	kg	0.80	kg
Tractor	5.54	kg	0.09	kg	0.10	kg
Agricultural tillage	12.4	kg	0.09	kg	0.10	kg
Transport						
Tractor and trailer	6.24	t·km	1.04	t·km	3.51	t·km
Resources						
Water	117	m^3			15.0	m^3
Products						
Straw	2.12	t	0.24	t	0.27	t
Emissions to air						
Ammonia	2.49	kg	0.26	kg	0.33	kg
Nitrous oxide	0.19	kg	0.02	kg	0.02	kg
Nitrogen	0.86	kg	0.10	kg	0.11	kg
Emissions to water						
Nitrate	4.64	kg	-		0.62	kg
Phosphate	0.09	kg	0.01	kg	0.03	kg

Table 4.5. Global inventory data regarding feedstock supply (SS2) per FU

	Plant 5		Plant 6	
Materials and fuels				
Straw	2.12	t	0.27	t
Animal waste	0.61	t	178	t
Food waste			76.7	t
Diesel	1.04	kg	0.12	kg
Tractor	0.06	kg	0.01	kg
Agricultural tillage	0.06	kg	0.01	kg
Transport				
Tractor and trailer	5.98	t·km	46.8	t·km
Products				
Silage	2.36	t	0.25	t
Animal waste	0.61	t	178	t
Food waste	-		76.7	t

Table 4.6. Global inventory data regarding bioenergy production (SS3) per FU

	Plant 5		Plant 6	
Materials and fuels				
Silage	2.36	t	0.25	t
Animal waste	0.61	t	178.08	t
Food waste			76.71	t
Diesel	0.33	kg	38.5	g
Tractor	0.05	kg	4.97	g
Agricultural tillage	0.03	kg	3.64	g
Lubricant oil	0.16	kg	0.02	kg
Anaerobic digestion plant	$1.4 \cdot 10^{-4}$	p	$1.3 \cdot 10^{-4}$	p
Co-generation unit	$4.0 \cdot 10^{-5}$	p	$4.0 \cdot 10^{-5}$	p
Energy				
Electricity	50.2	kWh	235	kWh
Products				
Electricity	1,000	kWh	1,000	kWh
Heat	1,106	kWh	1,154	kWh
Digestate	2.11	t	6.21	t
Emissions to air				
Carbon dioxide	32.7	kg	18.7	kg
Methane	2.48	kg	2.62	kg
Carbon monoxide	14.2	g	8.11	g
Nitrogen oxides	4.44	g	2.54	g
NMVOC	0.59	g	0.34	g
Nitrous oxide	0.74	g	0.42	g
Sulphur dioxide	6.21	g	3.55	g

Table 4.7. Global inventory data regarding digestate management (SS4) per FU

	Plant 5		Plant 6	
Materials and fuels				
Raw Digestate	2.11	t	6.2	t
Products				
Digestate liquid fraction (own crops)	1.79	t		
Digestate solid fraction (own crops)	0.32	t		
Raw digestate for own crops			0.59	t
Raw surplus digestate			5.61	t
Emissions to air				
Ammonia	0.19	kg	0.18	kg
Nitrous oxide	0.19	kg	0.01	kg
Nitrogen	0.30	kg	0.03	kg
Nitric oxide	0.13	kg	$6.0 \cdot 10^{-4}$	kg
Methane	0.06	kg	0.21	kg
Carbon dioxide	0.03	kg	0.98	kg

Table 4.8. Global inventory data regarding surplus digestate use (SS5) per FU

	Plant 6	
Materials and fuels		
Digestate	5.61	t
Diesel	1.21	kg
Tractor	0.15	kg
Agricultural tillage	0.32	kg
Transport		
Tractor and trailer	44.9	t·km
Avoided mineral fertilisers production		
Ammonium nitrate	8.87	kg
Emissions to air		
Ammonia	2.94	kg
Nitrous oxide	0.22	kg
Nitrogen	1.01	kg
Emissions to water		
Nitrate	5.43	kg
Phosphate	0.38	kg
Avoided emissions to air and water		
Ammonia	0.21	kg
Nitrous oxide	0.17	kg
Nitrogen	0.78	kg

4.4. Life cycle impact assessment

4.4.1. Comparative assessment

The European Commission published a working document on the sustainability of solid and gaseous biomass used for bioenergy production in 2014 (European Commission, 2014). Regarding biogas, the report highlighted the environmental concerns of the use of energy crops and encouraged the use of alternative organic wastes. The report also established that existing bioenergy installations from biomass should achieve GHG savings of at least 70% compared to a fossil fuels reference system, defined as the benchmark value (fossil fuel comparator) to be considered sustainable (European Commission, 2014). The estimation equates to life cycle emissions of less than 201 kg CO_2 eq per MWh of electricity produced from biomass.

Figure 4.3 depicts a comparison of Plants 5 and 6 regarding the carbon footprint (i.e. CC), as well as the two reference systems: i) the fossil fuel comparator

(European Commission, 2014) and ii) the Italian electric profile updated for the year 2012 (Terna Rete Italia, 2015). It is important to notice that the methodological assumptions made in LCA studies affect the outcomes of the analysis. Therefore, the fossil fuel reference system and the results of this chapter should be compared carefully, since it is possible that the system boundaries of both studies could be different.

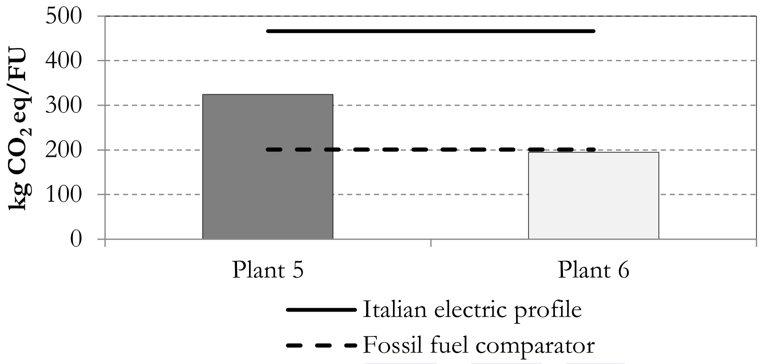

Figure 4.3. Characterisation results of Plants 5 and 6 regarding CC

Both biogas systems achieved better environmental results compared with the environmental profile of the Italian electric grid. However, only Plant 6 achieved results comparable to those proposed to consider bioenergy from biomass as environmental sustainable (i.e. fossil fuel reference). GHG emissions derived from feedstock supply and electricity consumption are higher in Plant 6 than in Plant 5. The impact of feedstock supply is higher due to higher transport distances, especially for the delivery of SSFW and food waste. In addition, once these waste streams are in the biogas plant, they need to be pre-treated, which entails electricity requirements of 25% of the electricity produced in Plant 6 whereas it is only 7% in Plant 5. However, as abovementioned, the cultivation of energy crops in Plant 5 is an important source of GHG emissions due to diesel consumption and direct emissions of nitrous oxide from the application of fertilisers (mineral and organic); therefore, the high ratio of energy crops digested in Plant 5 results in greater environmental impacts in CC compared to Plant 6.

Carbon footprint is the most widely used environmental indicator; however, addressing this indicator alone offers a very limited version of the overall environmental performance. Regarding this issue, Venkatesh and Elmi (2013) criticised the focus on climate change and pointed out the importance of avoiding problem shifting; i.e., reducing the environmental impacts produced in climate change by increasing them to other impact categories. The characterisation results per FU split per subsystem and electricity consumption in Plants 5 and 6 can be found in Table 4.9. The comparison of the environmental profile of both biogas systems showed important differences for the different impact categories. While POF followed a similar behaviour to CC, the environmental impacts of Plant 5 were higher than in Plant 6 for OD and FD. The main reasons behind these results are the higher electricity consumption required in Plant 6 for the pre-treatment of food waste and the transport distances.

Table 4.9. Characterisation results per FU in (a) Plant 5 and (b) Plant 6

(a)

		Total	SS1	SS2	SS3	SS4	EC[a]
CC	(kg CO_2 eq/FU)	316	144	7.67	89.8	51.0	23.4
OD	(kg CFC-11 eq/FU)	$1.7\cdot10^{-5}$	$1.1\cdot10^{-5}$	$1.2\cdot10^{-6}$	$1.5\cdot10^{-6}$	0	$2.9\cdot10^{-6}$
TA	(kg SO_2 eq/FU)	8.27	7.40	0.04	0.279	0.47	0.08
FE	(kg P eq/FU)	0.06	0.05	$9.9\cdot10^{-4}$	$4.5\cdot10^{-3}$	0	$5.8\cdot10^{-3}$
ME	(kg N eq/FU)	1.40	1.36	$2.4\cdot10^{-3}$	$8.0\cdot10^{-3}$	0.02	$2.9\cdot10^{-3}$
POF	(kg NMVOC/FU)	1.18	0.80	0.07	0.27	$6.3\cdot10^{-4}$	0.05
ALO	(m^2a/FU)	210	203	0.714	1.26	0	5.48
FD	(kg oil eq/FU)	39.5	26.3	2.47	3.89	0	6.88

(b)

		Total	SS1	SS2	SS3	SS4	SS5	EC[a]
CC	(kg CO_2 eq/FU)	234	17.8	50.6	89.6	8.18	-42.0	110
OD	(kg CFC-11 eq/FU)	$2.1\cdot10^{-5}$	$1.4\cdot10^{-6}$	$9.1\cdot10^{-6}$	$5.4\cdot10^{-7}$	0	$-4.1\cdot10^{-6}$	$1.4\cdot10^{-5}$
TA	(kg SO_2 eq/FU)	8.27	0.89	0.24	0.25	0.29	6.24	0.36
FE	(kg P eq/FU)	0.17	0.01	$2.1\cdot10^{-3}$	$5.5\cdot10^{-3}$	0	0.12	0.03
ME	(kg N eq/FU)	1.67	0.18	0.01	$6.1\cdot10^{-3}$	0.01	1.45	0.01
POF	(kg NMVOC/FU)	0.89	0.10	0.48	0.20	$2.1\cdot10^{-3}$	-0.09	0.23
ALO	(m^2a/FU)	47.3	20.6	0.30	0.94	0	-0.25	25.7
FD	(kg oil eq/FU)	45.6	3.38	17.4	2.16	0	-9.58	32.3

[a]EC – electrical consumption

Under a closer perspective, there are other factors that affect the performance of both biogas plants. The methane content of the biogas and the electric efficiency of the CHP affect the production of electricity. As shown in Figure 4.2, these parameters are different in the two plants under assessment. That is, for the production of 1 MWh of electricity, 560 m^3 of biogas are required in Plant 5 (45% CH_4) and 438 m^3 in Plant 6 (60% CH_4). Another issue is the feedstock, not only due to the different production and transport schemes but it will also influence the biogas yield and the composition of the digestate. As aforementioned, the production of energy crops was identified as the most important hotspot in Plant 5, accounting for the major share of the environmental impacts. However, it is important to note that they render into high biogas yield and it has influence on the overall environmental profile. More deeply, the SGPs of maize and triticale used as feedstock in these plants were 620 and 550 m^3 biogas/t TVS, respectively. Regarding Plant 6, SSFW and food waste from supermarkets represent 41% of the input flow fed into the digester, but also 75% of the TVS. The SGP of SSFW is reasonable (375 Nm^3 biogas/t TVS), but for food waste, this value is significantly higher (660 Nm^3 biogas/t TVS). Therefore, they can be considered a potential source of energy without including the environmental burdens of its production since they are considered a waste. Nevertheless, it is important to take into account the important energy requirements in the pre-treatment, which ended up into important impacts in energy-related categories such as OD and FD.

Regarding TA, FE and ME, Plant B also achieved worse environmental results than Plant A. It is important to notice that more feedstock is fed to the digester, mainly associated to the ratio of pig slurry digested, which implies low biogas yield, higher digestate production and derived emissions from its storage and application on land. Moreover, TA and ME are mainly influenced by emissions of nitrogen-based compounds (i.e. ammonia and nitrate, respectively). Ammonia emissions arise from the storage and application of digestate while nitrate leaching only occurs in the agricultural land. In the case of FE, it is highly affected by phosphate leaching from the application of digestate as fertiliser. Despite pig slurry has been identified as a moderate source of biogas, it is important to consider that the anaerobic digestion can be a sustainable option for manure management under the approach of waste-to-energy conversion. This issue will be further discussed in Chapter 6.

4.4.2. Methodological implications

Estimation of digestate derived emissions

As demonstrated in the results obtained in this study, the most important *hotspots* were associated to derived emissions. Specifically, emissions from the storage of the digestate as well as from the application of digestate and mineral fertilisers such as ammonium nitrate and urea played a key role in the environmental profile, specifically in terms of impact categories such as TA, FE and ME. In related LCA studies available in the literature, these emissions are usually estimated using different methodologies (De Vries et al., 2012b; Fantin et al., 2015). In a sensitivity analysis, the environmental results obtained in the base case study were compared with the results that would be obtained if other two internationally accepted methodologies were considered: the IPCC and the EMEP/EEA methods.

• **Option 1 – Combined method.** In the base case, a combination between two methods were adopted. Storage emissions were calculated according to emission factors provided by De Vries et al. (2012a). This methodology provides detailed emission factors for ammonia (0.04 kg NH_3-N/kg $TAN_{applied}$), nitrogen gas (0.01 kg N_2-N/kg $N_{applied}$) and nitric oxide (0.001 kg NO-N/kg $N_{applied}$). In addition, it allows differentiating methane and nitrous oxide emissions from the liquid and solid storage of organic substrates (0.001 kg N_2O-N/kg N and 0.17 kg CH_4/t for liquid storage and 0.02 kg N_2O-N/kg N and 0.004 kg CH_4/t for solid storage). Moreover, field emissions were computed according to Brentrup et al. (2000). This methodology takes into account different parameters that influence ammonia emissions such as average air temperature, infiltration rate and time between application and precipitation or incorporation into the soil. Additionally, emissions factors are defined for nitrous oxide (0.0125 N_2O-N/kg $N_{applied}$) and nitrogen gas (0.09 kg N_2-N/kg $N_{applied}$). Finally, nitrate is calculated by the balance between nitrogen entering the system (nitrogen from fertilisers and from atmospheric deposition) and leaving the system (nitrogen present in the crop as well as nitrogen as ammonia, nitrous oxide and nitrogen gas).

• **Option 2 – 2006 IPCC Guidelines for National Greenhouse Gas Inventories.** The methodology proposed in "IPCC Guidelines for National Greenhouse Gas Inventories" (IPCC, 2006) was used as an alternative method.

More deeply, Chapter 10 allows the calculation of methane and nitrous oxide emissions from manure storage. Indirectly, ammonia and nitrogen oxides can be also estimated. Specifically, using Tier 2 of the methodology, methane can be calculated considering temperature, TVS value, maximum methane producing capacity of the substrate and the type of storage (solid or liquid). Regarding the computation of nitrous oxide, Tier 2 provides emission factors for direct emissions depending on the type of storage (0.005 kg N_2O-N/kg N for solid storage and 0 kg N_2O-N/kg N for liquid storage). In order to calculate emissions of indirect nitrous oxide, this methodology also provides the percentage of nitrogen lost in the form of ammonia and nitrogen oxides (48% for liquid storage and 45% for solid storage); however, it does not allow differentiating between them. It has been considered that 90% of the nitrogen is lost in the form of ammonia, since it has been determined as the main emitted compound (Denier Van Der Gon and Bleeker, 2005). The chapter 11 of this report presents a methodology for the calculation of nitrogen-based emissions from the application of organic and inorganic substrates on land. In the same way, Tier 2 offers an emission factor for the calculation of direct nitrous oxide emissions (0.01 kg N_2O-N/kg $N_{applied}$); and through the calculation of indirect nitrous oxide emissions, ammonia, nitrogen oxides and nitrate can be also estimated (0.20 kg NH_3-N + NO_x-N/kg $N_{applied}$ for organic fertilisers such as digestate, 0.10 kg NH_3-N+NO_x-N/kg N for mineral ones and 0.30 kg NO_3^--N/kg $N_{applied}$ for nitrate leaching). In this context, the default emissions factors are 0.01 kg N_2O-N/ kg NH_3-N + NO_x-N and 0.0075 kg N_2O-N/kg NO_3^--N).

• **Option 3 – EMEP/EEA air pollution emissions inventory guidebook 2016.** The model presented in the EMEP/EEA Air Pollutant Emissions Inventory Guidebook 2009 enables the calculation of more accurate ammonia emissions (European Environment Agency, 2016). Tier 2 suggests an emission factor of 0.0266 kg NH_3-N/kg $N_{applied}$ for raw digestate storage, 0.0116 kg NH_3-N/kg $N_{applied}$ for liquid digestate storage and 0.0150 kg NH_3-N/kg $N_{applied}$ for solid digestate storage. The methodology also points out that other compounds should be quantified such as total suspended particles, particulate matter and non-methane volatile organic compounds, but that at present, there are no methods to calculate them. Therefore, other emissions from the storage of digestate such as methane and nitrous oxide has been completed (IPCC, 2006). Regarding the application of fertilisers on land, this methodology also gives emission factors for the calculation of ammonia and nitric oxide emissions. In more detail, emission factors for the application of organic wastes are 0.08 kg NH_3/kg $N_{applied}$ and 0.04 kg NO/kg $N_{applied}$; while emissions from the application of ammonium nitrate are 0.016 kg NH_3/kg $N_{applied}$ and from the application of urea are 0.159 kg NH_3/kg $N_{applied}$. Figure 4.4 shows the comparative environmental profiles of Plant 5 and 6 when applying the different options for the calculation of direct emissions from storage and fertilisers application.

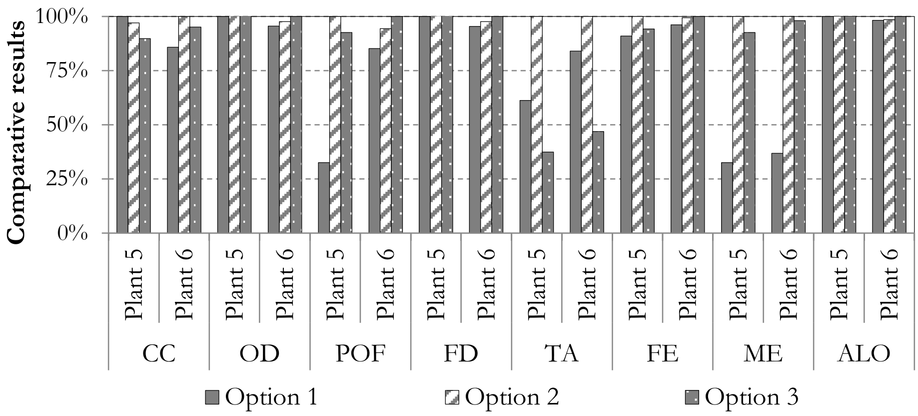

Figure 4.4. Comparative environmental profiles of Plants 5 and 6 applying different options for the estimation of storage and field emissions

As expected, important differences were identified in the environmental impacts produced in TA, ME and POF. In more detail, as explained before, the emission factors for ammonia are very different among methodologies. More deeply, the variability observed in TA is caused by differences in ammonia emissions. Specifically, the IPCC Guidelines for National Greenhouse Gas Inventories does not differentiate emissions from ammonia and nitrogen oxides and they are calculated from the TN of the substrate; while other methodologies such as the proposed in Option 1 estimate ammonia emissions according to its TAN content. This fact has been considered especially important since the ammonification process produced during anaerobic digestion is one of the main drawbacks of the process since it enhances the potential for ammonia emissions during digestate storage compared to the untreated substrate (Fantin et al., 2015). Moreover, Option 1 takes into account specific characteristics of the climate and agricultural field where fertilisers are applied. Additionally, differences observed in ME are the result of different nitrate leaching. Option 1 also estimates these emissions considering specific characteristics of the crop under study, including the yield and composition of the crop, other emissions and atmospheric nitrogen deposition; while Options 2 and 3 apply a direct emission factor. Finally, variability in POF is motivated by different considerations of the methodologies. More into detail, Option 1 considers emissions of nitric oxide from the storage of substrates, while Options 2 and 3 consider nitrogen oxides as NO_X. According to the LCA methodology applied, nitric oxide emissions produced impact only in ME (0.06 kg N eq/kg $NO_{emitted}$), while nitrogen oxides emissions affected on other impact categories such as TA (0.56 kg SO_2 eq/kg $NO_{Xemitted}$), ME (0.039 kg N eq/kg $NO_{Xemitted}$) and POF (1 kg NMVOC/kg $NO_{Xemitted}$).

Influence of functional unit selection

The primary purpose of a FU is to provide a reference to which the inputs and outputs are relatedThe potential environmental impacts calculated for the biogas plants are relative expressions, as they are related to the FU selected. Among the different biogas LCA studies available in the literature, there is a remarkable variability on the FU used, making the comparison between LCA studies difficult to perform. In fact, the selection of FU depends on the objectives of each study and not only one FU is appropriate (Cherubini and Strømman, 2011). For example, a FU related to inputs (i.e. tonne of biomass digested) make possible the comparison among different uses for a given feedstock; while FU related to outputs make possible the identification of the best way to supply a product such as biogas from different feedstock. In this sensitivity analysis, the influence of FU selection was analysed by comparing the performance of different FU available.

• **FU1 –** In the base case the FU selected was 1 MWh of electricity produced and injected into the national grid.

• **FU2 –** It is possible to consider that the common function of the system under study is the production of methane for its conversion into bioenergy. Therefore, it is possible to consider 1 m^3 of methane as FU.

• **FU3 –** Following the same approach, but also taking into account the quality of the biogas produced, 1 m^3 of biogas produced can be the FU selected.

• **FU4 –** If the function of the system is considered as the anaerobic digestion of organic matter for bioenergy purposes, 1 t of TVS fed into the digesters may be a suitable FU.

The comparative results among FUs are shown in Figure 4.5 and a summary of effects produced in parameters of each biogas plant can be found in Table 4.10.

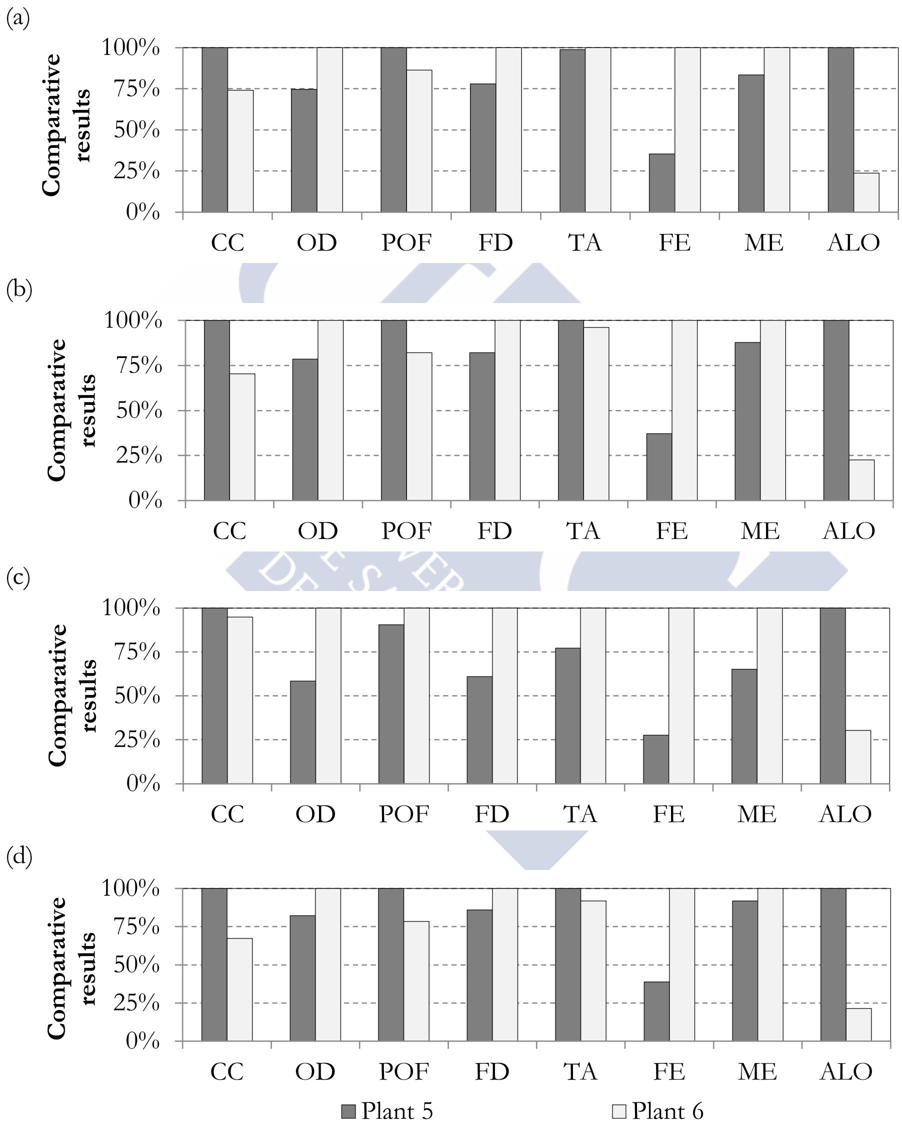

Figure 4.5. Comparative profiles of Plants 5 and 6 for (a) FU1, (b) FU2, (c) FU3 and (d) FU4

The main reason for the differences observed in Figure 4.5 is related to the environmental impacts produced within feedstock production regarding Plant 5 and surplus digestate use in Plant 6 (identified as environmental *hotspots* in section 4.4.1). This can be explained since, depending on the FU selected, the amount of feedstock (in terms of mass and TVS) and biogas (in volume of biogas or methane) required as well as electricity produced is different. Therefore, the main influence of the FU selected is the consideration of the different efficiencies of both biogas plants regarding the conversion of feedstock into biogas as well as in the conversion of biogas into electricity.

Table 4.10. Main input and output data related to the different FU under study

		FU1		**FU2**		**FU3**		**FU4**	
		Plant 5	Plant 6	Plant 5	Plant 6	Plant 5	Plant 6	Plant 5	Plant 6
Feedstock	(t/FU)	2.76	6.65	$5\cdot10^{-3}$	0.01	0.01	0.03	3.90	8.17
TVS in	(t TVS/FU)	0.71	0.81	$1\cdot10^{-3}$	$2\cdot10^{-3}$	$3\cdot10^{-3}$	$3\cdot10^{-3}$	1	1
Biogas	(m^3/FU)	560	438	1	1	2.24	1.67	790	538
Methane	(m^3/FU)	250	263	0.45	0.60	1	1	352	323
Digestate	(t/FU)	2.11	6.20	$4\cdot10^{-3}$	0.01	$6\cdot10^{-3}$	0.02	3.0	7.62
Electricity	(kWh/FU)	1,000	1,000	1.79	2.29	4.00	3.81	1,410	1,229
Heat	(kWh/FU)	1,106	1,154	1.97	2.64	4.43	4.40	1,560	1,418

As shown in Table 4.10, for the production of 1 MWh of electricity (FU1), more biogas is required in Plant 5 than in Plant 6; nevertheless, the amount of methane needed is lower in Plant 5 than in Plant 6. The main reasons for these differences are i) the quality of the biogas produced in each plant and ii) the electricity efficiency of the CHP of each plant. In terms of methane production, the quality of the biogas produced in Plant 5 is lower than in Plant 6 (44.6% and 60%, respectively). Regarding the CHP, the efficiency of the engine in Plant 5 is higher than in Plant 6 (41% and 39%, respectively). Moreover, for the production of the required biogas, much more feedstock is required in Plant 6 than in Plant 5, due to the high ratio of animal waste digested in Plant 5. This means that this FU considers the efficiency of the main two processes carried out in the biogas plant (biogas and bioenergy production).

The quality of the biogas produced was identified as a key parameter to have into account; this can be observed in the differences between FU2 and FU3. In more detail, more feedstock in terms of TVS is required in Plant 6 than in Plant 5 for the production of 1 m^3 of biogas (FU2); however, for the production of 1 m^3 of methane (FU3), the same amount of TVS is required in both plants. In addition, whenever FU2 is considered, Plant 6 resulted in more bioenergy production than Plant 5; though, when considering FU3, Plant 5 produces more bioenergy than Plant 6. The reasons for these results are the higher biogas potential of feedstock in Plant 5 in comparison with the biogas of Plant 6 and the lower quality of the biogas produced in Plant 5 than in Plant 6.

Regarding FU4, the digestion of 1 t of TVS renders in much more biogas using a lower amount of feedstock in Plant 5 (790 m^3 biogas/t feedstock) than in Plant 6 (538 m^3 biogas/t feedstock). However, in terms of methane, the production is more similar (352 and 323 m^3 CH_4/t feedstock, respectively), ending in quite similar bioenergy production (1,410 and 1,229 kWh electricity/t TVS).

4.5. Discussion

4.5.1. Potential biogas production of substrates

As shown in the previous section, the variety of substrates used entailed different environmental performance because of many factors such as different composition (entailing different digestate characteristics), different methane yield and the inclusion or not of the feedstock production (energy crops vs residues). The data provided in this chapter regarding the potential biogas production of the feedstock are specific for these substrates. As shown in Table 4.11, in the literature there are available different studies reporting different biogas productivities for the substrates under study, which can have an impact in the environmental results obtained.

Table 4.11. Methane yields for the substrates under study reported by different studies

	Methane yield (Nm^3 methane/t TVS)				
	Present study	Gissén et al. (2014)	Møller et al.(2004)	Schott et al. (2013)	Evangelisti et al. (2014)
Maize	322	361	-	-	-
Triticale	286	397	-	-	-
Pig slurry	252	-	356[a] 275[b]	-	-
Chicken manure	137	-	-	-	-
SSFW	191	-	-	443	378
Food waste	350	-	-	-	-

[a]Fatteners; [b]Sows

For example, Gissén et al. (2014) studied the potential biogas production from different energy crops in Sweden, ranging from 237 Nm^3 CH_4/t TVS for hemp to 408 Nm^3 CH_4/t TVS for beet root. Contrarily to our results, they reported significantly higher methane yield for maize and triticale than the values reported here (Table 4.11). With these results, it would suppose 14% and 2% higher methane production in Plant 5 and 6, respectively. Although a reduction of the environmental impacts would be expected, the influence of agricultural practices would still be predominant on the environmental impact. González-García et al. (2013) reported the environmental consequences of the cultivation of different energy crops in the same area of study, including wheat, triticale and different classes of maize. The authors concluded that the cultivation of different energy crops produced different environmental impacts due to different productivity yields, field requirements and direct emissions. Moreover, Dressler et al. (2012) studied maize cultivation in three different areas of Germany and they pointed out that the environmental impacts varied according to regional farming procedures and specific characteristics of the area such as soil type and climate conditions.

Besides, Møller et al. (2004) also provided higher methane production factors for pig slurry compared with this study (see Table 4.11). This higher methane production would mean an increase in methane and electricity production of 1% and 8% in Plant 5 and 6, respectively. From this minor contribution, it is important to highlight that the nitrogen and phosphorus content of the substrates has a strong influence on the environmental profile, especially in impact categories such as ME and FE due to nutrient leaching. Finally, the methane potential of food waste depends on its composition which highly differs among regions and collection schemes. For example, Schott et al. (2013) considered 443 Nm^3 methane/t TVS as the theoretical methane potential for separate collection of household food waste in Sweden. On the other hand, Evangelisti et al. (2014) took into account for their calculations a methane production of 378 Nm^3 CH_4/t TVS regarding SSFW in United Kingdom.

4.5.2. Requirements of sustainable biogas production

Some of the criteria needed to meet the requirements of sustainable biogas production includes secure energy supply, avoid competence with food production, socio-economic development including creation of local employment and reduced environmental impacts (Jacobs et al., 2016). Moreover, according to the European Commission (European Commmission, 2014), the environmental sustainability of biomass use for bioenergy production may be reduced by several factors such as unsustainable agricultural practices, land use changes, direct and indirect emissions and inefficient bioenergy generation. Considering the results of the present study, some suggestions and further improvements can be made in order to improve the sustainability of biogas production.

✓ Since climate change is an important concern for European environmental policy, LCA studies are becoming increasingly relevant for policy decision-making. Therefore, for the proper development of LCA studies, a higher level of harmonisation in the methodology application is required to make the studies comparable and transparent (Bacenetti et al., 2016). Therefore, as already stated by Bacenetti et al. (2016), there is a need for common and very specific guidelines for LCA studies to assess and communicate the environmental performance.

✓ It is also important to consider that, as shown in this study, climate change should not be the only focus of environmental concerns of policies regarding bioenergy production. Acidification and eutrophication impacts mainly linked to energy crops cultivation and digestate management should be integrated in the environmental policies.

✓ In addition, as proved by Dressler et al. (2012), local factors and regional parameters have a strong effect in LCA results. Therefore, it is necessary to consider regional parameters (e.g., transport distances, agricultural area for biomass production and digestate spreading, competition for cereal silage between biogas production and livestock activity) with the aim of performing a representative LCA study. Only if regional variations are considered, the results of environmental indexes will be representative, as the results could vary from one region to another.

✓ Regarding the selection of the feedstock, as suggested by Whiting and Azapagic (2014), economic incentives should include further requirements on feedstock type to promote the use of different types of wastes and to prevent the use of energy crops that may compete with other uses.

4.6. Conclusions

This chapter demonstrated that renewable energies can achieve savings of GHG when compared to conventional fossil reference systems. However, the environmental results obtained were strongly dependent of the specific substrate selected and the digestate management. In Plant 5, the use of energy crops was identified as an important source of bioenergy due to its high biogas potential; however, important environmental impacts arise from their production since they are cultivated exclusively for bioenergy purposes. The characterisation of the SSFW and food waste used Plant 6 proved that these substrates can be an alternative co-substrate able to improve the environmental profile of the biogas system. They have higher energy potential than pig slurry and no environmental burdens from their production are allocated to the biogas system. Nevertheless, electricity consumption in the biogas plant is increased due to the pre-treatment requirements of this type of waste. Moreover, in Plant 6 the use of substrates such as pig slurry with lower biogas potential than energy crops resulted in a higher amount of digestate per unit of electricity produced. In this sense, the spread of the produced digestate in agricultural land produced important environmental impacts of acidifying and eutrophying substances. With this regard, a sensitivity analysis was performed for the calculation of emissions derived from the storage and application of digestate.

The study proved how the environmental profile of both biogas plants would change considering different accepted methodologies. In addition, another sensitivity analysis was performed to analyse different FUs used in the literature. The use of electricity output as FU revealed as an appropriate FU since it takes into consideration the conversion efficiencies of the two most important processes within a biogas system: feedstock into biogas and biogas into bioenergy.

4.7. List of acronyms

ALO	Agricultural land occupation
CC	Climate change
CHP	Co-generation heat and power
FD	Fossil depletion
FE	Freshwater eutrophication
FU	Functional unit
GHG	Greenhouse gas
iLUC	Indirect land use changes
IPCC	International Panel on Climate Change
LCA	Life cycle assessment
LCI	Life cycle inventory
ME	Marine eutrophication
OD	Ozone depletion
POF	Photochemical oxidant formation
SGP	Specific gas potential
SS	Subsystem
SSFW	Source-segregated food waste
TA	Terrestrial acidification
TAN	Total ammonia nitrogen
TN	Total nitrogen
TP	Total phosphorus
TS	Total solids
TVS	Total volatile solids

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Chapter 5: Eco-efficiency assessment of agricultural biogas

Summary

Despite the fact that bioenergy production from agricultural biogas is considered an effective strategy to minimise the impact on climate change, operational choices in biogas plants can compromise its environmental sustainability. The aim of Chapter 5 was to analyse the eco-efficiency operation of 15 agricultural biogas plants located in Northern Italy. For this purpose, the selected methodology was a combination of Life Cycle Assessment (LCA) and Data Envelopment Analysis (DEA).

According to the results obtained, 9 out of 15 biogas plants were identified as efficient and benchmark values for inputs such as feedstock, electricity and transport were calculated. Thereafter, the environmental profile of the original and the virtual plants were compared to determine the net environmental gains linked with the inputs reduction proposed in the DEA model. The results differed among impact categories, being climate change the most favoured by the proposed reductions. Regarding acidification and eutrophication categories, environmental impacts were related to the management of the produced digestate, while agricultural land occupation was not only influenced by the ratio of energy crops used in the plants but also with the biomass production yield per hectare of each crop. These facts were not fully reflected in DEA results; therefore, the reduction targets were not always focused on the most polluting facility. Despite this, owing to the reduction targets, the overall environmental impacts were reduced below the impact produced by the Italian electric profile for all the plants under study.

Outline of Chapter 5

5.1. Contextualisation of the study

As mentioned in previous chapters, the number of agricultural biogas plants in Italy has grown exponentially in few years, and installed power increased from 24 MW in 2007 to 564 MW in 2012. Most of these plants are located in the Po Valley region, since it is the most important area regarding to industrial, agricultural and livestock sectors. In fact, 1,000 out of the 1,800 plants operating in Italy are located in the Po valley. In order to guarantee the best environmental performance, the efficient operation of the biogas plant is a key factor, ensuring maximum substrate utilisation and minimum residual methane potential to optimise bioenergy production while reducing emissions from the management of the digestate (Ruile et al., 2015). The fundamental parameters for this optimum conversion are the type of the feedstock, the operating temperature, the HRT, the OLR and the stability of the biological process (Naik et al., 2014). The great number of similar biogas plants in this region opens up the possibility of a systematic comparison among them to ascertain about the operational inefficiencies that have a negative impact on their environmental profile. As described in Chapter 2, this can be done with the combined application of the LCA+DEA methodology. DEA enables the identification of inefficient operating points, promoting technological improvements under the perspective of an efficient operation performance. The specific combination of LCA+DEA has been proposed to detect and sort out the technical inefficiencies that are sources of unnecessary environmental impact (Lozano et al., 2009). It has been applied to different processes such as wine production (Vázquez-Rowe et al., 2012), fisheries (Vázquez-Rowe et al., 2010) and even to WWTPs (Lorenzo-Toja et al., 2015a). However, the combined method LCA+DEA has not been applied to agricultural biogas production. Therefore, the objective of Chapter 5 was to assess and compare the operational and environmental performance of 15 different biogas plants, applying the combined LCA+DEA methodology. The analysis was conducted to accomplish a number of objectives: i) to detect operationally inefficient biogas plants, ii) to benchmark target input consumption levels for the inefficient biogas plants, iii) to quantify the environmental benefits of moving towards operational efficiency in biogas production and iv) to identify the best functioning plants to be used as operational and environmental references.

5.2. Materials and methods

5.2.1. Description of the biogas plants

As aforementioned, the aim of this chapter was to analyse 15 agricultural biogas plants from an eco-efficiency perspective. The assessment comprised some of the biogas plants already assessed in Chapters 3 and 4 (Plants 2 – 6) and 10 new plants. Plant 1 was not included in the study since it has been considered that the main purpose of this biogas plant is "the management of waste" rather than "the production of electricity", as considered for the other plants that use energy crops. The reasons supporting this decision are that it is a small biogas plant (electrical power of 250 kW), that only digests pig slurry. All the 15 biogas plants are located in the Po Valley (Northern Italy), a large flat area with special density of biogas plants due to its industrial, agricultural and livestock activities (Carrosio, 2013). The location of the plants is presented in Table 5.1 and in Figure 5.1.

Table 5.1. Location of each biogas plant under study

Plant	Region	District
2	Lombardy	San Giorgio di Lomellina
3	Piamont	Vercelli
4	Lombardy	Pavía
5	Piamont	Alzate di Momo
6	Lombardy	Castelleone
7	Lombardy	Brembio
8	Lombardy	Somaglia
9	Piamont	Mandello Vitta
10	Lombardy	Santa Cristina e Bissone
11	Lombardy	Montanaso Lombardo
12	Piamont	Casalvolone
13	Lombardy	San Giorgio di Lomellina
14	Piamont	Castelnuovo Scrivia
15	Piamont	Villata
16	Lombardy	Maleo

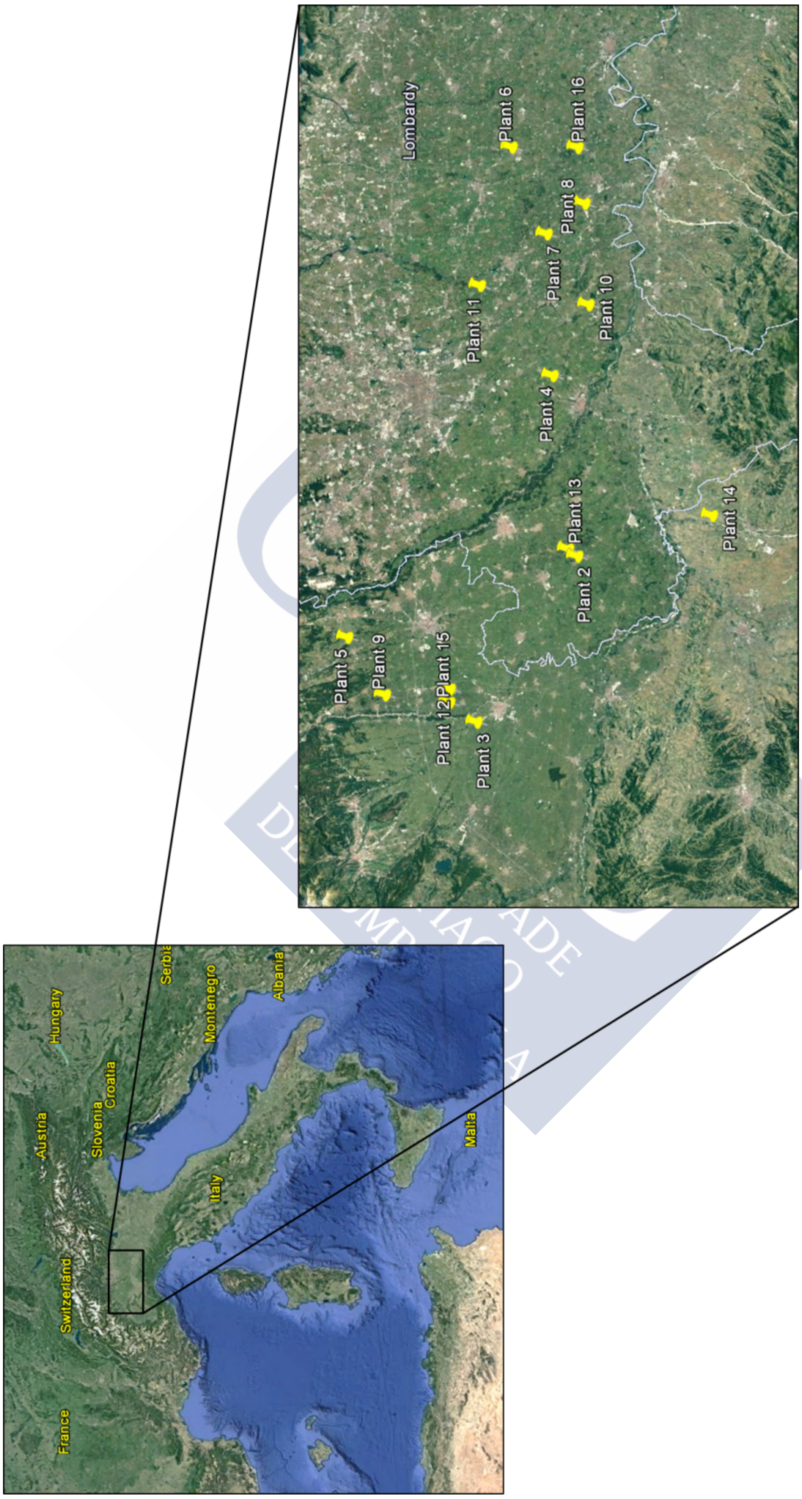


Figure 5.1. Location of the biogas plants under study

Most of the biogas plants under study adopt a co-digestion approach treating energy crops (mainly maize silage and triticale, but also ryegrass and sorghum) mixed with animal waste (pig, cow and chicken manure) in different ratios. A detailed description regarding the feedstocks used in each biogas plant can be found in Table 5.2. The biogas plants have been categorised depending on the ratio of energy crops/animal waste digested. Plants named as "A" mainly use energy crops as substrates for biogas production. In more detail, out of these 15 plants, 7 co-digest energy crops in a ratio higher than 75%; moreover, two of them perform anaerobic mono-digestion of maize. Furthermore, 6 plants, categorised as "B", rank in an intermediate position with 25% and 75% of energy crops. Therefore, only 2 biogas plants digest energy crops in a ratio lower than 25%.

Maize silage is by far the most used feedstock, being digested in 14 biogas plants with different ratios. In addition, other maize products such as maize flour and maize gluten are also consumed in four biogas plants. Regarding other energy crops, triticale is also co-digested in 5 biogas plants; while ryegrass and sorghum are only digested in one plant each. In the same way, pig slurry is widely used in the biogas plants under study, being consumed in different ratios in 12 plants, followed by cattle manure and chicken manure that are digested in 6 and 5 biogas plants, respectively. Additionally, other plants include the digestion of other organic residues or co-products such as SSFW, food waste from supermarkets and glycerol. It is important to highlight that the daily input mixture depends on the seasonal availability of each substrate. In this study, average data from the operation of one whole year was managed.

Table 5.2. Feedstock used in the biogas plants and plant grouping

Plant	Energy crops				Residues				Total			
	Maize	Triticale	Maize flour	Other	Pig waste	Cattle waste	Chicken waste	Food waste	Substrate	Energy Crops	Residues	Plant category [a]
	(t/d)	(t/d)	(t/d)	(t/d)	(t/d)	(t/d)	(t/d)	(t/d)	(t/d)	(%)	(%)	
2	19.0	13.0	-	-	28.0	-	-	-	60.0	53%	47%	B
3	54.0	-	-	-	45.0	-	-	-	99.0	55%	45%	B
4	36.5	-	-	-	-	-	-	-	36.5	100%	0%	A
5	43.9	4.98	-	-	12.6	-	1.35	-	62.9	78%	22%	A
6	9.86	-	-	-	178	-	-	76.7	265	4%	96%	C
7	45.0	-	-	-	45.0	10.0	-	4.00	104	43%	57%	B
8	50.0	-	-	-	50.0	-	-	-	100	50%	50%	B
9	47.1	2.95	-	-	-	14.2	0.39	-	64.6	77%	23%	A
10	38.0	-	0.02	-	7.00	3.00	-	-	48.0	79%	21%	A
11	-	-	7.00	-	30.0	8.60	-	-	46.1	15%	85%	C
12	45.3	1.88	-	3.77	1.06	2.37	0.80	-	55.2	92%	8%	A
13	46.3	4.32	-	0.10	-	6.41	1.35	-	58.5	87%	13%	A
14	58.5	-	-	-	44.0	-	-	-	102.5	57%	43%	B
15	45.2	-	-	-	23.4	-	-	-	69.9	65%	35%	B
16	10.0	-	3.00	-	-	-	-	-	13.0	100%	0%	A

[a] A = Energy crops > 75%; B = 75% > Energy crops < 25%; C = Energy crops < 25%

Regarding the operation of the anaerobic process, most plants operate in wet conditions and mesophilic temperature (40°C) while three plants work at thermophilic range (50°C). The OLR changes among plants but it is kept between 0.83 and 4.50 kg TVS/m^3·d and the HRT also varies among plants, from 20 to 90 days. The biogas produced is burned in a CHP unit, with different total installed electric and thermal power capacities as well as electric and thermal efficiencies. Table 5.3 includes a summary of the main operational parameters in each biogas plant. In addition, as outlined in Table 5.4, digestate management is different among the biogas plants. In 10 of the 15 biogas plants, the produced digestate is separated into its liquid and solid fractions. Biogas plants that digest high ratio of energy crops require the recirculation of part of the produced digestate (or the liquid fraction of the separated digestate) to keep the TS content inside the digester constant (around 6%). This is performed in 7 biogas plants out of 15. Before being applied on land as an organic fertiliser, digestate is stored in the plant, as shown in Figure 5.2. Most plants have open storage; however, some have closed facilities, which entail lower derived emissions.

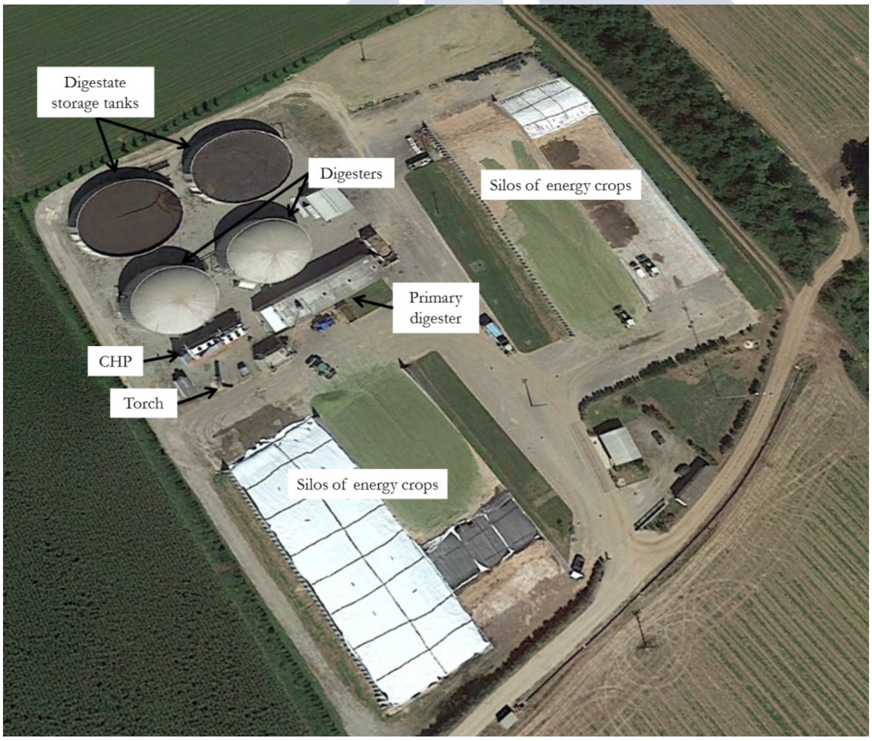


Figure 5.2. View of a typical biogas plant

Table 5.3. Main characteristics and operational parameters of the plants

Plant	Digesters (number)	AD (stages)	T (ºC)	TVS (TVS/d)	OLR (kg TVS/m^3·d)	HRT (d)	Biogas produced (m^3)	Electric power (kW)	Electric efficiency (%)
2	1	Single	40	10.60	3.65	35-45	6043	500	38.5
3	1	Single	40	18.24	3.38	60	11436	999	40.7
4	1	Single	40	10.41	3.85	23-35	6500	520	37.0
5	2+1	Double	44	16.14	2.48	40-55	12746	999	41.0
6	4	Single	40	32.41	4.50	20-30	17423	1664	39.0
7	1+1	Double	45-55	19.32	2.58	40-50	11302	1000	40.9
8	1+1	Double	40-48	17.13	2.45	70	10778	999	40.9
9	2	Single	40	16.85	3.12	66	11770	999	40.5
10	1+1	Double	53	12.36	2.47	31	8897	836	39.0
11	1+1	Double	40	7.68	4.27	45	4320	370	38.0
12	2	Single	40	16.60	3.27	45	10782	999	40.8
13	2	Single	40	17.46	3.36	45	11300	999	40.8
14	2	Single	40	19.62	1.78	20-35	11190	999	40.8
15	3	Single	55	15.36	1.71	50-90	11099	999	40.9
16	1	Single	45	5.58	4.29	30	4395	370	39.0

TVS – total volatile solids; OLR - organic loading rate; HRT – hydraulic retention time

Table 5.4. Digestate management strategies among biogas plants

Plant	Digestate separation	Digestate recirculation	Digestate storage	Surplus digestate
2	no	yes	open	5.22 t/d
3	yes	yes	open	-
4	yes	yes	open	-
5	yes	no	covered (liquid fraction)	-
6	no	no	covered	217 t/d
7	no	no	open	15.2 t/d
8	no	no	open	2.47 t/d
9	yes	yes	covered	-
10	yes	no	open	-
11	yes	no	open	32.9 t/d
12	yes	yes	open	-
13	yes	yes	open	-
14	yes	yes	open	-
15	no	no	open	-
16	yes	no	open	-

5.2.2. The five step LCA + DEA method

The environmental profiles of the 15 biogas plants were computed by the LCA + DEA method to understand their environmental efficiency and to propose a series of benchmarks to improve their environmental performance. The combination of LCA and DEA methodologies has evolved since it first appeared in 2009 (Lozano et al., 2009). The first integrations proposed a 3 steps procedure. Nevertheless, this was upgraded later to a 5 step framework that allowed the environmental evaluation of current and virtual DMUs (Vázquez-Rowe et al., 2010). The five-step LCA + DEA, which was finally the selected option to understand the eco-efficiency of the aforementioned biogas plants, is structured as follows:

- ✓ Data collection of each plant in order to build the LCI.
- ✓ Calculation of the environmental burdens of each plant (LCIA).
- ✓ Computation of the DEA model to obtain the efficiency scores and target projections for each entity.

- ✓ Environmental evaluation of the inefficient plants based on the results obtained in the previous step.
- ✓ Eco-efficiency verification, original and virtual DMUs comparison and results evaluation.

Nevertheless, a slight modification of the method was necessary in order to fit the extensive variety of feedstock used in the 15 biogas plants. As explained before, most plants perform the co-digestion of energy crops with organic residues (see Table 5.1). This means that there is a wide compendium of feedstock with diverse characteristics, entailing different environmental profiles. For the sake of an accurate assessment and to compute the different co-digestion substrates, a homogenisation step was made prior to the DEA computation. This step called direct environmental benchmarking was introduced by Avadí et al. (2014) and consists on the use of the weighted ReCiPe Endpoint single indicator (Goedkoop et al., 2009) to obtain a single value that summarises the environmental profiles of the production of all the feedstocks used in each plant.

5.2.3. LCA methodology

As explained before, the LCA methodology was selected to analyse the environmental performance of the original and the virtual biogas plants.

Goal and scope definition

The main objective of the LCA studies performed was to calculate the environmental profile of each original and virtual plant to quantify the potential environmental gains of applying the LCA + DEA methodology. The approach of the LCA methodology and the assumptions made were in accordance with the previous chapters. Therefore, the FU selected was the production of 1 MWh of electricity and the system boundaries included five subsystems defined: SS1: deedstock production; SS2: feedstock supply; SS3: bioenergy production, SS4: digestate management and SS5: surplus digestate use. As presented in Figure 5.3, they included from the cultivation of energy crops to the production of bioenergy and the use of digestate. Nevertheless, the production of wastes (animal, food and industrial) was excluded since their production would not be influenced by a change in their management.

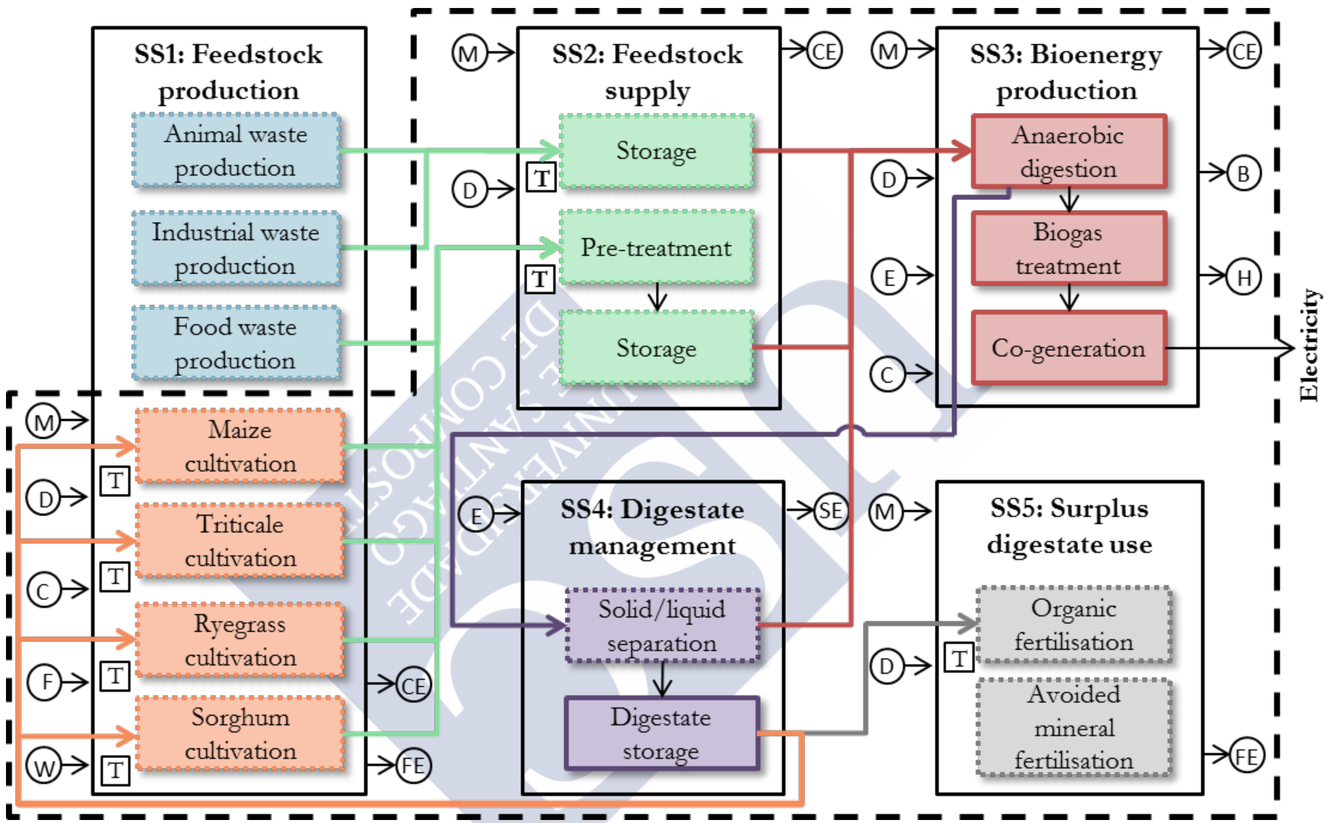


Figure 5.3. Flowchart and system boundaries of the biogas systems. Dotted boxes indicate processes not performed in all biogas plants. Acronyms: T – transport; M – machinery; D – diesel; C – chemicals; F – fertilisers; W – water; E – electricity; B – biogas losses; H – heat dissipated; CE – combustion emissions; FE – fertilisation emissions; SE – storage emissions

The CHP engine has two main co-products. In all the biogas plants, the electricity produced is injected in the grid whereas heat is partially used in the digester; therefore, all impacts were allocated to electricity production. Only one biogas plant (Plant 7) uses the surplus heat for heating nearby greenhouses. In this particular case, an avoided product perspective was considered in this study including the environmental credits derived from avoided heat production. Moreover, in the anaerobic digestion process, biogas and digestate are also co-produced; once again, biogas was considered as the main product. Whenever the production of digestate was higher than the required in the cultivation of energy crops, a system expansion was performed to include the use of the produced digestate in a different agricultural system, including the credits from the avoided use of ammonium nitrate.

Life cycle inventory

The LCI of each plant was developed using primary data from the plant as well as following the assumptions explained in previous chapters. In detail, regarding the production of the cereals silage (maize, triticale, ryegrass and sorghum), input and output data regarding the agricultural stage were collected by means of interviews with the farmers. More detailed information concerning these agricultural steps can be found in Bacenetti et al., (2014); González-García et al., (2013) and Noya et al., (2017). The amount of digestate applied on land was calculated according to the nitrogen supplied by the organic fertilisation practices of the crop. Emissions derived from the application of both mineral and organic fertilisers were computed according to Rossier and Charles (1998) and Brentrup et al. (2000).

Furthermore, primary specific data concerning the operation of each DMU were provided by plant workers for the year 2012. Regarding feedstock supply to the facilities, different transport distances were considered depending on the specific biogas plant. In the case of animal waste, it is usually transported by tractors or directly pumped from the nearby farm. Despite it is usually performed on a daily basis, outdoor storage for 3 days was accounted for the manure. Concerning energy crops, they are transported to the plant after being harvested; where they are usually ensiled and stored. As a result of ensiling, 10% of the original mass is lost (Bacenetti and Fusi, 2015). In addition, maize flour and gluten are also used as substrates in some biogas plants; in this case, the corresponding treatment of the maize outside the biogas plant was taken into account.

Direct emissions of biogas and nitrogen compounds to air from the storage of digestate were also computed according to De Vries et al. (2012b). Finally, as mentioned, some plants produce more digestate than the required for the cultivation of their own crops. In this case, the avoided mineral fertilisation using ammonium nitrate was estimated according to the nitrogen replacement value of 65% (Bacenetti et al., 2016a; De Vries et al., 2012b). It has built the specific LCI of each plant using average data from the operation of 1 year. Due to the number of plants dealt, only the average LCI of the three categories of biogas plants in terms of their feedstock is presented in Tables 5.5, 5.6, 5.7, 5.8 and 5.9.

Finally, background data regarding the production of all required inputs such as diesel fuel, chemicals, lubricant oil and mineral fertilisers as well as CHP emissions were taken from ecoinvent® version 3 database (Wernet et al., 2016).

Table 5.5. Global inventory data regarding feedstock production

	A		B		C		
Materials and fuels							
Digestate	3.79	(±1.29)	4.03	(±0.52)	0.76	(±0.25)	t
Seeds	0.77	(±0.33)	0.67	(±0.10)	0.17	(±0.13)	kg
Pesticides	0.22	(±0.04)	0.24	(±0.05)	0.05	(±0.04)	kg
Urea	2.22	(±0.45)	2.43	(±0.60)	0.54	(±0.40)	kg
Ammonium nitrate	0.18	(±0.16)	0.32	(±0.79)			kg
Diesel	28.1	(±15.2)	16.8	(±14.6)	3.30	(±3.49)	kg
Tractor	3.19	(±1.85)	1.81	(±1.79)	0.34	(±0.38)	kg
Agricultural tillage	7.74	(±5.10)	2.22	(±3.55)	0.78	(±0.98)	kg
Transport							
Tractor and trailer	351	(±291)	1,065	(±2,203)	18.0	(±20.0)	t·km
Resources							
Water	124	(±26.6)	126	(±20.7)	32.2	(±24.2)	m^3

Table 5.6. Global inventory data regarding feedstock production (cont.)

	A		B		C		
Products							
Straw	2.35	(±0.48)	2.48	(±0.35)	0.58	(±0.44)	t
Emissions to air							
Ammonia	2.73	(±0.61)	6.04	(±7.96)	0.70	(±0.53)	kg
Nitrous oxide	0.21	(±0.05)	5.00	(±11.7)	0.54	(±0.64)	kg
Nitrogen	0.96	(±0.21)	1.33	(±0.85)	0.24	(±0.18)	kg
Emissions to water							
Nitrate	5.72	(±1.14)	4.56	(±1.97)	2.81	(±3.04)	kg
Phosphate	0.10	(±0.03)	0.20	(±0.07)	0.11	(±0.10)	kg

Table 5.7. Global inventory data regarding feedstock supply

	A		B		C		
Materials and fuels							
Straw	2.35	(±0.48)	2.48	(±0.35)	0.58	(±0.44)	t
Animal waste	0.32	(±0.27)	1.96	(±0.48)	4.47	(±0.00)	t
Food waste			0.67	(±1.63)	1.21	(±1.01)	t
Diesel	1.03	(±0.21)	1.09	(±0.15)	0.26	(±0.19)	kg
Tractor	0.06	(±0.01)	0.07	(±0.01)	0.02	(±0.01)	kg
Agricultural tillage	0.05	(±0.01)	0.06	(±0.01)	0.01	(±0.01)	kg
Transport							
Tractor and trailer	137	(±250)	380	(±804)	34.0	(±19.0)	t·km
Products							
Silage	2.14	(±0.44)	2.26	(±0.32)	0.53	(±0.40)	t
Animal waste	0.32	(±0.27)	1.96	(±0.48)	4.47	(±0.00)	t
Food waste			0.67	(±1.63)	1.21	(±1.01)	t

Table 5.8. Global inventory data regarding bioenergy production

	A		B		C		
Materials and fuels							
Silage	2.17	(±0.44)	2.33	(±0.28)	0.53	(±0.40)	t
Animal waste	0.32	(±0.27)	1.96	(±0.48)	4.47	(±0)	t
Food waste			0.67	(±1.63)	0.96	(±1.36)	t
Industrial residues					0.25	(±0.35)	t
Digestate (recirculated)	3.98	(±3.77)					t
Water	2.19	(±3.29)					t
Diesel	0.34	(±0.07)	0.36	(±0.04)	0.08	(±0.06)	kg
Tractor	0.04	(±0.01)	0.05	(±0.01)	0.01	(±0.01)	kg
Agricultural tillage	0.03	(±0.01)	0.03	(±0)	0.01	(±0.01)	kg
Chemicals	0.02	(±0.05)	0.02	(±0.03)			kg
Lubricant oil	0.33	(±0.14)	0.25	(±0.08)	0.19	(±0.25)	kg
Biogas plant	$1 \cdot 10^{-5}$	($\pm 1 \cdot 10^{-5}$)	$1 \cdot 10^{-4}$	($\pm 6 \cdot 10^{-6}$)	$1 \cdot 10^{-4}$	($\pm 1 \cdot 10^{-5}$)	p
Co-generation unit	$4 \cdot 10^{-5}$	(±0)	$4 \cdot 10^{-5}$	(±0)	$4 \cdot 10^{-5}$	(±0)	p
Energy							
Electricity	73.5	(±18.5)	74.5	(±14.3)	160	(±106)	kWh
Products							
Electricity	1,000	(±0)	1,000	(±0)	1,000	(±0)	kWh
Heat	1,180	(±169)	1,007	(±175)	1,252	(±138)	kWh
Digestate	6.70	(±3.47)	6.22	(±3.45)	5.51	(±0.98)	t
Avoided products							
Avoided heat			119	(±291)			kWh
Emissions to air							
Carbon dioxide	38.8	(±15.0)	33.1	(±10.3)	44.8	(±36.9)	kg
Methane	1.80	(±1.25)	2.13	(±1.05)	2.67	(±0.10)	kg
Carbon monoxide	19.6	(±9.16)	16.1	(±5.37)	22.8	(±20.7)	g
Nitrogen oxides	6.11	(±2.86)	5.02	(±1.68)	7.11	(±6.47)	g
NMVOC	0.81	(±0.38)	0.67	(±0.22)	0.95	(±0.86)	g
Nitrous oxide	1.02	(±0.48)	0.84	(±0.28)	1.19	(±1.08)	g
Sulphur dioxide	8.56	(±4.01)	7.03	(±2.35)	9.96	(±9.07)	g

Table 5.9. Global inventory data regarding digestate management

	A		B		C		
Materials and fuels							
Raw Digestate	6.70	(±3.47)	6.22	(±3.45)	5.51	(±0.98)	t
Products							
Digestate (for own crops)	2.72	(±1.73)	2.90	(±1.78)	0.76	(±0.77)	t
Surplus digestate			0.12	(±0.25)	4.74	(±1.23)	t
Emissions to air							
Ammonia	0.32	(±0.55)	0.37	(±0.18)	0.27	(±0.22)	kg
Nitrous oxide	0.24	(±0.23)	0.18	(±0.23)	0.17	(±0.23)	kg
Nitrogen	0.44	(±0.32)	3.41	(±7.77)	0.31	(±0.39)	kg
Nitrogen oxide	0.16	(±0.15)	0.12	(±0.16)	0.12	(±0.16)	kg
Methane	0.82	(±1.69)	0.60	(±0.13)	0.45	(±0.34)	kg
Carbon dioxide	3.53	(±6.85)	1.91	(±0.40)	1.84	(±1.21)	kg

Table 5.10. Global inventory data regarding surplus digestate use

	A	B		C		
Materials and fuels						
Digestate		0.19	(±0.27)	4.74	(±1.23)	t
Diesel		0.04	(±0.06)	1.02	(±0.27)	kg
Tractor		0.01	(±0.01)	0.13	(±0.03)	kg
Agricultural tillage		0.01	(±0.01)	0.27	(±0.07)	kg
Transport						
Transport and trailer		0.92	(±1.42)	30.2	(±20.8)	t·km
Emissions to air						
Ammonia		0.14	(±0.20)	2.56	(±0.54)	kg
Nitrous oxide		0.01	(±0.01)	0.19	(±0.04)	kg
Nitrogen		0.05	(±0.07)	0.88	(±0.19)	kg
Emissions to water						
Nitrate		0.29	(±0.40)	5.11	(±1.09)	kg
Phosphate		0.01	(±0.03)	0.24	(±0.21)	kg
Avoided materials and fuels						
Ammonium nitrate		0.42	(±0.59)	7.71	(±1.64)	kg
Avoided emissions to air						
Ammonia		0.01	(±0.01)	0.19	(±0.04)	kg
Nitrous oxide		0.01	(±0.01)	0.15	(±0.03)	kg
Nitrogen		0.04	(±0.05)	0.68	(±0.14)	kg

Life cycle impact assessment

The life cycle inventory gathered for each individual biogas plant was converted into life cycle environmental impacts. Firstly, CC was determined by considering the characterisation factors provided by IPCC (2013). Moreover, the ReCiPe Midpoint H methodology (Goedkoop et al., 2009) has been also applied to quantify other environmental impacts produced in four additional impact categories: FE, ME, TA and ALO.

5.2.4. DEA methodology

DEA model selection

DEA methodology offers a range of different models with different characteristic and properties. For this case study, two different models were tested for the available dataset: CCR model and SBM model (Charnes et al., 2013). The aforementioned models were run under different conditions (i.e. input/output oriented, constant/variable return to scale) and the results were evaluated to determine the best fitting one. Finally, the SBM model was selected based on its advantages for the matrix computation, since it allows calculating the efficiency scores regardless the units of measure used for the set of inputs and outputs (Thrall, 1996). Another intrinsic characteristic of the SMB model is that it follows a non-radial approach. Convexity, scalability and free disposability of inputs and outputs are assumed for the determination of the efficient production frontier. On top of that, this characteristic makes it more suitable to analyse matrices with low correlation within their inputs (Charnes et al., 2013). Finally, the SBM model provides projections for the minimisation of inputs and/or outputs, based on the slacks with respect to the production frontier for the inefficient DMUs, to attain the target theoretical environmental profile of the sample (Cooper et al., 2007).

Regarding the conditions under the model was finally run, an input-oriented approach was selected for this case study, based on two main factors. Firstly, the study aims at minimising the use of resources (i.e. inputs) and related environmental impacts without a reduction in final electricity production (i.e. output). Secondly, the model selection is based on the fact that electricity production per cubic meter of biogas is limited by the features of the technology selected.

Another important decision with regard to the model conditions for its computation is the selection of the return-to-scale approach: constant or variable. When applying constant return to scale (CRS), the model will identify the most efficient DMU in the sample and will make a straight line from the origin and through the efficient DMU (in the more simplistic scenario, a single input and a single output). With this approach, it is difficult to integrate economies of scale; and thus, it will be difficult for small and large-sized DMUs to be deemed under CRS conditions (Thanassoulis, 2001). Nevertheless, the variable return-to-scale (VRS) approach incorporates economies of scale and detects both increasing and decreasing economies of scale on the efficient frontier. In the current case study, VRS was assumed for the final DEA matrix, based on the fact that the operational sizes within the analysed biogas plants were very different. Thus, the calculated production frontier will be created based on the best performing DMUs at different sizes (Banker et al., 1984).

Input/output selection

The DEA matrix employed for the study was composed by 3 inputs and one single output, as shown in Table 5.11: I1) feedstock production; I2) consumed electricity; I3) transport; and O1) produced electricity. The input selection was based on their operational importance and environmental load. In the first iterations of the study, another input (infrastructure) was included; nevertheless, for the final computation of the matrix, this input was discarded for two main reasons. The first one was the little insight that this input provided to the results due to the similarity among the different plants. The second one was the proximity between the sample size (15) and the rule of thumb (12), which is used to determine the minimum size of the sample to elaborate the DEA matrix. Inputs 2 and 3 involve direct computation from the LCIs used for the environmental assessment (step 2) to the matrix, which is a common procedure in all LCA+DEA studies (Vázquez-Rowe and Iribarren, 2015). I2 represents the energy taken from the grid consumed for the operation of the plant taking into account the pre-treatment operations, mixing, pumping and sludge dewatering. With regard to I3, it gathers the transport of the different feedstocks from their production centre to the biogas plant; it considers the amount of feedstock delivered and the transport distance. As explained before, in the case of input 1 (i.e. feedstock) and due to the complexity of this issue, a previous homogenisation step was necessary. Thus, I1 does not refer to a single inventory item, but to a compendium of environmental impacts related to the production of the different substrates. With regard to the output selection, which should reflect the main function of the production system, the gross electricity produced in the CHP units was chosen. Within the 15 plants included in the sample, all of them are characterised for selling all the energy produced by the combustion of the biogas to the grid. Self-consumption of a fraction of the produced energy is not performed; the energy required to cover the plant operation (i.e. input 2) comes directly from the grid, because this allows maximising the benefits from the energy sale. The final selection of input and outputs encompass to a great extent the principal items and environmental *hotspots* extracted from the environmental characterisation of the DMUs performed in step 2. The final matrix used for the computation of the efficiency of the plants is presented in Table 5.11.

Table 5.11. DEA matrix (inputs and outputs referred per day)

	Inputs			**Output**
Plant	**Feedstock** (Pt/d)	**Consumed electricity** (kWh/d)	**Transport** (t·km/d)	**Produced electricity** (kWh/d)
2	428	659	87.8	11,965
3	630	1,854	387	22,224
4	384	1,175	110	11,353
5	572	1,143	136	22,759
6	117	9,366	1,249	39,820
7	511	1,675	2,257	23,930
8	565	1,594	365	22,777
9	616	1,659	467	23,696
10	437	1,758	93.7	20,064
11	147	734	54.2	8,636
12	827	1,784	718	22,893
13	629	1,256	398	23,265
14	915	1,706	5,299	23,906
15	521	2,289	108	23,568
16	177	630	127	8,854

5.3. Results

5.3.1. Environmental assessment of current DMUs

The obtained characterisation results of each biogas plant can be found in Table 5.12 referred to the FU, which is 1 MWh_e.

Table 5.12. Characterisation results of each plant per FU

Plant	CC (kg CO_2 eq)	FE (kg P eq)	ME (kg N eq)	TA (kg SO_2 eq)	ALO (m^2a)
2	346	0.10	1.52	11.9	358
3	478	0.11	1.83	10.5	213
4	522	0.09	2.15	11.6	256
5	316	0.06	1.40	8.27	210
6	234	0.17	1.68	8.37	49.9
7	161	0.10	1.64	9.14	165
8	295	0.07	1.59	9.23	183
9	306	0.07	1.47	8.50	200
10	310	0.07	1.41	8.01	168
11	349	0.12	1.68	9.71	81.2
12	428	0.10	2.11	9.27	321
13	431	0.07	1.52	9.06	213
14	612	0.14	1.87	11.3	231
15	283	0.05	1.40	7.59	171
16	307	0.05	1.09	6.33	131

Regarding CC, the cultivation of energy crops (SS1) represented an important source of GHGs, especially for the plants digesting higher ratio of energy crops, categorised as A and B, accounting between 104 and 264 kg CO_2 eq/FU and representing between 34% and 80% of the total burdens in this impact category. Digestate management (SS4) was also identified as an important *hotspot* concerning CC due to direct emissions derived from the storage of the digestate. Biogas plants that perform the separation of the digestate and store the solid fraction in open piles achieved higher GHG emissions in this subsystem due to nitrous oxide emissions (see Table 5.4). GHG emissions derived from the production of bioenergy (SS3) did not differ significantly among biogas plants; with the exception of Plant 7 that achieved better results since it is the only one that takes advantage of the surplus heat in a nearby greenhouse. Furthermore, feedstock supply (SS2) had no remarkable environmental impacts, aside from Plant 14 (111 kg CO_2 eq/FU) due to particularly long transport distances (maize is transported 55 km and pig slurry 40 km). In the same way, the consumption of electricity was only relevant in Plant 6 due to the required pre-treatment of the food waste entering the plant.

Phosphate and nitrate leaching due to digestate application on land were the most important contributors to FE and ME, respectively. As expected, these emissions were mainly produced during the cultivation of energy crops (SS1), especially for those biogas plants categorised as A and B, because they consume a high ratio of energy crops. Whereas phosphate emissions were associated to the composition of the applied digestate, nitrate emissions depended on the nitrogen balance, which is different for each energy crop. More specifically, these environmental burdens caused in these biogas plants, in average, 79% of the impacts in FE and 95% in ME. Despite the fact that Plants 6 and 11 consume much lower amount of energy crops (plants named as C), the surplus digestate was also applied on agricultural land (SS5), which also entailed environmental impacts derived from phosphate and nitrate leaching. Linked with the previous, the cultivation of the energy crops used in the biogas plants was identified as the key environmental *hotspot* regarding TA. For those biogas plants that digest high ratio of energy crops (categories A and B), the cultivation of cereals represented 85.3% of the environmental burdens produced in TA. Whenever more digestate is produced than the required in the cultivation of the energy crops, ammonia is also emitted in the use of the digestate in other agricultural systems but it is partially offset due to the avoided use of mineral fertilisers. The storage of the digestate in each facility also entails ammonia emissions, which directly depends on the TAN content of the produced digestate. It is linked with the feedstock selected as well as the degradation of organic nitrogen produced inside the digester.

Finally, ALO was directly influenced by the use of energy crops in each biogas plant. The results in this impact category essentially depended on the amount of energy crops used per MWh of electricity produced and the yield of each cereal per hectare. In more detail, while maize and sorghum yields are 65.1 t/ha and 51.4 t/ha, respectively, triticale and ryegrass yields are much lower, being 37 t/ha and 11.4 t/ha, respectively, meaning that for the production of the same amount of energy crops, different cultivation areas are needed. In addition, the cultivation of winter crops is longer in time compared with summer crops.

5.3.2. DEA analysis

The DEA matrix used in the analysis was prepared based on the available LCI and the homogenisation step described in section 5.2.2. The matrix (Table 5.11) was introduced in the DEA-Solver Professional Release 10.0 software (Cooper et al., 2007) and ran under an input-oriented SBM model as described in section 5.2.4. The main results extracted from the model computation presented in Table 5.13 are the efficiency scores for each of the units under assessment and the proposed reductions for the operational inputs in those DMUs deemed as inefficient.

Table 5.13. Target reduction percentages for operational inputs and efficiency scores

Plant	Reduction in inputs			Efficiency
	Feedstock	Consumed electricity	Transport	
	(%)	(%)	(%)	(%)
2	0	0	0	100
3	11.9	39.2	65.7	61.1
4	40.3	30.9	36.2	64.2
5	0	0	0	100
6	0	0	0	100
7	0	0	0	100
8	0	24.4	61.4	71.4
9	11.2	3.91	57.8	75.7
10	0	0	0	100
11	0	0	0	100
12	31.3	31.3	79.8	52.2
13	0	0	0	100
14	40.9	0.62	96	54.2
15	0	0	0	100
16	0	0	0	100

Among the 15 plants included in the study, a total of 9 were found fully efficient (i.e. efficiency score of 100%). The percentage of efficient DMUs with respect to the whole sample (60%) is relatively high in comparison with previous studies of LCA+DEA applied to similar systems. Lorenzo-Toja et al. (2015) performed an eco-efficiency evaluation of 113 wastewater treatment plants (WWTPs) and only 9.7% were benchmarked as efficient, this can be explained due to the clear heterogeneity of the WWTPs sample. In addition to the number of efficient DMUs, the average efficiency for the sample was 85% (±19%), which show a high operational efficiency among the sample, with only one plant operating close to 50% of efficiency (Plant 12).

On top of that, the correlation between the different inputs and the output was also calculated (Table 5.14), only in the case of I2 (electricity consumed) and O1 (electricity produced), the correlation coefficient (0.7673) showed a linear tendency between both variables. For the other variables, no linearity among the inputs and the output was found and the correlation coefficient was close to zero.

Table 5.14. Correlation indexes among inputs and outputs of the DEA matrix

	Input 1	**Input 2**	**Input 3**	**Output**
Input 1	1.000	-0.301	0.443	0.251
Input 2	-0.301	1.000	0.174	0.791
Input 3	0.443	0.174	1.000	0.348
Output	0.251	0.791	0.348	1.000

In the case of the 6 non-efficient plants, the target reduction percentages for the corresponding inputs are gathered in Table 5.13. It has to be noted that these reductions are based on the theoretical operational efficiency frontier and sometimes they could be unachievable in reality due to technical or practical limitations. Nevertheless, the reduction percentages are moderate and only in the case of I3, (transport) reductions over 50% are proposed. Thus, the application of these reductions to the operation inputs will allow the inefficient DMUs to operate at full efficiency without hindering the output production. As it can be observed, no inputs reductions are available for the efficient plants, because their target operational benchmarks already match with the actual operating conditions.

5.3.3. Environmental assessment of virtual DMUs

The final stage of the five-step LCA + DEA methodology consists on the application of the life cycle impact assessment to the virtual DMUs. Accordingly, the virtual DMUs were calculated by applied the reduction target values obtained by DEA for the inefficient plants. In this way, the environmental savings due to an efficient operation could be estimated by comparing the environmental profile of the virtual DMUs with the original plants. The objective of this final step is to prove that an increase in the efficiency of the operation of the biogas plants can lead to an improvement on their environmental performance. For comparison purposes, the same inventory assumptions and impact assessment methodology were applied in the virtual DMUs. In more detail, the Midpoint results of the original and the virtual DMUs were compared in terms of CC (Figure 5.4), FE and ME (Figure 5.5) and TA and ALO (Figure 5.6). In addition, Figure 5.4 also shows the profile of the Italian electric profile in terms of GHGs as reference.

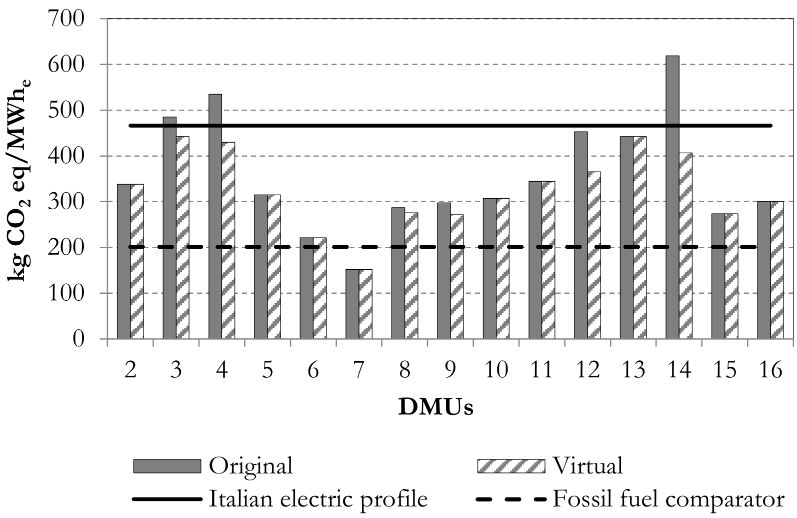


Figure 5.4. Midpoint results regarding CC for original plants (plain) and virtual targets (striped)

As shown in Figure 5.4, the production of 1 MWh$_e$ entailed higher GHG emissions in three original DMUs than the Italian electric profile (Plants 3, 4 and 14). These DMUs have in common that they are the biogas plants that digest more maize per FU (between 2.67 and 3.22 t energy crops/MWh$_e$). There are other biogas plants (Plant 2) that digest higher ratio of energy crops (2.94 t/MWh$_e$); however, the impact is lower since an important share consists on triticale (1.20 t/MWh$_e$). In addition, these plants store the solid fraction of the digestate separately, entailing higher direct emissions of nitrous oxide, which helped to increase the environmental impacts in CC.

Nevertheless, it can be observed in Figure 5.4 that all virtual plants calculated after the suggested reductions in inputs achieved lower GHG emissions than the reference system, proving that the reductions calculated in the DEA would allow the most polluting DMUs to improve their environmental profile.

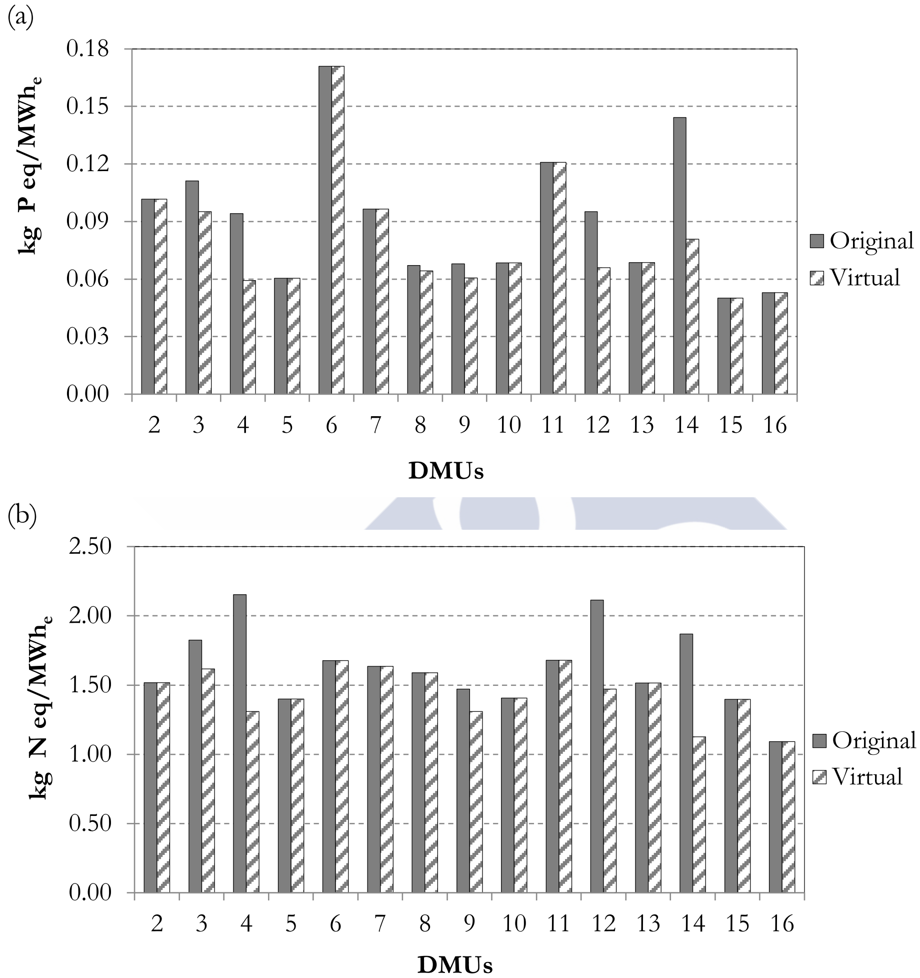
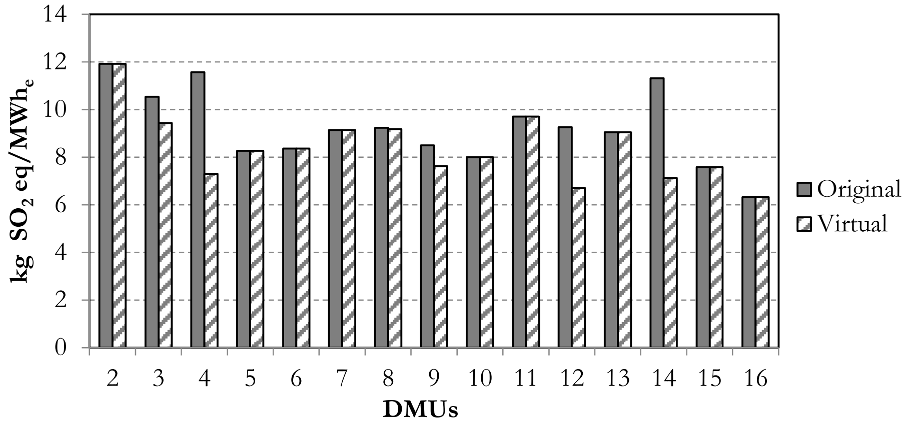


Figure 5.5. Midpoint results regarding (a) FE and (b) ME for original plants (plain) and virtual targets (striped)

From Figure 5.5 it can be seen that the DMU causing the largest impact in FE (Plants 6) was identified as efficient; therefore, no reductions were established for this DMU. The reason for these results is related to the phosphorus content of the digestate, which is directly linked with the type of feedstock digested. In more detail, pig slurry and food waste are the substrates with higher content of phosphorus. With regard to ME, the target reductions affected the DMUs with higher environmental impacts. These biogas plants consume a high ratio of energy crops for the production of electricity, especially maize which requires greater amount of digestate, even from other biogas systems, entailing higher nitrate leaching.

(a)



(b)

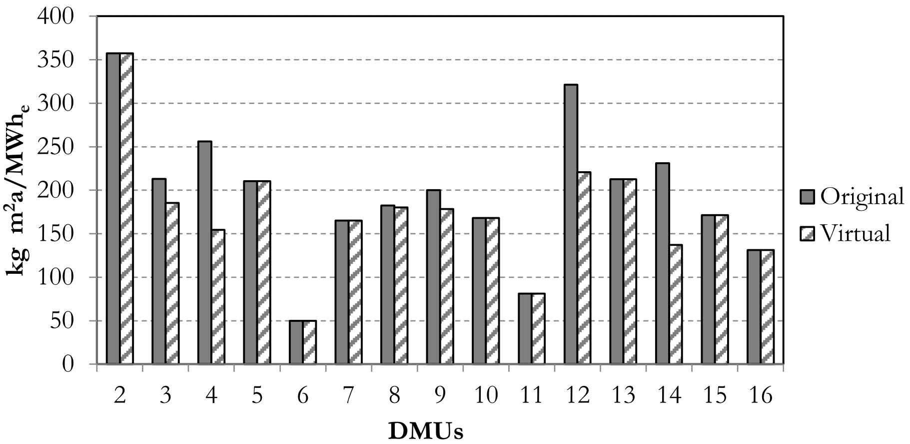


Figure 5.6. Midpoint results regarding (a) TA and (b) ALO for original plants (plain) and virtual targets (striped)

Concerning TA and ALO, the target reductions were again not always linked with the DMUs that cause larger environmental impacts. For example, Plant 2 produced the highest environmental burdens in these impact categories; however, it has been identified as efficient. In the case of TA, environmental impacts were not only linked with high energy crops consumption, but also with the storage of the raw digestate in open tanks, which has not been included in the DEA matrix. With regard to ALO, this DMU consumes high ratio of triticale and this crop requires less digestate application and is less energy intensive than maize. However, the productivity per hectare and year is much lower, causing higher environmental impacts. Due to the variability of the results among impact categories, the ReCiPe Endpoint H methodology was applied to provide a full view of the environmental performance of each DMU and the environmental gains achieved. The comparative results obtained are presented in Figure 5.7.

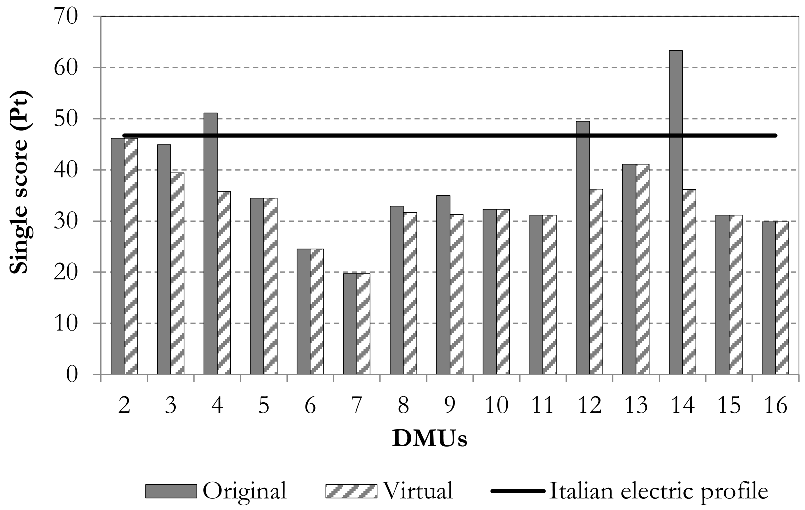


Figure 5.7. Endpoint results expressed as single score for original plants (plain) and virtual targets (striped)

As displayed in Figure 5.7, the reduction targets affected the DMUs showing the worst environmental impact. More deeply, the environmental profiles of inefficient DMUs were improved between 3.64% and 42.8%. Moreover, Plants 4, 12 and 14, which exhibited the highest environmental impacts in terms of single score, were able to improve their environmental profile enough to attain values below the reference of the Italian electricity mix. However, it is important to consider that the methodology still lacks on Endpoint models for ME.

5.4. Discussion

5.4.1. Parameters influencing environmental efficiency

On top of the discussion regarding the environmental performance of the original and virtual biogas plants presented in the results section, a statistical analysis was conducted to cast some light on those parameters influencing their eco-efficiency. It should be noted that the eco-efficiency is calculated by means of the DEA computation taking into account the inputs and outputs considered in the matrix. As explained in section 5.2.4, the input/output selection was performed based on the operational importance and environmental load. Nevertheless, the limitation in the number of inputs/outputs may lead to neglect some environmental impacts in the eco-efficiency assessment. With regard to the underlying factors affecting the biogas plants eco-efficiency, the analysis was divided into two different lines: the role of the type of feedstock and the influence of the design and operational parameters of the plant (TVS, OLR, HRT, temperature of operation and CHP power).

In order to understand the connection between the type of feedstock and the plants efficiency, two groups were formed based on the data available in Table 5.1: mixed plants (<75% energy crops) and energy crop plants (>75% energy crops). An ANOVA analysis was performed to clear up if there were significant differences between the eco-efficiency results of both groups. Prior to apply the ANOVA statistical method, it is important to check that the following conditions are fulfilled: independence of the observations, homogeneity of variances and normality of the sample. On one hand, Levene test results (p-value: 0.968) were used to confirm the homogeneity of variances. On the other hand, Shapiro-Wilk normality test results for both groups (p-values: 0.016 and 0.027) showed that normality does not hold in both cases. As normality is rejected, ANOVA test cannot be applied; thus, the U-Mann Whitney test (a non-parametric alternative) is used. The p-value extracted from the U-Mann Whitney method (0.838) confirms the null hypothesis, indicating that there are no significant differences between the groups. The previous statement indicates that within the studied sample, the effect of the main feedstock used (i.e. energy crops or residues) is not relevant in the eco-efficiency levels attained by the different biogas plants. Despite that the production of energy crops is more environmentally demanding that the use of animal wastes or other residues, the higher biogas production efficiency of the former offset the whole system results.

Regarding the influence of the design and operational parameters of the anaerobic digesters on the eco-efficiency results, Pearson product-moment correlation and its significance test were used. Table 5.15 shows the corresponding results for all the parameters under assessment.

Table 5.15. Pearson product-moment correlation p-values and significance

	Digester temperature	TVS	OLR	HRT	CHP power
Efficiency (Person coefficients)	0.447	-0.113	0.170	-0.097	-0.093
Significance test (p-values)	0.095	0.688	0.544	0.731	0.745

TVS – total volatile solids; OLR – organic loading rate; HRT – hydraulic retention time; CHP – cogeneration heat and power

P-values from the significance test all over 0.05 (interval of confidence, 95%) indicate that the null hypothesis (correlation equals 0) is not rejected in any case. Therefore, no clear influence of any of the aforementioned parameters on the eco-efficiency levels was found. Previous LCA studies on biogas production from the co-digestion of different substrates indicated that some of the operational parameters showed clear influence on the environmental burdens of the different subsystems (Rodriguez-Verde et al., 2014). Nevertheless, in the case of the eco-efficiency assessment, none of these parameters has been highlighted as an important driving force of the whole system efficiency. In fact, it should be noted that the biogas plants, their associated feedstock production and the digestate management constitute complex systems whose eco-efficiency is controlled by a compendium of underlying factors (plant size, type of feedstock, digestate management and composition) rather than by a single operational parameter. This is in line with the results found by Lorenzo-Toja et al. (2015) for a sample of 113 WWTPs.

5.4.2. The role of digestate in LCA of biogas

As a result of the anaerobic digestion process, several properties of the manure substrate changes (Anderson-Glenna et al., 2013; Möller and Müller, 2012), including higher values of pH, ammonium/total nitrogen ratio and mineral nitrogen content as well as lower total organic carbon. One of the strengths of anaerobic digestion is that nutrients are preserved and can be used as fertiliser in agricultural production. Nowadays, crops are cultivated far from farms and animal feed is often brought in from other areas, even imported from other countries. This means that balancing nutrients available from biogas plants to nutrient needs in agricultural land is a significant challenge and requires careful management (IEA Bioenergy, 2016). In the past years, due to the very favourable policy framework, the spread of agricultural biogas plants performing co-digestion schemes based on manure and other co-substrates such as energy crops has been observed. Co-digestion offers the possibility of controlling the feedstock composition to enhance the quality of the produced digestate. However, the dominance of efficiency criteria for methane production can lead to a shorter HRT than the time necessary for full stabilisation of the digestate (Nkoa, 2014). In any case, the introduction of a wide range of possible substrates for co-digestion entails the increase of digestate production, and therefore, the available nutrients. This fact, together with the accumulation of biogas plants in certain regions, might lead to an oversupply of digestate in specific areas (Rehl and Müller, 2011).

Specifically for the Po Valley, Mela and Canali (2014) estimated that around 1.9 billion m^3 of biogas were produced in the Po Valley from around 4.9 million tons of maize silage in 2012; meaning that more than 80,000 ha were cultivated for bioenergy purposes. These authors also pointed out that the upward trend of this phenomenon: in 2007 the total maize area used for biogas production in the Po Valley represented just 0.4%, it grew up to 5.2% in 2011 and in 2012 it as accounted for more than 10% (Negri et al., 2014). In accordance with the authors, this sudden increase in biogas production may have consequences on the local agricultural production structure, since government subsidies are granted for biogas plants using substrates transported from a maximum distance of 70 km (Mela and Canali, 2014).

As a result, water pollution due to nitrate leaching in certain Italian areas is considerable. The European Nitrates Directive, implemented in 1991, intended to improve water quality in Europe by preventing pollution of ground and surface water by nitrate leaching from agriculture (EEC, 1991). Member States are required to designate as NVZ all the land areas in their territory that drain into polluted waters or waters at risk of pollution if no action is taken. The identification of polluted water includes ground and surface water; in particular, those used or intended for the abstraction of drinking water, containing or that could contain (if no action is taken to reverse the trend) nitrates concentrations higher than 50 mg/L. This has forced local administrators and government to introduce stringent regulations regarding the use of manure and digestate resulting in national action plans. The Italian Regulation concerns both NVZ and non NVZ; the amount of nitrogen from organic substrates to be spread is limited to 170 kg/ha in NVZ. In Northern Italy, although the high amount of nitrate leaching, the average nitrate concentration in the groundwater of Po plain in 2012 was relatively low (64% of wells have concentrations lower than 25 mg/L). In Lombardy, wells with nitrates concentration superior than 40 mg/L are located in the upper sector of the plain, where soils are very permeable and the impact of urban wastewater is high. Surface water in Po plain shows nitrates concentration lower than 10 mg/L in 62% of the monitored sites. The European Commission granted to 5 regions of Northern Italy (Emilia-Romagna, Lombardia, Piemonte and Veneto) with a derogation (EU, 2011), which increased the nitrogen from organic substrates that can be spread up to 250 kg/(ha·year). In addition, 2/3 of the effluent must be spread by June 30 every year, while the remaining 1/3 must be used by November 1. It is also agreed that techniques of manure spreading resulting in low emission of nitrogen must be used. Nutrient leaching potential from application of digestate depends on factors such as fertilisation strategies (e.g. time and methods of application), soil texture, topography, meteorology and cropping systems (Nkoa, 2014). Nutrient leaching could be mitigated by better management practices such as adjustment of digestate dosage according to the nutrient demand of the specific crop, synchronisation of nutrient release with crop requirements, injection into the soil, avoidance of applications in autumn, long time gaps between digestate application and incorporation into the soil (Nkoa, 2014).

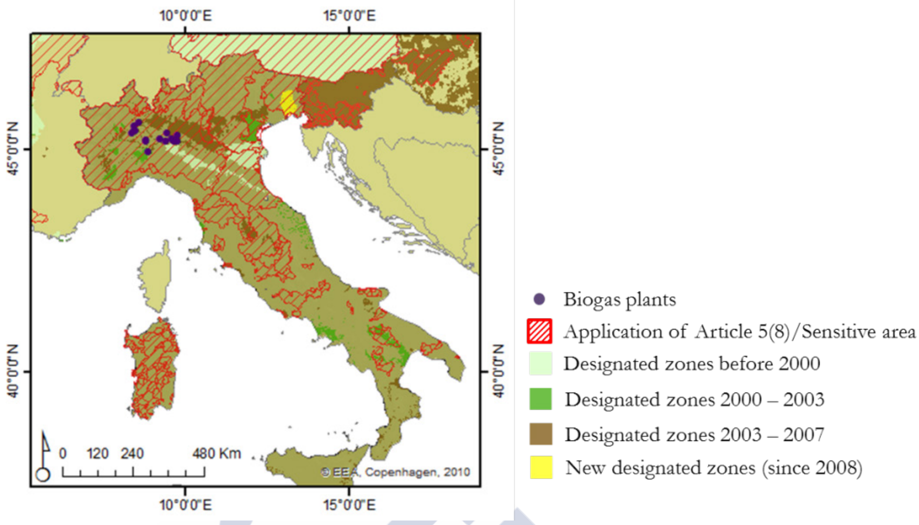


Figure 5.8. Situation of the biogas plants under study regarding the designated NVZ in Italy

With regard to LCA studies, allocation between biogas and digestate production is avoided by following the recommendations of ISO 14040 (2006); therefore, a system expansion is applied to include the use of digestate as an organic fertiliser (Lansche and Müller, 2012; Poeschl et al., 2012). In the system expansion, the environmental loads from the application of the digestate are included and the production and use of the equivalent amount of mineral fertilisers are subtracted. In other words, the system is expanded to include the use of the digestate and the substitution of mineral fertilisation. However, this should be considered carefully since digestate produced from energy crops may be used in their cultivation and digestate from manure have the same NPK content than the untreated manure. Additionally, this avoided process typically includes the production of nitrogen-, phosphorus- and potassium-based fertilisers as well as derived emissions from their application. Nevertheless, each specific case should be considered; for example, it has been considered that the use of digestate does not entail the avoided use of phosphorus and potassium fertilisers, since they would not be applied in any case. In addition, it should be considered that for accounting avoided credits from digestate use, different issues may arise from a methodological perspective:

- The calculation of emissions derived from the application of both mineral and organic fertilisers. There are different options for their calculation including from direct emissions factors to more complex methodologies (Brentrup et al., 2000; Brockmann et al., 2014; De Vries et al., 2012; IPCC, 2006). The problem of using emission factors is that they do not include specific particularities of the area under study, while the problem of more complex methodologies is that they require extensive specific data for their implementation. In addition, as discussed in detail in Chapter 4, different methodologies also imply different results.

- The calculation of the equivalent amount of replaced mineral fertilisers. The fertilising value of the digestate is lower than for mineral fertilisers, because organic fertilisers contain both forms of mineral and organic nitrogen (Brockmann et al., 2014). Despite there are different methodologies available, its calculation for each specific case may require additional information on the composition of the digestate that may be difficult to obtain.

5.5. Conclusions

The joint implementation of LCA and DEA was used to assess the eco-efficiency of 15 real agricultural biogas plants operating in Northern Italy. The DEA analysis included three inputs: feedstock production, transport and electricity consumption as well as one output: the electricity produced. The efficient plants were identified and benchmarks for the biogas plants which are operating with some level of inefficiency were estimated. According to the results, 60% of the biogas plants were found to operate with full efficiency (score of 100%). In addition, all the other plants operate with efficiencies above 50%, showing the high operating efficiency of the sample (average efficiency of 85.43%). For inefficient plants, reduction targets were applied to estimate the virtual plants. The comparison of the environmental profile of the original and the virtual plants showed that the environmental gains differed among impact categories. In more detail, reduction targets allowed the improvement of the environmental profile of the most polluting plants regarding CC and ME, since they are the biogas plants with larger use of energy crops per FU. However, the plants causing the highest impacts in FE, TA and ALO were identified as efficient; therefore, no reductions were established for these DMUs. This is because the ratio of energy crops digested was not the only driving reason for the results. In more detail, the environmental impacts of FE were directly related to the phosphorus content of the digestate; the storage of the digestate contributed to TA and the crop production per hectare and year was relevant for ALO. However, regarding the total environmental impact, the reduction targets based on the eco-efficiency principles influenced the plants exhibiting the worst overall environmental impact, evidencing the effectiveness of the combined LCA+DEA methodology.

5.6. List of acronyms

ALO	Agricultural land occupation
CC	Climate change
CCR	Charnes-Cooper-Rhodes
CHP	Co-generation heat and power
CRS	Constant return to scale
DEA	Data envelopment analysis
DMU	Decision making unit
FE	Freshwater eutrophication
GHG	Greenhouse gas
HRT	Hydraulic retention time
I	Input
IPCC	International Panel on Climate Change
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
ME	Marine eutrophication
NMVOC	Non-methane volatile organic compound
NVZ	Nitrate vulnerable zone
O	Output
OLR	Organic loading rate
SBM	Slacks-based measure of efficiency
SS	Subsystem
SSFW	Source-separated food waste
TA	Terrestrial acidification
TS	Total solids
TVS	Total volatile solids
VRS	Variable return to scale
WWTP	Wastewater treatment plant

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Chapter 6: Sustainable management of manure – The LiveWaste Project

Summary

The production of livestock waste from animal farms is one of the largest pressures to the environment in Cyprus. Currently, most small-scale facilities use open anaerobic lagoons for the management of animal waste; nevertheless, there is an important number of biogas plants treating animal manure. The environmental sustainability of an innovative treatment system for the management of livestock waste proposed in the LiveWaste project (LIFE12 ENV/CY/000544) was analysed in Chapter 6. The integrated treatment system includes anaerobic digestion in two stages as the core process and the treatment of all the gaseous, liquid and solid streams produced, to produce bioenergy, water reuse, struvite and compost. The analysis of the environmental impacts of the pilot plant operating in Cyprus was performed analysing four possible configurations regarding the treatment of the anaerobic effluent. The results also indicated that the best environmental results were obtained when struvite recovery was maximised. Another analysis was performed to compare the performance of the LiveWaste project with the existing scenarios for animal waste management. Anaerobic lagoons were identified as a major source of greenhouse gas emissions (404 kg CO_2 eq/t manure); while the better results were obtained for a simple biogas plant (-42 kg CO_2 eq/t manure). However, nutrients recovery/removal performed in the LiveWaste project considerably reduced impacts in eutrophication and acidification categories (40-80% lower impacts). Finally, a multicriteria analysis was applied by integrating environmental, social and economic indicators in the analytical hierarchy process to determine the most sustainable management strategy for animal waste in Cyprus. The results of the analysis showed the environmental benefits of the conversion of animal waste into a source of bioenergy, water and nutrients.

Outline of Chapter 6

6.1. Introduction to the LiveWaste project

The production of livestock waste from animal breeding facilities is one of the largest pressures to the environment in Cyprus, which has received relatively little attention so far (Insam et al., 2014). The introduction of intensive farming operations has increased the amount of animal waste generated in specific areas, aggravated by the insular nature of the country (Insam et al., 2014). Currently, most small-scale pig and cattle farms use open anaerobic lagoons for the management of their animal waste (Kythreotou et al., 2012). In parallel, the direct spread of animal manure on land is still a common practice in the country. However, the treatment of animal waste in biogas plants notably increased in Cyprus in the last years linked to the promotion of anaerobic digestion as a way to reduce environmental impacts and produce energy and a stabilised organic fertiliser (Al Seadi et al., 2013). By installing an on-farm anaerobic digester, farm owners can simultaneously comply with different European Directives while diversifying incomes by selling excess energy (Bangalore et al., 2016). Firstly, when operating in co-digestion conditions, anaerobic digestion of organic waste has the potential to divert municipal waste from landfills; thus, fostering compliance with the Landfill Directive (European Union, 1999). Secondly, they contribute with both the 1st and 2nd Renewable Energy Directives, since they classify biogas as a renewable energy source (European Parliament, 2009). In this sense, the production of renewable energy from organic waste is especially interesting in Cyprus. The country has an isolated energy system, which depends on fuel imports and therefore, costs of primary energy import are high (Kythreotou et al., 2012). Another issue that has to be dealt with is the large fluctuation in energy demand between seasons, which is caused by the high temperatures during the summer and the large tourist population arriving in the country.

With regard to the produced digestate, after its removal from the digester, it can be used with no further treatment than storage until the spread season (Crolla et al., 2013). The storage, transport, handling and application of raw digestate as an organic fertiliser results in significant costs for farmers, higher than its fertiliser value due to its large volume and low dry matter content (Rehl and Müller, 2011). The treatment of the produced digestate can also help to fulfil the specifications established in the Nitrates Directive (EEC, 1991). NVZs have been designated in Cyprus as a result of their high nitrate concentrations on water, as shown in Figure 6.1. As explained in Chapter 1, digestate processing can be performed in different ways and several methods for nutrients recovery/removal are available nowadays. On one hand, phosphorus can be recovered through struvite crystallisation. On the other hand, it is possible to remove nitrogen by performing the BNR process in a sequencing batch reactor (SBR) (Frison et al., 2013a). However, due to the substantial removal of organic matter attained in the anaerobic treatment, the addition of an external carbon source is required in the subsequent aerobic process for effective BNR (Frison et al., 2013b).

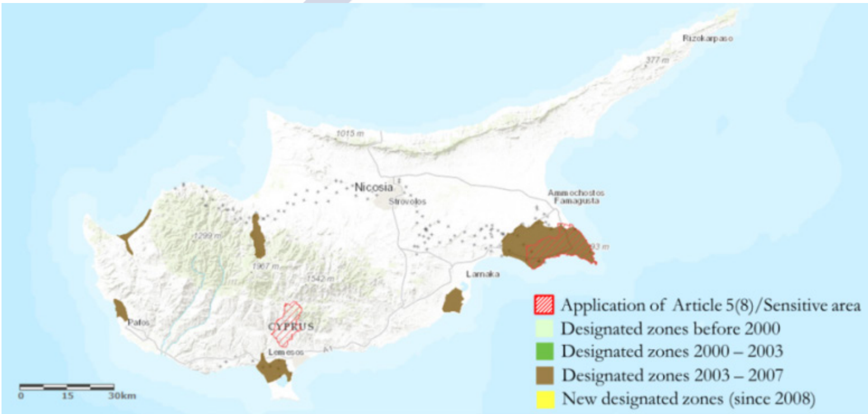


Figure 6.1. Nitrate vulnerable zones established in Cyprus

The LiveWaste project (LIFE12 ENV/CY/000544) was co-financed by LIFE+ EU financial instrument under the thematic area of LIFE+ Environmental Policy and Governance in the priority area of Waste and Natural Resources. The LiveWaste was coordinated by Cyprus Technology University and the consortium was composed of three universities including University of Verona (Italy), National Technical University of Athens (Greece) and the University of Santiago de Compostela (Spain), one private company (Animalia Genetics Ltd) as well as the Environmental Department of the Ministry of Agriculture, Nature Resource and Environment of Cyprus. The LiveWaste project aimed to develop, demonstrate, optimise and evaluate an innovative combined system for the treatment of livestock waste with the aim of resource recovery. Consequently, the treatment scheme was designed following the principles of circular economy, converting organic waste into a number of valuable products that can find their way back to the market, considering also the necessity of reducing negative environmental and social impacts associated with its management.

Therefore, the objective of Chapter 6 was to assess the environmental impacts and benefits associated with the implementation of the LiveWaste project. More deeply, the performance of the pilot plant and the potential full implementation in Cyprus were evaluated. Finally, the three different alternatives for the management of animal waste in Cyprus were compared from an environmental, social and economic point of view by applying the AHP method.

6.2. Environmental assessment of the LiveWaste treatment scheme

6.2.1. Goal and scope definition

The objective of this analysis is the quantification of the environmental impacts associated with the conversion of animal waste into a source of bioenergy, nutrients, water and compost in the pilot plant installed within the framework of the LiveWaste project (Figure 6.2), by means the LCA methodology. In addition, the four possible configurations that the pilot plant offers for its operation were also compared to identify the most sustainable scheme that maximises the conversion of animal waste into a source of energy, water and nutrients.

Figure 6.2. LiveWaste pilot plant operating in Cyprus

Integrated treatment system

The pilot plant treats 250 kg/d of livestock waste consisting of a mixture of pig slurry mixed with other animal waste such as poultry and horse manure for increased biogas production. It includes the anaerobic digestion of the animal waste as the core process, as well as the treatment of all the derived gaseous, liquid and solid streams. The specific treatment scheme of the LiveWaste prototype is shown in Figure 6.3.

The anaerobic digestion is the first treatment step and it is carried out in two tanks separately. The objective is to perform the hydrolysis and acidogenesis phases in the dark fermentation (DF) unit and the acetogenesis and methanogenesis in another separate tank where methane is produced (Guo et al., 2010). Both tanks are CSTR operating at different operational conditions under mesophilic conditions (37°C); thus, the first reactor has an OLR of approximately 18.6 kg TVS/m^3·d and a HRT of only 3.6 days while the second reactor works with a OLR of 2.42 kg TVS/m^3·d and a HRT of 32 days. As a result, around 7 m^3 of biogas are produced per day with an average composition of 57%-62% in methane, 38%-43% in carbon dioxide and 0.05% of hydrogen sulphide.

Since it is necessary to have a biogas with adequate characteristics to ensure efficient and clean production of bioenergy, the hydrogen sulphide present in the produced biogas needs to be removed due to its toxic and corrosive characteristics (Abatzoglou and Boivin, 2009). In the LiveWaste project, a biological desulphurisation was selected as it is especially indicated to treat emissions characterised by large flows and low concentration of pollutants. Specifically, a biotrickling filter (BTF) was selected due to the high solubility of hydrogen sulphide in the aqueous phase (Sander, 2015). The BTF is adapted to anoxic conditions, which accomplishes the oxidation of hydrogen sulphide through the biochemical pathway known as autotrophic denitrification. The BTF packing material is a combination of open pore polyurethane foams in the upper parts and polypropylene pall rings of in the bottom section. As a result, up to 75% of the hydrogen sulphide produced contained in the biogas is removed by the BTF.

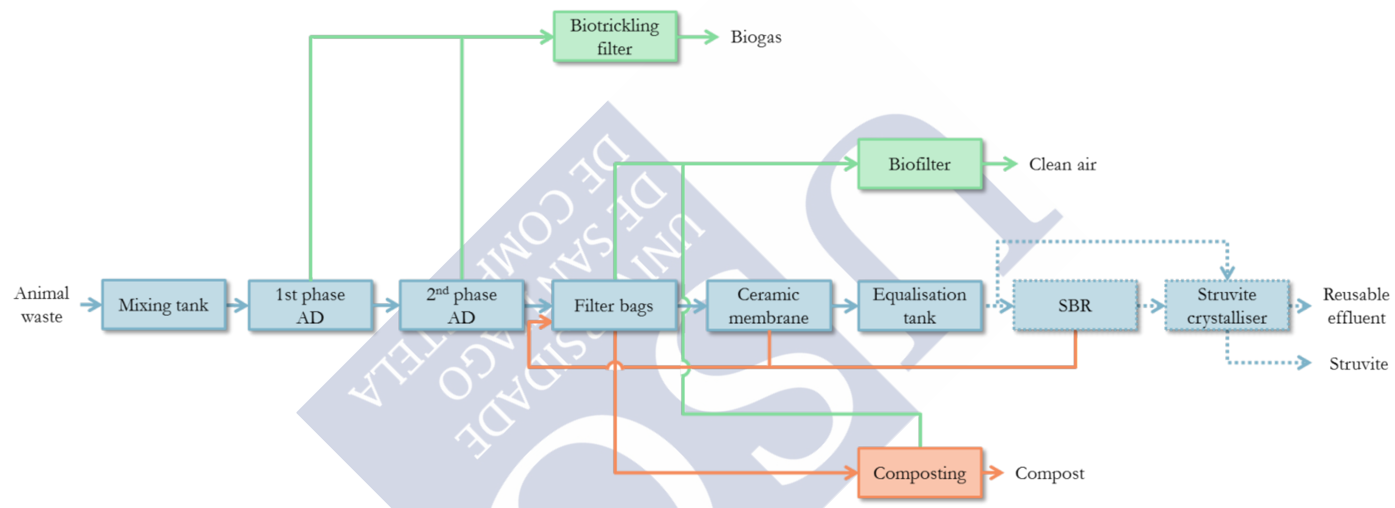


Figure 6.3. Combined system proposed in the LiveWaste project for the management of livestock waste in Cyprus. Dotted boxes and arrows indicate the possible configurations of the treatment scheme. Acronyms: AD – anaerobic digestion; SBR – sequencing batch reactor

The produced digestate, which is highly diluted (4-5% TS), is separated into its liquid and solid fractions. The filtration system is composed of two elements: a filter bag and an ultrafiltration membrane. The bag filters are two bags of 100 L which retain the solid particles above 3 mm of diameter. In order to promote the formation of aggregates and favour the separation performance, a polyelectrolyte is mixed with the digestate before filtration. The ultrafiltration membrane consists of a ceramic membrane which removes the smallest solids (pore diameter of 0.2 μm), forcing the pass of the liquid through the membrane operating at high pressure. The inlet flow ranges from 6 to 20 m^3/h, with an inlet pressure of 2.5-3 bar.

The treatment of the produced permeate is flexible and different configurations can be followed, including the possible combinations between the struvite crystalliser and the SBR. The interest of struvite precipitation relies on the possibilities of accomplishing nutrient recovery from organic waste streams and the use of this material as slow-release fertiliser (Zhang et al., 2010). It is a physico-chemical process in which nitrogen and phosphorus are recovered as struvite crystals ($MgNH_4PO_4 \cdot 6H_2O$). In this case, the process requires the addition of magnesium. The resulted effluent can be also sent to the SBR for the removal of the remaining nutrients to achieve a quality effluent, suitable for its reuse for irrigation. In the SBR, the biological nutrient removal via nitrite is performed in a biological reactor with a HRT of 8 days. As indicated in Table 6.1, each cycle alternates different anaerobic, aerobic and anoxic phases. It is important to remark the necessity of adding an external carbon source such as acetic acid since some of the bacteria governing this process are heterotrophic microorganisms.

Table 6.1. Operation characteristics of the SBR

SBR stage	Value
Filling	7.5 min/cycle
Anaerobic phase	25 min/cycle
Aerobic phase	160 min/cycle
Anoxic phase	80 min/cycle
Sedimentation	15 min/cycle
Discharge	7.5 min/cycle

The composting unit is included to treat all the produced sludge along the process. The main objective of this stage is the maturation of this solid fraction to obtain high-quality marketable compost. For this, the produced sludge treated is mixed with straw to improve its C/N ratio and aeration.

In order to minimise the emissions of volatile organic compounds (VOCs) and odours, the gaseous streams produced during composting process as well as those collected from the headspace of the different tanks were treated in the bio-filtration hybrid unit. This method consists of a conventional biofilter coupled with a post-treatment of activated carbon as a polishing step. The use of the activated carbon allows high efficiency of odour and VOC removal even if the biofilter is not working properly. An overall flow of 1.2 m^3/h with an average VOC concentration of 545 mg C/m^3 and a clear perception of odours was treated.

As mentioned, the design of the pilot plant allows flexibility in the treatment of the produced permeate. One possibility relies on its treatment in the SBR for the biological removal of nutrients through the short-cut nitrification/denitrification process. Another one is the production of struvite, a phosphate-based fertiliser, in a struvite crystallisation unit. Moreover, the combination of both treatment systems offers two additional configurations of the LiveWaste prototype. The four possible configurations of the LiveWaste prototype under assessment and the system boundaries considered are shown in Figure 6.4.

Function and functional unit

The function of this system is the sustainable treatment of livestock waste to convert it into biogas, treated effluent, struvite and compost. Since the prototype treats 250 kg/d of a mixture of pig, poultry and horse manure, the FU chosen was 250 kg of livestock waste mixture treated each day in the LiveWaste pilot plant.

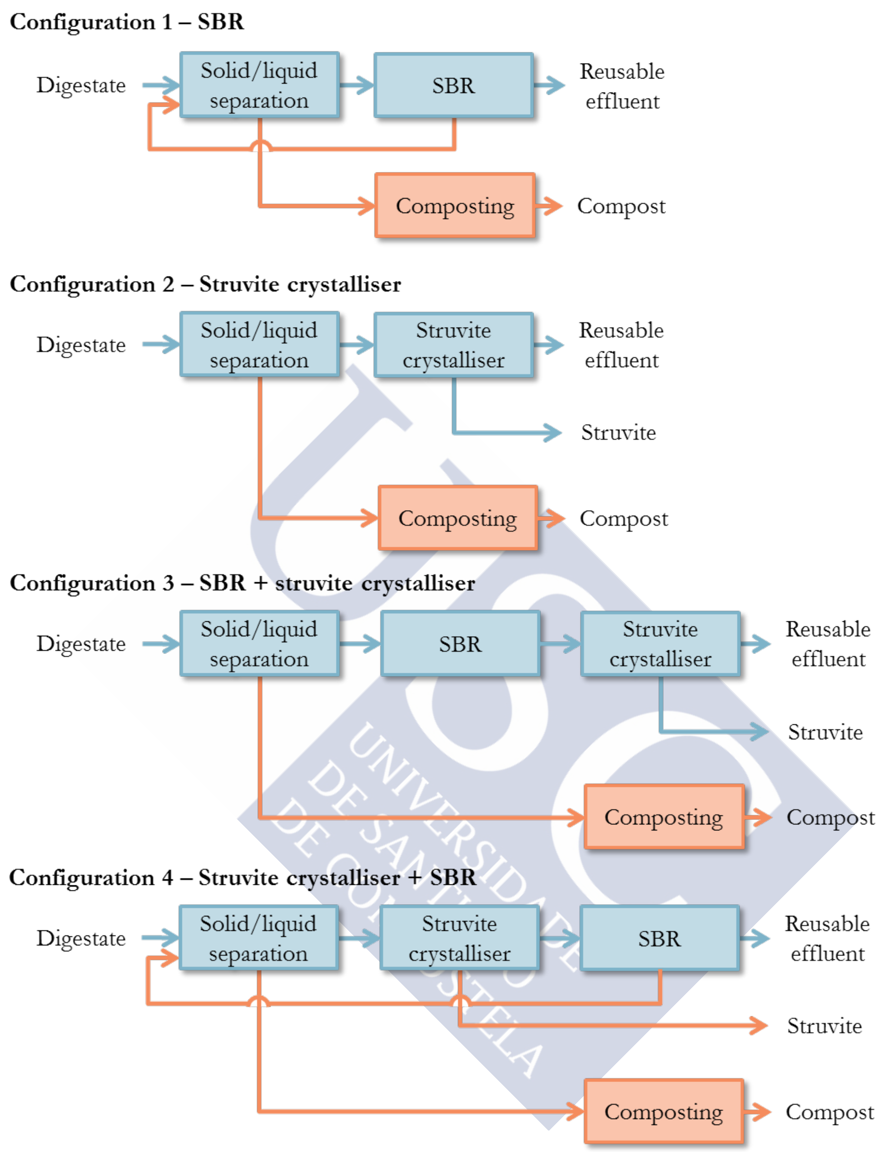


Figure 6.4. Different possible configurations of the treatment scheme. Acronyms: SBR – sequencing batch reactor

Description of the system boundaries

As mentioned, the treatment scheme proposed in the LiveWaste project converts the livestock waste into valuable products (i.e. biogas, treated effluent, struvite and compost). In this study, a system expansion was performed to account the environmental benefits of the production of these products, since they can reduce the need of producing other products with the same function. Therefore, the boundaries of the system under study were extended to include the production and use of these substituted processes. It has been shown previously that the choice of the avoided function can have a decisive influence on the results (Finnveden et al., 2005). This perspective has been considered by including that the produced effluent can be used for irrigation in agricultural land, reducing the requirements of water, nitrogen and phosphorus-based mineral fertilisers (Figure 6.5). In addition, it has been taken into account that the compost can be used as a soil conditioner, also providing nutrients to agricultural land; therefore, avoiding the use of peat and mineral fertilisers (Saer et al., 2013). In the same way, the produced struvite also substitutes the production of nitrogen and phosphorus-based mineral fertilisers. Regarding biogas, it has been considered that, due to the scale of the pilot plant, the biogas produced is used to produce heat in a boiler to heat up the anaerobic digestion process; accordingly, no environmental credits were accrued for its production in this analysis.

6.2.2. Life cycle inventory

In order to complete the LCI, mass balances regarding TS, TVS and nutrients (TN and TP) were computed based on design parameters and the results from the operation of the pilot plant. The amount of livestock treated and the different derived products for each configuration are presented in Table 6.2.

Table 6.2. Summary of the main input and outputs of the four configurations

Input/outputs	Units	Conf 1	Conf 2	Conf 3	Conf 4
Livestock waste treated	ton/year	91.3	91.3	91.3	91.3
Biogas production	m^3/year	2,570	2,570	2,570	2,570
Struvite recovery	kg/year	0	129	61.4	103
Final effluent	m^3/year	72.1	73.1	72.1	72.0
Compost	ton/year	10.7	10.6	107	10.7

The infrastructure was accounted considering that the pilot plant is constructed in stainless steel with a life time of 20 years. The amount of polymer added to the filter bags is controlled by the TS content of the influent (~0.25 kg polymer/kg TS). In addition, it was considered that sodium hypochlorite is used every three months to clean the ceramic membrane (Wang et al., 2010).

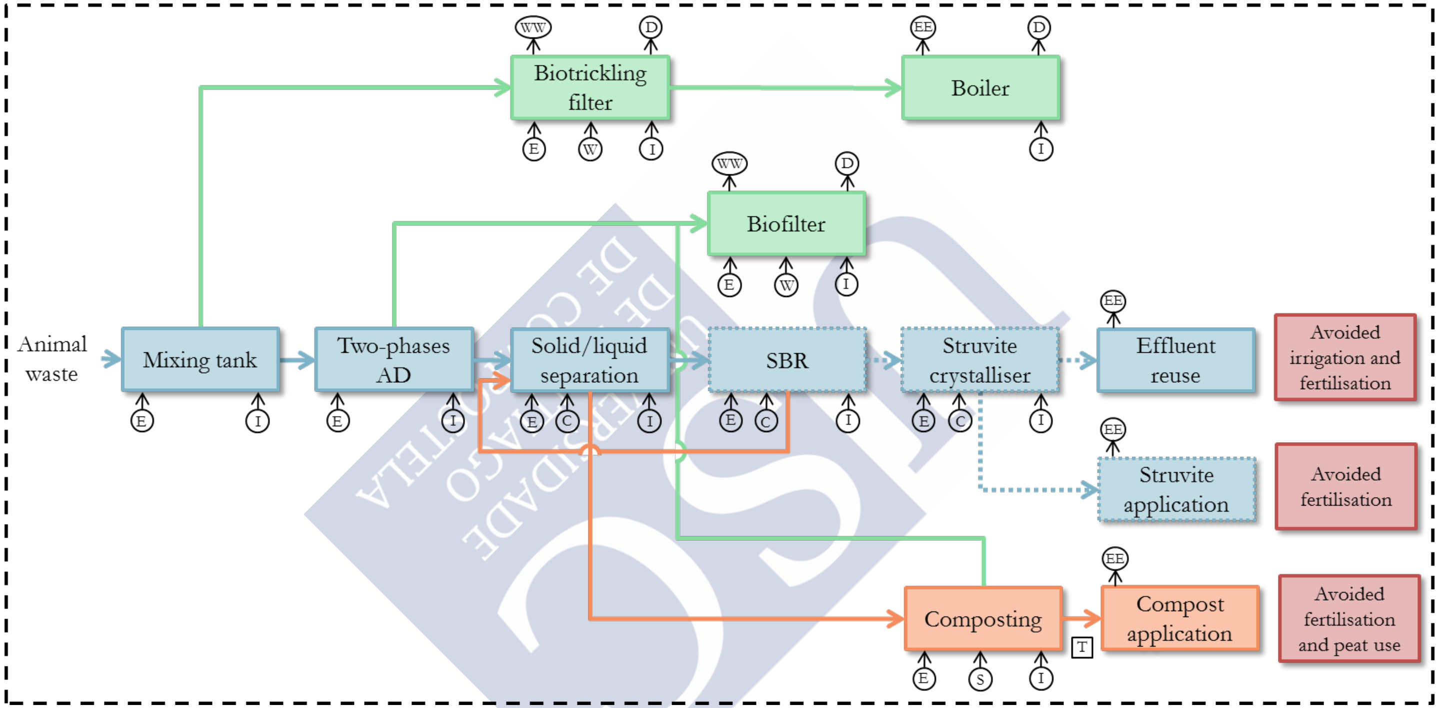


Figure 6.5. Unit processes included in the system boundaries of the treatment system under study. Dotted processes are not included in the configurations. Dotted boxes indicate processes not performed in all configurations. Acronyms: T – transport; E – electricity; I – infrastructure; W – water; C – chemicals; S – Straw; EE – emissions; WW – wastewater production; L – disposal of waste

Regarding the SBR, acetic acid was needed as carbon source depending on the chemical oxygen demand (COD) of the effluent, in a range of 2.2 kg COD/kg $N_{removed}$. In the struvite crystalliser, magnesium oxide was added according to a Mg/P molar ratio of 1.5. The specific Cypriot conditions were considered for the production of electricity, including electricity production from different sources: 79.5% heavy fuel oil, 12.6% diesel, 1.1% biomass, 5.4% wind energy and 1.4% solar energy (Statistical Service of Cyprus, 2013).

Among the different methodologies available in the literature, the methodology described in the IPCC report "Guidelines for National Greenhouse Gas Inventories" regarding emissions of nitrogen-based compounds was undertaken in this study (IPCC, 2006). In more detail, direct emissions of nitrous oxide were calculated considering 0.01 kg N_2O-N/kg N input from organic fertilisers. Emissions of ammonia and nitrogen oxides from the application organic substrates were with an emission factor of 0.20 kg N/kg N input, as ammonia and nitrogen oxides (NH_3-N + kg NO_X-N), respectively, being mainly in the form of ammonia (90%) and a small fraction as nitrogen oxides (10%) (Van Der Gon and Bleeker, 2005). An emission factor of 0.30 kg N-NO_3^-/kg N input for nitrate leaching is also provided. Regarding struvite, its low solubility prevents it from producing nutrient rich run-off (Tao et al., 2016). The nitrogen and phosphorus fertilising replacement values of these organic substrates provided by De Vries et al. (2012) were used to calculate the avoided fertilisers production. Avoided emissions from the application of mineral fertilisers were also computed using the emissions factors provided by IPCC (2006), which considers an emission factor of 0.10 kg NH_3-N + kg NO_X-N/kg N input. Moreover, the agricultural machinery used for compost or fertilisers application (tractor, implement, diesel and derived emissions from diesel consumption) was also accounted. To do so, an average input of 220 kg N/ha was considered (Eurostat, 2014). Regarding the transport of the end-products to the agricultural land, a transport distance of 25 km was considered in all scenarios under assessment, according to an analysis of the current situation in Cyprus. Finally, the ecoinvent® database version 3 (Wernet et al., 2016) was used to include background data of peat production (Dones et al., 2007), chemicals (Althaus et al., 2007), transport (Spielmann et al., 2007) and waste disposal (Doka, 2007). The complete LCI used is given in Table 6.3.

Table 6.3. LCI of the four configurations under study

	Conf 1		Conf 2		Conf 3		Conf 4	
Inputs from technosphere								
Materials								
Livestock waste	250	kg	250	kg	250	kg	250	kg
Stainless steel	1.23	kg	1.10	kg	1.28	kg	1.28	kg
Methacrylate	2.43	g	2.43	g	2.43	g	2.43	g
Fibre glass	0.33	g	0.33	g	0.33	g	0.33	g
Polyurethane foam	0.39	g	0.39	g	0.39	g	0.39	g
Polypropylene	0.20	g	0.20	g	0.20	g	0.20	g
Pine bark chips	0.70	g	0.70	g	0.70	g	0.70	g
Perlite	2.10	g	2.10	g	2.10	g	2.10	g
Granular activated carbon	1.47	g	1.47	g	1.47	g	1.47	g
Polymer	2.71	kg	2.71	kg	2.71	kg	2.71	kg
Sodium hypochlorite	0.08	g	0.07	g	0.08	g	0.08	g
Acetic acid	0.62	kg	0	kg	1	kg	1	kg
Magnesium oxide	0	kg	0.09	kg	0.04	kg	0.07	kg
Tap water	1.44	kg	1.44	kg	1.44	kg	1.44	kg
Tractor	1.63	g	1.60	g	1.63	g	1.63	g
Agricultural implement	5.24	g	5.16	g	5.24	g	5.24	g
Diesel	14.7	g	14.5	g	14.7	g	14.7	g
Avoided materials due to effluent application								
Ammonium nitrate	53.3	g	240	g	47.0	g	51.1	g
Phosphate fertiliser	52.3	g	19.7	g	9.42	g	15.8	g
Tractor	0.09	g	0.41	g	0.08	g	0.08	g
Agricultural implement	0.25	g	1.18	g	0.22	g	0.24	g
Diesel	1.92	g	9.05	g	1.72	g	1.81	g
Avoided materials due to struvite application								
Ammonium nitrate	0	g	20.1	g	9.60	g	16.1	g
Phosphate fertiliser	0	g	90.5	g	43.3	g	72.7	g
Avoided materials due to compost application								
Peat	29.4	kg	29.0	kg	29.4	kg	29.4	kg
Nitrogen fertiliser	88.7	g	87.0	g	88.7	g	67.4	g
Phosphate fertiliser	298	g	224	g	298	g	246	g
Tractor	0.13	g	0.13	g	0.13	g	0.13	g
Agricultural implement	0.38	g	0.37	g	0.38	g	0.38	g
Diesel	2.95	g	2.81	g	2.95	g	2.95	g
Transport								
Tractor and trailer	1,036	kg·km	1,024	kg·km	1,036	kg·km	1,036	kg·km
Electricity								
Electricity	53.6	kWh	13.6	kWh	53.6	kWh	53.6	kWh
Inputs from nature								
Avoided natural resources due to effluent application								
Water, natural origin	198	L	200	L	198	L	197	L

Table 6.4. LCI of the four configurations under study (cont.)

	Conf 1		Conf 2		Conf 3		Conf 4	
Outputs to technosphere								
Products								
Heat (from biogas)	34.7	kWh	34.7	kWh	34.7	kWh	34.7	kWh
Effluent	198	L	200	L	198	L	197	L
Compost	29.4	kg	29.0	kg	29.4	kg	29.4	kg
Waste streams to treatment								
Wastewater	1.44	m^3	1.44	m^3	1.44	m^3	1.44	m^3
Solid waste	34.9	g	34.9	g	34.9	g	34.9	g
Outputs to nature								
Emissions from effluent application								
Ammonia	17.9	g	80.7	g	15.8	g	17.2	g
Nitrogen oxides	4.45	g	20.0	g	3.93	g	4.27	g
Nitrous oxide	1.29	g	5.80	g	1.14	g	1.23	g
Nitrate	109	g	491	g	96.1	g	70.9	g
Phosphate	0.80	g	0.30	g	0.14	g	0.08	g
Avoided emissions due to effluent application								
Ammonia	5.82	g	26.2	g	5.14	g	5.58	g
Nitrogen oxides	1.45	g	6.52	g	1.28	g	1.39	g
Nitrous oxide	0.84	g	3.77	g	0.74	g	0.80	g
Nitrate	109	g	491	g	96.1	g	67.9	g
Phosphate	0.79	g	0.30	g	0.14	g	0.08	g
Emissions from compost application								
Ammonia	129	g	127	g	129	g	98.2	g
Nitrogen oxides	32.1	g	31.5	g	32.1	g	24.4	g
Nitrous oxide	9.29	g	9.11	g	9.29	g	7.06	g
Nitrate	785	g	770	g	785	g	597	g
Phosphate	4.54	g	3.40	g	4.54	g	3.75	g
Avoided emissions due to compost application								
Ammonia	9.69	g	9.50	g	9.69	g	7.36	g
Nitrogen oxides	2.41	g	2.36	g	2.41	g	1.83	g
Nitrous oxide	1.39	g	1.37	g	1.39	g	1.06	g
Nitrate	118	g	116	g	118	g	89.5	g
Phosphate	4.49	g	3.37	g	4.49	g	3.71	g

6.2.3. Life cycle impact assessment

The impact categories selected from the ReCiPe Midpoint methodology were common to the other chapters (CC, OD, TA, FE, ME, POF and FD) (Goedkoop et al., 2009). In addition, the analysis was completed considering water depletion (WD) from the ReCiPe Midpont methodology due to the specific problem of water scarcity in Cyprus as well as malodours air (MA) as an additional impact category from the CML 2001 method (Hischier et al., 2009).

The characterisation results related to the FU selected for each configuration under assessment can be found in Table 6.4.

Table 6.4. Characterisation results of the four configurations under study

		Conf 1	Conf 2	Conf 3	Conf 4
CC	(kg CO_2 eq/FU)	57.0	14.9	57.2	57.8
OD	(kg CFC-11 eq/FU)	$8.0\cdot10^{-6}$	$1.8\cdot10^{-6}$	$8.0\cdot10^{-6}$	$8.1\cdot10^{-6}$
TA	(kg SO_2 eq/FU)	0.65	0.46	0.65	0.60
FE	(kg P eq/FU)	0.004	0.001	0.002	0.003
ME	(kg N eq/FU)	0.08	0.12	0.08	0.06
POF	(kg NMVOC/FU)	0.17	0.07	0.17	0.16
WD	(m^3/FU)	-197	-200	-197	-197
FD	(kg oil eq/FU)	19.5	5.73	19.5	19.6
MA	(m^3 air/FU)	858,523	298,982	858,658	837,826

As it can be seen in Table 6.4, the environmental results change among the impact categories considered since different emissions of hazardous substances and extractions of natural resources have different influence on the impact category indicators at the midpoint level (Goedkoop et al., 2009). Each impact category was studied in detail to identify the most influential processes that result in the highest environmental burdens.

Climate change

As displayed in Figure 6.6, Configurations 1, 3 and 4 present similar environmental behaviour in this impact category; while Configuration 2 achieved the best results. Among the processes contributing to this impact category, the consumption of electricity was identified as the most important *hotspot*, accounting for 74% to 82% (52.9 kg CO_2 eq/FU) in the environmental impacts of Configurations 1, 3 and 4; however, it only represented 39% of the impacts in Configuration 2. Electricity is consumed in different stages of the treatment scheme, especially for pumping, mixing and aeration operations; nonetheless, the most energy-consuming process is the aeration of the SBR to complete the aerobic phase. Therefore, the reason for the different results in Configuration 2 is related to the fact that in this scheme the produced permeate from the ceramic membrane is treated in a struvite crystalliser; being the only configuration that does not include an SBR. It is important to remark that the environmental impact of the consumption of electricity is directly linked with the electricity mix of the specific country under study. In this case, the Cypriot electricity profile is highly dependent on fossil energy sources, producing 0.98 kg CO_2 eq/$kWh_{produced}$. It is also important to highlight that biological nitrogen removal via nitrite has several benefits compared to the conventional nitrification/denitrification via nitrate such as 25% of oxygen savings during nitrification and 40% less requirement for organic carbon source during heterotrophic denitrification (Galí et al., 2007). Therefore, the environmental impacts of energy consumption could be much higher if biological nutrients removal is performed under the conventional scheme.

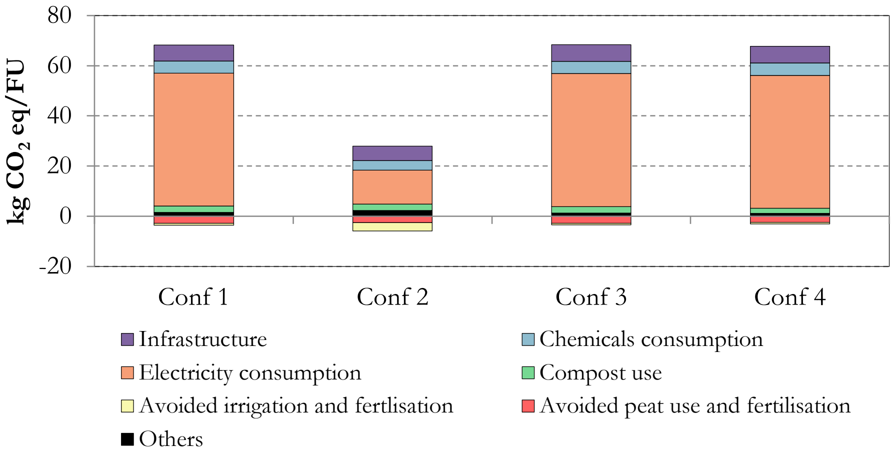


Figure 6.6. Characterisation results in CC per FU

Regarding other sources of environmental burdens, impacts related to chemicals and infrastructure production as well as compost application on land show a similar behaviour in all configurations. The chemicals consumed in the pilot plant, producing between 3.9 and 4.9 kg CO_2 eq/FU, are the polymer for the solid/liquid separation, sodium hypochlorite for cleaning the ceramic membrane, acetic acid as a carbon source in the SBR and magnesium oxide to provide magnesium to the struvite crystallisation reactor. The infrastructure of the pilot plant, mainly stainless steel, produced around 6.4 kg CO_2 eq/FU. Moreover, direct nitrous oxide emissions derived from the application of organic substrates on land were the main contributor to the impact produced by the use of compost

(~2.6 kg CO_2 eq/FU). Finally, environmental credits mostly due to the production of compost helped to offset the environmental impacts. Avoided fertilisers production and derived emissions due to the fertiliser potential of the compost reduced the environmental impacts by 6.4-6.7 kg CO_2 eq/FU.

Ozone depletion, photochemical oxidant formation and fossil depletion

A similar behaviour to that observed in CC was identified in OD, POF and FD. Regarding OD and FD, the emissions of ozone depleting substances and the consumption of fossil fuels are mainly linked with the production of electricity; representing between 67% and 87% of the impacts in OD and between 49% and 77% in FD. As in CC, chemicals and infrastructure production contributed together with 16%-28% of the impacts in OD and 18%-36% in FD. In addition, avoided peat use and fertilisation reduced the impacts by 3%-8%. As shown in Figure 6.7, the distribution of relative impacts was a slightly different in POF.

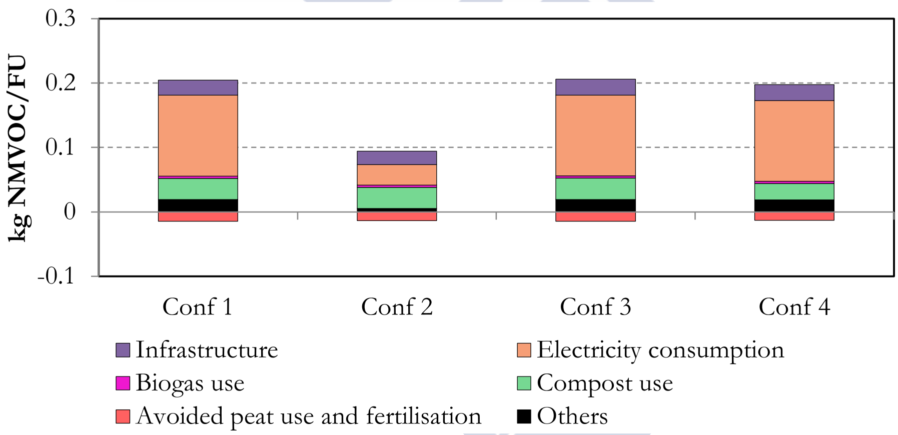


Figure 6.7. Characterisation results in POF per FU

In this impact category, electricity production is the main contributor although the relative impact produced is lower in this case, representing between 21% and 57% of the impacts. Once again, chemicals and infrastructure production contributed with 17%-21% of the impacts, and avoided peat use and fertilisation reduced the impacts by 6%-9%. Nevertheless, different to OD and FD, in this impact category compost use had remarkable impacts. In more detail, these impacts were mainly due to nitrogen oxides emitted during compost application on land with 11% to 21% of the impacts.

Terrestrial acidification, freshwater and marine eutrophication

The production of the electricity consumed in the pilot plant was again identified as an important *hotspot* in TA, producing between 12% and 48% of the impacts, mainly due to emissions of sulphur dioxide. As presented in Figure 6.8, this impact category was also largely influenced by direct emissions of ammonia that occur during the application of compost on land (31%-41% of the impacts). In the same way, ammonia emission from the use of the final effluent for irrigation produced an important impact in Configuration 2 (~26%), different from Configurations 1, 3 and 4 (< 6%). The emissions of ammonia derived from the application of the effluent are directly linked with the quality of the final effluent. Since Configuration 2 does not include the SBR, the removal of nitrogen cannot be accomplished and the concentration of nitrogen in the final effluent is higher.

In the systems under study, phosphate leaching occurs in the application of organic fertilisers (such as compost) affecting FE. However, according to De Vries et al. (2012a), almost all the phosphorus contained in organic substrates are available for the plants. This meant that the impacts related to phosphate leaching from compost application were mostly counteracted by avoided phosphate leaching from mineral fertilisers. Moreover, struvite is a high quality fertiliser since it slowly release the nutrients contained, helping to prevent leaching and enhancing the environmental profile of the system (Tao et al., 2016). Therefore, the maximisation of struvite recovery is a key aspect in the improvement of impact categories such as FE.

Finally, impacts produced in ME are connected with nitrogen emissions, being nitrate the most contributing compound in the systems under study. In contrast to phosphorus, the availability for plants of nitrogen in organic substrates is not the same than in mineral fertilisers (De Vries et al., 2012a). Therefore, in this case nitrate leaching derived from the final effluent and the compost were only partially offset by avoided nitrate emissions from the application of mineral fertilisers. The environmental impact produced by the application of the compost was similar in all configurations. Furthermore, Configuration 2 presented a higher impact due to the use of the final effluent for irrigation. This fact is linked with the quality of the final effluent, since this configuration does not include a SBR for nitrogen removal, resulting in higher nitrate leaching.

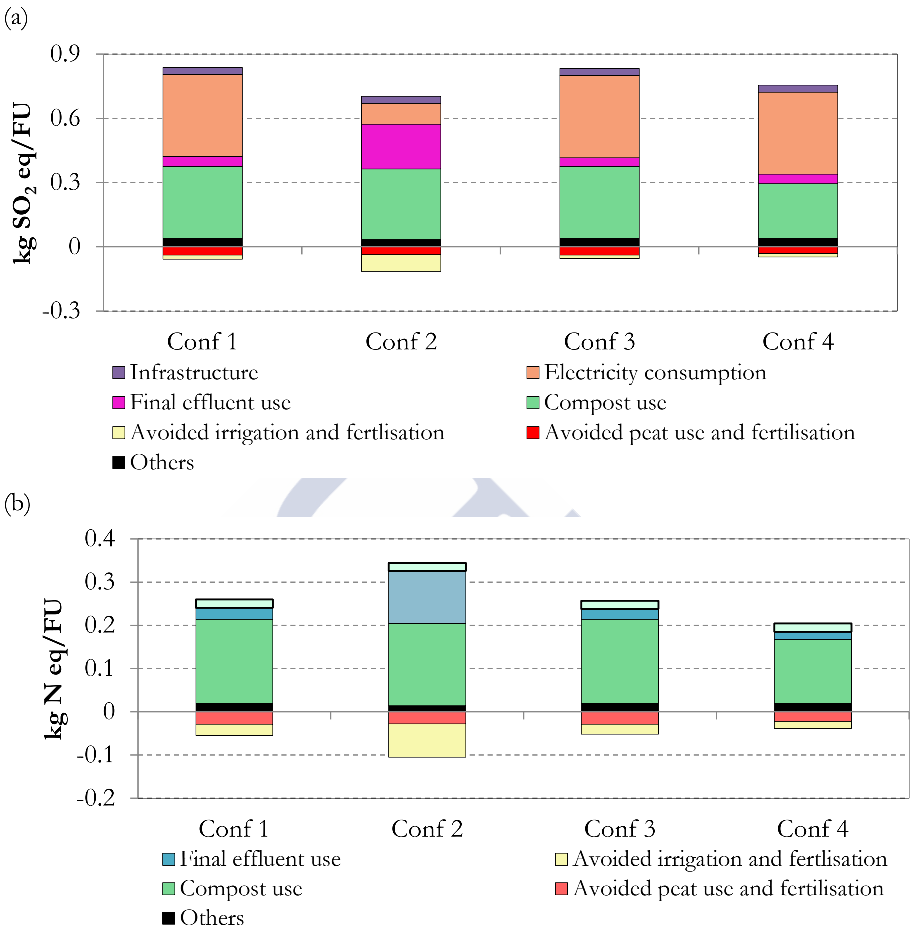


Figure 6.8. Characterisation results in (a) TA and (b) ME per FU

Water depletion and malodours air

WD accounts for the consumption of water in the system under study. The production of chemicals, especially the polymer used in the solid/liquid separation, has a remarkable effect in this impact category due to the water required for its production. However, the overall results in WD showed the positive effect of the production of a treated effluent that can be reused for irrigation in the four configurations. Therefore, it has been considered within the system boundaries that it can avoid the use of water from natural origin.

MA measures the emissions of compounds that potentially can produce malodours. In this case, ammonia emissions were identified as a key contributor to this impact category, producing from 34% to 53% of the environmental impacts in MA (Figure 6.9). However, the production of electricity also entails emissions of acetaldehyde and hydrogen sulphide in the production of electricity from different fuels, representing from 5% in the case of Configuration 2 up to 27% in Configuration 1, explaining the reason for the superior environmental results of Configuration 2.

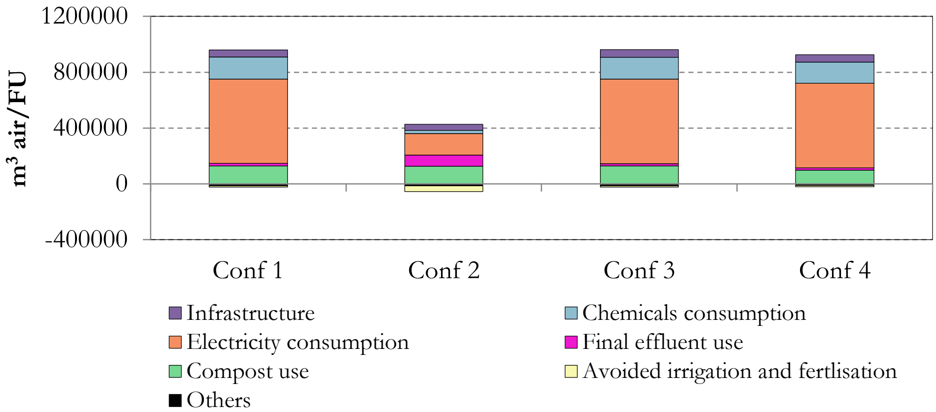


Figure 6.9. Characterisation results in MA per FU

6.2.4. Sensitivity analysis

As explained before, for the proper biological removal of nutrients in the SBR, the addition of a carbon source is required. The objective of this analysis is to compare the use of different carbon sources in Configuration 4, including acetic acid, methanol and the effluent from the DF reactor, since it is rich in VFAs. In general terms, around 2.2 g of COD are required for the removal of 1 g of nitrogen. For the removal of the same amount of nitrogen, the use of methanol would imply lower amount of substance used in comparison with the acetic acid, due to their different chemical nature (1.07 g COD/g acetic acid and 1.50 g COD/g methanol). Moreover, when comparing the environmental profile of both substrates, per kilogram of substance, the production methanol entails lower environmental impacts compared with the production of acetic acid in all categories, with reductions ranging from 27% to 77%. The use of of part of the effluent from the DF would entail a reduction in the chemicals needed in the plant as well as a sligth reduction of the potential biogas produced. According to the results obtained in this sensitivity analysis, the use of methanol or the partial recirculation of DF effluent would enhance the environmental profile of the treatment system. This would have minor consequences in impact categories such as CC, OD, TA, ME and WD (<2%). Regarding POF and FD the impact would be decreased by 6%. In the case of MA, the use of methanol or DF effluent instead of acetic acid would improve the impact category by 27%, due to emissions to air of acetic acid and acetaldehyde associated to the production process of acetic acid. The low overall impact achieved in FE due to the recovery of nutrients emphasised the relative improvement achieved by the change in carbon source. The impact produced in FE would be reduced by 106% with the use of methanol and by 114% with the use of the effluent from the DF, since acetic acid production entails 5 times more impacts in FE than methanol production.

6.3. Environmental assessment of manure management in Cyprus

This section includes the quantification of the environmental impacts of implementation of the LiveWaste system at full-scale in Cyprus. In order to perform this analysis, mass and energy balances were also developed to simulate the entire treatment train based on both data from the pilot plant as well as from literature on full-scale plants that perform some of the processes under assessment (Mininni et al., 2015).

6.3.1. Goal and scope definition

The LCA methodology allows the assessment of the proposed innovations before these reach full scale in order to anticipate possible problems and related solutions (Mininni et al., 2015). The potential environmental performance of the full implementation of the LiveWaste system was compared with the environmental performance of the current available practices for livestock waste management in Cyprus. Considering the results from section 6.2, Configuration 4 has been selected on the basis of struvite recovery and high removal of nitrogen rates. To do so, it was modelled a full-scale biogas plant with the technology proposed in the LiveWaste project with a capacity of 200 t of livestock manure per day and daily biogas of production of 5,600 m^3. Additionally, four existing alternatives for the management of livestock waste in Cyprus were considered and compared with the scheme proposed in the LiveWaste project:

• **Scenario 1. Anaerobic lagoon –** In this scenario all livestock waste is directly pumped to an anaerobic lagoon. Direct emissions to air and water derived from this practice have been identified.

• **Scenario 2. Anaerobic lagoon after solid/liquid separation –** Animal waste is separated into its liquid and solid fractions. The liquid is pumped to an anaerobic lagoon, while the solid is stored and then applied on land as an organic fertiliser. Therefore, it has been considered that this solid fraction avoids the proportional use of mineral fertilisers.

• **Scenario 3. Direct application –** Animal manure is stored in the farm, transported to the agricultural land and applied as an organic fertiliser. As in Scenario 2, environmental credits from the replacement of mineral fertilisers have been included.

• **Scenario 4. Conventional biogas plant –** Animal waste is digested in an anaerobic reactor to produce biogas, which is used in a co-generation heat and power unit. Electricity is injected into the national grid, while heat is used in the plant. In addition, the produced digestate is applied on land as an organic fertiliser. Therefore, two avoided processes are associated to this scenario: electricity production from the grid and mineral fertilisation.

• **Scenario 5. LiveWaste treatment scheme –** Animal waste is treated in a two-phase anaerobic digestion for the production of high quality biogas. The biogas is treated in a biotrickling filter for the removal of hydrogen sulphide. For comparison purposes with Scenario 4, it has been considered as base case that it is used to produce electricity in a co-generation unit. The produced digestate is separated into its liquid and solid fractions for the recovery of phosphorus, the removal of nitrogen and the production of compost. The produced effluent can be used for irrigation, replacing the use of water and mineral fertilisers. The produced compost is applied on land as a soil conditioner, reducing the need of peat and mineral fertilisers. Finally, all gaseous streams produced in the whole process are treated in a biofilter unit for the removal of odours.

The system boundaries of the five scenarios under study can be found in Figure 6.10.

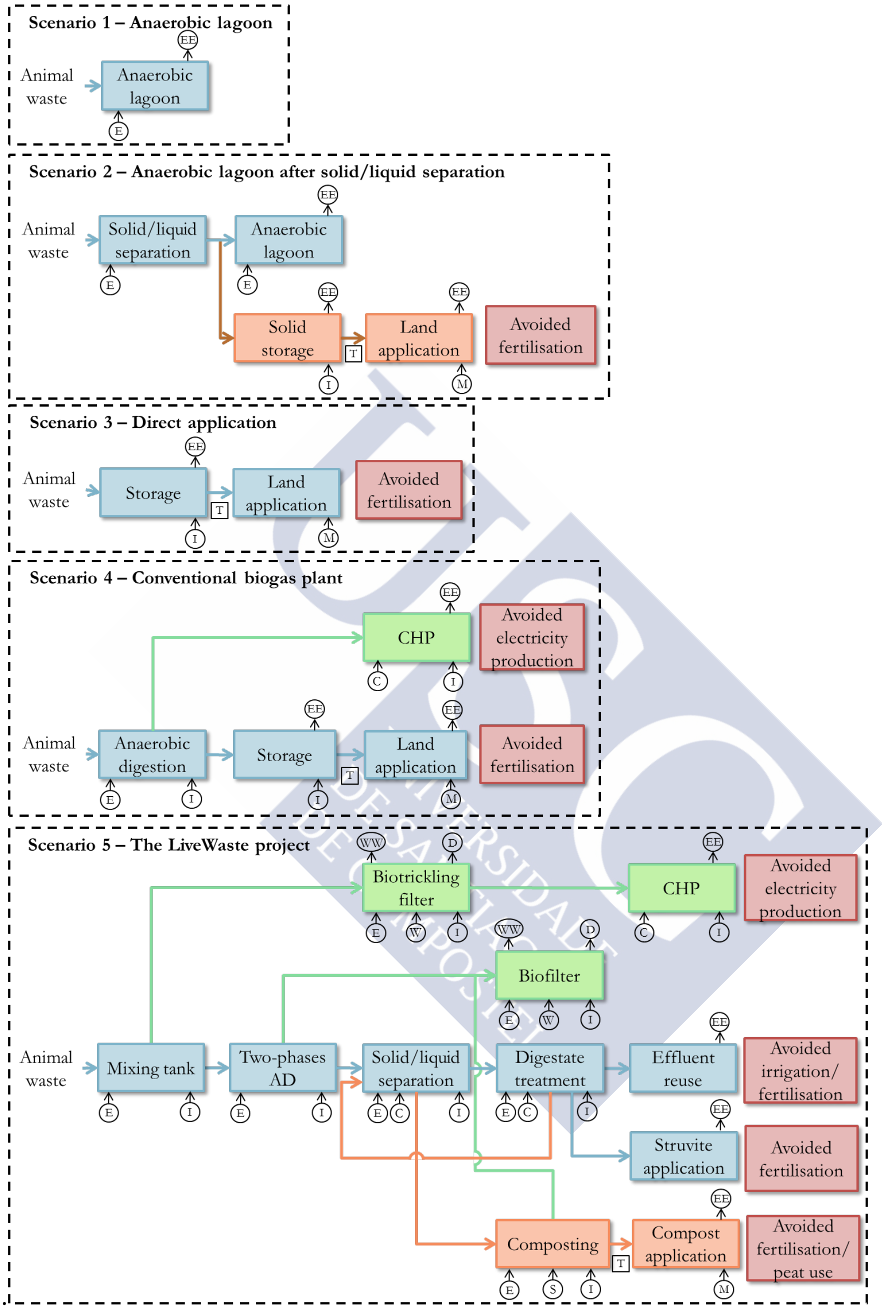


Figure 6.10. Five possible scenarios for animal waste management in Cyprus. Acronyms: EE – emissions; E – electricity; M – machinery; I – infrastructure; C – chemicals; T – transport; WW – wastewater; W – water, D – landfill disposal; S – straw

6.3.2. Life cycle inventory

As in the previous analysis, the LCI was compiled from mass balances (TS, TVS, TN and TP) for each scenario. For better comparison of the results, it is important to avoid the variability due to different input waste characteristics (Mininni et al., 2015). Therefore, in this study, each treatment system under study has been simulated treating animal waste with the same characteristics. This allows assessing all different solutions based on a common basis and conditions. Methane, ammonia, nitrogen oxides and nitrous oxide emissions from anaerobic lagoons as well as liquid and solid storage have been computed according to IPCC (2006). This methodology does not differ between ammonia and nitrogen oxides; however, since nitrogen is mostly emitted as ammonia, it has been considered that 90% is emitted as ammonia and 10% as nitrogen oxides. From the total methane production potential of the animal manure, expressed in terms of m^3 CH_4/kg TVS: 77%, 39% and 4% are emitted in the anaerobic lagoon, the liquid and the solid storage, respectively. Regarding emissions of nitrogen-based compounds, in the anaerobic lagoon 40% of the nitrogen content in the animal waste is emitted mainly as ammonia but also as nitrogen oxides; whereas in the liquid and solid storage, 48% and 45% of the nitrogen input is emitted as ammonia and nitrogen oxides. This methodology considers that anaerobic lagoons and liquid storage do not emit nitrous oxide; while in the solid storage, 0.5% of the nitrogen contained in the waste is emitted as nitrous oxide. In addition, this methodology was also applied when accounting for ammonia, nitrogen oxides, nitrous oxide and nitrate emissions derived from the application of mineral and organic substrates into the soil. Concerning organic substrates, this methodology establishes that 20% of the nitrogen applied is emitted as ammonia and nitrogen oxides, 1% as nitrous oxide and 30% as nitrate. With regard to mineral fertilisers, it considers that nitrous oxide and nitrate emissions are the same; however, ammonia and nitrogen oxides emissions only represent 10% of the nitrogen applied. In addition, a transport distance of 25 km has been considered from the site of manure/digestate production to the agricultural land where it is applied, according to an analysis of the current situation in Cyprus. Electricity requirements have been computed considered the consumption factors provided in Tchobanoglous et al. (2014). The LCI used to perform the environmental assessment of each scenario is given in Table 6.5

Table 6.5. LCI of the five scenarios under study

	Scenario 1		Scenario 2		Scenario 3		Scenario 4		Scenario 5	
Inputs from technosphere										
Materials, transport and energy										
Livestock waste	1	t	1	t	1	t	1	t	1	t
Polymer									10.8	kg
Sodium hypochlorite									0.30	g
Magnesium oxide									0.28	kg
Acetic acid									2.36	kg
Tractor			0.009	kg	0.03	kg	0.03	kg		
Agricultural implement			0.03	kg	0.06	kg	0.06	kg		
Diesel			0.08	kg	0.21	kg	0.21	kg		
Tractor and trailer			3.72	t·km	24.6	t·km	24.2	t·km		
Electricity	0.77	kWh	1.10	kWh	0.39	kWh	2.76	kWh	6.40	kWh
Avoided materials										
Nitrogen fertiliser			0.26	kg	0.84	kg	0.84	kg	0.51	kg
Phosphorus fertiliser			0.63	kg	1.06	kg	1.06	kg	1.34	kg
Electricity							53.9	kWh	53.9	kWh
Peat									94.2	kg
Inputs from nature										
Avoided resources										
Water									789	L

Table 6.5. LCI of the five scenariosunder study (cont.)

	Scenario 1		Scenario 2		Scenario 3		Scenario 4		Scenario 5	
Outputs to nature										
Emissions to air and water from anaerobic lagoon										
Methane, biogenic	14.5	kg	5.81	kg						
Hydrogen sulphide	0.02	kg	0.009	kg						
Ammonia	1.08	kg	0.76	kg						
Nitrogen oxides	0.20	kg	0.19	kg						
Nitrogen to water	1.48	kg	1.04	kg						
Phosphorus to water	0.53	kg	0.21	kg						
Emissions to air and water from manure/ digestate storage										
Methane, biogenic			0.45	kg	7.35	kg	0.17	kg		
Hydrogen sulphide			0.001	kg	0.01	kg	0.28	g		
Ammonia			0.36	kg	1.30	kg	1.30	kg		
Nitrogen oxides			0.07	kg	0.24	kg	0.24	kg		
Nitrous oxide			0.006	kg				kg		
Emissions to air and water from land application										
Ammonia			0.09	kg	0.28	kg	0.28	kg	0.43	kg
Nitrogen oxides			0.02	kg	0.07	kg	0.07	kg	0.11	kg
Nitrous oxide			0.006	kg	0.02	kg	0.02	kg	0.03	kg
Nitrate			0.54	kg	1.71	kg	1.71	kg	2.49	kg
Phosphate			0.01	kg	0.02	kg	0.02	kg	0.03	kg
Avoided emissions to air and water from mineral fertilisation										
Ammonia			0.03	kg	0.09	kg	0.09	kg	0.06	*kg*
Nitrogen oxides			0.004	kg	0.01	kg	0.02	kg	0.01	kg
Nitrous oxide			0.003	kg	0.008	kg	0.01	kg	0.008	kg
Nitrate			0.35	kg	1.11	kg	1.11	kg	0.68	kg
Phosphate			0.01	kg	0.02	kg	0.02	kg	0.03	kg

6.3.3. Life cycle impact assessment

The ReCiPe H Midpoint methodology and the CML 2001 was selected to perform the LCIA (Goedkoop et al., 2009; Hischier et al., 2009), in order to analyse the same impact categories than in section 2.3: CC, OD, TA, FE, ME, POF, WD, FD and MA. The characterisation results of each scenario are presented in Table 6.6 per FU, that is 1 tonne of animal manure treated. The most influential processes that result in the highest environmental burdens were identified by studying each impact category in detail.

Table 6.6. Characterisation results of Scenarios 1, 2, 3, 4 and 5

		Scenario 1	**Scenario 2**	**Scenario 3**	**Scenario 4**	**Scenario 5**
CC	(kg CO_2 eq/FU)	404	176	209	-41.9	-28.3
OD	(kg CFC-11 eq/FU)	$1.1 \cdot 10^{-7}$	$9.7 \cdot 10^{-8}$	$5.6 \cdot 10^{-7}$	$-7.0 \cdot 10^{-6}$	$-6.3 \cdot 10^{-6}$
TA	(kg SO_2 eq/FU)	2.76	3.04	3.82	3.46	0.76
FE	(kg P eq/FU)	0.53	0.21	$6.5 \cdot 10^{-5}$	0.01	0.001
ME	(kg N eq/FU)	1.59	1.20	0.29	0.79	0.51
POF	(kg NMVOC/FU)	0.35	0.34	0.43	0.24	0.02
WD	(m^3/FU)	0.002	-0.05	-0.07	-0.18	-0.73
FD	(kg oil eq/FU)	0.25	0.16	1.48	-15.0	-7.19
MA	(m^3 air/FU)	54,728,688	24,306,317	28,658,306	1,555,748	439,472

Climate change

As shown in Figure 6.11, Scenarios 1, 2 and 3 achieved important environmental impacts in CC; while Scenarios 4 and 5 attained environmental benefits. In more detail, since the IPCC methodology considers that there are no nitrous oxide emissions, methane emissions from the anaerobic lagoon in Scenarios 1 and 2 and from the storage of liquid manure in Scenario 3 were the main contributor to these impacts, being responsible of 98% of the impacts produced. In Scenario 3, the produced manure is stored in an open tank, where important methane emissions are produced, resulting in higher environmental impacts than Scenario 2. The separation of the manure into liquid and solid fractions in Scenario 2 helped to improve the environmental profile of this management scheme since it diverted part of the manure from anaerobic lagoons, reducing the derived environmental impacts.

Regarding Scenarios 4 and 5, the environmental benefits attained are linked with the production of electricity from manure, which is especially beneficial in Cyprus due to the high ratio of fossil fuels in its electricity mix. While Scenario 4 produces electricity from manure with well-established technologies, Scenario 5 does it using more recent and advanced technologies; this also implied lower electricity consumption in Scenario 4 compared with Scenario 5, resulting in higher overall GHGs savings.

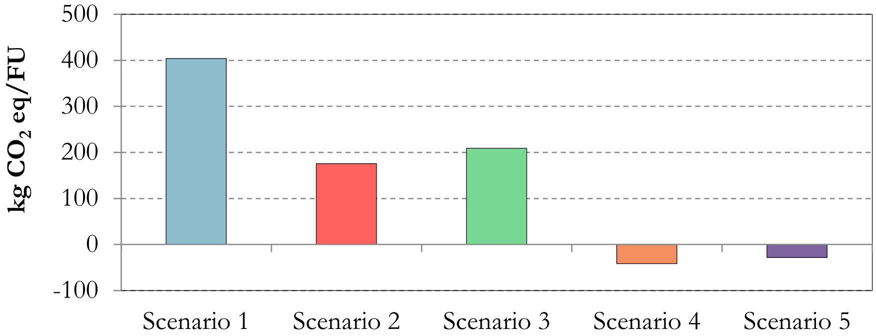


Figure 6.11. Characterisation results in CC per FU

Ozone depletion, photochemical oxidant formation and fossil depletion

OD and FD show a similar behaviour regarding the environmental impacts produced by the scenarios under study. On one hand, the environmental impacts of Scenarios 1, 2 and 3 in OD and FD are very low (almost negligible) compared with Scenarios 4 and 5. The reason is that these scenarios have a very low mechanisation level, which implies low environmental burdens associated to the production of electricity, infrastructure or diesel. On the other hand, the avoided electricity production from the Cypriot electricity grid resulted in environmental savings for both impact categories in Scenarios 4 and 5. Scenario 4 achieved higher environmental savings than Scenario 5 in these categories since it produces electricity from biogas with lower requirements of infrastructure and consumption of electricity. The reason is that Scenario 5 presents a more complex technology for the treatment of the produced anaerobic effluent, resulting in higher consumption of infrastructure and electricity.

In the case of POF, the environmental impacts, presented in Figure 6.12, followed a different behaviour. The impacts produced were related to direct emissions of methane and nitrogen oxides derived from the anaerobic lagoon and the storage and application on land of the manure or digestate. In Scenarios 3 and 4, the most important source of nitrous oxide emissions was the storage of liquid manure and digestate. Nevertheless, in the case of Scenario 4, these impacts are partially reduced due to the avoided electricity production. In the case of Scenario 1 and 2, nitrous oxide emissions derived mainly from use of anaerobic lagoons for the management of animal manure.

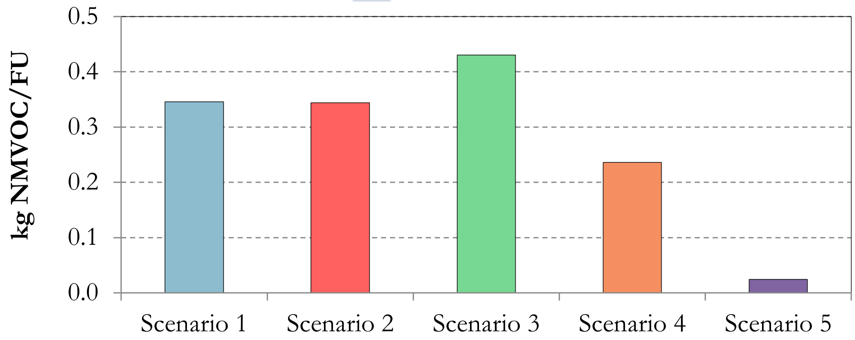


Figure 6.12. Characterisation results in POF per FU

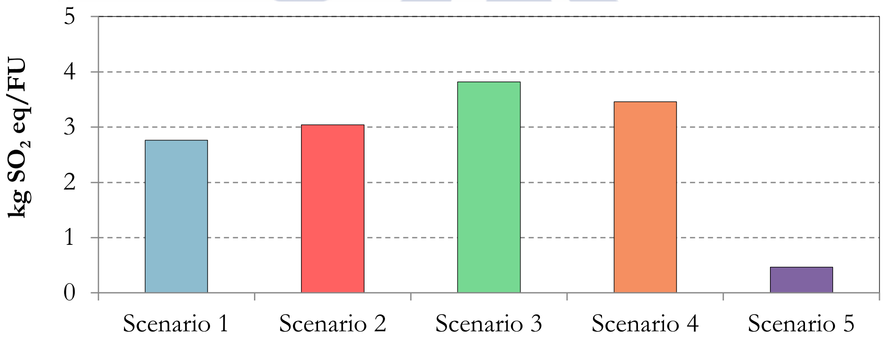
Terrestrial acidification, freshwater and marine eutrophication

In terms of TA, Scenario 3 and 4 reached the worst environmental results since they include the storage of the raw manure and digestate in open tanks, which entails the highest nitrogen lost as ammonia (3.82 and 3.46 kg SO_2 eq/FU), as depicted in Figure 6.13. In addition, Scenario 1 and 2 reached similar impacts (2.76 and 3.04 kg SO_2 eq/FU); in this case, ammonia is mainly emitted as consequence of the degradation of the organic matter in the anaerobic lagoon. Finally, Scenario 5 achieved the best environmental results in this impact category (0.47 kg SO_2 eq/FU). This treatment scheme includes the biological removal of nitrogen performed in the SBR resulted in lower nitrogen content in the final products, reducing the amount of ammonia emissions when applying these products to agricultural land.

Regarding FE and ME, Scenarios 1 and 2 achieved the worst environmental results due to the management of manure in anaerobic lagoons, as presented in

Figure 6.13 for the case of ME. In this practice, the nutrients contained in the manure cannot be taken by any plant and it does not result in the production of any valuable products that help to offset the environmental impacts produced. The low impacts in Scenarios 3, 4 and 5 are the consequence of the use of manure and digestate as organic fertilisers. In more detail, phosphate and nitrate leaching occurs during the application of both organic (manure and digesate) and mineral fertilisers on land. Therefore, phosphate and nitrate emissions due to the use of manure or digestate as an organic fertiliser are partially offset by avoided emissions from the substitution of mineral fertilisers. In addition, the avoided production of phosphate and nitrogen mineral fertilisers also helped to counteract these impacts. Furthermore, the recovery of phosphorus and the removal of nitrogen in Scenario 5 also reduced the derived eutrophication impacts.

(a)



(b)

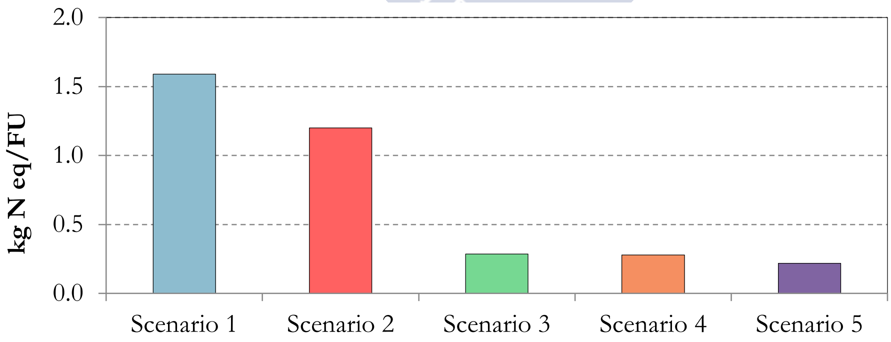


Figure 6.13. Characterisation results in (a) TA and (b) ME per FU

Water depletion and malodours air

The environmental results in WD presented in Figure 6.14 are related to water savings due to the replacement of products such as mineral fertilisers. In addition, the production of a high quality effluent in Scenario 5 results in the avoidance of the same quantity of water from natural origin used for irrigation.

In terms of MA, the environmental impacts produced are mostly related to direct emissions of hydrogen sulphide. These emissions are produced during the decomposition of organic matter and linked to the production of methane in the anaerobic lagoons. On the other hand, Scenario 5 achieved the best environmental results for this impact category because this treatment scheme treats all derived emissions from the plant in a biofilter as well as the produced biogas in a biotrickling filter.

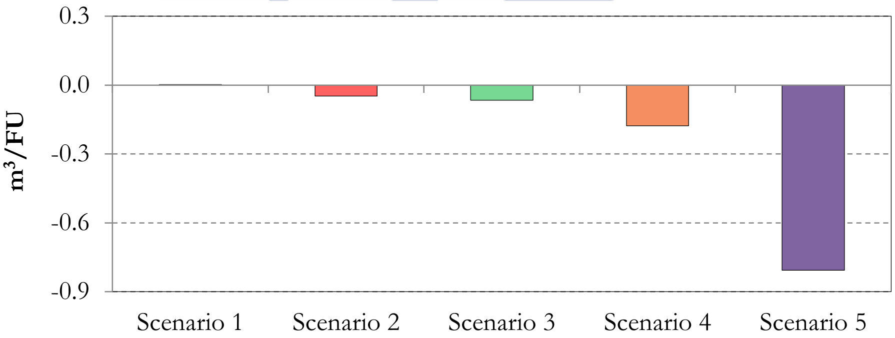


Figure 6.14. Characterisation results in WD per FU

Normalisation

Due to the disparities in the environmental results, normalisation has been performed to obtain a single result for each scenario under study. In more detail, the normalisation factors provided by the ReCiPe Midpoint methodology for Europe have been used. The obtained results are shown in Figure 6.15. According to the results, the normalised results of Scenario 1 and 2 are significantly higher than those of Scenario 3, 4 and 5, proving the environmental damage caused by the important emissions to air and water that derived from the use of anaerobic lagoons.

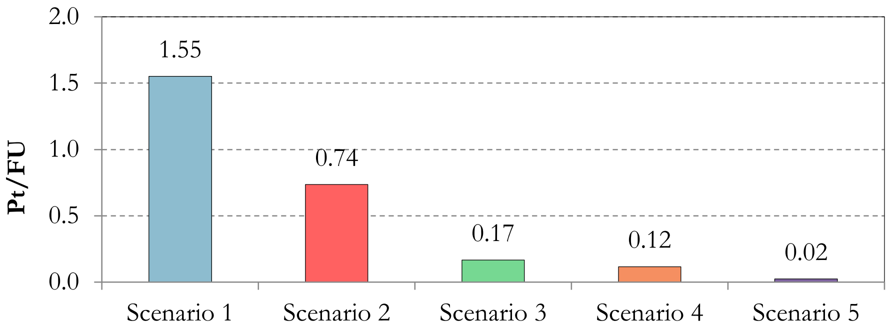


Figure 6.15. Normalised results for the five scenarios under study per FU

Scenario 3 achieved better results than Scenario 4, since despite that the total content of nutrients in the manure and the digestate are similar, Scenario 4 produces biogas which has a very positive effect on the environmental profile due to bioenergy production. Nevertheless, despite anaerobic digestion preserves the nutrient content of manure, a number of properties changed (Anderson-Glenna et al., 2013; Möller and Müller, 2012), including higher pH, higher ammonium/total nitrogen ratio, higher mineral nitrogen content and lower total organic carbon. Therefore, the methodology for the calculation of derived emissions from these management systems should have into account the specific characteristics of the organic substrates.

Finally, Scenario 5 achieved the best environmental profile of all scenarios under study. Despite the higher requirements in electricity and infrastructure, the innovative treatment scheme proposed in the LiveWaste project produces a quality biogas that can be used as a source of renewable energy. In addition, it performs the recovery and removal of nutrients from the produced digestate, which have been identified as a major source of emissions that cause environmental burdens not only in eutrophication but also in other impact categories such as acidification. In addition, the treatment of all gas streams produced during the process assures the reduction of direct emissions, improving the environmental profile of the process.

6.3.4. Sensitivity analysis

Different analysis were performed regarding: i) the use of the produced biogas and ii) the methodology for estimating derived emissions

Different biogas use pathways

The energy contained in biogas can be used for the production of heat and/or electricity in a co-generation unit. In addition, after proper upgrading to biomethane, biogas can be injected in the natural gas grid or be used as vehicle fuel after being upgraded (Petersson, 2013). In more detail, the efficiency regarding bioenergy production from biogas can greatly vary among the different biogas utilisation pathways (Poeschl et al., 2012a). Therefore, a sensitivity analysis was conducted to analyse different possible options for the use of the produced biogas at full-scale integration of the LiveWaste project to identify the best potential pathway from an environmental point of view (Poeschl et al., 2012b). In more detail, three different biogas utilisation pathways have been considered:

• **Option 1. Biogas upgrading (natural gas grid)** – In this case, the produced biogas is upgraded into biomethane (>96%) using a pressure swing adsorption technology (PSA) as described in Jungbluth et al., (2007). The produced biomethane is injected into the natural gas grid and it can be used for the production of heat in substitution of natural gas.

• **Option 2. Biogas upgrading (vehicle fuel)** – As in the previous case, biogas is upgraded into biomethane through the PSA process (>96%). The produced biomethane can be used as a vehicle fuel, considering that 1 Nm^3 of enriched biogas replaces 0.7 L of diesel (Murphy and McCarthy, 2005) and that the fuel consumption of a car for a distance of 100 km is 6 L.

• **Option 3. Electricity production** – In this option, biogas is used in a co-generation unit for the production of electricity and heat, with efficiencies of 35% and 50%, respectively. As in the base case, the electricity produced is fed into the Cypriot national grid, while the heat produced is used in the facilities and the surplus is emitted as a waste to the atmosphere. Therefore, it has been considered that only electricity substitutes the equivalent amount of electricity in the Cypriot grid.

The FU selected to compare the different options for the use of biogas is 1 m^3 of biogas produced. Within the system boundaries, the production of the biogas is not included since it is the same in the three possible options. Moreover, the avoided environmental loads due to the substitution of different fossil energy carriers would be included in the analysis to consider the specific case study. The comparative results obtained for each option of biogas use are shown in Figure 6.16.

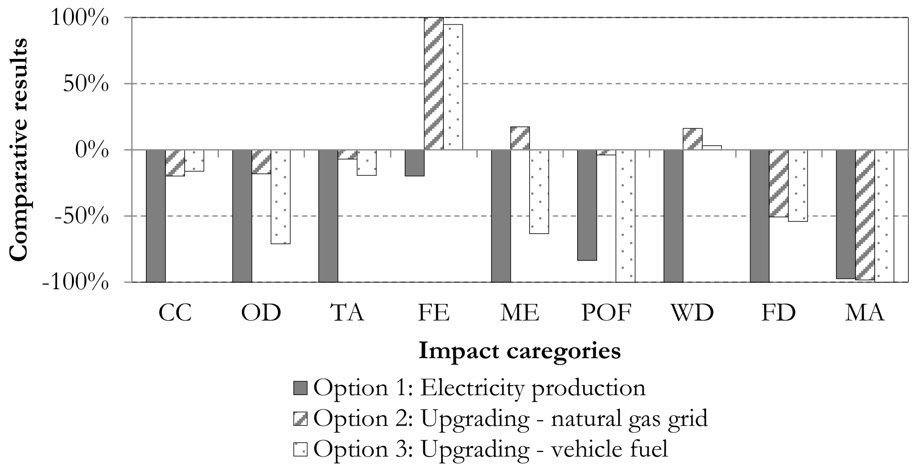


Figure 6.16. Comparative environmental results for the different biogas use pathways

The results highly change among the impact categories under study. However, electricity production and biogas upgraded for its use as a vehicle fuel appears as viable options for the use of the produced biogas. The most common use for biogas in Europe is the generation of electricity and heat through a cogeneration unit (Patterson et al., 2011). The main reason is the economic incentives granted for selling the electricity to regional suppliers. However, the environmental results depend on each specific country since credits from the avoided products play an important role in offsetting the environmental impacts of the applied treatment scheme. For example, the production of renewable energy is more interesting in a country as Cyprus with an electricity mix highly dependent on fossil fuels. In other countries, the replacement of petrol in vehicles can appear as an interesting option for the reduction of GHGs emissions. Furthermore, the choice of the biogas use depends on many factors beyond the environmental ones. Patterson et al. (2011) analysed the potential use of biogas for vehicle fuel and co-generation in United Kigndom. They concluded that the capital costs required to produce biomethane transport fuel were 19% higher than for co-generation; however, operational costs were around 26% lower due to the replacement of diesel.

Methodology for emissions estimation

As discussed before, the calculation of derived emissions from the system under study is a major issue in LCA studies. There are different available methodologies which entail different assumptions. In the base case, the IPCC (2006) methodology was applied to calculate emissions from the anaerobic lagoons, liquid and solid storage of manure and digestate as well as from the application on land of mineral and organic fertilisers (Methodology 1). As shown, this methodology allows the quantification of a varied range of emissions, which is especially important when comparing a wide variety of treatment systems, as in this case. In this sensitivity analysis, the results obtained when applying this methodology were compared with the ones achieved using a combined methodology between the emission rates proposed by IPCC (2006) and by Rotz, (2004) (Methodology 2). Rotz (2004) considers that in the anaerobic lagoon 70% of the TN is lost as emissions to the atmosphere, as ammonia (50%), as nitrogen gas (45%) and, differently from the other method, as nitrous oxide (5%). Regarding liquid storage, a 30% loss of nitrogen as ammonia is considered. Finally, for the solid storage of manure, a loss of 20% is suggested, mainly as ammonia (85%) but also as nitrate (10%) and nitrous oxide (5%). Nevertheless, in order to complete the assessment, IPCC (2006) methodology was used to calculate the methane emissions from anaerobic lagoons and storage tanks as well as emissions of nitrogen-based compounds derived from the application of the mineral and organic substrates on agricultural land.

The comparative results obtained using the two methodologies for accounting direct emissions are shown in Figure 6.17 for each scenario with the FU of 1 tonne of manure treated.

It can be noticed that Scenario 5 did not experience any change when using Methodology 1 or 2. The reason for these results is because in this scenario the direct emissions are only related to the application of the final products in agricultural land, since this system treats all in situ emission and does not include any open storage tank

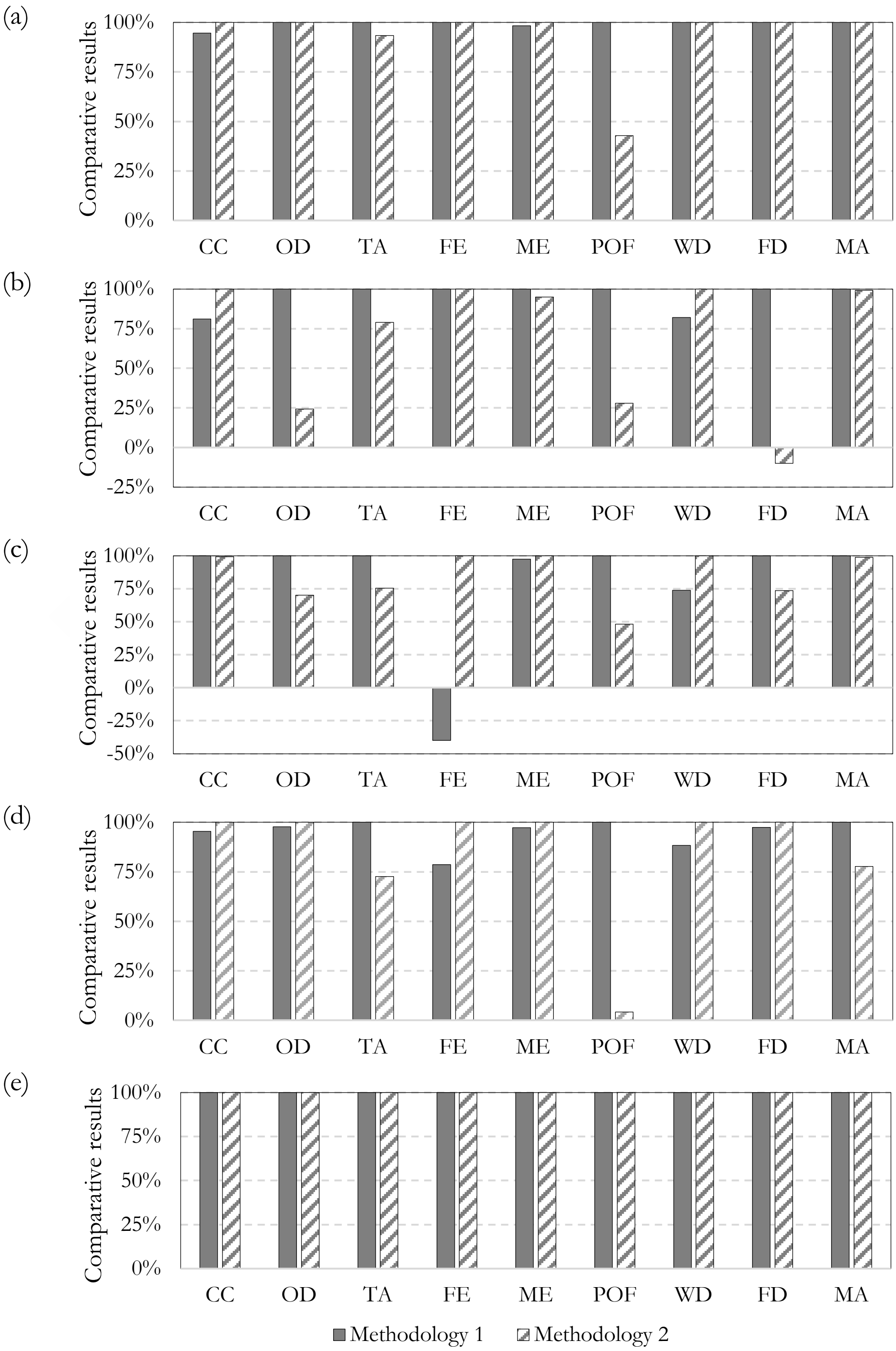


Figure 6.17. Comparative results obtained for (a) Scenarios 1, (b) 2, (c) 3, (d) 4 and (e) 5

The use of Methodology 1 entailed higher impacts in CC in Scenarios 1, 2 3 and 4 due to the account of nitrous oxide emissions from the solid storage of manure as well as from anaerobic lagoons. On the contrary, Scenarios 1, 2 3 and 4 presented a reduction in the environmental impacts produced in POF when using Methodology 2 (from 52% to 96% of the impacts). The reason is related to the emissions of nitrogen oxides; different from IPCC (2006), Rotz (2004) does not include emission factors for these emissions in anaerobic lagoons and in storage tanks.

In addition, OD and FD also experienced reductions despite these impact categories are not influenced by direct emissions from the treatment systems. In more detail, lower emissions of nitrogen during the treatment process ends up in higher nitrogen content in the final product that is applied on land. In the same way, the results obtained in FE in Scenarios 1 and 2 are the same since in these scenarios the impacts were produced by direct emissions of phosphorus to water. However, differences were found in Scenarios 3 and 4 for this impact category. Despite the environmental burdens produced in this impact category are related to phosphate leaching from the application of organic fertilisers in the soil, these impacts were almost counteracted by avoided phosphate emissions from the application of mineral fertilisers, since the replacement value was 99%. Therefore, the differences found are related to different nitrogen losses as emissions to air that produced different content of nitrogen in the final product that is applied on land, resulting in different amount of avoided production of mineral fertilisers.

Finally, TA also presents a reduction when using Methodology 2 (from 7% to 27%) due to lower direct ammonia emissions from anaerobic lagoons and liquid and solid storage.

6.4. Multicriteria analysis of the manure practices in Cyprus

This section integrates the environmental, social and economic indicators studied in the LiveWaste project by applying the AHP (Saaty, 2008). The objective of the analysis is to select the most sustainable alternative for the management of the livestock waste in Cyprus.

6.4.1. Goal and formulation of alternatives

As mentioned, the goal of the assessment is to select the most sustainable waste management for the specific context of Cyprus. To do so, the multicriteria analysis AHP was applied to compare the two most common practices for animal waste management in Cyprus with the one proposed in the LiveWaste project.

• **Alternative 1** – It includes the separation of waste and the treatment of the liquid fraction in an anaerobic lagoon, which is the most spread waste management scheme in Cyprus. This alternative entails low capital and operational costs; however, it is a source of pollution and does not produce any valuable product.

• **Alternative 2** – This option deals with the anaerobic digestion of animal waste in a conventional biogas plant. This simple treatment scheme converts the animal waste into bioenergy; the produced digestate is suitable for its reuse in agriculture as an organic fertiliser; however, the produced digestate is not a marketable product.

• **Alternative 3** – This alternative is the one proposed in the LiveWaste system. It involves higher technological level (higher infrastructure and energy requirements) compared with Alternatives 1 and 2. The system entails higher capital and operating costs; however, it results in several high-quality marketable products in line with the concept of circular economy.

6.4.2. Sustainable indicators selection and evaluation

The next step is the selection of the criteria and sub-criteria to be used for the assessment. In this case, the criteria include environmental, social and economic indicators. For each criteria category, four specific indicators were selected. The selection of the indicators is made according to which of the indicators better translates a comprehensive and meaningful assessment of waste management sustainability (Milutinović et al., 2014). The hierarchy tree where all criteria and sub-criteria are presented according to their importance is shown in Figure 6.18.

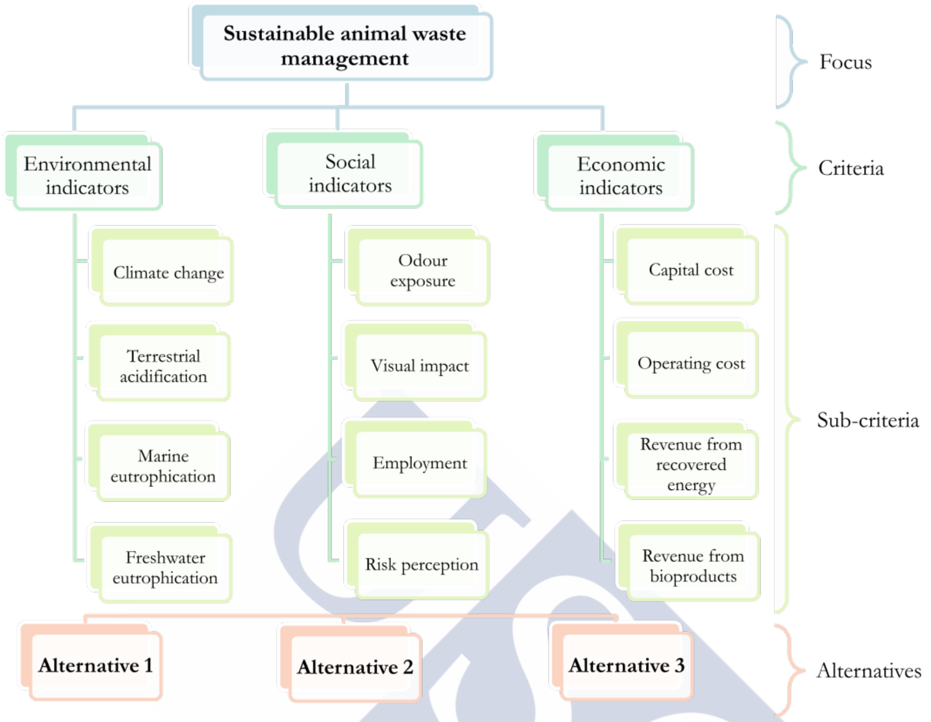


Figure 6.18. The hierarchy tree for the selection of the most sustainable alternative

The indicators were selected in accordance with the results obtained in the LiveWaste project regarding the environmental and socio-economic impacts of the project. The definition of each indicator can be found in Table 6.7. In addition, the summary of the results of each sub-criterion under study according to the results of the project regarding each alternative are presented in Table 6.8.

6.4.3. Determination of global priority vectors

The most important step of these decision-making processes is a correct pair-wise comparison, whose quantification is the most crucial step in multi-criteria methods which use qualitative data (Milutinović et al., 2014).

A pair-wise comparison matrix should be developed for each of the criteria selected by setting the rating of relative importance between the two alternatives considered.

Table 6.7. Definition of the indicators selected

Environmental indicators	Climate change	The impact (positive or negative) produced in climate change is defined as the weighted sum of the life cycle emissions of greenhouse gases, being the most important ones carbon dioxide, methane and nitrous oxide.
	Terrestrial acidification	It considers the environmental impacts related to the atmospheric deposition of acidifying inorganic substances that cause a change in acidity in the soil.
	Marine eutrophication	This indicator measures eutrophication of aquatic bodies which can be defined as nutrient enrichment of the oceans, especially in terms of nitrogen.
	Water depletion	This indicator quantifies the water saved or consumed in each alternative under study.
Social indicators	Employment	This indicator is defined as the amount of direct and indirect labour required for each livestock waste management scenario.
	Visual impact	This indicator measures the visual impact of the waste treatment plants, taking into account the visibility, fragility, and contour quality.
	Odour exposure	This indicator measures the most common residents' complaint associated with manure management.
	Risk perception on human health	This indicator is based on citizens' fear of negative health effects due to normal operation of the electricity generation technology.
Economic indicators	Capital cost	The investment costs of each alternative under study include land cost, the costs of the necessary buildings and of all the necessary equipment.
	Operating cost.	This indicator considers the costs of maintenance and operation of each alternative.
	Revenue from energy recovery	This indicator represents the income derived from the sale of electricity from renewable sources that is produced when the biogas produced is used in a co-generation engine.
	Revenue from bioproducts	This indicator is defined as the economic revenues from the sales of struvite as fertiliser and the compost as soil conditioner

Table 6.8. Results of each indicator regarding the alternatives under study

			Alternative 1	Alternative 2	Alternative 3
Environmental indicators	Climate change	(kg CO_2 eq/t waste)	202	-40	-22
	Terrestrial acidification	(kg SO_2 eq/t waste)	2.48	2.52	0.24
	Marine eutrophication	(kg N eq/t waste)	1.17	0.29	0.16
	Water depletion	(m^3/t waste)	-0.07	-0.22	-0.41
Social indicators	Odour exposure		High	Low	Very low
	Visual impact		High	Low	Low
	Employment		Low	Medium	Medium-high
	Risk perception on human health		High	Low	Low
Economic indicators	Capital cost	(€/t waste)	10	66.8	71.0
	Operating cost	(€/t waste)	~0*	3.20	7.67
	Revenues from energy recovery	(€/kWh)	-	0.135	0.135
	Revenues from bioproducts	(€/t struvite) (€/t compost)	-	-	0.40 7.50

* The operating costs of anaerobic lagoons are related to the removal of sludge from the bottom once per year; therefore, they have been considered minimal (EPA, 2002).

Pair-wise comparisons are quantified by using the scale shown in Table 6.9. In addition, a reciprocal rating (e.g. 1/3, 1/5, 1/7, 1/9) applies when the second alternative is preferred to the first. Finally, the value 1 is always assigned to an alternative comparison itself.

Table 6.9. AHP measurement scale

Intensity of importance	Definition
1	Equal importance
3	Moderate importance
5	Strong importance
7	Very strong importance
9	Extreme importance
2,4,6,8	Intermediate values

The relative importance of each alternative for each indicator selected was based on the expertise of a panel of experts. Then, it is required the calculation of the normalised matrix, this is done by dividing each number of a column of the matrix of pairwise comparison for the total sum of the column. Then the priority vector is determined by estimating the average of each row of the normalised matrix. This average value for each row represents the priority vector of the alternative with respect to the criteria considered. Finally, the consistence ratio is quantified to measure the consistence of each pair-wise comparison matrix. As a result, a priority matrix is developed summarising the results obtained in the previous steps, as shown in Tables 6.10, 6.11 and 6.12.

Table 6.10. Priority matrix including the environmental factors

	Climate change	Terrestrial acidification	Marine eutrophication	Water depletion
Alternative 1	0.33	0.40	0.40	0.25
Alternative 2	0.17	0.20	0.20	0.25
Alternative 3	0.17	0.20	0.20	0.25

Criteria weights denote the importance of each criterion and sub-criterion when synthesising the scoring of the three alternatives for the management of animal waste (Chatzimouratidis and Pilavachi, 2009).

Table 6.11. Priority matrix including social indicators

	Odour exposure	Visual impact	Employment	Risk perception
Alternative 1	0.21	0.33	0.27	0.19
Alternative 2	0.05	0.08	0.07	0.11
Alternative 3	0.11	0.17	0.13	0.14

Table 6.12. Priority matrix including economic factors

	Capital cost	Operating cost	Revenues - energy	Revenues - bioproducts
Alternative 1	0.13	0.09	0.17	0.12
Alternative 2	0.25	0.18	0.17	0.18
Alternative 3	0.25	0.36	0.33	0.35

Finally, global priority vector is quantified to select the best alternative. This is done by multiplying the priority of the criteria and the priority matrix of alternatives. The obtained results can be found in Table 6.13.

Table 6.13. Global priority vectors for Alternative 1, 2 and 3

	Global priority vector
Alternative 1	0.11
Alternative 2	0.36
Alternative 3	0.53

As shown, Alternative 3 would be selected as the most sustainable option.

6.4.4. Sensitivity analysis

While objective data is difficult to alter, subjective assessments can vary among decision makers with different culture, education and experiences. To overcome this obstacle, sensitivity analysis can be used to analyse how a variation of criteria weights would affect the partial and global results (Chatzimouratidis and Pilavachi, 2009). Within this sensitivity analysis, different priorities have been given to the criteria to determine if a change in the weights given would change the obtained results.

- ✓ SA1: all criteria have the same importance
- ✓ SA2: environmental factors are the most important, with an intensity of 7.
- ✓ SA3: economic factors are the most important, with an intensity of 7.
- ✓ SA4: social factors are the most important, with an intensity of 7.

The results obtained for the four options under study can be found in Table 6.14.

Table 6.14. Results of the sensitivity analysis

	SA1	SA2	SA3	SA4
Alternative 1	0.15	0.10	0.10	0.24
Alternative 2	0.33	0.36	0.37	0.28
Alternative 3	0.52	0.54	0.54	0.48

In any case, Alternative 3 was identified as the best sustainable option for the management of animal manure, regardless the weight provided to the criteria under study.

6.5. Conclusions

This chapter analysed from an environmental perspective the four possible configurations of the LiveWaste system. Despite the characterisation results changed among the impact categories under study, the energy consumed process as well as the removal or not of nutrients have been identified as the key parameters in defining the environmental profile of each configuration. Configuration 2, which only includes the struvite crystalliser, shows the best environmental results in terms of CC, OD, POF and FD. However, the role of the SBR in the removal of nitrogen showed positive effect in several impact categories such as TA and ME. In this sense, it can be concluded that Configuration 4, which considers the struvite crystalliser before the SBR, was identified as the best scheme for the effective removal of nutrients.

Furthermore, the chapter also analysed the environmental performance of a full-scale biogas plant which performs the treatment scheme proposed in the LiveWaste project. In more detail, the environmental profile of this plant was compared with four conventional treatment schemes available for the management of livestock waste in Cyprus. The use of anaerobic lagoons was identified as the worst manure management practice, producing large impacts in CC, POF and eutrophication categories due to the important emissions that arise from the anaerobic decomposition of the manure. The conventional biogas plant analysed in Scenario 4 show positive performance in impact categories such as CC, OD and FD since it is able to produce biogas without the level of technology presented in the Livewaste alternative. However, it produced higher environmental impacts in TA due to the higher ammonia emissions associated to the application of the digestate on land, suggesting the need to properly treat the produced anaerobic effluent. Finally, Live-Waste plant (Scenario 5) achieved the best overall environmental results (normalisation) not only due to the production of valuable products such as biogas, a reusable effluent and compost, but also due to the removal of nitrogen and the treatment of the gas streams, which helped to reduce the derived environmental impacts. Regarding the produced biogas, the upgrading for vehicle use and co-generation were identified as good options for its use. However, the best utilisation pathway can vary among different countries since the replaced products have a high impact.

Moreover, the sustainability analysis by means of the AHP methodology considered the management of livestock waste in anaerobic lagoons, conventional biogas plant or in a facility according to the LiveWaste approach under environmental, social and economic indicators. The results showed that, even if the capital and operational costs of the LiveWaste treatment are higher, the environmental, social and economic benefits made it the most sustainable animal waste management option.

6.6. List of acronyms

BNR	Biological nitrogen removal
BTF	Biotrickling filter
CC	Climate change
COD	Chemical oxygen demand
CSTR	Continuous stirred tank reactor
DF	Dark fermentation
FD	Fossil depletion
FE	Freshwater eutrophication
FU	Functional unit
HRT	Hydraulic retention time
IPCC	Intergovernmental Panel on Climate Change
LCA	Life cycle assessment
LCI	Life cycle inventory
MA	Malodours air
ME	Marnie eutrophication
NVZ	Nitrate vulnerable zone
OD	Ozone depletion
OLR	Organic loading rate
POF	Photochemical oxidant formation
SBR	Sequencing batch reactor
TA	Terrestrial acidification
TN	Total nitrogen
TP	Total phosphorus
TS	Total solids
TVS	Total volatile solids
VFA	Volatile fatty acid
VOC	Volatile organic compound
WD	Water Depletion

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Section II:

Sewage biogas

Chapter 7: Anaerobic co-digestion of urban organic waste for enhanced biogas yield

Summary

Co-digestion of sewage sludge and source-segregated food waste (SSFW) is one of the different waste management schemes in many European countries. However, its implementation in the United Kingdom is limited by regulatory and management regimes. The aim of Chapter 7 was to analyse the potential environmental advantages of the integration of SSFW management within the facilities of a wastewater treatment plant (WWTP). Different integration rates were proposed: i) mono-digestion of sewage sludge as the base case; ii) the co-digestion of source segregated food waste (SSFW) considering the spare capacity of the existing digester and iii) the co-digestion of the total amount of food waste produced by the community in an additional digester of the WWTP.

According to the results obtained, the implementation of SSFW in an existing WWTP can be considered a suitable option to integrate the management of both waste streams while increasing biogas production (to be converted to electricity and heat). From an environmental point of view, the total integration of the SSFW produced by the community within the WWTP entailed significant benefits, particularly in energy-related categories, reducing the environmental impact compared with the base case. While eutrophication was not significantly affected by the changes proposed, acidification impacts were negatively influenced by the integration of SSFW in the WWTP. These impacts were related to the additional load of nutrients to the digester.

Outline of Chapter 7

7.1. Introduction to anaerobic digestion of sewage sludge

One possibility for the proper management of waste in the framework of a circular economy is to recover valuable products from municipal sewage sludge. In this sense, anaerobic digestion has been applied in the United Kingdom for over 100 years to treat sewage sludge (Houses of Parliament, 2011). However, the use of sewage sludge as a single substrate in anaerobic digestion facilities (mono-digestion) typically results in low loaded and low efficiencies (Cavinato et al., 2013). This is mainly related to low TVS removal efficiencies (20-30%) and low SGP (~0.35 m^3 biogas/kg TVS_{fed}). Recently, the government of the United Kingdom committed to an increase in energy from biogas with the "Anaerobic Digestion Strategy and Action Plant" published in 2011. In this context, there is a growing interest in processing a wider range of substrates including food waste, animal manure and agricultural biomass. As a result, by the end of 2014 over 91 biogas plants using food waste as feedstock were running. Food waste is applied as organic substrate in biogas plants due to its high SGP (~0.6 m^3 biogas/kg TVS_{fed}). However, the use of SSFW for biogas production is limited due to the need of implementing an efficient separate collection scheme (Cavinato et al., 2011). The composition of SSFW varies among regions and seasons (Zhang et al., 2013), which may affect the stability of the operation of biogas plants using SSFW as the only substrate.

Within the water industry, the co-digestion of sewage sludge with SSFW can be enforced to utilise the surplus capacity that often exists in digestion units, improving WWTP economy and allowing waste management at local level at the same time (Braun and Wellinger, 2009; CIWEM, 2011; Fitamo et al., 2016). In addition, anaerobic co-digestion has several benefits, such as better supply of macro and trace elements, potential of using available extra capacities, approaching energy self-sufficiency and outweighing constraints associated with the variability of food waste, including seasonality issues (Dai et al., 2013; Fitamo et al., 2016; Koch et al., 2016, 2015). However, additional pre-treatment equipment has to be installed for the integration of co-digestion in existing WWTPs. Sewage sludge is being co-digested in many WWTPs together with food waste at European level, and full-scale facilities can be found in Denmark, Germany, Switzerland and Italy (Bolzonella et al., 2006; Braun and Wellinger, 2009; Koch et al., 2016). However, in the United Kingdom the current operational and regulatory framework for organic waste anaerobic digestion does not favour anaerobic co-digestion of sewage sludge with SSFW, because they fall into different regulatory regimes (CIWEM, 2011; Iacovidou et al., 2012). More specifically, the land application of the digestate produced using as sewage sludge as substrate is governed by the Sludge Regulations and the Safe Sludge Matrix; while when SSFW is used as feedstock, the Anaerobic Digestate quality protocol and the Publicly Available Specification (PAS) 110 is applied. The Quality Protocol for anaerobic digestate establishes the EoW criteria for digestate, according to which it can be concluded if the digestate is no longer classified as a waste and therefore, be used for agricultural land. Therefore, the aim of Chapter 7 was to provide evidence in terms of environmental benefits of the combined management of these streams to address legal barriers and obstacles for the co-digestion of sewage sludge and SSFW in the United Kingdom. In this way, the study encourages the development of integrated circular value chains towards more sustainable waste management options by the application of LCA. Specific objectives include the identification of potential environmental consequences derived from their co-digestion at different integration rates within a WWTP. Furthermore, it also proposes possible solutions that could increase energy return, reduce GHG emissions and promote better nutrient management. Finally, the political constraints for the implementation of this treatment scheme driven by the political regimes in the United Kingdom were identified and discussed.

7.2. Materials and methods

The integration of SSFW management within municipal wastewater treatment facilities was analysed to identify advantages and drawbacks of wider adoption. Policy barriers that prevent the adoption of this scheme in the United Kingdom were also questioned and discussed in detail.

7.2.1. Schemes for resource recovery from urban organic waste

The proposed schemes and the size of the community were selected to represent the most typical cases of urban wastewater treatment that apply anaerobic digestion in the United Kingdom (DEFRA, 2012). The integration of a co-substrate in the anaerobic digester applied in the sludge line also has an impact on the water line; thus the performance of the whole WWTP was analysed. Three treatment schemes that include different integration rates of SSFW in the sludge digester were evaluated for a community of 150,000 population equivalent (PE) in the United Kingdom.

• **Scheme 1. Anaerobic mono-digestion** – This scheme reflects the current situation in the United Kingdom, where legislation encourages the separate digestion of sewage sludge and food waste. Therefore, sewage sludge is treated according to a mono-digestion scheme in a conventional facility to produce biogas that can be valorised for heat and electricity.

• **Scheme 2. Anaerobic co-digestion when surplus capacity is available** – Usually, anaerobic digesters installed in WWTP have a spare capacity that could be further exploited. In this scheme, the surplus capacity in the digester is filled with SSFW that is used as a co-substrate. Thus, a proportional amount of SSFW needs to be delivered to the WWTP and suitable pre-treatment is required. This modification is accompanied by higher biogas yields increasing the efficiency of the WWTPs.

• **Scheme 3. Anaerobic co-digestion of the total amount of waste** – The co-digestion of sewage sludge with SSFW within the WWTPs can be taken further with the construction of an additional digester that provides the surplus volume required to treat the total amount of sewage sludge and SSFW produced by the community. Therefore, in this scheme both sewage sludge and SSFW are co-digested in the WWTP.

Baseline scheme applying anaerobic digestion

All different schemes were assessed based on a common set of boundary conditions with the purpose of minimising the introduction of variability errors/ uncertainties due to different influent characteristics, process conditions and effluent quality (Mininni et al., 2015). The latter allowed better comparison of treatment schemes. The base scenario includes a conventional municipal WWTP that applies primary and secondary treatment in the water line and anaerobic digestion in the sludge line, as presented in Figure 7.1. Since it is a comparative study, the treatment scheme was identical for all schemes.

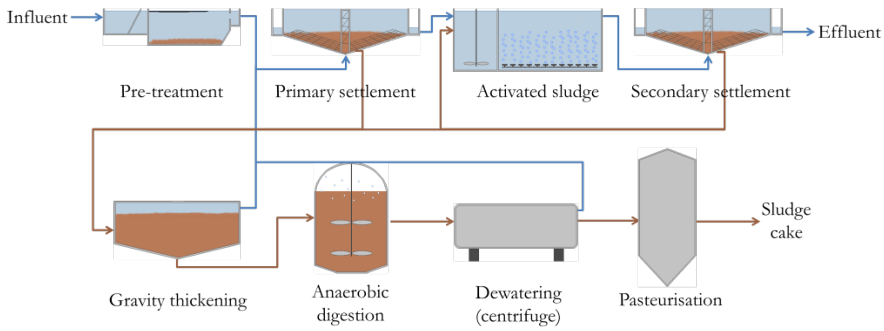

Figure 7.1. Treatment scheme considered in the current study

After pre-treatment, the heavier organic matter is then separated in the primary settler by gravity, and primary sludge is removed for further treatment in the sludge line. The remaining organic matter is removed in a conventional activated sludge system, which also performs the conventional nitrification-denitrification process. The operational conditions considered were 15 days of solids retention time (SRT) and 12 h of HRT. The activated sludge produced in the secondary settler is also treated in the sludge line. Primary and secondary sludge are mixed and then thickened to achieve a concentration of total solids around 6-10%. The surplus effluent is recirculated to the head of water line. Anaerobic digestion takes place in a CSTR, operating at 36°C. The average HRT is between 16 and 27 days and the average OLR is 2.5 kg TVS/m^3·d. The digester has a total volume of 3,916 m^3, filled up to 80% of its capacity. The SGP for the mixed sludge is 0.35 m^3 biogas/kg TVS$_{fed}$, with an average methane content of 60%. Biogas is burnt in a CHP unit (Heimersson et al., 2017). The CHP unit has electrical and thermal efficiency of 35% and 55%, respectively. Heat and electricity are recirculated in different processes of the plant (Heimersson et al., 2016). It is assumed that nutrients are not lost during anaerobic digestion; therefore, the total nutrient content of the digestate and waste (in terms of nitrogen and phosphorus) is the same (Møller et al., 2009). The sludge needs proper sanitation before its application to arable land to fulfil legislative requirements (Heimersson et al., 2017). In this study, digestate is pasteurised at 70°C for 1 h (European Commission, 2001) and it is dewatered in a centrifuge. The liquid fraction is recirculated back to the water line, while the solid fraction is stored to be added to land as organic fertiliser (Holm-Nielsen et al., 2009).

Integration of management schemes in the water treatment facilities

SSFW comes from separated collection assuming that the quality of the substrate is such that there is almost no need to perform pre-treatment (Zhang et al., 2013). Then it is transferred to a primary shredder where the particle size is reduced to 12 mm. In addition, an SGP of 0.60 m^3 biogas/kg TVS_{fed} for SSFW is considered (Banks et al., 2011; Evangelisti et al., 2014). As previously mentioned, the use of SSFW as co-substrate affects the characteristics of anaerobic supernatant that is recirculated to the plant (water line) due to its high nutrient content.

7.2.2. Environmental assessment methodology

Goal and scope definition

The objective of this LCA study is to quantify the environmental implications of each examined scheme for the management of urban organic waste. The function of the system is to valorise the sewage sludge produced by a community of 150,000 inhabitants. Thus, the FU is 1 tonne of sewage sludge valorised through the anaerobic digestion process.

The processes within the system boundaries are shown in Figure 7.2. The production of food waste and sewage was excluded from the analysis, since it does not affect their valorisation potential. The water line was also included in the system boundaries, since the recirculation of the dewatered liquid has an impact on the whole facility.

Additionally, the surplus electricity produced from biogas substitutes the equivalent amount of electricity from the British electric mix; thus, resulting in environmental credits (Heimersson et al., 2016). Similarly, the digestate that is used as an organic fertiliser in agricultural land can substitute mineral fertilisation considering its equivalent nutrient value (Brockmann et al., 2014; Heimersson et al., 2016).

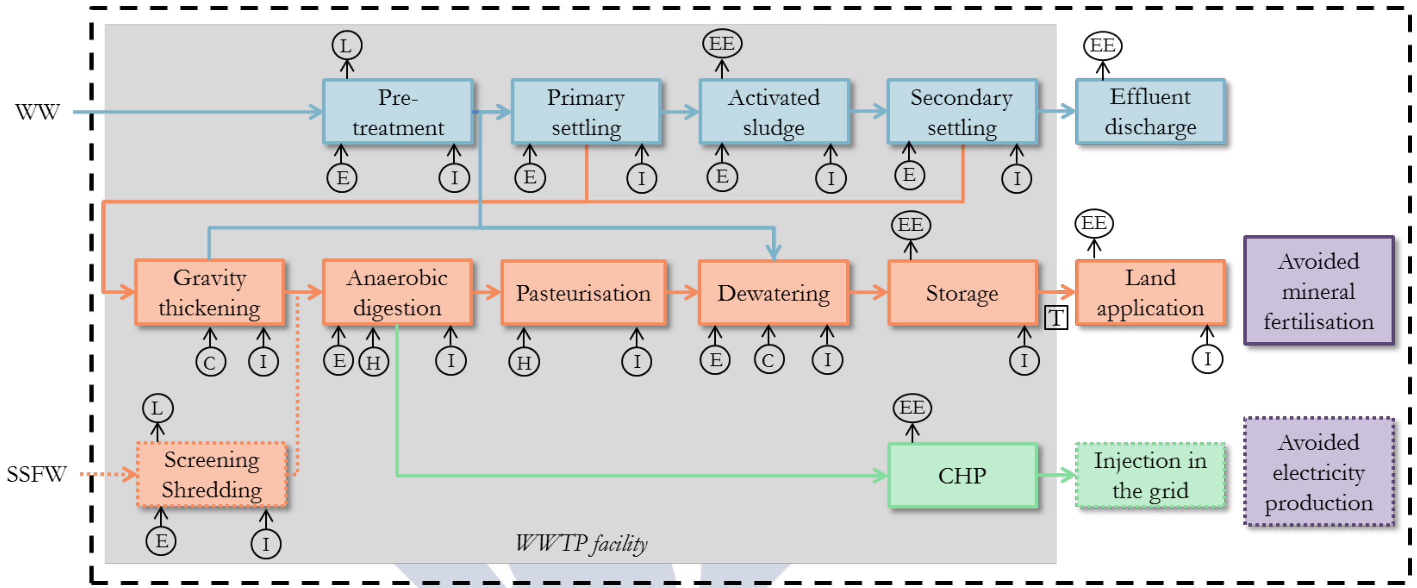

Figure 7.2. Unit processes included in the system boundaries. Dotted boxes indicate processes not performed in all schemes. Acronyms: WW – wastewater; SSFW – source segregated food waste; T – transport; E – electricity; H – heat; I – infrastructure; C – chemicals; EE – emissions; L – waste disposal

Life cycle inventory

Data from full-scale plants and mass balance calculations were considered for each management scheme, which allowed to simulate the entire treatment train (Garrido et al., 2013; Mininni et al., 2015). The WWTP treatment capacity is 0.26 m^3/PE·d, resulting in a total flow of 25,000 m^3/d. Typical wastewater characteristics were used, including COD of 825 mg/L, TN of 31 mg/L and TP of 7.6 mg/L. Primary and secondary sludge production was 55 g TS/PE·d and 25 g TS/PE·d, respectively (Tchobanoglous et al., 2014); while the organic content in terms of TVS was 70% TS and 75% TS, respectively. The mixture of primary and secondary sludge resulted in 524 t/PE·d. Concerning food waste, the production rate in the United Kingdom is 260 kg/household year, considering an average household size of 2.4 people (Quested et al., 2013). However, it has been considered that 20% of the food waste generated is lost during the source segregation process; therefore, a total amount of 35.6 t SSFW/d is potentially available for anaerobic digestion. The composition of the SSFW was accounted according to Zhang et al. (2013); 26% TS, out of which 92% are TVS, 30.8 kg TN/t TS and 4.5 kg TP/t TS.

Inventory data regarding all inputs and outputs for the different schemes based on available reported data, specifically for the United Kingdom when available, to handle a representative situation. In more detail, municipal solid waste, grit and grease removed in the pre-treatment and sent to landfill were estimated according to Lorenzo-Toja et al. (2016). Solid/liquid separation was modelled according to reported dewatered sludge characteristics (Tchobanoglous et al., 2014). The required polyelectrolytes to improve separation efficiency during thickening were 2.2 kg/t TS, while the polymer demand during dewatering was 10 kg/t TS (Mills et al., 2014).

Electricity and heat that is produced in co-generation from the biogas produced were calculated considering the electrical and thermal efficiencies of the engine (35% and 55%, respectively) as well as the methane composition of the biogas (60%), which resulted in a calorific value 5.5 kWh/m^3 biogas. In addition, a small percentage (around 1%) of the produced biogas is released to the atmosphere due to leakages in valves and pipes (Heimersson et al., 2017). The produced heat is used to maintain mesophilic conditions in the digester and 70°C at the pasteurisation tank. If the heat requirements of the plant are higher than the heat produced in the CHP, the combustion of natural gas can fulfil the heat requirements. The produced electricity is also used internally in the WWTP, while in cases where more electricity is produced than the required one in the plant, the surplus is fed to the National grid. However, electricity can be consumed from the grid for plant operation during CHP maintenance and brokerages. Electricity requirements in the WWTP were calculated according to reported data presented in Table 7.1. Heat consumption in the anaerobic digestion process and during pasteurisation was calculated according to Banks et al. (2011).

Table 7.1. Reported data for electricity consumption

Process	Electricity consumption	Source
Screens	0.0004 kWh/m^3	Tchobanoglous et al. (2014)
Grit removal	0.008 kWh/m^3	Tchobanoglous et al. (2014)
Biological reactor	11 kWh/kg $N_{removed}$	Tchobanoglous et al. (2014)
Secondary settler	0.0035 kWh/m^3	Tchobanoglous et al. (2014)
Thickening	60 kWh/t TS	Mills et al. (2014)
Anaerobic digestion	40 kWh/t TS	Mills et al. (2014)
Dewatering	0.0055 kWh/m^3	Tchobanoglous et al. (2014)
Pumping and mixing	0.08 kWh/m^3	Campos et al. (2016)

The transport distance considered for SSFW collection was 20 km, while the distance between the WWTP and the agricultural field where the final digested products is applied was 50 km (Heimersson et al., 2017). Lubricant oil consumption in the CHP as well as the infrastructure of the plants were computed according to the ecoinvent® database, using the scaling factor provided by Whiting and Azapagic (2014). Direct emissions from the biological reactor were considered as 0.01 kg N_2O-N/kg $N_{denitrified}$ (Foley and Lant, 2007) and 2.8 kg CH_4/t COD_{in} (Daelman et al., 2012). Biogas losses, nitrous oxide and ammonia emissions produced during the storage of sludge in the plant were estimated with the emission factors used by Heimersson et al. (2017). In the same way, emissions of ammonia, nitrous oxide and nitrate from the application of sludge on arable land were also taken into account according to Heimersson et al. (2017). The same emission factors regarding nitrous oxide and nitrate were considered for the application of mineral fertilisers, while ammonia emissions were calculated according to IPCC (2006). In addition, with the aim of quantifying the amount of nitrogen-based mineral fertilisers that are potentially replaced by the digestate, a replacement value of 50% was considered for nitrogen coming from sewage sludge (Heimersson et al., 2017) and 80% for nitrogen coming from food waste (Evangelisti et al., 2014). The primary life cycle data used in the study for each treatment scheme can be found in Table 7.2.

Table 7.2. LCI of the target technological solutions

	Scheme 1		**Scheme 2**		**Scheme 3**	
Inputs from Technosphere						
Materials and fuels						
WWTP	$1.4\cdot10^{-4}$	p	$1.4\cdot10^{-4}$	P	$1.4\cdot10^{-4}$	p
CHP	$1.7\cdot10^{-4}$	p	$1.9\cdot10^{-4}$	P	$4.6\cdot10^{-4}$	p
Polyelectrolyte	110	kg	106	kg	148	kg
Lubricant oil	1.00	kg	1.14	kg	2.77	kg
Tractor	1.97	kg	1.97	kg	2.28	kg
Agricultural implement	6.34	kg	6.33	kg	7.32	kg
Diesel	17.8	kg	17.8	kg	20.6	kg
Energy						
Electricity from the grid	3,208	kWh	2,798	kWh		
Heat from natural gas	9,825	kWh	8,685	kWh		
Transport						
Lorry	219	t·km	1895	t·km	2157	t·km
Outputs to Technosphere						
Products						
Electricity	4,975	kWh	5,629	kWh	13,701	kWh
Digestate	33.6	t	33.5	t	38.7	t
Wastes						
MSW to landfill	822	kg	822	kg	822	kg
Grit to landfill	220	kg	220	kg	220	kg
Grease to landfill	54.4	kg	54.4	kg	54.4	kg
Lubricant oil to landfill	1.00	kg	1.14	kg	2.77	kg
Avoided products						
Electricity from the grid					4,473	kWh
Nitrogen fertiliser	39.1	kg	43.2	kg	92.1	kg
Phosphorus fertiliser	272	kg	276	kg	321	kg

Table 7.2. LCI of the target technological solutions (cont.)

	Scheme 1		Scheme 2		Scheme 3	
Outputs to Nature						
Emissions to air						
From biological reactor						
Methane, biogenic	57.9	kg	57.9	kg	70.2	kg
Nitrous oxide	7.07	kg	7.33	kg	10.39	kg
Biogas losses						
Methane, biogenic	11.3	kg	19.3	kg	39.5	kg
Co-generation						
Methane, biogenic	0.77	kg	0.87	kg	2.12	kg
Nitrous oxide	0.08	kg	0.10	kg	0.23	kg
Nitrogen oxides	0.50	kg	0.57	kg	1.38	kg
NMVOC	0.07	kg	0.08	kg	0.18	kg
CO	1.61	kg	1.82	kg	4.43	kg
Digestate storage						
Methane, biogenic	2.75	kg	2.94	kg	5.11	kg
Ammonia	4.40	kg	4.07	kg	9.15	kg
Nitrous oxide	0.22	kg	0.24	kg	0.41	kg
Digestate application						
Ammonia	6.45	kg	7.42	kg	16.7	kg
Nitrogen oxides	1.23	kg	1.32	kg	2.27	kg
Avoided emissions to air						
Mineral fertilisers application						
Ammonia	4.75	kg	5.29	kg	11.2	kg
Nitrous oxide	0.62	kg	0.68	kg	1.45	kg
Emissions to water						
Effluent discharge						
Suspended solids	213	kg	213	kg	212	kg
COD	949	kg	949	kg	946	kg
TN	250	kg	250	kg	249	kg
TP	19.6	kg	19.6	kg	19.6	kg
Digestate application						
Nitrate	34.7	kg	36.9	kg	63.9	kg
Phosphate	4.14	kg	4.20	kg	4.89	kg
Avoided emissions to water						
Mineral fertilisers application						
Nitrate	17.3	kg	19.1	kg	40.8	kg
Phosphate	4.14	kg	4.20	kg	4.89	kg

The ecoinvent® database version 3.2 (Wernet et al., 2016) was also used for background data concerning generation of electricity, heat from natural gas and peat (Dones et al., 2007), production of chemicals (Althaus et al., 2007), transportation (Spielmann et al., 2007) and waste disposal (Doka, 2007). However, the United Kingdom electricity mix taken from the grid was modelled based on the ecoinvent® database but updated with data for the average electricity production and import/export data for the United Kingdom in 2016 (Department of Business Energy & Industrial Strategy, 2017).

7.3. Results

7.3.1. Performance of the technology solutions

A summary of the main technical parameters of the examined schemes is given in Table 7.3. Sewage sludge is a diluted substrate; since after thickening the TS content is 6% out of which 72% is TVS; the latter explains the low biogas potential (0.35 $m^3/kg\ TVS_{fed}$). On the contrary, SSFW is a more concentrated substrate with 25% of TS, out of which 92% are TVS, making it a suitable co-substrate (0.60 $m^3/kg\ TVS_{fed}$). The total volume of the digester in the existing WWTP is 3916 m^3 (Scheme 1), while only 80 % is used. In Scheme 2, the surplus capacity of the digester is filled with SSFW. Considering 10% spare capacity for safety reasons, 261 m^3 is used. Thus, 2.83 t/d of SSFW can be co-treated to maintain an OLR of 2.5 kg TVS/m3·d. In mass terms, SSFW accounted only for 1% of the mass added to the digester; however, it is supposed 7% of the TVS digested. The use of SSWF increased the SGP up to 0.37 $m^3/kg\ TVS_{fed}$ in Scheme 2 producing 392 m^3/d more biogas than in Scheme 1 (14%). The SGP of the mixture was calculated as the proportional ratio of the SGP of sewage sludge and SSFW according to the TVS provided by each substrate; therefore, no additional biogas production due to the co-digestion effect was considered (Fitamo et al., 2016; Kuglarz and Mrowiec, 2007).

In Scheme 3, the total amount of SSFW produced in the community was integrated into the sludge line; thus a second digester is applied. Considering the same OLR and 10% of spare capacity, an additional digestion volume of 3327 m^3 is required. In that case the feasibility of the integration of additional process within the existing WWTPs infrastructure (Bertanza et al., 2015) should be taken into consideration. However, the co-digestion of the total amount of food waste produced by the community entailed the additional digestion of 34.9 t/d of SSFW that is accompanied with an extra load of 8.21 t TVS/d. Thus, the mass share of SSFW is 15%, which is 50% in terms of TVS. This led to an increase of SGP up to 0.48, resulting in 2.7 times higher biogas yields than Scheme 1 (4929 m^3 more biogas each day). Finally, the increased in electricity and heat generation is proportional to the additional biogas production, with an increase to electricity generation to 679 and 8545 kWh/d in Schemes 2 and 3, respectively.

However, in schemes (Schemes 2 and 3) where co-digestion is applied the dewatered liquid that is recirculated from the centrifugation has higher nutrient levels than in the baseline scheme (Scheme 1). Higher energy is required to maintain the nitrification/denitrification process, while the level of nitrous oxide emissions is higher. In the base case (Scheme 1), the electricity consumption of the plant is 0.327 kWh/m^3 of treated wastewater. The electricity consumption increased up to 0.429 kWh/m^3 of wastewater treated (increase by 24%) when SSFW was co-treated in the plant. An extra load in the biological reactor is expected due to the additional load of TN coming from the recirculated dewatered liquid (29% higher in Scheme 3 compared to Scheme 1). In more detail, 16 and 206 kg TN/d was received in the biological reactor in Schemes 2 and 3, respectively. However, the electricity produced through biogas generation in Scheme 1 and 2, is not enough to cover the requirements of the whole plant; while Scheme 3 exhibits a surplus electricity production that can be injected into the national electric grid. Finally, the amount of digestate produced in Scheme 3 is by15% higher than in the baseline scheme. The digestate should be stored before its application to land for agricultural purposes.

Table 7.3. Summary of the performance parameters of each scheme

Parameter	Unit	Scheme 1	Scheme 2	Scheme 3	
				Digester I	Digester II
Treated wastewater	(m^3/d)	25,056	25,056	25,056	
Treated SSFW	(t/d)		2.83	35.6	
Digested sludge	(t/d)	200	200	200	
	(t TVS/d)	8.16	8.16	8.16	
Digested SSFW	(t/d)		2.77	34.9	
	(t TVS/d)		0.66	8.21	
Used volume	(m^3)	3,263	3,524	3,524	2,924
Total volume	(m^3)	3,916	3,916	3,916	3,217
SGP	(m^3/kg TVS_{fed})	0.35	0.37	0.47	0.47
Biogas production	(m^3/d)	2,855	3,247	4,189	3,595
Electricity production	(kWh/d)	4,950	5,629	13,495	
Heat production	(kWh/d)	8,508	9,675	23,194	
Digested substrate to land	(t/d)	33.6	33.6	38.6	

SS – sludge; SSFW – source-segregated food waste; SGP – specific gas production

7.3.2. Environmental impact of the technological solutions

The characterisation results corresponding to the FU chosen, which is 1 t of sewage sludge valorised through the anaerobic digestion process, for the three schemes under study are presented in Table 7.4.

Table 7.4. Characterisation results corresponding to Schemes 1, 2 and 3 per FU.

		Scheme 1	Scheme 2	Scheme 3
CC	(kg CO_2 eq)	18.6	18.5	15.0
OD	(kg CFC-11 eq)	$9.8 \cdot 10^{-7}$	$8.9 \cdot 10^{-7}$	$4.8 \cdot 10^{-8}$
TA	(kg SO_2 eq)	0.05	0.05	0.07
FE	(kg P eq)	0.04	0.04	0.04
ME	(kg N eq)	0.49	0.49	0.50
POF	(kg NMVOC)	0.03	0.03	0.02
FD	(kg oil eq)	2.77	2.54	0.34

Significant differences are observed among the impacts of each scheme, as well as among the impact categories selected. The influence of different hazardous substances emissions and primary materials extractions varies depending on the impact categories.

Climate change

Schemes 1 and 2 have similar behaviour in terms of CC since the extra energy consumption and higher emissions in the biological reactor are counterbalanced by the additional biogas produced (Figure 7.3). Scheme 3 presented higher GHGs emissions but also higher avoided impacts due to replaced processes such as electricity production and mineral fertilisation. Various processes affect CC; however, the most important source of GHG emissions is the biological reactor, generating between 9.9 and 13.7 kg CO_2 eq/FU. The increased emissions produced in Scheme 3 were attributed to the higher nitrogen content in the recirculated dewatered liquid. Figure 7.3 shows that the environmental impacts related to heat production do not exist in Scheme 3. The heat produced in the CHP engine in Scheme 3 is enough to meet heat requirements of the plant. Similarly, electricity production has a negative impact in Schemes 1 and 2, while a positive effect is observed for Scheme 3. The surplus electricity in the latter case can be injected into the national grid, reducing the consumption of the electricity from other sources. The identified differences among schemes were attributed to higher electricity production when co-digestion is performed (Table 7.3). Finally, avoided mineral fertilisation reduces the environmental impacts from 2 - 3.5 kg CO_2 eq/FU. The environmental credits are higher in Scheme 3 due to the higher amount of digested substrate produced.

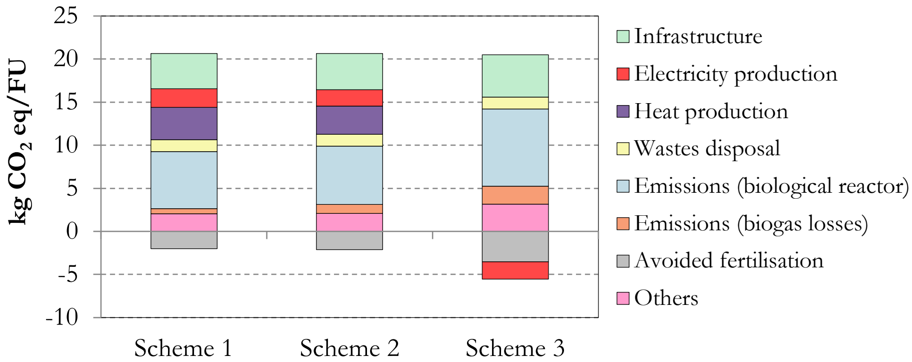

Figure 7.3. Relative contributions to CC for the three schemes

Ozone depletion, photochemical oxidant formation and fossil depletion

With regard to other energy-related categories such as OD, POF and FD, the processes exhibiting higher contributions are infrastructure, electricity and heat generation, digestate land application and derived avoided mineral fertilisation (Figure 7.4).

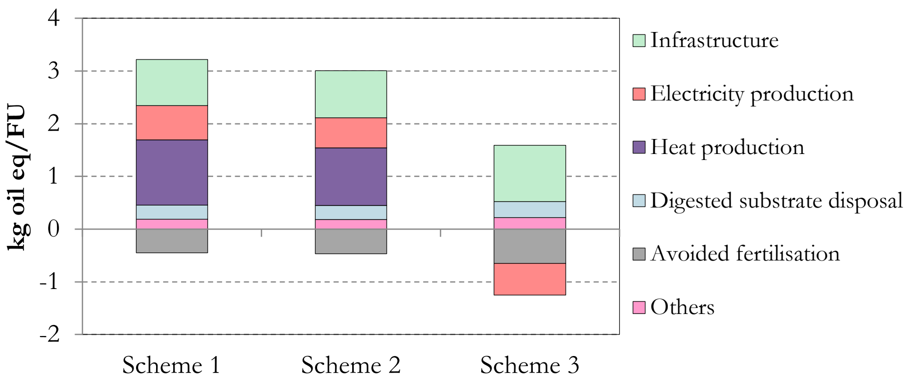

Figure 7.4. Relative contributions to FD for the three schemes

Infrastructure contributed by 21%-34% of the impacts in OD, with 39%-45% in POF and with 2%-37% in FD. Similarly to CC, electricity and heat production had a negative impact in Scheme 1 and 3 for the three examined impact categories; while in Scheme 3, electricity production has a positive impact and heat production has no impact. The reason is that Scheme 3 produces more electricity and heat than required in the plant; while electricity surplus is injected into the national grid, surplus heat is not further used. Electricity from the national grid and heat from natural gas accounted for 51% to 55% of the impacts in OD, 22% to 25% for POF and 48% to 52% for FD. The application of the cake (digestate) on land has moderate impact in these energy categories (lower than 15%) due to the machinery and diesel required for transport and spreading. Finally, the avoided production of mineral fertilisers positively affected these categories (between 12% and 24% of the impacts produced). The results are in line with the findings of Heimersson et al. (2017), who concluded that the substitution approach in LCA studies for the produced biogas as well as for the sludge used on arable land have an important effect in impact categories such as CC and POF.

Terrestrial acidification, freshwater and marine eutrophication

These impact categories are mainly influenced by emissions to air and water of nutrients associated to water and sludge treatment processes. However, the potential recovery of nutrients, which in this case is performed through the application of the digested substrate on land, can have important influence in these categories due to the avoided production and use of mineral fertilisers. With this regard, Heimersson et al. (2016) also emphasised the importance account for nitrogen and phosphorus flows in LCA studies, since they can be available as potential resources or leave as emissions. Environmental impacts of the eutrophication related categories (FE and ME) are due to the discharge of the final effluent from wastewater treatment, accounting for 78%-81% of the impacts in FE and 89%-93% of the impacts in ME. Other contributing processes include the application of the digestate on land as well as avoided mineral fertilisation. Concerning the TA related impacts, digestate management had the main contribution as revealed in Figure 7.5. Ammonia emissions from the treated digestate land application had the main share for this impact category. The emissions are directly related to the amount of organic fertiliser and its composition in terms of nitrogen. Schemes 1 and 2 produce less digestate (Table

7.3). Thus, the farming use of the produced digested sludge entailed 0.034 and 0.038 kg SO_2 eq/FU in Schemes 1 and 2, respectively, while in Scheme 3 the emissions increased to 0.082 kg SO_2 eq/FU. Proportional to the amount of substrate applied on land is the quantity of mineral fertilisers replaced by this system.

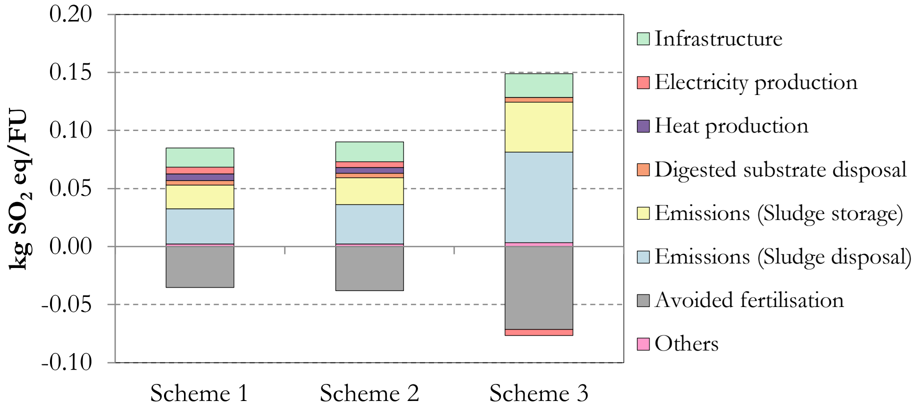

Figure 7.5. Relative contributions to TA for the three schemes

7.3.3. Sensitivity analysis

Several studies have evaluated the increased biogas yields from the addition of food waste to sewage sludge digestion units (Fitamo et al., 2016; Kuglarz and Mrowiec, 2007). The main advantages of co-digestion of sewage sludge with food waste include improved biogas generation, enhanced degradation efficiency and acceleration of hydrolysis of sewage sludge; thus, overall improved digestion performance, more efficient use of digestion equipment and an additional revenue for WWTPs (Kim et al., 2004; Kuo-Dahab et al., 2014).

Nielfa et al. (2015) compared the application of co-digestion and mono-digestion for the treatment of organic fraction of municipal solid waste (OFMSW) and secondary sludge. Co-digestion achieved by 14-18% more methane production than the co-digestion of both substrates separately. Dai et al. (2013) demonstrated 10 and 20% higher methane yields when sludge is co-digested with food waste compared to the respective yields from sludge mono-digestion. However, the ratio of food waste added in the digester within WWTPs impacts on the process efficiency resulting in different synergistic effects (Koch et al., 2015). Food waste used in ratios up to 13% were found to increase methane yield up to 7%; however, higher ratios of food waste decrease methane production up to 3%. The latter was also validated in full-scale application of the process by Koch et al. (2016). Increased biogas production, volatile solids reduction and digester stability was obtained by Kuo-Dahab et al. (2014) with the addition of food waste up to 50%TS in sewage sludge pilot digestion units. The average biogas production is up to 30% higher when 20% of food waste and 80% sewage sludge mixture is used as substrate in the examined process.

The current section analyses the effects of SSFW and sludge co-digestion on biogas generation yields when food waste is added at various ratios:

• **Co-digestion 0 –** As in the base case, mass balances were performed assuming that the biogas potential of the mixture is the same than the substrates separately (sewage sludge 0.35 $m^3/kg\ TVS_{fed}$; SSFW 0.60 $m^3/kg\ TVS_{fed}$).

• **Co-digestion 10** – It was assumed that synergies produced in the digester between the two substrates produced a positive effect on biogas yield, being 10% higher than in Co-digestion 0.

• **Co-digestion 20** – As in the previous case, it was assumed that the biogas yield increased by 20% in comparison with Co-digestion 0.

The assessment focuses on the system performance (efficiency) and the environment impact. The results obtained from the mass balances and the LCA are shown in Table 7.5.

Table 7.5. Comparative analysis of SSFW and sludge co-digestion effects on the biogas yield

		Co-digestion 0			Co-digestion 10		Co-digestion 20	
		Scheme 1	**Scheme 2**	**Scheme 3**	**Scheme 2**	**Scheme 3**	**Scheme 2**	**Scheme 3**
System performance – efficiency								
Mass to digester	(t/d)	200	203	236	203	236	203	236
SGP	(m^3/kg TVS_{fed})	0.35	0.37	0.477	0.41	0.52	0.44	0.572
Biogas production rate	(m^3/d)	2,855	3,247	7,885	3,572	8,673	3,896	9,462
Electricity production	(kWh/d)	4,975	5,629	13,701	6,192	15,072	6,755	16,442
Heat production	(kWh/d)	8,551	9,675	23,549	10,642	25,904	11,610	28,259
Digestate land application	t(/d)	33.6	33.5	38.7	33.4	38.6	33.4	38.5
Environmental performance								
CC	(kg CO_2 eq/FU)	18.6	18.5	15.0	17.9	14.4	17.3	13.8
OD	(kg CFC-11 eq/FU)	$9.8\cdot10^{-7}$	$8.9\cdot10^{-7}$	$4.8\cdot10^{-8}$	$8.1\cdot10^{-7}$	$-4.8\cdot10^{-10}$	$7.2\cdot10^{-7}$	$-1.4\cdot10^{-7}$
TA	(kg SO_2 eq/FU)	0.05	0.05	0.07	0.05	0.08	0.06	0.09
FE	(kg P eq/FU)	0.04	0.04	0.04	0.04	0.04	0.04	0.04
ME	(kg N eq/FU)	0.49	0.49	0.50	0.49	0.50	0.49	0.49
POF	(kg NMVOC/FU)	0.03	0.03	0.02	0.03	0.02	0.03	0.02
FD	(kg oil eq/FU)	2.77	2.54	0.34	2.31	0.08	2.08	-0.18

In Scheme 2, where the ratio of food waste in the digester is only 1.4% of the mass 2, the SGP was 0.369 and 0.387 m^3 biogas/kg TVS_{fed} for an increase of 5% and 10%, respectively. In Scheme 3 where the ratio of food waste is 15% of the mass (50% TVS in the digester), the SGP increased up to 0.501 and 0.524 m^3 biogas/kg TVS_{fed}. In both co-digestion schemes, biogas increased SGP is accompanied in increased biogas yield compared to the mono-digestion case (Table 7.5). The daily additional (compared to the base case) biogas production is 162 and 325 m^3/d for 5% and 10% food addition, respectively in Scheme 2; while in Scheme 3, 394 and 788 m^3/d more biogas is produced for 5% and 10% SSFW addition (Figure 7.6).

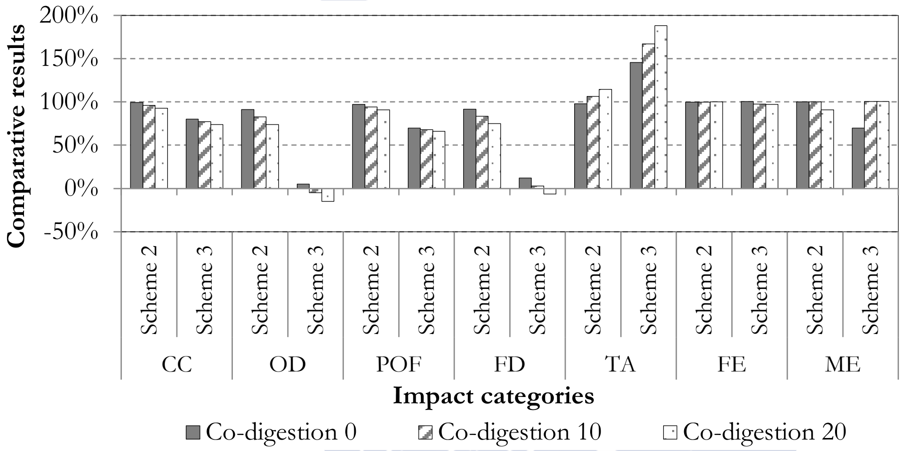

Figure 7.6. Comparative results of the sensitivity analysis

Increased biogas yield has a small impact on eutrophication categories (FE and ME). Moreover, CC effect is slightly improved when co-digestion is applied. Higher biogas yield has a positive impact; however, the emissions derived from the combustion of the biogas negatively affects. The benefits were more prevailed for FD, since environmental credits resulted from avoided electricity production increase the environmental performance for the specific impact category. Adverse environmental impact (in comparison with the base case) was obtained for TA category. Higher degradation of organic matter in the digester results in higher ammonium nitrogen levels in the digestate due to the degradation of organic nitrogen. Therefore, the derived ammonia emissions from the storage and application of the digestate are also higher.

7.4. Discussion

7.4.1. Regulatory context

The treatment of municipal wastewater results to the production of large quantities of sewage sludge (Kelessidis and Stasinakis, 2012). Sludge management remains an open and challenging issue at European level. From an economic point of view, sewage sludge management remains a key issue in WWTPs (Bertanza et al., 2015); sludge treatment and disposal accounts for an important amount of the total operating costs (Bertanza et al., 2015). Sludge treatment through anaerobic digestion followed by digestate spreading on arable land is the most commonly applied way to deal with sludge (Heimersson et al., 2017). This is in accordance with the Sludge Directive 86/278/EEC that encourages the recycle of sewage sludge in agricultural land, providing that it is not harmful to the soil, vegetation, animals or humans. Sewage sludge production in the United Kingdom was 1,137 thousand tons of TS in 2012 (Eurostat, 2016). Anaerobic digestion is the most common option for sewage sludge treatment (73%) and agricultural land recycling the most used way for sludge disposal (74%) (Eurostat, 2016). Moreover, approximately 60% of the biogas produced in the water industry is used in a CHP to co-produce heat and electricity, since it allows the water utilities access to Government incentives for renewable energy generation (Newton, 2008). The anaerobic digestion of sludge provides benefits from an economic point of view, as demonstrated by Bertanza et al. (2015). The authors studied the economic performance of WWTPs considering alternative sludge treatment options. They concluded that, for a facility performing the pasteurisation and anaerobic digestion of sewage sludge, the sludge disposal cost as well as the power feed-in income was the two items with the greatest impact on the final economic outcome of the plant. Due to electricity production and derived economic incentives, many countries of the European Union tend to increase the use of anaerobic digestion for waste management, despite the low commercial value of digestate, even partially replacing the composting practice (Mininni et al., 2014).

In this context and according to the results, the use of food waste in co-digestion with sewage sludge is a sustainable and cross-sectorial solution of urban waste management (Iacovidou et al., 2012). The addition of a co-substrate with high

SGP delivers economic benefits from the higher renewable energy generation and its associated incentives. Co-digestion may even result in increased bioenergy production due to synergies between both substrates. In addition, costs related to the infrastructure required for the exclusive digestion of food waste are also saved due to the use of the spare capacity of WWTPs. Both digestate and bioenergy from sewage sludge and SSFW are controlled under different regulations. Therefore, in order to move forward, to simplify regulations and make the quality criteria clearer is almost a prerequisite (Iacovidou et al., 2012).

The financial incentives for the anaerobic digestion of organic waste in the United Kingdom depend on the application scale and the use of the energy produced. In addition, there are differences in the incentives depending on the feedstock. For example, production of biogas by anaerobic digestion is eligible for ROC. Electricity from digested sludge qualifies for lower ROCs compared with energy from food waste, since the anaerobic digestion of sludge is considered as an established technology (Iacovidou et al., 2012). FiTs, introduced in 2010, provide a guaranteed payment for renewable electricity producers (<5 MWe). Smaller generators, with higher capital costs per MW, receive a higher price. However, the water industry does not receive support for electricity generated under this scheme. The situation changes regarding the Renewable Heat Incentive (RHI), introduced recently, providing a guaranteed payment for heat used from biogas combustion (<200 kWth) and all biomethane injected into the grid. This would improve the environmental performance of the energy mix of the United Kingdom, since around 42% of the energy was produced from natural gas in 2016 (Department for Business Energy & Industrial Strategy, 2017). However, the abovementioned financial support mechanisms are not clear when dealing with co-digestion. For example, as mentioned, the allocation of ROCs varies considerably depending on the source of biogas. In more detail, the energy generated from digested sewage sludge qualifies for 0.5 ROCs per MWh of electricity generated, whereas the energy from food waste qualifies for 2 ROCs per MWh, but how to make the allocation of ROCs when dealing with mixed feedstock is not specified (Iacovidou et al., 2012).

Regarding the management of the digestate, all plants in the United Kingdom must comply with regulations concerning environmental protection, animal by-products, duty of care, health and safety and waste handling. The Environmental Permitting scheme requires all biogas plants to obtain a permit or exemption to operate and to spread digestate, entailing different charges. When a feedstock is considered as a waste, the digestate deriving from it is classified as waste until they meet PAS 110 and Quality Protocol standards. The Quality Protocol for anaerobic digestate establishes the EoW criteria for digestate (Environment Agency, 2014). The EoW criteria establish the standards to guarantee that produced materials do not endanger human health and harm the environment when used as product (Environment Agency, 2014). Once these criteria are met, the material is no longer classified as waste and can be used without going through waste management monitoring. Source segregated biodegradable waste, such as SSFW falls into the input material for the production of digestate to that can be used outside the waste directive. However, in the United Kingdom, as in other countries (e.g. Sweden, Germany), sewage sludge is excluded from this list (Mininni et al., 2014). Therefore, digestate produced from sewage sludge is not covered by any approved quality protocol and its use in agricultural land is regulated under a different structure. The application of digestate from sewage sludge in agricultural land is currently controlled under the Sludge (Use in Agriculture) Regulations. The existing regulations do not define quality criteria for the proper management of digestate from sewage sludge and food waste. If sewage sludge is mixed with biodegradable waste, the farmers should obtain environmental permit to apply the digestate to land. This creates markets barriers for the adoption of the integrated management processes within the water utilities at larger scale. In addition, waste management license requirements create regulatory constraints and uncertainties that currently makes co-digestion unattractively complex and expensive, preventing the use of food waste within the water industry (Iacovidou et al., 2012).

Accordingly, the simplification of these regulatory schemes would also assist the United Kingdom to meet its targets for the reduction of landfilled waste, decrease GHGs related to waste management and increase renewable energy production. The identification and overcoming of regulatory, governance, financial and legal drivers and barriers for technology implementation and use of recovered resources remains a challenge. The results of the current work demonstrated environmental benefits and increased efficiency of the co-management of SSFW and sludge through anaerobic digestion in existing WWTPs. The latter provides evidence supporting adoption of the integrated approach and promoting the transition of water utilities towards circular economy. New marketing potential and financing incentives and strategies to maximise the multiple value of the recovered products and energy will increase attractiveness and facilitate the adoption of the target schemes.

7.5. Conclusions

This study demonstrated that, if carefully applied, co-digestion of sewage sludge and SSFW can deliver beneficial synergies for the water industry. In more detail, the study evaluated and compared different integration rates of the management of SSFW within the facilities of a WWTP serving 150,000 PE from an environmental point of view. According to the results obtained, the co-digestion of SSFW in an existing WWTP can multiply by a factor of 2.7 the electricity from biogas; while increasing the produced digestate by 15%. However, it is important to consider that it is necessary to integrate the equipment required for the pre-treatment of the SSFW as well as an additional digester for the co-digestion of the total amount of both wastes produced by the community under study, which may be not feasible in all cases. From an environmental perspective, the total integration of SSFW within the water facilities (Scheme 3) appeared as an excellent opportunity to achieve environmental benefits. These benefits were particularly important in energy-related categories, reducing the environmental impact compared with the base case where sewage sludge was digested alone (Scheme 1). Specifically, comparing Scheme 1 and 3, the environmental impact was reduced by 20% in CC, 95% in OD, 30% in POF and 88% in FD. FE and ME performed unaffected by these changes, mainly due to the high contribution of the discharge of the final effluent and the consideration of avoided mineral fertilisation, which helped to offset the impacts related to nutrients leachate. On the contrary, TA presented an increased in the impact produced (30% higher in Scheme 3 than in Scheme 1). It has been identified that the additional impacts in TA came from the additional load in nitrogen to the anaerobic process, which ended up in higher ammonia emissions during the storage and application of the digested substrate.

7.6. List of acronyms

CC	Climate change
CHP	Co-generation heat and power
COD	Chemical oxygen demand
CSTR	Continuous stirred tank reactor
EoW	End-of-waste
FD	Fossil depletion
FE	Freshwater eutrophication
FIT	Feed-in tariff
FU	Functional unit
GHG	Greenhouse gas
HRT	Hydraulic retention time
LCA	Life cycle assessment
ME	Marine eutrophication
OD	Ozone depletion
OLR	organic loading rate
PE	Population equivalent
POF	Photochemical oxidant formation
RHI	Renewable Heat Incentive
ROC	Renewable Obligation Certificate
SGP	Specific gas potential
SRT	Solids retention time
SSFW	Source segregated food waste
TA	Terrestrial acidification
TN	Total nitrogen
TP	Total phosphorus
TS	Total Solids
TVS	Total volatile solids
WWTP	Wastewater treatment plant

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Chapter 8: Decentralised treatment of domestic wastewater and organic waste

Summary

A technical and environmental evaluation of an innovative scheme for the co-treatment of domestic wastewater and domestic organic waste (DOW) was undertaken by coupling an up-flow anaerobic sludge blanket (UASB), a sequencing batch reactor (SBR) and a fermentation reactor. Alternative treatment configurations were evaluated with different waste collection practices as well as various schemes for nitrogen and phosphorus removal. All treatment systems fulfilled the required quality of the treated effluent in terms of chemical oxygen demand (COD) and total suspended solids (TSS) concentrations. However, only the configurations performing the short-cut nitrification/denitrification with biological phosphorus removal met the specifications for water reuse. A functional unit (FU) of 2,000 people receiving treatment services was considered. The most relevant sources of environmental impacts were associated to the concentration of dissolved methane in the UASB effluent that is emitted to the atmosphere in the SBR process, electricity consumption (mainly for aeration in the SBR), sludge management and the discharge of the treated effluent in receiving waters. The scheme of separate waste collection together with biological nitrogen removal and phosphorus uptake via nitrite was identified as the best configuration, with satisfactory treated effluent quality and environmental impacts lower than those of the other examined configurations.

Outline of Chapter 8

8.1. Introduction to decentralised waste treatment

Centralised wastewater treatment may be not feasible or the most cost-effective option for all sites. For instance, due to geographical conditions and dispersed settlements, more than 9,000 WWTPs in Italy are designed for 2,000 PE or lower (Libralato et al., 2012). The European legislation on urban wastewater treatment defines discharge limits for biochemical oxygen demand (BOD_5), COD and total suspended solids (TSS) for WWTPs serving PE higher than 2,000 PE; while for lower agglomerations, it only states that appropriate treatment must be implemented (EEC, 1991). When it comes to nutrient concentrations, limitations for TP and TN are only specified for treated effluents from facilities with a treatment capacity larger than 10,000 PE discharging into sensitive recipients. The option of reusing the treated water from small scale WWTPs in agriculture is interesting, provided that the treated effluent is available near the potential points of use; thus, decreasing the costs of reclaimed water distribution systems (Hophmayer-Tokich, 2000). Currently, there is no European Union legislation concerning the use of reclaimed water, being each country the responsible for the implementation of national or regional regulations (Norton-Brandão et al., 2013).

In order to guarantee sustainability, the applied treatment process should imply relatively low energy consumption and enhance the potential reuse of water and other valuable by-products, such as biogas. The application of anaerobic processes, i.e. up-flow anaerobic sludge blanket (UASB), appears as a robust and attractive technology (Latif et al., 2011). Compared to aerobic wastewater treatment, the UASB process has several advantages, such as low operating expenses, high efficiency, simplicity, flexibility, low requirements of space, energy and chemicals as well as biogas generation and reduced sludge production (Latif et al., 2011). However, there are still some barriers that limit the use of anaerobic processes, including the process instability at temperatures below 20°C, low pathogen removal, negligible nutrient removal, odours, long start-ups and the need for adequate post-treatment (Latif et al., 2011). In addition, it is important to consider the concentration of dissolved methane in the anaerobic effluent since low temperature raises methane solubility, reducing methane recovery for bioenergy production and promoting its release into the environment (Cookney et al., 2016; Matsuura et al., 2015). BNR from the low strength anaerobic effluent can be applied as a further polishing step (Frison et al., 2013b; Malamis et al., 2013). Biological nitrogen removal via nitrite has several benefits compared to conventional nitrification/denitrification such as 25% of oxygen savings during nitrification and 40% less need for organic carbon source during heterotrophic denitrification (Galí et al., 2007). Enhanced biological phosphorus removal (EBPR) can be also performed using nitrite as electron acceptors (Katsou et al., 2015). Denitrifying via nitrite biological phosphorus removal (DNBPR) offers the possibility of integrating phosphorus and nitrogen removal in a robust process. In the presence of nitrite and lack of oxygen, nitrite is denitrified to gaseous nitrogen and, simultaneously, phosphate is taken by denitrifying phosphorus accumulating organisms (DPAOs) (Peng et al., 2011). DPAOs are able to accumulate significant amounts of polyphosphate under anoxic conditions, similarly to the phosphorus accumulating organisms (PAOs) in the conventional EBPR process. Due to the substantial organic matter removal attained in the anaerobic treatment, the addition of an external carbon source is required in the subsequent aerobic process for effective BNR (Frison et al., 2013a). The latter opens up the possibility of integrating the management of domestic organic waste (DOW) with sewage. The use of organic waste (i.e. fermented liquids) from households as external carbon source achieves satisfactory rates of denitrification and phosphorus accumulation, while decreasing operational costs (Frison et al., 2013a). Food waste disposers (FWDs) are being promoted as an alternative practice for the collection of DOW (Iacovidou et al., 2012). Specifically, the implementation of FWDs entails reduced transport requirements and odours when compared to the conventional collection (Battistoni et al., 2007; Bernstad et al., 2013). However, the environmental assessment of FWD use is required, with specific focus on energy demand, water consumption and increased organic loads in the WWTP (Battistoni et al., 2007; Marashlian and El-Fadel, 2005).

The objective of Chapter 8 was to evaluate the feasibility of an integrated system designed for the decentralised co-management of wastewater and DOW in a small community of 2,000 PE. Various scenarios were evaluated including (i) alternatives in the collection of DOW regarding the integration rates of FWDs within the community, (ii) different nitrogen removal processes and (iii) the potential of including biological phosphorus removal in the treatment scheme.

8.2. Materials and methods

8.2.1. Base case UASB-SBR configuration

The selection of the treatment configuration was based on the results of a pilot scale UASB-SBR system operating at the premises of the University of Verona, taking into consideration cost criteria, legislative aspects for the treated effluent and DOW management in Italy, as well as the characteristics of the small community in terms of waste collection and sewage management. The treatment scheme proposed included: i) an UASB reactor to treat sewage and to produce biogas, ii) a fermentation process to produce VFAs from DOW, iii) a SBR to remove nutrients from the UASB effluent and iv) a composting process to treat the excess sludge and convert it to compost to be applied as soil conditioner. Nutrient removal in small scale WWTPs is not required by European Legislation; however, depending on the relevant National and or Regional Law of countries, compliance to specific nitrogen and phosphorus limits of the treated effluent before discharge to specific water recipients may be required.

It is important to assess the proposed innovations before these reach the demonstration or full scale to anticipate possible problems and related solutions (Mininni et al., 2015). To this aim, mass and energy balances can be used for evaluating the potential effects of the proposed solution in comparison. Therefore, mass balances were developed for the whole treatment scheme regarding TS, TVS, COD and nutrients (TN and TP) to model the different streams of the treatment system. The flowchart of the baseline treatment scheme is shown in Figure 8.1.

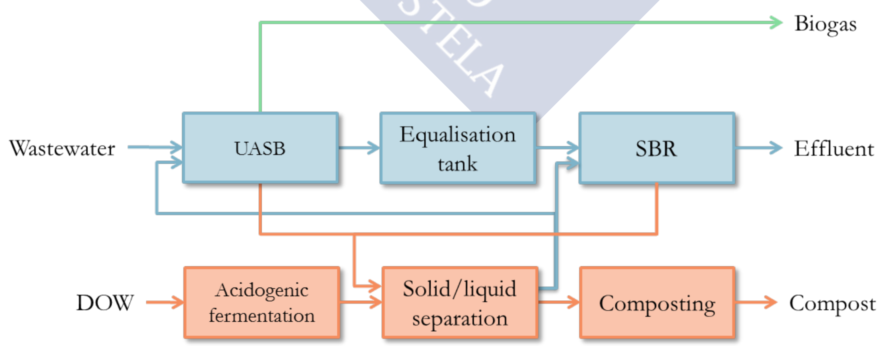

Figure 8.1. Integrated treatment system for the management of wastewater and DOW

• **UASB unit** – The average sewage flow was 400 m^3/d, assuming a wastewater production rate of 200 L/capita·d for a community population of 2,000 people. The production rates of COD, TN and TP were taken as 120 g COD/capita·d, 12 g TN/capita·d and 1.8 g TP/capita·d, respectively. The UASB reactor was operated at ambient temperature (22±2°C), at an average OLR of 1.4-2.1 kg COD/m^3 reactor·d, a HRT of 8 h and an up-flow velocity of 1 m/s. According to the experimental results, the UASB produced 7.2-13.2 L/d of biogas with a methane content ranging between 60-65%, which corresponded to an average experimental methane yield of 0.26 m^3 CH_4/kg $COD_{removed}$. This value is lower compared to the theoretical value of methane yield of 0.35 m^3 CH_4/kg $COD_{removed}$ since it does not include the dissolved methane present in the UASB effluent which is not recovered. For calculation purposes, global removal efficiencies of 77% and 70% were considered for COD and TSS respectively, assuming that 1 kg of COD degraded produced 0.26 m^3 methane. The dissolved methane derived from the operation of the UASB at moderate temperature was also taken into account in terms of its environmental impact, by considering concentrations of 20 mg CH_4/L under supersaturation conditions (Souza et al., 2011). The biogas produced in the UASB was treated in a biotrickling filter to remove hydrogen sulphide. The biogas was burnt in a boiler for heat production as considered appropriate for small and decentralised systems. The boiler had a thermal efficiency of 90% and 10% of losses.

• **Fermentation unit** – Considering that the UASB effluent had a very low COD/N ratio (2.5 kg COD/kg N), fermentation of DOW was applied to produce VFAs, which would then be fed to the SBR to promote the removal of nutrients contained in the UASB effluent. Furthermore, the surplus of fermented DOW was sent to the UASB to increase the OLR and, therefore, biogas production. A production rate of 0.30 kg DOW/capita·d was considered (Bolzonella et al., 2003). Assuming a collection efficiency of 83%, 500 kg of DOW are separately collected at household level and transported to the treatment facility on a daily basis for a community of 2,000 inhabitants. Regarding its physicochemical composition, TS of DOW was 25%, including 1,200 mg COD/g TS, 25 mg N/g TS and 3 mg P/g TS. Moreover, its content in carbohydrates was 600 g/kg TVS, in proteins 200 g/kg TVS and in sugars 160 g/kg VS. In the baseline configuration, DOW was firstly ground to produce a homogeneous mixture and then diluted with water up to 6% TS. The fermenter was fed at an average OLR of 11 kg COD/m^3·d and operated at 35°C and at a HRT of 5.2 d (Katsou et al., 2015; Lee et al., 2014; Traverso et al., 2000). During the fermentation process, organic matter is converted to acids (e.g. acetic acid, butyric acid, lactic acid etc.) while CO_2 is also released as a result of metabolic processes. Furthermore, hydrolysis of organic nitrogen and subsequent ammonification takes place, pH increases and some ammonia is released into the atmosphere. It was assumed that 8.5% of COD was converted into carbon dioxide and methane and that losses of TN (2%) as ammonia and TS (2%) also take place (Battistoni et al., 2002).

• **Dewatering unit** – The fermented DOW and the excess sludge from the UASB and SBR were separated into a liquid and a solid fraction by applying a screw-press. Therefore, a liquid fraction rich in VFAs was produced to be used in the SBR and a solid fraction to be further treated in the composting unit. The separation efficiencies of fermented DOW and sludge are different. More specifically, in the case of fermented DOW 65% of TS, 40% of COD and TN and 50% of TP were transferred to the solid stream and the remaining in the liquid fraction; when sludge was separated, 95% of TS, COD, TN and TP ended up in the solid fraction (Albertson et al., 1991; Battistoni et al., 2002). The produced solid fraction was sent to the composting unit, while the liquid fraction was stored in an equalisation tank with a HRT of 10 h before being fed to the SBR.

• **SBR unit** – The SBR was applied as a post-treatment stage for the UASB effluent and for the liquid stream generated from the screw-press. The SBR cycle comprised of filling, the sequential operation under anaerobic, aerobic and anoxic conditions, settling and decanting. The system operated at low dissolved oxygen level (around 1 mg/L) to perform short-cut nitrification/denitrification (scND) instead of the conventional nitrification/denitrification (ND) process. Previous work has shown that the combination of a suitable volumetric nitrogen loading rate (vNLR) and low DO can result in effective via nitrite nutrient removal from domestic sewage (Katsou et al., 2015). The calculation of the oxygen demand was based on the organic carbon and ammonia load.

• **Composting unit** – Sludge composting took place in an enclosed system equipped with a biofilter (Colón et al., 2009). Wheat straw was used as a bulking agent and was mixed with sludge to improve aeration, to provide a C/N ratio in the range of 25:1-35:1 and adjust the moisture content of the mixture in the range of 60-65% (Hernandez et al., 2006; Tremier et al., 2005). The addition of the bulking agent also prevented the compost mixture from excessive compacting.

The straw had the following characteristics: 90% of TS out of which 90% were TVS, 60% of total carbon (TC), 0.9% of TN and 0.1% of TP (Rihani et al., 2010).

8.2.2. Alternative approaches for the base case

Alternative approaches were examined to identify the best treatment configuration from a technical and environmental point of view. More specifically, three options were analysed considering different integration levels of FWDs in the community, diverse nitrogen removal options in the SBR and the possibility of including phosphorus removal.

• **DOW collection** – The collection of DOW in the community was considered with various FWDs integration rates. These disposal units are equipped with a shredding system, allowing effective collection of DOW, which is pumped together with wastewater to the treatment plant. Three different situations were considered, including i) Configuration 1, which involved the separate collection of wastewater and DOW (0% FWDs integration) by pumping wastewater from households to the WWTP and delivering DOW by trucks to the WWTP after separately collected at households; ii) Configuration 2, which included the integration of FWD in 50% of the households in the community (Evans et al., 2010), being the remaining DOW transported by trucks to the treatment plant; and iii) Configuration 3, which considered complete integration of FWDs (100%) in the community. The introduction of FWDs has been reported to cause an additional load of 60 g TSS/capita·d, 95 g COD/capita·d, 2.1 g TN/capita·d and 0.3 g TP/capita·d in the influent wastewater (Bernstad et al., 2013; De Koning, 2003). The application of FWDs leads to an increase in tap water consumption (up to 4.5 L/capita·d) required for pumping wastewater and DOW to the WWTP (Bernstad et al., 2013; Rosenwinkel and Wendler, 2001). A primary settler was implemented before the UASB to receive the mixture of wastewater and shredded DOW for the effective settling of the primary sludge. The removal efficiencies of COD, TSS, TP and TN in the primary settler were assumed to be 30%, 50%, 10% and 5%, respectively (Tchobanoglous et al., 2014). The produced sludge was fed to the fermentation unit to produce VFAs, while the supernatant was fed into the UASB.

• **Alternative processes for nitrogen removal** – Biological nitrogen removal was integrated in the scheme by applying conventional nitrification/denitrification (ND) and short-cut nitrification/ denitrification (scND) in the SBR.

• **Phosphorus removal** – Biological phosphorus removal using oxygen and nitrate or nitrite as electron acceptors in the SBR was also evaluated. Nitrogen and phosphorus removal accomplished under anoxic conditions require lower amounts of external carbon source and energy compared to aerobic conditions (Malamis et al., 2013).

The different 12 configurations under study are summarised in Table 8.1.

Table 8.1. Summary of the examined schemes for DOW and wastewater co-treatment

Treatment scheme	Waste collection	Nitrogen removal	Phosphorus removal
Conf 1-scND	Conventional 100%	Shortcut nitrification/ denitrification	No
Conf 1-scND-P	Conventional 100%	Shortcut nitrification/ denitrification	Yes
Conf 1-ND	Conventional 100%	Conventional nitrification/denitrification	No
Conf 1-ND-P	Conventional 100%	Conventional nitrification/denitrification	Yes
Conf 2-scND	Conventional 50% FWDs 50%	Shortcut nitrification/denitrification	No
Conf 2-scND-P	Conventional 50% FWDs 50%	Shortcut nitrification/denitrification	Yes
Conf 2-ND	Conventional 50% FWDs 50%	Conventional nitrification/denitrification	No
Conf 2-ND-P	Conventional 50% FWDs 50%	Conventional nitrification/denitrification	Yes
Conf 3-scND	FWDs 100%	Shortcut nitrification/denitrification	No
Conf 3-scND-P	FWDs 100%	Shortcut nitrification/denitrification	Yes
Conf 3-ND	FWDs 100%	Conventional nitrification/denitrification	No
Conf 3-ND-P	FWDs 100%	Conventional nitrification/denitrification	Yes

8.2.3. Environmental analysis

The objective of the LCA study was to quantify the environmental impact of each configuration to identify the most favourable one from an environmental point of view. The FU selected was the treatment of the wastewater and DOW produced by a community of 2,000 PE per day.

System boundaries

The processes considered within the system boundaries of the tested configurations are outlined in Figure 8.2. The generation of waste streams (wastewater and DOW) was excluded from the environmental analysis, since the way how they are treated/valorised does not affect earlier stages.

The sewer network has an important contribution to the total environmental impact of wastewater management (Doka, 2007). However, in this work, the sewer system was excluded since it was considered to be similar regardless the specific treatment scheme proposed.

In LCA studies, when waste treatment systems are converted into alternatives for resource recovery, they are usually credited by considering the avoided environmental impacts of producing a different product with the same function (Finnveden et al., 2005). In this manner, the environmental benefits of the production of valuable products can be quantified. The produced heat from biogas was partially used to heat the fermentation reactor, while the surplus heat can be exploited for heating nearby households 8 months per year. Fuel oil was assumed as the fuel used for accounting the environmental credits, since it was considered the most appropriate for a small and decentralised community. Moreover, numerous studies have demonstrated the horticultural properties of compost, being able to substitute peat in the production of ornamental plants (Ceglie et al., 2015; Russo et al., 2011), although its fertiliser capacity is usually considered lower than that of other organic substrates such as manure or digestate (De Vries et al., 2012a). Therefore, it was assumed that the produced compost can be used as soil conditioner avoiding the extraction, transport and use of a similar quantity of peat (Boldrin et al., 2009; Saer et al., 2013).

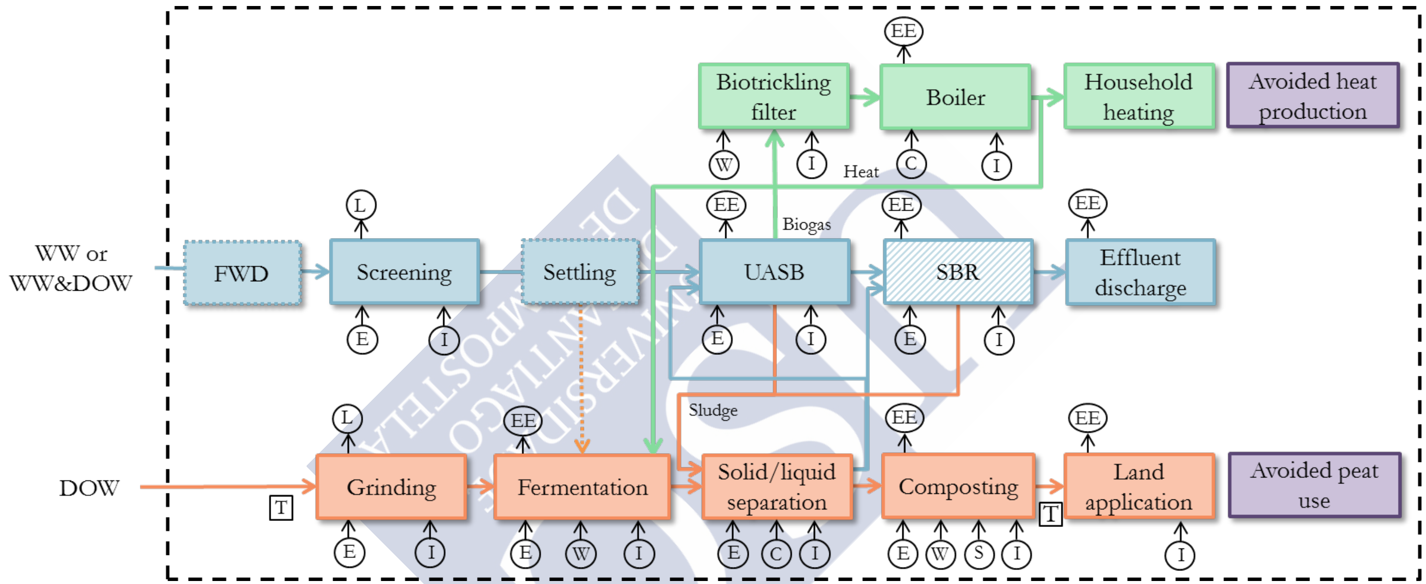

Figure 8.2. Unit processes included in the system boundaries. Dotted boxes indicate processes not performed in all configurations and striped boxes indicate that the process have modification among configurations. Acronyms: WW – wastewater; DOW – domestic organic waste; FWD – food waste disposer; T – transport; E – electricity; I – infrastructure; W – water; C – chemicals; S – Straw; EE – emissions; L – disposal of waste

Inventory data

Inventory data regarding all inputs and outputs for each configuration were based on experimental results from the UASB-SBR pilot plant and mass balances. A description of the bibliographic sources used to build the life cycle inventory is provided in Table 8.2.

The ecoinvent® database version 3 (Wernet et al., 2016) was used to introduce background data regarding the production of electricity, heat from fuel oil and peat, manufacture of chemicals, transportation and waste disposal. Concerning the production of electricity, the process included in the database has been updated using data for the average electricity generation and import/export data for Italy in 2014 (Terna Rete Italia, 2015).

Table 8.2. Life cycle inventory sources

Inputs
Energy consumption
• Wastewater pumping: 0.0385 kWh/m^3 wastewater (Tchobanoglous et al., 2014)
• Sludge pumping: 0.0008 kWh/m^3 sludge (Tchobanoglous et al., 2014)
• Screening: 0.0004 kWh/m^3 wastewater (Tchobanoglous et al., 2014)
• Mixing: 0.8424 kWh/m^3 reactor (Tchobanoglous et al., 2014)
• Sludge dewatering (screw press): 0.0009 kWh/m^3 wastewater (Tchobanoglous et al., 2014)
• Settling: 0.00095 kWh/m^3 wastewater (Tchobanoglous et al., 2014)
• SBR aeration: 0.320 kWh/m^3 wastewater (scND), 0.448 kWh/m^3 wastewater (ND)
• Fermentation heating: 14 kWh/m^3 fed (Energy balance)
• DOW grinding: 0.00051 kWh/kg (Zeeman et al., 2008)
• FWDs use: 0.51 kWh/m^3 DOW (Evans et al., 2010)
• Composting: 9 kWh/t sludge (Fisher, 2006)
Plastic bags in DOW collection: 7.2 kg/t DOW (Blengini, 2008)
Tap water in FWDs: 4.5 L/capita·d (Rosenwinkel and Wendler, 2001)
Polyelectrolyte (screw press): 20 g/t TS in (Tchobanoglous et al., 2014)
WWTP infrastructure
• Concrete: 8.8·10-5 m^3/m^3 wastewater (calculated according to the reactor volume)
• Related construction materials (e.g. aluminium, polyethylene, water…) (Foley et al., 2010)

Table 8.2. Life cycle inventory sources (cont.)

Outputs
UASB (Biogas losses): 1.5% biogas produced (Poeschl et al., 2012)
Heater (boiler emissions) (Jungbluth et al., 2007)
Fermentation tank • Carbon dioxide: 90% COD removed • Methane: 10% COD removed • Ammonia: 100% N removed
SBR emissions (scND) (Frison et al., 2015) • Nitrous oxide: 0.24% N in • Ammonia: 0.0016% N in • Carbon dioxide: 53.2% COD in • Methane: 0.81% COD in + 100% of dissolved methane from the UASB
SBR emissions (ND) • Nitrous oxide: 3.1 g/kg N removed • Carbon dioxide:1 kg/ kg N removed
Composting emissions (Boldrin et al., 2009)
Compost application emissions (Bruun et al., 2006)

8.3. System performance

The SBR operated under a HRT of 10 days, a solids retention time (SRT) of 18 days, a vNLR of 0.15 kg N/m^3·d and a volumetric phosphorus loading rate (vPLR) of 0.022 kg P/m^3·d. These parameters were considered to be invariable among all the configuration schemes. Regarding DO concentration in the aerobic phase, the SBR performing BNR via nitrate operated at DO concentrations of 2 mg/L; whereas the DO level was kept close to (and even below) 1 mg/L in the process via nitrite. It was observed that under these conditions the ratio of NO_2-N/NO_X-N gradually increased and was steadily maintained above 99% during the operation of the SBR. In the ND configurations, sNUR was on average 2.02 g N/kg VSS·h, while in the scND configurations, the sNUR was on average 4.93 g N/kg VSS·h, as supported by experimental results. The lower needs of external carbon source in the BNR process via nitrite can maintain higher average sNUR in the reactor. When enhanced biological phosphorus removal was performed, the pathway schemes integrating processes via nitrite resulted in slightly higher specific phosphorus uptake rates (sPUR) compared to the processes via nitrate: 3.85 g P/kg VSS·h and 3.19 g P/kg VSS·h, respectively.

Table 8.3 shows the characteristics of the treated effluent as it is calculated for each scenario in terms of COD, TSS, TN and TP.

Table 8.3. Calculated characteristics of the treated effluent after SBR for each alternative scheme under assessment

Treatment scheme	**COD** (mg/L)	**TSS** (mg/L)	**TN** (mg/L)	**TP** (mg/L)
Configuration 1-scND	36.4	14.1	9.48	7.46
Configuration 1-scND-P	40.8	14.9	9.94	1.94
Configuration 1-ND	39.9	15.7	14.9	7.49
Configuration 1-ND-P	40.4	15.8	21.4	5.34
Configuration 2-scND	49.6	17.8	9.20	7.80
Configuration 2-scND-P	58.8	22.0	9.86	1.82
Configuration 2-ND	61.3	23.1	9.71	8.24
Configuration 2-ND-P	65.7	24.5	9.84	6.27
Configuration 3-scND	55.2	20.2	9.69	8.02
Configuration 3-scND-P	65.1	23.5	9.96	1.16
Configuration 3-ND	67.6	25.6	10.0	7.89
Configuration 3-ND-P	68.2	25.0	12.7	6.70

All the scenarios achieved a treated effluent with COD levels between 36.4 and 69.5 mg/L and TSS concentrations from 14.1-25.9 mg/L; therefore, meeting the EU limits of COD and TSS for discharge into water bodies. However, the quality of the treated effluent regarding nutrients was not appropriate for water reuse. Regardless of the waste collection strategy, only the systems which performed the BNR through the short-cut nitrification/denitrification together with biological phosphorus uptake via nitrite (scND-P configurations) were able to reduce the nutrients to the levels required by existing National standards in Europe. In any case, it should be noticed that to comply with the reuse criteria, tertiary treatment (coagulation and sand filtration) followed by appropriate disinfection is required. The configurations applying scND achieved 85-86% nitrogen removal, while the nitrogen removal efficiency was 67-85% for ND configurations. Phosphorus removal, when applied, was higher than 80% for BNR via nitrite and around 43-73% via nitrate. The relation between the carbon source supplied and the one required for the BNR process justifies these results, since in some configurations such as Configuration 1-ND, the carbon source required for the denitrification process is significantly higher than the one that is available by the system. In this case, high levels of external carbon source were required for BNR in the conventional treatment system. The COD consumed for denitritation ranged from 49.6-57.2 kg COD/day, while the COD required for conventional denitrification via nitrate varied from 61.2-99.8 kg COD/day. The latter was not enough to remove nitrogen and this is the reason why the nutrient concentrations of the treated effluent are higher in the conventional nitrification/denitrification processes.

As presented in Table 8.4, diverting fermented DOW liquid from the UASB to the SBR resulted in lower biogas production in the UASB. More specifically, the application of conventional ND allowed recirculation rates of fermented liquid to the UASB from 0%-11% of the amount of fermented liquid produced, while the respective recirculation rates were up to 45% for the scND scheme. As a result, when scND was performed, the average biogas production was usually higher. Regarding the food waste collection options, the use of FWDs (Configurations 2 and 3) increased the COD levels at the head of the plant. After primary settling, the settled sludge was fed to the fermentation unit to produce VFAs, while the clarified effluent was sent to the UASB. Part of the fermented liquid was sent to cover the BNR needs of the SBR and the remaining part to the UASB to increase biogas recovery. More specifically, 59% and 52% of the inlet COD was fed to the UASB in configurations 2 and 3, respectively. The treatment and disposal of sludge is an important issue in WWTPs (Wei et al., 2003). The most commonly applied methods for sludge disposal at EU level include landfills, land application and incineration. In the examined systems, sludge was valorised through composting. The compost properties must be in line with the quality assurance protocol. As seen in Table 8.4, sludge production was directly affected by the food waste collection system. The partial or total application of FWDs (Configurations 2 and 3) resulted in higher sludge production compared to the separate collection schemes (Configuration 1) (around 18-19% increase). The increase in sludge production is attributed to the operation of the primary settler required when FWDs are used. The implementation of the primary settler implies the separation of primary sludge that is further sent to the fermentation reactor. In addition, the amount of sludge produced was 2-4% lower in the scND configurations. Finally, the schemes with EBPR produced more sludge than the ones lacking EBPR (3-7% increase).

Table 8.4. Calculated production rates of methane, sludge and compost in the different treatment schemes

Treatment scheme	Biogas production (m^3/d)	Sludge production (kg/d)	Compost production (kg/d)
Configuration 1-scND	50.0	760	586
Configuration 1-scND-P	45.2	820	630
Configuration 1-ND	45.2	838	634
Configuration 1-ND-P	45.2	805	619
Configuration 2-scND	49.0	909	698
Configuration 2-scND-P	43.1	975	751
Configuration 2-ND	41.0	951	732
Configuration 2-ND-P	38.4	995	762
Configuration 3-scND	53.1	901	694
Configuration 3-scND-P	47.0	970	746
Configuration 3-ND	44.8	944	705
Configuration 3-ND-P	44.0	973	747

8.4. Environmental profile

8.4.1. Base case UASB-SBR configuration

Table 8.5 summarises the LCA characterisation results for the baseline configuration (Configuration 1-scND) per FU. According to the results, despite the differences in environmental results among the examined impact categories, it is important to highlight the general positive effect of avoided processes. Avoided heat production from fuel oil played an important role in offsetting the environmental impacts in energy related impact categories, while avoided peat use has a more modest contribution in reducing environmental impacts. Generally speaking, it can be noticed that CC and TA are influenced by an important number of processes, including direct emissions from the system and electricity production. It can be also observed the important impact of energy demanding processes such as electricity production and transport in impact categories such as OD, POF and FD. Finally, direct emissions from the system including the discharge of the effluent mainly affected FE and ME.

Table 8.5. Environmental profile of the UASB-SBR configuration per FU

	CC (kg CO_2 eq)	**OD** (kg CFC-11 eq)	**TA** (kg SO_2 eq)	**FE** (kg P eq)	**ME** (kg N eq)	**POF** (kg NMVOC)	**FD** (kg oil eq)
Total	468	$-5.6\cdot10^{-6}$	1.47	3.04	6.67	0.49	5.76
Electricity production	76.3	$9.7\cdot10^{-6}$	0.28	0.03	0.01	0.24	24.3
Chemicals production	13.0	$4.2\cdot10^{-7}$	0.04	$8.5\cdot10^{-4}$	$1.1\cdot10^{-3}$	0.04	8.33
Transport	17.9	$2.9\cdot10^{-6}$	0.09	$1.7\cdot10^{-3}$	$4.8\cdot10^{-3}$	0.16	5.88
Infrastructure	24.5	$1.8\cdot10^{-6}$	0.08	0.01	$3.8\cdot10^{-3}$	0.09	5.35
Biogas use	1.14	$3.6\cdot10^{-8}$	0.25	$5.3\cdot10^{-3}$	$1.8\cdot10^{-3}$	0.06	0.15
Waste disposal	58.1	$3.9\cdot10^{-7}$	0.01	$1.6\cdot10^{-3}$	0.33	0.04	0.73
Dissolved methane	179	0	0	0	0	0.08	0
Fermentation emissions	37.9	0	0.19	0	0.01	0.02	0
Biogas losses	11.0	0	0	0	0	0.01	0
SBR emissions	40	0	$1.0\cdot10^{-3}$	0	$3.8\cdot10^{-5}$	0.01	0
Effluent discharge	0	0	0	3.00	3.84	0	0
Composting emissions	77.3	0	0.48	0	0.02	0.01	0
Land application emissions	48.0	0	0.33	0.01	2.49	0	0
Avoided heat production	-101	$-1.8\cdot10^{-5}$	-0.22	-0.01	-0.01	-0.14	-34.7
Avoided peat production	-19.3	$-2.7\cdot10^{-6}$	-0.09	$-2.0\cdot10^{-3}$	-0.01	-0.15	-5.14
Others	2.81	$3.8\cdot10^{-7}$	0.02	$9.5\cdot10^{-4}$	$9.9\cdot10^{-4}$	0.09	1.00

In order to better understand the results obtained, the treatment scheme under assessment was evaluated excluding environmental credits. Therefore, each impact category was examined in detail considering only negative loads to identify the system components with greater environmental impacts.

Climate change

Figure 8.3 summarises the relative contributions of each process to CC for the baseline scenario.

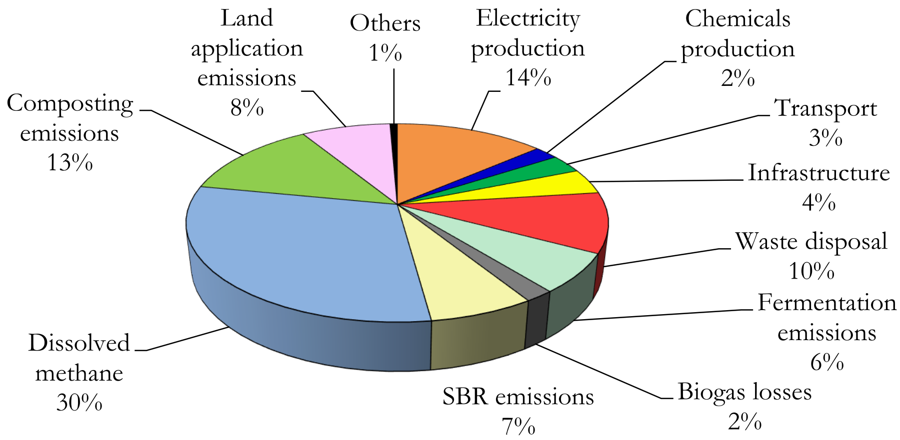

Figure 8.3. Relative contributions to CC of the processes involved

Regarding the environmental impact in CC (468 kg CO_2 eq/FU), the electricity requirements contributed up to 14% of the global impact in CC. From the total electricity consumed, aeration in SBR accounted for 80%. However, it is important to consider that the environmental impact of the consumption of electricity is directly linked to the electricity mix of the specific country under study. Dissolved methane in the anaerobic effluent entailed direct emissions which were identified as the main contributor (180 kg CO_2 eq/FU), representing 30% of the impact produced in CC. Emissions derived from the composting unit also contributed with 77 kg CO_2 eq/FU to the impacts in CC (~13%); these environmental impacts were related to direct emissions of methane and nitrous oxide that were generated during biomass decomposition. Despite the fact that composting is an aerobic process, methane emissions may occur, especially for enclosed systems, in anaerobic pockets of the mixture that is composted (Boldrin et al., 2009). Methane and nitrous oxide emissions derived from the application of compost on land contributed with 48 kg CO_2 eq/FU to the total impact produced in CC (representing 8% of the total impacts). In addition, direct emissions produced in the SBR unit contributed with 40 kg CO_2 eq/FU to the environmental profile (meaning around 7% of the impacts).

Ozone depletion, photochemical oxidant formation and fossil depletion

The relative contributions to the most important processes affecting other energy-related categories such as OD, POF and FD are presented in Figure 8.4.

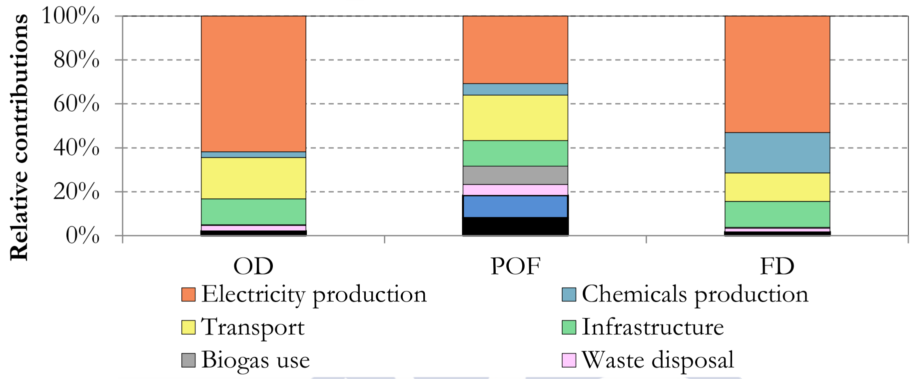

Figure 8.4. Relative contributions to OD, POF and FD of the processes involved

As it can be observed, mostly of the processes affecting these impacts categories are similar. Electricity consumption was identified as the main contributor, accounting for 62%, 31% and 53% of the impacts produced, respectively. Transport requirements among the system, including the transport of DOW from the households to the facilities, the transport of chemicals as well as the transport of compost to agricultural land, accounted for 13%-21% of the impacts. The production of the infrastructure also brought about 12% of the impacts. In addition, direct emissions produced from the dissolved methane contained in the anaerobic effluent produced 10% of the impacts in POF; while the production of chemicals were responsible for 18% of the impacts in FD.

Terrestrial acidification, freshwater and marine eutrophication

The relative contributions of the most important process to TA, FE and ME are outlined in Figure 8.5.

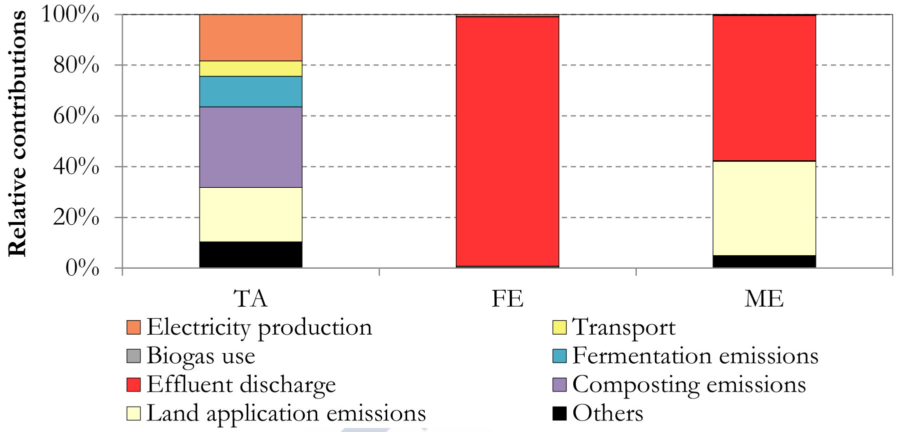

Figure 8.5. Relative contributions to TA, FE and ME of the processes involved

Regarding TA, direct emissions of ammonia produced during the fermentation and composting processes as well as during the application of the compost were the main contributors, producing 12%, 32% and 21% of the total impacts, respectively. In addition, the production of electricity also produced 18% of the burdens in this impact category. Furthermore, the discharge of treated effluent was the main source for eutrophication emissions, contributing up to 98% in FE and 57% in ME. Emissions of phosphorus from the treated effluent were responsible for FE, while nitrogen emissions from the treated effluent were related to ME. In addition, leachates of nitrate derived from the application of compost on land had an important contribution in ME (37%).

8.4.2. Alternative approaches

In order to increase environmental sustainability, several treatment configurations were analysed and compared in terms of their environmental profile to identify the most suitable scheme. Characterisation results for each configuration are given in Table 8.6.

Table 8.6. Characterisation results for wastewater and DOW treatment configurations per FU

	CC (kg CO_2 eq)	**OD** (kg CFC-11 eq)	**TA** (kg SO_2 eq)	**FE** (kg P eq)	**ME** (kg N eq)	**POF** (kg NMVOC)	**FD** (kg oil eq)
Conf 1- scND	468	$-4.9 \cdot 10^{-6}$	1.49	3.04	6.70	0.51	7.47
Conf 1-scND-P	500	$-3.0 \cdot 10^{-6}$	1.58	0.85	6.71	0.52	11.2
Conf 1- ND	519	$-5.2 \cdot 10^{-7}$	1.67	2.95	8.26	0.58	17.4
Conf 1-ND-P	518	$3.7 \cdot 10^{-8}$	1.65	2.21	11.3	0.59	18.9
Conf 2- scND	565	$9.4 \cdot 10^{-6}$	1.77	3.28	6.36	0.74	40.0
Conf 2-scND-P	565	$1.2 \cdot 10^{-5}$	1.91	0.85	6.84	0.75	44.2
Conf 2- ND	600	$1.5 \cdot 10^{-5}$	1.88	3.43	6.50	0.83	53.4
Conf 2-ND-P	609	$1.6 \cdot 10^{-5}$	2.02	1.43	9.06	0.83	55.2
Conf 3- scND	502	$5.1 \cdot 10^{-6}$	1.33	3.39	6.41	0.55	28.4
Conf 3-scND-P	547	$8.9 \cdot 10^{-6}$	1.71	0.58	6.75	0.66	36.7
Conf 3- ND	567	$1.1 \cdot 10^{-5}$	1.74	3.31	6.72	0.71	42.5
Conf 3-ND-P	556	$1.2 \cdot 10^{-5}$	1.73	1.20	9.72	0.68	43.5

Climate change

In terms of CC, the profile achieved by each configuration varied depending on the collection scheme, as shown in Figure 8.6.

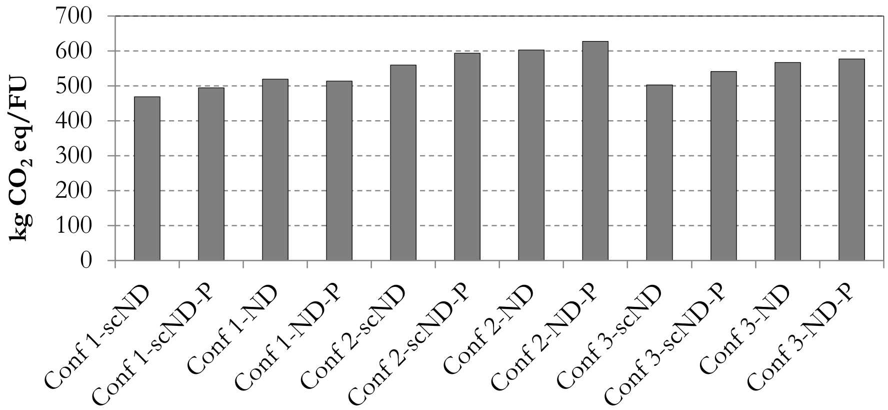

Figure 8.6. Environmental performance of the different configurations concerning CC

The partial use of FWDs (Configuration 2) resulted in an average increase in GHG emissions of 1.20 in comparison with the separate collection of DOW and wastewater (Configuration 1). Due to the implementation of a primary settler, the use of FWDs (Configuration 2 and 3) resulted in less methane generation and, subsequently, in lower environmental credits due to avoided heat production that highly affected this category (Table 8.4). Additionally, this collection system also produced more sludge in comparison with Configuration 1, increasing the environmental impacts from direct emissions of nitrous oxide and methane from composting and land application processes. Finally, the implementation of FWDs was associated with additional energy consumption compared to the separate collection of DOW. Furthermore, partial implementation of FWDs (Configuration 2) achieved the worst results since in households where FWDs are not installed, DOW needs to be collected by trucks and sent to the treatment facility. Therefore, this scheme is burdened by the technology/infrastructure for the FWDs (i.e. settler after sewage screening) and the separate DOW collection and transportation (i.e. waste collection bins and trucks).

Concerning the removal of nitrogen, conventional denitrification processes (Configuration ND) entailed different emissions and aeration requirements, resulting in different electricity consumption. In particular, energy consumed for air supply was 14% higher in the configurations performing nitrogen removal via nitrate (ND) than via nitrite (scND). As a consequence, ND configurations exhibited 11% more environmental impact on average regarding CC than scND configurations. Finally, due to higher demand of carbon source in the SBR, the removal of phosphorus slightly increased the environmental profile, by 3% on average.

Ozone depletion, photochemical oxidant formation and fossil depletion

Since the driving factors for the differences found among collection schemes were linked to energy, transport and infrastructure requirements, the environmental profiles regarding OD, POF and FD followed the same trend as CC. Therefore, the partial implementation of FWDs (Configuration 2) exhibits the worst environmental performance in terms of OD, TA and FD. As mentioned, this waste collection practice results in less heat production and thus, lower environmental credits, which highly affect OD, POF and FD. In addition, the use of FWDs produces more sludge and consequently, compost which has to be delivered and spread in agricultural land as a soil conditioner.

As presented in Figure 8.7, for the specific case of OD, Configurations 1 achieved environmental benefits due to the avoided heat production from fuel oil. As mentioned before, a detailed analysis was performed to analyse the influence of this type of methodological assumptions.

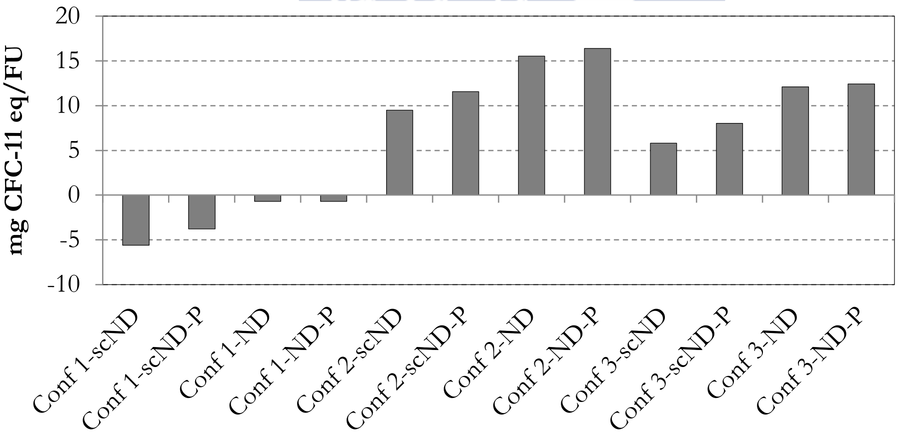

Figure 8.7. Environmental performance of the different configurations concerning OD

Terrestrial acidification, freshwater and marine eutrophication

In terms of TA, the differences among configurations are attributed to differences in the quantities of the produced sludge (Table 8.4), due to direct ammonia emissions resulting from the composting process and from the final use of the compost. Regarding impacts produced in eutrophication categories, the differences found among configurations can be explained by the quality of the treated effluent. As displayed in Figure 8.8, a reduction of 25%-83% in the FE category was observed in the systems that perform EBRP, since this impact category is only influenced by emissions of phosphorus compounds. Finally, regarding ME, the configurations that perform nitrogen removal via nitrate together with EBPR (Configurations ND-P) resulted in higher impacts in comparison with the remaining configurations. The reason is linked with the levels of nitrogen in the final effluent because the carbon source was not enough for complete denitrification. This adversely affected ME, which was 33-69% higher compared to the configuration performing denitrification via nitrite.

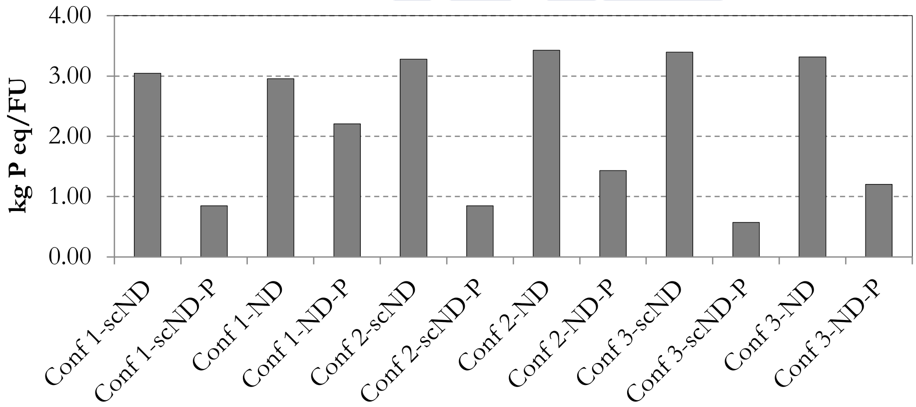

Figure 8.8. Environmental performance of the different configurations concerning FE

8.5. Sensitivity analysis

8.5.1. Design parameters

A sensitivity analysis was performed to analyse the influence of three selected key parameters in the performance of the suggested treatment scheme: (i) the COD removal efficiency in the UASB process, (ii) the efficiency in the separate collection of DOW at household level and (iii) the bulking agent used for composting.

COD removal efficiency in UASB

In the base case, the UASB removal efficiency in terms of COD was considered as 77% of the COD entering the reactor. In this sensitivity analysis, it has been considered that only around 50-55% of the COD input was removed.

As expected, the lower removal efficiency implied lower biogas and sludge production in the UASB. According to the results obtained, biogas and sludge production decreased by 29-35% compared to the base case and, therefore, derived heat production. In addition to higher sludge production, energy consumption for aeration in the SBR was also 13%-21% higher compared with the base case. Nevertheless, total sludge production in the treatment scheme increased between 13% and 27%, which was attributed to the sludge production in SBR, resulting in a larger production of compost (15-27%).

From an environmental perspective, as shown in Table 8.7, all these variations worsened the environmental profile of the treatment configurations. More in detail, the environmental impacts in CC increased between 6% and 19% mainly due to lower environmental credits from heat production, higher emissions of fossil carbon dioxide due to electricity consumption, and greater emissions of nitrous oxide from the production and application of the compost. In addition, higher impacts related to electricity consumption and lower environmental credits from heat production increased the environmental impacts regarding the remaining energy-related categories. In more detail, on average, OD increased $7.3 \cdot 10^{-6}$ kg CFC-11 eq/FU, POF decreased to 0.09 kg NMVOC/FU and FD raised to 16 kg oil eq/FU. With regard to other impacts, acidification also increased due to the higher direct of ammonia emissions linked with the production of sludge and, therefore, compost which increased the environmental impacts produced in TA by 9-27%. Nevertheless, the results in eutrophication categories (FE and ME) were similar to those obtained for the base case since, despite of lower COD removal efficiency, the treatment configurations were able to achieve similar quality of the final effluent, especially regarding nutrients concentration.

Table 8.7. Characterisation results for the sensitivity analysis on COD removal efficiency per FU

	CC (kg CO_2 eq)	**OD** (kg CFC-11 eq)	**TA** (kg SO_2 eq)	**FE** (kg P eq)	**ME** (kg N eq)	**POF** (kg NMVOC)	**FD** (kg oil eq)
Conf 1- scND	557	$1.4\cdot10^{-6}$	1.73	2.84	5.84	0.54	20.6
Conf 1-scND-P	532	$2.4\cdot10^{-6}$	1.78	1.07	5.95	0.53	22.4
Conf 1- ND	564	$5.8\cdot10^{-6}$	1.83	2.91	8.34	0.62	30.9
Conf 1-ND-P	565	$7.1\cdot10^{-6}$	1.87	2.12	11.0	0.63	33.4
Conf 2- scND	614	$1.6\cdot10^{-5}$	2.01	3.10	5.92	0.77	52.7
Conf 2-scND-P	634	$1.7\cdot10^{-5}$	2.07	0.51	7.43	0.76	54.9
Conf 2- ND	675	$2.2\cdot10^{-5}$	2.21	3.31	7.18	0.97	68.3
Conf 2-ND-P	667	$2.1\cdot10^{-5}$	2.14	2.24	7.39	0.84	64.9
Conf 3- scND	572	$1.4\cdot10^{-5}$	1.69	3.20	5.64	0.63	45.8
Conf 3-scND-P	609	$1.7\cdot10^{-5}$	2.02	0.53	7.20	0.80	53.5
Conf 3- ND	635	$2.0\cdot10^{-5}$	2.07	3.19	7.16	0.89	62.8
Conf 3-ND-P	638	$2.1\cdot10^{-5}$	2.09	2.98	7.21	0.88	63.5

DOW collection efficiency

In the base case, it was assumed that 83% of DOW produced at household level was separately collected and delivered to the treatment facility. In Configurations 1, this meant that out of the total 600 kg DOW produced each day, 500 kg were delivered to the treatment facility; while in Configurations 2 this meant that 250 kg DOW/d were delivered to the facility because the remaining 300 kg were collected through FWDs. Configurations 3 were not influenced since all DOW produced was collected in FWDs. However, the values of collection efficiency can vary from one community to another. Therefore, in this sensitivity analysis it was assumed that only 40% of DOW was successfully separated in the households and delivered by trucks to the treatment facility.

According to the results obtained, the lower collection efficiency resulted in lower amount of DOW sent to the fermentation reactor, implying not only less available carbon source and/or biogas production, but also greater amount of DOW sent to landfill. Under this assumption, although 100% of the fermented liquid is sent to the SBR in Configurations 1, it is not enough for efficient nutrient removal and the treated effluent is characterised by high nutrient concentration. Moreover, in Configurations 1, the total amount of sludge produced and compost are lower compared with the base case due to the lower amount of DOW handled. Conversely, the collection efficiency had a lower influence in Configurations 2, where the supply of DOW is guaranteed by the implementation of FWDs in 50% of the households. In these cases, the quality of the effluent achieved was similar to the base case due to the effective nutrient removal allowed by the proper supply of carbon source in the SBR.

The environmental results obtained are summarised in Table 8.8. Concerning CC, the environmental impacts in Configurations 1 were between 17 and 56 kg CO_2 eq/FU higher compared with base case, mainly due to higher organic waste send to landfill. The environmental impacts in OD, POF and FD showed a slight decrease due to low waste collection and energy consumption. Nevertheless, the environmental impacts in ME increased by 51% and 105%, since TN is not effectively removed. Regarding phosphorus, the environmental impacts in Conf 1-scND-P increased from 0.849 kg P eq/FU to 3.14 kg P eq/FU since the biological removal of TP is not performed.

Table 8.8. Characterisation results for the sensitivity analysis on DOW collection per FU

	CC	**OD**	**TA**	**FE**	**ME**	**POF**	**FD**
	(kg CO_2 eq)	(kg CFC-11 eq)	(kg SO_2 eq)	(kg P eq)	(kg N eq)	(kg NMVOC)	(kg oil eq)
Conf 1- scND	524	$-5.7·10^{-6}$	1.08	3.14	12.1	0.48	1.08
Conf 1-scND-P	524	$-5.7·10^{-6}$	1.08	3.14	12.1	0.48	1.08
Conf 1- ND	540	$-2.7·10^{-6}$	1.17	3.14	17.1	0.56	8.69
Conf 1-ND-P	537	$-2.7·10^{-6}$	1.17	3.14	17.1	0.56	8.69
Conf 2- scND	596	$4.7·10^{-6}$	1.54	3.32	6.82	0.68	28.3
Conf 2-scND-P	629	$7.1·10^{-6}$	1.65	0.75	7.18	0.69	33.1
Conf 2- ND	684	$1.2·10^{-5}$	1.92	3.46	8.22	0.86	44.5
Conf 2-ND-P	644	$1.0·10^{-5}$	1.68	3.45	8.21	0.76	40.4
Conf 3- scND	507	$5.8·10^{-6}$	1.35	3.39	6.48	0.56	30.0
Conf 3-scND-P	541	$8.1·10^{-6}$	1.68	0.573	6.75	0.64	34.6
Conf 3- ND	572	$1.2·10^{-5}$	1.77	3.31	6.73	0.72	44.2
Conf 3-ND-P	577	$1.2·10^{-5}$	1.80	1.21	9.84	0.72	44.9

Bulking agent used for composting

In the base case, wheat straw was used as bulking agent in the composting process. In the sensitivity analysis 3, wheat straw was substituted by sawdust.

The change in the bulking agent meant different compost mixture composition, resulting in different emissions from composting and from compost application. Wheat straw had a composition in terms of 10% moisture, TC, TN and TP of 60%, 0.9% and 0.1%, respectively as percent of dry solids; the composition of sawdust was 20% moisture, 60% TC, 0.2% TN and 0.03% TP. The lower content in nutrients resulted in i) lower amount of sawdust required to achieve the appropriate C/N ratio and ii) lower emissions of nutrient-based compounds derived from the composting process and the application of compost on land (including emissions of nitrous oxide and ammonia and leachates of nitrate and phosphate).

In terms of environmental impacts, this change meant a reduction in GHG emissions of 7% in average due to lower nitrous oxide emissions during composting, while in OD, POF and FD no differences were observed. Regarding TA and ME, the acidification impacts were reduced, on average, by 17% and 12% owing to lower ammonia emissions and nitrate leaching, respectively. However, no significant changes occurred in terms of FE, since almost all of the effects produced in this impact category (>98%) were allocated to the discharge of the treated effluent.

8.5.2. LCA assumptions

The influence of the selection of important parameters in the environmental balance was assessed. A comparison between the baseline case and alternative scenarios was performed to identify sensible variations in the results. The parameters considered in the sensitivity analysis were the fugitive biogas emissions rate, alternative avoided products as well as the reuse (or not) of the treated effluent.

Biogas losses

Fugitive biogas emissions from anaerobic processes are usually included in the environmental analysis. These emissions directly affect CC, not only due to direct methane emissions, but also by decreasing the potential heat production from biogas. In the baseline scenario of the current study, 1.5% of biogas produced was taken into account as biogas losses in accordance to De Vries et al. (2012b). Poeschl et al. (2012) considered that these losses can vary from 1 to 1.8%. Therefore, a sensitivity analysis was performed to assess the influence of different rates of biogas losses in the environmental profile (i.e. 1% and 1.8%).

As expected, the differences in biogas losses have no effect in impact categories such as TA, FE and ME. Regarding CC, the decrease of the emissions to 1% of the biogas produced can save from 4-5 kg CO_2 eq/FU, whereas when the biogas losses were 1.8%, the environmental profile can increase by 5-7 kg CO_2 eq/FU; in both cases representing less than 1% of the impacts. In the same way, the differences found in OD, POF and FD accounted for less than 1% of the impacts. The results proved the slight influence of this methodological assumption.

Avoided products

As described in Section 3.2., credits from the avoided products played an important role in offsetting the environmental impacts of the applied treatment configurations, especially regarding CC. Alternative avoided products were analysed to identify their impact. The baseline case, where the avoided heat was produced from fuel oil at small-scale in Europe, was compared with the substitution of heat produced from different fuels, such as natural gas and hard coal (Dones et al., 2007). The substitution of peat for compost is usually performed on a 1:1 volume basis (Boldrin et al., 2009). In this study, identical density was assumed for compost and peat. However, Boldrin et al. (2009) stated that compost and peat densities are very variable and can be different; it is possible that 1 tonne of compost can replace the use of 0.2-1 tonne of peat. Accordingly, an equivalence of 0.2, 0.6 and 0.8 t peat/t compost was considered.

The fuel used for the production of heat has a main role in the derived environmental impacts. For example, in terms of CC the production of heat from fuel oil generates 0.34 kg CO_2 eq/kWh (base case), while the environmental impact of heat production from natural gas and hard coal is 0.25 and 0.58 kg CO_2 eq/kWh, respectively. Therefore, if natural gas is considered as the substitute fuel, the environmental profile of the treatment configurations would increase around 13-24 kg CO_2 eq/FU, representing an increase of 2%-6% in the environmental impacts; whereas, when considering hard coal, it can be improved by 38-68 kg CO_2 eq/FU, meaning an environmental profile 6%-17% lower compared with the base case. Therefore, the substitution of heat from fuel oil to heat from hard coal could suppose a considerable change in the environmental profile of the treatment scheme. However, this performance has not been identified in all impact categories. For example, the environmental impacts derived from the production of heat from hard coal are 16 and 7 times higher (0.002 kg NMVOC/FU) compared with light fuel oil and natural gas ($1.2 \cdot 10^{-4}$ and $2.8 \cdot 10^{-4}$ kg NMVOC/FU, respectively). Finally, the lowest environmental impacts produced in FD were achieved by heat produced from natural gas (0.09 kg oil eq/FU); while the impacts produced from the production of heat from light fuel oil and hard coal were similar (~0.11 kg oil eq/FU).

Regarding avoided peat, the effect of different replacement ratios implies diverse environmental impacts among categories. Therefore, the lowest replacement ratio, that is 0.2 t peat/t compost, meant an increase of the environmental impacts by 27%, 22% and 22% in OD, POF and FD; while it only implied an increase of 3% and 5% in CC and TA. Finally, other impact categories such as FE and ME were not influenced by this assumption.

Treated effluent reuse

The quality of the treated effluent in the scND-P configurations met the specifications for water reuse provided that effective tertiary filtration and appropriate disinfection take place (Section 7.3.1). Therefore, the treated water can be reused for irrigation instead of being discharged in water bodies. This practice reduces the impact of direct discharge of nutrients; however, it entails other potential environmental burdens from the filtration and disinfection as well as the use of agricultural machinery and emissions derived from the treated effluent discharge on land. In the sensitivity analysis, it has been considered that the treated effluent is further treated in a sand filter using aluminium sulphate as coagulant, followed by UV disinfection, as described in Meneses et al. (2010). In addition, derived emissions were computed using the methodology described in IPCC (2006). The obtained results are depicted in Figure 8.9.

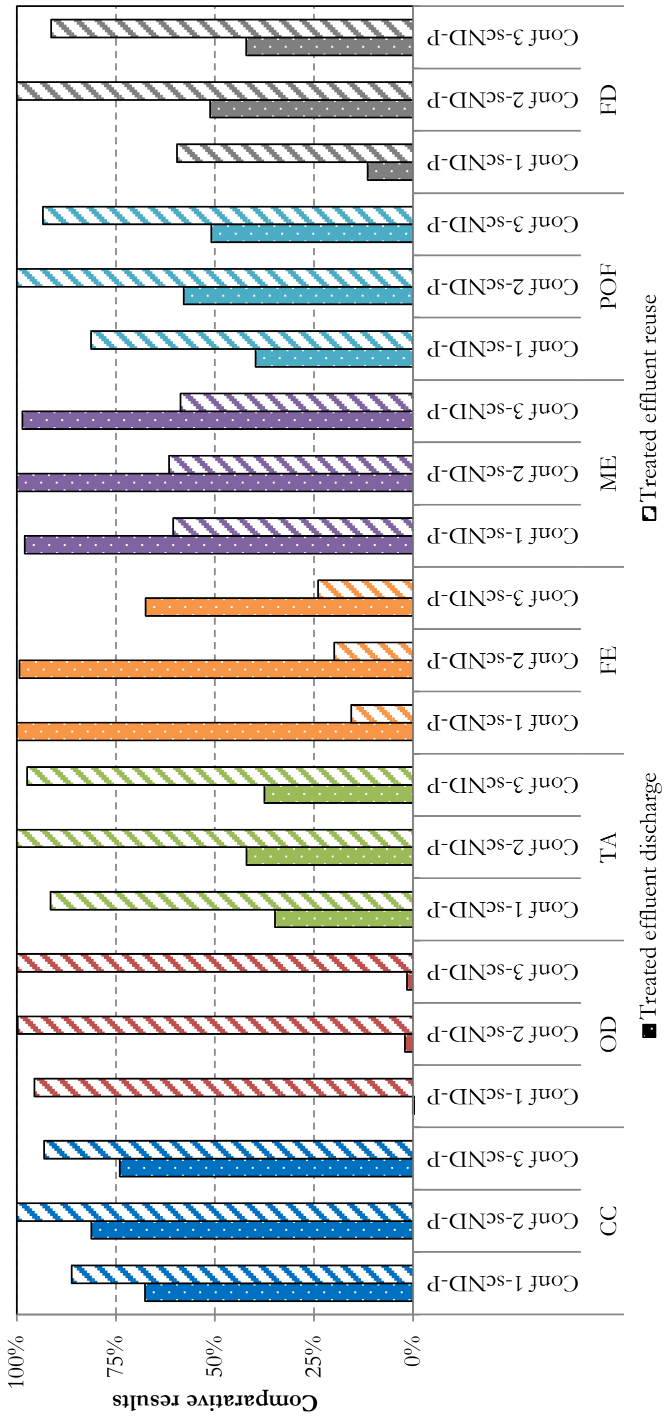

Figure 8.9. Comparative results of the sensitivity analysis regarding treated effluent reuse

As presented in Figure 8.9, the use of treated effluent for irrigation had an adverse impact in the environmental profile of energy-related impact categories (CC, OD, POF and FD) due to the tertiary treatment as well as the use of agricultural machinery for irrigation. Regarding TA, the environmental impacts also increased due to direct ammonia emissions produced during irrigation. On the contrary, the performance in FE and ME was improved by 64%-84% and 38%-40%, due to the reduction of direct P and N emissions into water recipients. In ND configurations, the nutrient limits for the treated effluent are not met due to the lack of carbon source and thus, water reuse is not possible. LCA works dealing with wastewater treatment have identified further environmental benefits (i.e. replacement of mineral fertilisers) from the use of reclaimed water for agricultural purposes (Meneses et al., 2010). The application of scND-P configurations can result in savings of 3.85-4.10 kg N/FU and 0.95-1.59 kg P_2O_5/FU of nitrogen and phosphorus based fertilisers. This would result in a reduction of approximately 44.7 kg CO_2 eq/FU, 8.24 g P eq/FU and 10.5 g N eq/FU for CC, FE and ME, respectively, which would enhance the environmental profile of the systems by 11-17%.

8.6. Proposal of alternative configurations

An alternative configuration was also evaluated in this section. In this case, it has been proposed the possibility of coupling the UASB with membranes in an anaerobic membrane bioreactor (AnMBR) for the solid/liquid separation. The proposal of this alternative technology is related to the large energy requirements previously observed in the previous configurations. Regardless the alternative scenario, SBR can be considered as an environmental hotspot; while this process has lower energy requirements than the aerobic membrane bioreactor and produces less amount of sludge (Li et al., 2013). The main barriers for the application of AnMBRs for domestic wastewater are related with the operating cost for membrane fouling control and mitigation (Li et al., 2013).

Therefore, for the three collection schemes under study, the use of a SBR applying denitrification via nitrite and biological phosphorus removal (Conf 1-scND-P, Conf 2- scND-P and Conf 3-scND-P) were compared from an environmental perspective with the use of AnMBR (Conf 1-AnMBR, Conf 2-AnMBR and Conf 3-AnMBR).

Within the employment of AnMBR, the total liquid stream produced from the screw-press is fed to the UASB since there is no requirement for carbon source. Consequently, the OLR increases from 1.8 to 2.4 kg COD/m^3·d, resulting in increased biogas production. In addition, coupling the treatment scheme with a membrane results in the production of a final effluent free of TSS, but rich in nutrients. The main characteristics of the configurations with AnMBRs are summarised in Table 8.9.

Table 8.9. Calculated characteristics of the treated effluent as well as production rates of methane, sludge and compost in the treatment schemes with membrane

		Conf 1-AnMBR	**Conf 2-AnMBR**	**Conf 3-AnMBR**
Treated effluent				
COD	(mg/L)	80	69	83
TSS	(mg/L)	0	0	0
TN	(mg/L)	63	64	66
TP	(mg/L)	8.5	8.9	9.2
Methane production (as biogas)	(m^3/d)	71	69	74
Sludge production	(kg/d)	496	689	669
Compost production	(kg/d)	344	436	423

The characterisation results of each configuration under study regarding the seven impact categories selected for the study can be found in Table 8.10. In addition, Figure 8.10 shows the comparative results obtained. Regardless the collection scheme, configurations implementing AnMBR perform better compared to scND-P in terms of CC, OD, TA, POF and FD. Regarding energy-related categories, these results can be explained considering that electricity consumption and avoided heat production from fuel oil were identified as the most influential parameters. Configurations scND-P consume more electricity than Configurations AnMBRs to treat the same amount of sewage and DOW, mainly due to aeration requirements during the aerobic phase of the SBR. In addition, in Configurations AnMBR, biological nutrient removal is not practiced; thus, the entire available carbon source of the fermentation process is fed to the AnMBR. The latter results in enhanced biogas production and thus, increased bioenergy generation; hence, more environmental credits are obtained due to the avoided heat production from natural gas.

Table 8.10. Characterisation results of the schemes under comparison

	CC ($kg\ CO_2$ eq)	**OD** (kg CFC-11 eq)	**TA** ($kg\ SO_2$ eq)	**FE** (kg P eq)	**ME** (kg N eq)	**POF** (kg NMVOC)	**FD** (kg oil eq)
Conf 1-scND-P	490	$-3.8 \cdot 10^{-6}$	1.55	0.85	6.64	0.50	9.31
Conf 2-scND-P	585	$1.2 \cdot 10^{-5}$	1.84	0.84	6.58	0.75	44.2
Conf 3-scND-P	533	$8.0 \cdot 10^{-6}$	1.63	0.57	6.61	0.64	34.5
Conf 1-AnMBR	315	$-1.3 \cdot 10^{-5}$	0.99	3.43	26.2	0.49	-8.64
Conf 2-AnMBR	318	$-9.9 \cdot 10^{-6}$	0.95	3.64	26.7	0.43	-4.70
Conf 3-AnMBR	264	$-1.3 \cdot 10^{-5}$	0.73	3.81	27.7	0.34	-13.4

Concerning TA, since ammonia emissions were identified as the main contributor, the differences found among configurations are related with the amount of sludge produced. Configurations scND-P include a SBR and generate higher sludge quantities and consequently, higher ammonia emissions during the composting process. In addition, higher amount of compost is produced, resulting in higher ammonia emission levels when compost is applied on land as a soil conditioner. With regard to eutrophication-related categories (FE and ME), the results are directly affected by the concentration of TP and TN in the discharged final effluent. Configurations scND-P that apply BNR obtained better environmental results for these two impact categories due to the lower nutrient concentration of the treated effluent that is discharged (Table 8.3).

The selection of the most suitable treatment configuration depends on the specific characteristics of the small community and the final use of the treated effluent. When the final purpose is the treated effluent discharge into water recipients, the treated water should meet the limits set by the EU legislation for urban wastewater treatment (EEC, 1991). As mentioned, in the case of a decentralised WWTP serving a community of 2,000 PE, restrictions for biochemical oxygen demand (25 mg/L), COD (125 mg/L) and TSS (35 mg/L) are set. Therefore, the absence of limits for TP and TN allows the application of the scheme that includes AnMBR. The reuse of the final effluent for agricultural purposes limits the application of the system. The Italian Decree regulates water reuse of the treated effluent considering parameters such as salinity, pathogenicity, nutrients, heavy metals and micropollutants (Decreto Ministeriale n. 185, 2003). The maximum allowable concentrations of TP and TN in the treated effluent are 2 mg/L and 15 mg/L, respectively. Thus, the effluent from the UASB reactor requires post-treatment to reduce the nutrients level. In this case, since BNR must be applied, Configurations scND-P are suitable if effective tertiary filtration and appropriate disinfection take place.

8.7. Discussion

8.7.1. Wastewater treatment in small communities

The most common wastewater treatment scheme for small communities is constructed wetlands (Barros et al., 2008; Chan et al., 2008; Wu et al., 2011; Ye and Li, 2009). Other configurations have also been proposed, such as trickling filter, activated sludge, membrane bioreactor (Molinos-Senante et al., 2012) and an integrated step-feed biofilm process (Liang et al., 2010). A review of different schemes designed for the treatment of domestic wastewater at decentralised level can be found in Table 8.11. Nogueira et al. (2009) compared the economic and environmental profile of energy-saving and intensive wastewater treatment systems. Energy-saving technologies such as slow rate infiltration plants and constructed wetlands exhibited better results compared to the activated sludge processes. Yildirin and Topkaya (2012) evaluated the environmental behaviour of constructed wetlands, vegetated land and activated sludge (with and without phosphorus removal), reporting similar results in CC impact but also in terms of the eutrophication-related categories. Regarding more advanced treatment technologies, Zeeman et al. (2008) analysed the operational performance of UASB for the separate treatment of both grey and black water. Grey water was treated in a UASB-SBR system, while a struvite precipitation process was applied after the UASB process for black water. The comparison of the proposed treatment configurations with conventional sanitation showed energy savings of 200 MJ/PE year and phosphorus recovery via struvite of 0.14 kg P/PE year. In comparison with our study, important water reductions related with the use of vacuum toilets are shown. The production of grey and black water was in the range of 60-90 L/PE day and 6.8-7.5 L/PE day, respectively, while a production of 200 L/PE day was considered in the current work. Energy consumption was estimated as 151 MJ/PE year. Despite the differences among the treatment systems examined in the current work, similar results were obtained in Configurations 1 (160 MJ/PE·year), while in configurations 2 and 3, energy consumption was higher (300 MJ/PE·year). Regarding more advanced treatment technologies, Zeeman et al. (2008) analysed the operational performance of UASB for the separate treatment of both grey and black water. Grey water was treated in a UASB-SBR system, while a struvite precipitation process was applied after the UASB process for black water.

Table 8.11. Summary of performances of different decentralised wastewater systems

			Wu et al. (2011)	Ye and Li (2009)	Barros et al. (2008)	Chan et al. (2008)	Liang et al. (2012)	Our study (C1-scND-P)
Technology			CW	Towery hybrid CW	Anaerobic digester-CW	Vegetated sequencing batch coal slag bed	Integrated step-feed biofilm process	UASB + SBR
Description			Sedimentation tank + vertical flow CW (with one willow)	3 stages: stage 1 and 3 are rectangle subsurface horizontal flow CW and stage 2 is a circular CW	2 UASB in series + subsurface-flow CW +surface- flow CW	A CW working as a batch reactor using coal slag as substrate and Umbrella grass as the plant component	Drop-aeration biofilm combined with the step-feed process: 3 stages anoxic/oxic reactors + secondary clarifier	UASB for biogas production + SBR for nutrient removal using fermented DOW as carbon source + composting
PE			1 household	40	30	Pilot scale		2,000
Flow					3.2			400
RE	COD	%		84.5	92.2	63.6	80.0	85.0
	BOD_5	%	95.7		90.3	59.7		
	TSS	%	96.8	88.6	92.7	80.0	90.0	86.0
	TN	%		82.9	53.1	50.0		
	NH_4-N	%	87.3	83.1	52.3	50.5	90.0	82.0
	TP	%	87.9	64.2	51.4	40.0	15.0	72.4

Acronyms: PE: population equivalent; RE: removal efficiency; CW: constructed wetland

The comparison of the proposed treatment configurations with conventional sanitation showed energy savings of 200 MJ/PE year and phosphorus recovery via struvite of 0.14 kg P/PE year. In comparison with our study, important water reductions related with the use of vacuum toilets are shown. The production of grey and black water was in the range of 60-90 L/PE day and 6.8-7.5 L/PE day, respectively, while a production of 200 L/PE day was considered in the current work. Energy consumption was estimated as 151 MJ/PE year. Despite the differences among the treatment systems examined in the current work, similar results were obtained in Configurations 1 (160 MJ/PE·year), while in configurations 2 and 3, energy consumption was higher (300 MJ/PE·year). Alternatives of the conventional SBR were also analysed in the literature, including the performance of a sequencing batch membrane bioreactor (Krampe, 2013). One of the advantages of coupling a membrane to a SBR is the reduced cycle time as a result of the elimination of the settling phase and complete elimination of suspended solids in the treated effluent. However, they are associated with higher operating costs due to membrane fouling.

8.7.2. Comparative evaluation of environmental results

The results of this work are in agreement previous LCA studies on wastewater treatment. However, only qualitative comparison can be performed since the schemes examined in our work include the treatment of wastewater together with DOW. Hospido et al. (2004) assessed the potential environmental impacts that are associated with a municipal WWTP designed for 90,000 PE. The discharge of the treated effluent and land application of sludge were the main environmental hotspots of the treatment system. Gallego et al. (2008) analysed the environmental results of alternative technologies for wastewater treatment in small communities of less than 20,000 PE. Both the discharge of the treated effluent and the disposal of sewage sludge were identified as the most important environmental hotspots due to the presence of nutrients and heavy metals, respectively. The environmental and economic performance of 24 WWTPs was evaluated in the study of Rodriguez-Garcia et al. (2011). Nutrient emissions in the treated effluent were again the main hotspot for the eutrophication related categories, while electricity consumption for CC. Furthermore, LCA has been applied for the comparison of alternative schemes that apply integrated processes for organic waste and sewage sludge management. Nakakubo et al. (2012) compared the conventional incineration of food waste with the separate treatment of sewage sludge followed by anaerobic co-digestion of both waste streams, examining different processes for the digestate treatment. The authors demonstrated that from an environmental point of view, the combined management of both waste streams performed better than the separate scheme. Righi et al. (2013) analysed the environmental profile of decentralised sewage sludge and DOW management through anaerobic co-digestion.

8.8. Conclusions

The technical evaluation of the systems revealed that the co-management of wastewater and DOW is feasible for a small community since, regardless the applied collection scheme, the treated effluent met the discharge requirements. In addition, the removal of nitrogen via nitrite with EBPR in the SBR upgraded the treated effluent quality allowing its reuse for agricultural purposes provided tertiary filtration and disinfection take place. Those configurations performing denitrification via nitrite allowed higher levels of fermented liquid recirculation in the UASB, resulting in higher biogas generation. The environmental assessment of the alternative processes in the integrated systems showed that energy-related categories achieved the lowest results in Configuration 1-scNSD and the highest environmental impacts were produced in Configuration 2-ND-P. The environmental impacts were mainly attributed to the energy requirements for FWD operation and SBR aeration. The use of FWDs increased the environmental impacts compared to the separate collection, while denitrification via nitrate entailed higher impacts in energy-related categories compared to nitrogen removal via nitrite. Moreover, impacts in eutrophication related categories were derived from the discharge of the treated effluent. Thus, the collection scheme did not affect the environmental performance. The systems which perform nitrogen removal via nitrite and EBPR via nitrite resulted in better environmental profile concerning FE and ME. Considering technical and environmental aspects, it can be concluded that the separate collection of waste combined with nitrogen removal and phosphorus uptake via nitrite is the best configuration for the combined treatment of wastewater and DOW in a small community of 2,000 PE.

8.9. List of acronyms

AnMBR	Anaerobic membrane bioreactor
BNR	Biological nitrogen removal
BOD	Biochemical oxygen demand
C/N	Carbon to nitrogen ratio
CC	Climate change
COD	Chemical oxygen demand
DNBPR	Denitrifying via nitrite biological phosphorus removal
DO	Dissolved oxygen
DOW	Domestic organic waste
DPAO	Denitrifying phosphorus accumulating organism
EBPR	Enhanced biological phosphorus removal
FD	Fossil depletion
FE	Freshwater eutrophication
FWD	Food waste disposers
HRT	Hydraulic retention time
ME	Marine eutrophication
ND	Conventional nitrification/denitrification
OD	Ozone depletion
OLR	Organic loading rate
PAO	Phosphorus accumulating organism
PE	Population equivalent
POF	Photochemical oxidant formation
SBR	Sequencing batch reactor
scND	Short-cut nitrification/denitrification
TA	Terrestrial acidification
TC	Total carbon
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UASB	Up-flow anaerobic sludge blanket
VFA	Volatile fatty acid
vNLR	Volumetric nitrogen loading rate
WWTP	Wastewater treatment plant

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Section IV: Conclusions

Chapter 9: General findings and conclusions

9.1. Conclusions of the thesis

The purpose of this thesis was to analyse the environmental aspects associated with the production of biogas by the application of the anaerobic digestion process in several real and potential scenarios in the European Union. The importance is linked with the fact that biogas is being promoted as a sustainable form of energy in line with the concept of circular economy since it is able to convert biomass into valuable products such as energy, water and nutrients. LCA methodology has proved its usefulness as an environmental management tool that provides strategic information for the assessment and improvement of well stablished processes as well as for the design of novel ones. The main findings and general conclusions that were obtained from the research activities presented in Sections II and III are detailed below.

Section II. Agricultural biogas

This section focussed on the assessment of environmental sustainability of biogas production from agricultural biomass. Several real biogas plants operating in Italy were analysed to identify the most polluting stages in well-established biogas processes. An innovative treatment process for the management of manure in Cyprus that includes anaerobic digestion and the treatment of the produced digestate to produce several valuable products was also analysed.

The key findings of the LCA performed in four conventional agricultural biogas plants operating in Italy were:

✓ Biogas plants based on the treatment of manure achieved lower impacts on CC compared with the reference system, which was the environmental profile of the current scheme of electricity production in Italy. Biogas plants performing the extensive use of energy crops involved higher GHG emissions due to the cultivation of cereals.

✓ Impacts in acidification and eutrophication categories were higher in all biogas plants than in the reference system due to emissions derived from the management of digestate as an organic fertiliser, intensified by the extensive cultivation of energy crops owing to the use of mineral fertilisers. No defined tendency was identified for these categories regarding the type of substrate digested.

The conclusions related to the strategies proposed to mitigate environmental impacts in these plants were:

✓ The exploitation of the surplus heat in a nearby greenhouse improved the overall system profile, especially regarding energy-related categories, due to the avoided production of heat from diesel (conventional system).

✓ The covering of the storage tank reduced methane, nitrous oxide and ammonia emissions, which decreased the total GHG emissions and reduced acidification impacts; though, the high impact produced in the agricultural step hid this measure for improvement in the overall profile of the biogas system.

✓ The injection of digestate on land entailed different impacts among categories compared with the surface spreading due to higher emissions of nitrous oxide but lower emissions of ammonia.

A deeper analysis performed in two additional biogas plants in Italy with the aim of recognising the environmental consequences of substrate selection in agricultural biogas drew the main conclusions:

✓ Both plants achieved lower GHG emissions compared with the Italian electric profile; however, only the one performing the co-digestion of large proportion of wastes attained results compared with the requirements established by the European Commission to consider bioenergy from biogas as sustainable.

✓ Substrate selection has an outstanding impact in the performance of biogas systems, not only due to environmental burdens of their production and methane potential, but also because it influences the quantity and quality of the digestate.

✓ Food waste was identified as an interesting co-substrate, since it has a high energy potential (especially in comparison with pig slurry) and no environmental burdens from their production are allocated to the biogas system (different from energy crops). Nevertheless, electricity consumption in the biogas plant was increased due to the pre-treatment requirements of this type of waste.

The assessment performed to analyse the influence of methodological assumptions resulted in the following findings:

✓ The selection of the method used for accounting emissions from the management of digestate significantly affected the results, being ammonia and nitrate the emissions that varied the most.

✓ The use of electricity output revealed as an appropriate FU since it takes into consideration the conversion efficiencies of feedstock conversion into biogas and biogas conversion into bioenergy.

Operational inefficiencies leading to additional environmental impacts were studied in 15 Italian biogas plants concluding that:

✓ The combined implementation of LCA+DEA methodologies allowed the assessment of biogas plants from an ecoefficiency point of view, connecting operational parameters with environmental results.

✓ A high ratio of biogas plants operated under ecoefficient conditions (60% achieved a score of 100%); showing a general high operational efficiency in the whole sample (average efficiency of 85%).

✓ The reduction targets calculated based on the eco-efficiency principles for the inefficient plants enhanced the profile of the plants exhibiting the worst overall environmental impact.

The analysis of the sustainability of a novel treatment process for the treatment of manure in Cyprus showed that:

✓ The LCA of the pilot plant treating animal waste in Cyprus permitted the identification of the most contributing processes of the treatment scheme and the best configuration for the management of the liquid digestate.

✓ The biological removal of nutrients from the liquid digestate revealed as the most important energy-demanding process due to aeration requirements, affecting CC and other energy-related categories.

✓ The agricultural use of compost produced from the solid digestate was identified as the most contributing process to acidification and eutrophication categories; however, no substantial differences were found among configurations since they are only related to the treatment of the liquid digestate.

✓ The combination that allows the maximisation of struvite recovery and nitrogen removal resulted in treatment configuration was selected as the most appropriate from an environmental point of view.

✓ The replacement of acetic acid by methanol as carbon source for the operation of the biological nutrients removal process improved the performance of the system in terms of MA. The use of the DF effluent as carbon source would decrease the consumption of chemicals in the pilot plant.

The comparison of the performance of the selected configuration was compared with the four most common practices for animal waste management in Cyprus, finding that:

✓ The use of anaerobic lagoons was the recognised as the worst management practice since the disposal of manure under anaerobic conditions without the appropriate use of their organic matter and nutrients derived in significant methane emissions to air and nutrients discharged to water.

✓ The production of electricity from manure through biogas, either in a conventional biogas plant or following the novel treatment scheme proposed, improved significantly the environmental profile of these systems due to the high ratio of fossil fuels use in the Cypriot electric profile.

✓ The production of several valuable products from manure in the novel treatment system proposed, not only bioenergy but also compost, struvite and a reusable effluent, positively impacted the results due to the avoided production and use of fossil fuel-based alternatives.

✓ The sensitivity analysis revealed that electricity generation from biogas was a special beneficial option in Cyprus due to the characteristics of the electric profile. In addition, biogas upgrading to be used as vehicle fuel presented as an interesting alternative. The replacement processes selected has a remarkable influence on the results, showing that the best use of biogas depends on the specific case study.

✓ The use of various methodologies for the calculation of emissions from several manure management strategies derived in different emissions, which affected the results of the whole process, influencing even the amount of replaced mineral fertilisers.

A sustainable assessment was also performed to analyse the potential benefits of the treatment scheme in comparison with the two most common practices in Cyprus. The main findings were:

✓ AHP was presented as a suitable methodology able to integrate environmental, social and economic impacts to identify the most sustainable system among a group of alternatives.

✓ The selection of criteria used in the analysis greatly influence the results and the evaluation of them should be performed by a multidisciplinary expert panel.

✓ Even if the capital and operational costs of the LiveWaste treatment are higher, the environmental, social and economic benefits made it the most sustainable animal waste management option.

Section III. Sewage biogas

This section analyses the environmental consequences of anaerobic digestion as a waste valorisation option in the context of wastewater treatment, including different WWTP in terms of treatment capacities and treatment processes.

The assessment focused on the potential environmental consequences of the co-digestion of food waste and sewage sludge in a large WWTP (150,000 PE) showed that:

✓ The integration of food waste treatment in an existing WWTP entailed higher biogas production (up to 2.7 times), allowing the production of the electricity and heat required for the operation of the plant.

✓ In environmental terms, derived GHG emissions increased with the integration of food waste due to extra energy consumption and higher emissions in the biological reactor, but also higher environmental credits due to replacement electricity production and mineral fertilisers use.

✓ The sensitivity analysis performed to assess the potential effects of synergetic effect in co-digestion showed that increased biogas yield enhanced the performance of energy-related categories but increased acidification impacts due to the ammonification process.

✓ The barriers in regulations hamper the implementation of the co-management of sewage sludge and food waste at full-scale in the United Kingdom, preventing the implementation of circular management chains that helps to move towards sustainability in waste management strategies.

The assessment of sewage and food waste co-management in a decentralised community (2,000 PE) was assessed for an innovative integrated system that includes digestate treatment. The major findings of the study were:

✓ The technical evaluation of the systems revealed that the co-management of wastewater and food waste is feasible for a small community since, regardless the applied collection scheme, the treated effluent met the discharge requirements.

✓ The biological removal of nitrogen and phosphorus via nitrite upgraded the treated effluent quality, allowing its reuse for agricultural purposes provided tertiary filtration and disinfection take place. In addition, the lower requirement of carbon source allowed higher levels of fermented liquid recirculation in the UASB, resulting in higher biogas generation.

✓ The best environmental results in terms of energy-related impact categories were achieved by nutrients removal via nitrite due to lower aeration requirements and higher biogas potential due to lower consumption of fermented liquid as carbon source.

✓ The sensitivity analysis on design parameters showed that the efficiency of food waste collection and COD removal in the anaerobic process were the most important parameters analysed since they had an impact on the performance of the whole process.

✓ The sensitivity analysis on LCA assumptions evidenced the environmental benefits in eutrophication categories related to the use of the treated effluent in agricultural land due to the reuse of nutrients. However, impacts associated with other impact categories increased due to the tertiary treatment required.

9.2. Recommendations and future outlook

As evidenced along the thesis, anaerobic digestion can play a central role in the long-standing commitment towards sustainable development, mitigation of climate change and energy security. Anaerobic digestion also shares the principles of circular economy, allowing a "win-win" situation that permits the development of a prosper economy, while preserving the environment.

The pace of growth largely depends on the political and legal conditions, being able to either boost or hinder the implementation and further development of biogas technologies. General policy framework should promote biogas as a suitable energy production system as well as feasible waste valorisation strategy. In this sense, efforts should delve more deeply into the integration of circular value chains that allow the integrated management of different available organic wastes whose management still represent an environmental concern. To achieve success, well-defined goals and dissemination of information to all involved stakeholders is crucial. Moreover, policy context should also enable biogas suppliers to continue with research to make biogas technologies fit for the challenges of the future, preventing problem shifting, i.e. not only allowing savings GHG emissions, but also preventing acidification and eutrophication impacts.

Biogas industry and policy makers should strive on communication of the positive role of biogas in a future sustainable power supply system to improve public acceptance of biogas production. In this respect, it would be interesting to develop a decision-making scheme that systematically evaluates biogas technologies integrating social, economic and environmental criteria with the aim of boosting the spread of biogas production systems. Regarding the environmental dimension, it is necessary the development of common and very specific guidelines for LCA studies to assess and communicate the environmental results.

9.3. List of acronyms

AHP	Analytical hierarchy process
COD	Chemical oxygen demand
DEA	Data envelopment analysis
DF	Dark fermentation
FU	Functional unit
GHG	Greenhouse gas
LCA	Life cycle assessment
MA	Malodours air
PE	Population equivalent
UASB	Up-flow anaerobic sludge blanket
WWTP	Wastewater treatment plant

Publications

Journal publications

[1] **L Lijó**, S González-García, J Bacenetti, M Fiala, G Feijoo, JM Lema and MT (2014). Life Cycle Assessment of electricity production in Italy from anaerobic co-digestion of pig slurry and energy crops. Renewable energy, 68, 625-635. doi:10.1016/j.renene.2014.03.005

[2] **L Lijó**, S González-García, J Bacenetti, M Fiala, G Feijoo and MT Moreira (2014). Assuring the sustainable production of biogas from anaerobic mono-digestion. Journal of Cleaner Production, 72, 23-34. doi:10.1016/j.jclepro.2014.03.022

[3] **L Lijó**, S González-García, J Bacenetti, M Negri, M Fiala, G Feijoo and MT Moreira (2015). Environmental assessment of farm-scaled anaerobic co-digestion for bioenergy production. Waste management, 41, 50-59. doi:10.1016/j.wasman.2015.03.043

[4] S González-García, C Lacoste, T Aicher, G Feijoo, **L Lijó**, MT Moreira (2016). Environmental sustainability of bark valorisation into biofoam and syngas. Journal Cleaner Production, 125, 33-43. doi: 10.1016/j.jclepro.2016.03.024

[5] **L Lijó**, S Malamis, S González-García, MT Moreira, F Fatone and E Katsou (2016). Decentralised schemes for integrated management of wastewater and domestic organic waste: the case of a small community. Journal of Environmental Management. doi:10.1016/j.jenvman.2016.11.053.

[6] **L Lijó**, S Malamis, S González-García, F Fatone, MT Moreira and E Katsou (2017). Technical and environmental evaluation of an integrated scheme for the of wastewater and domestic organic waste in small communities. Water Research, 109, 173-185. doi:10.1016/j.watres.2016.10.057

[7] **L Lijó**, Y Lorenzo-Toja, S González-García, J Bacenetti, M Fiala and MT Moreira (2017). Eco-efficiency assessment of farm-scaled biogas plants. Bioresource Technology. doi:10.1016/j.biortech.2017.01.055

[8] **L Lijó**, S González-García, J Bacenetti and MT Moreira (2017). The environmental effect of substituting energy crops for food waste as feedstock for biogas doi:10.1016/j.energy.2017.04.137

Conference proceedings

Conference	**EcoTechnologies for Wastewater Treatment (EcoSTP2014)** (Verona, Italy)
Contribution I	Poster - The poster won the third award in the Conference "Environmental assessment of organic waste and domestic wastewater management in decentralised communities"
Authors	L Lijó, S González-García, E Katsou, S Malamis, F Fatone, G Feijoo and M T Moreira
Contribution II	Oral presentation "Environmental assessment of bioenergy production through suitable wastewater and waste management"
Authors	L Lijó, S González-García, D Renzi, F Fatone, G Feijoo and MT Moreira.
Conference	**Biogas Science 2014** (Vienna, Austria)
Contribution	Poster "The life cycle approach for the assessment of pig slurry management"
Authors	L Lijó, S González-García, J Bacenetti, M Fiala, G Feijoo and MT Moreira
Conference	**Balkan Young Water Professionals 2015** (Thessaloniki, Greece)
Contribution	Oral presentation "Assessing environmental consequences of the co-management of wastewater and domestic organic waste"
Authors	L Lijó, S González-García, E Katsou, S Malamis, F Fatone, G Feijoo and MT Moreira
Conference	**World Water Congress XV** (Edinburgh, United Kingdom)
Contribution	Oral presentation "Environmental assessment of an innovative scheme for the co-management of wastewater and domestic organic waste in small communities"
Authors	L Lijó, S González-García, E Katsou, S Malamis, F Fatone, G Feijoo and MT Moreira
Conference	**Sustainable Solid Waste Management 2015** (Tinos Island, Greece)
Contribution	Oral presentation "Integrated management of wastewater and domestic organic waste in small communities"
Authors	L Lijó, S González-García, E Katsou, S Malamis, F Fatone, G Feijoo and MT Moreira
Conference	**Life Cycle Management 2015** (Bordeaux, France)
Contribution	Poster presentation "Introducing pig slurry in full-scale biogas plants to reduce environmental footprint"
Authors	**L Lijó**, S González-García, J Bacenetti, M Fiala, G Feijoo, MT Moreira
Conference	**SUM2016** (Bergamo, Italy)
Contribution	Oral presentation "Environmental assessment of domestic food waste management alternatives"
Authors	**L Lijó**, MT Moreira, E Katsou and S González-García
Conference	**LET2016** (Jerez de la Frontera, Spain)
Contribution	Poster presentation "Environmental assessment of alternative domestic wastewater and organic waste management in small communities"

Authors	**L Lijó**, S González-García, E Katsou, S Malamis and MT Moreira
Conference	**Cyprus2016** (Limassol, Cyprus)
Contribution I	Oral presentation: “Environmental impacts of different options for the management of livestock waste in Cyprus”
Authors	**L Lijó**, S González-García, G Feijoo, MT Moreira
Contribution II	Oral presentation: “Comparative analysis of anaerobic mono-digestion and co-digestion of organic waste”
Authors	**L Lijó**, N Voulvoulis, N Frison, S González-García, MT Moreira, E Katsou
Contribution III	Poster presentation: “Impact assessment of alternative food waste schemes in a small community from a life cycle perspective”
Authors	**L Lijó**, E Katsou, MT Moreira, S González-García
Conference	**SWWS2016** (Athens, Greece)
Contribution I	Oral presentation: “Environmental assessment of alternative treatments for wastewater and domestic organic waste”
Authors	**L Lijó**, MT Moreira, E Katsou, S Malamis, S González-García
Contribution II	Oral presentation: “Environmental consequences of including food waste management in agricultural biogás production”
Authors	**L Lijó**, S González-García, J Bacenetti, M Fiala, MT Moreira
Conference	**Biorestec2016** (Sitges, Spain)
Contribution	Keynote “Eco-efficiency assessment of biogás production systems from biomass sources: Joint implementation of life cycle assessment and data envelopment analysis”
Authors	S González-García, **L Lijó**, Y Lorenzo, MT Moreira
Conference	**EcoSTP2016** (Cambridge, UK)
Contribution	Oral presentation: “Anaerobic co-digestion of sewage sludge and food waste. Environmental analysis and political issues in the UK”
Authors	**L Lijó**, N Voulvoulis, N Frison, S González-García, S Malamis, F Fatone, MT Moreira, E Katsou
Conference	**Lisbon2016** (Lisbon, Portugal)
Contribution	Oral presentation: “The environmental effect of substituting energy crops for food waste as feedstock for biogas production”
Authors	**L Lijo**, MT Moreira, J Bacenetti, M Fiala, S González-García