Life Cycle Assessment of shale gas, LNG and waste in the future UK energy mix

Carla Tagliaferri

Department of Chemical Engineering
University College London

A thesis submitted for the degree of
Doctor of Philosophy of University College London

May 2016

I, Carla Tagliaferri, confirm that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been indicated in the thesis.

Abstract

This thesis investigates the environmental impacts of novel energy sources and technologies developing in the UK in the near future. The life cycle assessment methodology (LCA) is applied to advanced energy systems in order to develop a comprehensive framework able to identify the most promising energy supplies, in the context of an increased focus on low carbon technologies and the requirement of a stable and secure energy supply. The outcomes of this study provide valuable information to stakeholders and policy makers to be correctly informed, and can help in planning new policy legislations or tune the existing ones.

The evolution of the UK energy mix through the recent past till the current times is analysed. Key sources and technologies for the future energy supply are identified and reviewed. Their environmental burdens are not currently quantified; hence, this study develops a number of different LCA models for a future, aware, energy development.

First, the study uniquely approaches the LCA analysis of shale gas and Liquefied Natural Gas (LNG) for the UK as they are expected to play an important role in the future UK energy mix.

After that, within the framework of diverting waste from landfill and produce renewable energy, the environmental impacts of an advanced gasification-plasma technology for electricity production are analysed for the treatment of different feedstocks. The technology is compared to other advanced and conventional waste-to-electricity technologies, including pyrolysis and combustion.

Bio-Substitute natural gas (Bio-SNG) production from waste through advanced thermal technologies is then studied within the context of de-carbonising the gas grid. This process is compared to biological processes for biomethane production from waste according to current and future energy mixes.

The outcomes of this research do not identify a unique trend. The context in which the analysed technologies operate, the basis of the comparisons between different alternatives and the approached perspective of the study, characterize the interpretation of the obtained results.

The environmental models developed in this study are suitable for the environmental assessments of energy mixes of different countries. The framework developed also identifies the boundaries, the flows and the alternative scenarios to be considered in parallel social and economic life cycle thinking studies.

Acknowledgments

First of all, I would like to express my gratitude to Prof. Paola Lettieri for her continuous support and guidance throughout this project and beyond. I am grateful for her encouragement to think freely and for all the opportunities that she has offered me. My gratitude goes also to Prof. Roland Clift for his support and for many fruitful discussions.

I would also like to thank Chris Chapman from Advanced Plasma Power (APP) for giving me the opportunity to join the team at the company and for the many valuable discussions on the project. I am also thankful to the entire APP team for providing technical data and support.

Thank you to all my friends and colleagues at UCL who made this PhD journey much more enjoyable.

My deepest gratitude goes to my family, Mum, Dad, Brother and Cats for their infinite support. A special thank you goes also to Dr C. D. for always being there for me and for stimulating scientific questions and discussions.

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Chapter 1. **Introduction**

The development of modern societies is strictly related to the types and amount of energy used. The energy sector is responsible for the majority of the total greenhouse gas emissions locally and globally because it has been mainly relying on the use of fossil fuels during the last decades. Therefore, shifts in energy sources and technologies can affect global warming and environmental quality. This is also true for the particular case of the UK where in 2013 more than 47% of the total domestic greenhouse gas emissions were caused by energy production and its use. To develop ‘sustainable energy systems’, the UK has to face challenges related to the current decrease in domestic gas production, increasing population and uptake of renewable technologies to decrease the total impact of the energy sector. The UK government has been providing support to unconventional gas development, import of Liquefied Natural Gas and also development of waste to energy technologies to restructure the energy mix. This thesis employs the life cycle environmental analysis to assess how the aforementioned technologies may contribute to the current debate on the evolution of the future energy mix. This introductory chapter presents the motivations and objectives of this research and reports the research outline.

1.1 Background

Massive energy production and consumption are the basis on which modern societies are developing. Our quality of life and the entire society’s wealth crucially rely on a readily available energy supply because energy can be usefully transformed into work. Political decisions, alliances and sometimes also wars are often driven by the gain of energy advantages. In the present time, a society primarily based only on animal, human, wind and water power, as happened in the past, can even be barely imagined.

We are sustainably developing only if we “meet the needs of the present without compromising the ability of future generations to meet their own needs” [1]. Sustainable development is currently considered the best approach to address the complex and interrelated threats that the world is facing today [2], including both a secure energy supply and environmental pressure. Present and future development based on sustainability is deemed highly imperative as an unsustainable development can seriously threaten the environment’s ‘health’ that is a key factor for the well-being of humans and other species on earth [2].

The production of many forms of energy determines the deployment or degradation of natural resources, and this is even truer when energy is produced from fossil fuels. Many countries worldwide are today mainly concerned with the future (short term and long term) availability of energy required by industrial affluent societies [3]. The reserves belong only to a few producing countries which many other countries have to rely on and the majority of the fossil energy sources are limited in their total amount deployable [4] (or at least the time needed for their natural production is much longer than the time of human deployment). In addition to this, the energy sector is significantly contributing to the global environmental impact [5]. In 2014, it was responsible for more than 35 % of the total anthropogenic Green House Gas emissions (GHG) and compared to the 2010 level, these emissions are expected to increase by 80-130% by 2050, if effective policies are not put in place [6,7]. However, it is still unclear the extent by which current societies have already exceeded the sustainable levels of resource deployment and the environmental impacts.

Given that rising energy demand seems inevitable in the future, the interrelated fossil fuel scarcity and environmental threats are pushing for the search for alternative paths and solutions to depart from the current state. Improved technologies that lead to higher energy efficiencies are one of the proposed solutions to alleviate the environmental burden caused by the energy sector. Some claim that the world will inevitably need to rely on fossil fuel and the improvement of energy efficiency is the only solution to substantially limit the environmental problems [8] and secure the energy supply of fossil resources. However, as humanity is nowadays facing unprecedented far-reaching anthropogenic-induced environmental problems, a combination of alternative solutions may actually provide a possible sustainable solution for future developments [9]. The underlining assumption in this thesis is that a combination of improved energy technologies from fossil fuels and novel technologies based on renewable energy sources are both the keys to reduce the environmental impacts of the energy sector.

1.2 Total GHG emissions in the UK

Over the last decades, energy production and use have mainly relied on fossil fuels. In the UK, the latest complete emissions inventory refers to the year 2013 (Figure 1.2.1). The energy supply sector, including the production of electricity and natural gas, was responsible for 33% of the total domestic emissions. If the use of natural gas in residences is also added to the picture, up to 47% of the total domestic emissions can be associated to energy. The transport sector is also still strongly relying on fossil fuels and it caused up to 21% of the total emissions. By contrast, the emissions from the waste sector were only responsible for 5% of the total; emissions from waste disposal have fallen by 52% since 2007, thanks to the reduction in the amount of biodegradable waste sent to landfill and improvements in the proportion of methane captured at landfill sites and used for energy production.

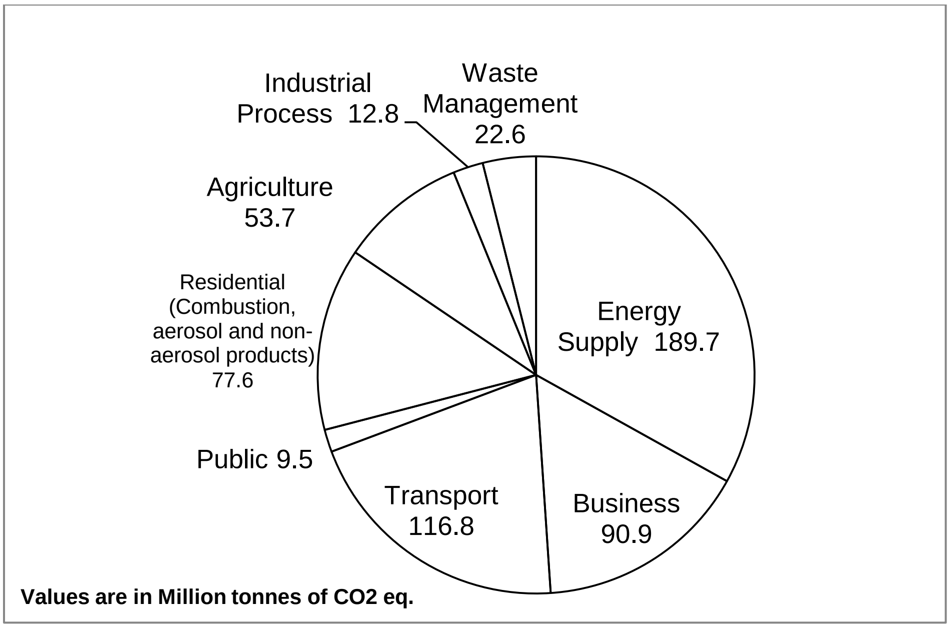

Figure 1.2.1. Total UK greenhouse gas emissions in 2013 by sector (adapted from [10]).

The emissions from the energy sector strongly depend on the sources and technologies used to supply the energy demand. As analysed in more details in Chapter 2, the UK energy mix has undergone a number of important changes during the last centuries that has strongly affected the environment. Wood and animal power were the primary sources of energy but those were soon substituted by coal that throughout the 19th and 20th century was the main source of energy. The 21st century saw a big diversification of the energy mix that started to include natural gas, nuclear, oil and also renewables. The development of the current energy mix is determined by a complex interconnection of engineered changes, type of energy technologies, energy stability and other factors.

Energy policies have to play a major part towards the modification of the energy mix in order to globally and locally decrease the impact of this sector. This is deemed high priority for a balanced development of the energy supply as specified by the energy trilemma.

1.3 The energy trilemma and the UK energy challenges

The complex links between politics, environment and economics are yearly addressed by the World Energy Council [11] that support policy makers as they set climate and development goals and design policies at international and local levels. The support to policymakers is founded on the 'sustainable energy development system'; this is seen as the base for the energy prosperity and competitiveness of individual countries. Three dimensions are considered fundamental for all economies (Figure 1.3.1): i) energy security, ii) energy affordability, iii) energy sustainability. A country needs to put in place reliable energy infrastructure able to meet the current and future energy requirements. Energy should be accessible and affordable to the entire population and should be produced with high efficiency and low carbon impact fuels including renewables.

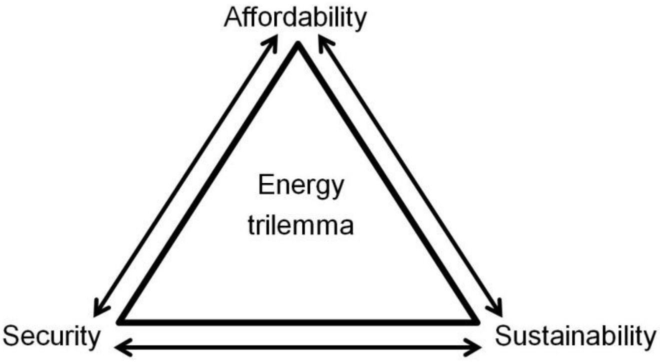

Figure 1.3.1. The energy trilemma.

The three aspects of the energy trilemma are strongly interconnected; therefore, a balanced approach needs to be taken. For example, a focus on the GHG reduction may impede energy security and access, while a focus on increasing affordability may impact energy security and environmental sustainability.

The Energy Trilemma Index is used to rank all countries in terms of their potential ability to provide energy policies based on the 3 dimensions of the energy trilemma. During the last three years (2013-2015), the UK ranked fourth in the world for the energy trilemma index showing a stable position [12]. However, to maintain the delivery of a balanced energy system in line with the trilemma, the UK has still to face a number of substantial challenges for the future:

1. The UK will become the most populated EU country by 2050, with a growth of more than 24% by this date (more than 77 million people are expected to be living in the UK in 2050 [13]). Hence, it will face a significant challenge to deliver an affordable energy supply.
2. The energy infrastructure in the UK is rapidly ageing and many old coal power plants will be forced to close and leave the space to lower carbon and more efficient energy sources and technologies.
3. Currently, the UK still strongly relies on fossil fuels for energy production (69% of the electricity production is based on fossil fuels) unlike the other trilemma index leaders (Switzerland, Sweden, Norway, etc.) that have fully developed hydro, nuclear and wind power in their energy mix. International, European and local agreements are pushing towards a drastic cut of greenhouse gas emissions and energy savings (the energy policies are analysed in more detail in chapter 2). Given the high share of fossil fuels in the current electricity mix, achieving a massive reduction of greenhouse gas emissions appears unrealistic [14] for the UK, unless the energy mix is strongly reformed to meet growing demand at reduced environmental impacts.
4. Natural gas is seen as a reliable, flexible and clean fuel as it can reduce greenhouse gas emissions when compared to coal and can provide a competitive energy price for a number of purposes [15]. Therefore, switching to gas is believed [15] to help meeting short-term emission targets for power generation, heating and transport while restructuring the electricity mix towards renewables. Hence, the UK government plans to use gas to meet long term emissions target but this would increase gas demand, which in turn would affect energy prices as the UK is heavily dependent on gas imports. In fact, in 2014 the UK imported 45% of its total natural gas consumption. In this context, the decline in the domestic gas production raises questions about possible new UK gas exploration to continue delivering secure and affordable energy.

It is clear that the points regarding the energy security at short term and the energy sustainability are contrasting and this complicates even further the energy challenges in the UK. The following sections discuss the strategy developed in the UK to meet future energy demand sustainably.

1.4 The UK energy strategy

The UK National Grid has projected future gas, power and heat demand and supply trends according to different scenarios ranging from low/high effort put on sustainability to low/high economic growth (chapter 2). Carbon targets are met on time or missed depending on the predicted economic prosperity. The demand for gas and power combined across residential, commercial, industrial and transport sectors is expected to be 975-1210 TWh/year by 2035, from current value of 1150 TWh/year (Figure 1.4.1).

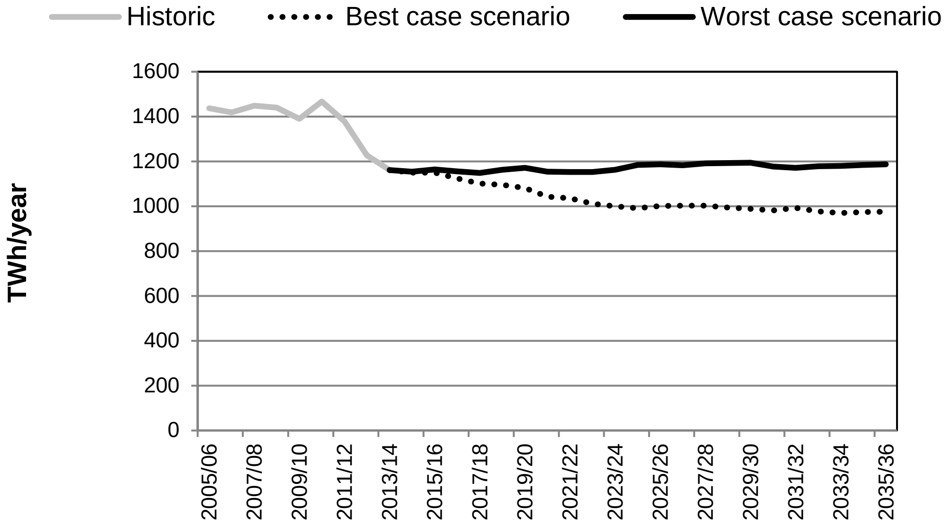

Figure 1.4.1. UK energy demand. Adapted from [16]

The UK Government is developing and adopting a broad range of measures to respond to the challenges discussed in the previous section and to supply the aforementioned requirements.

First of all, higher *energy efficiency and savings* are addressed by the Government in all sectors [17], including, for example, the supply chain of energy production itself, residential heating, industry and manufacturing as a means of reduction in the energy demand and environmental impacts of the energy systems.

The UK government is pushing towards the *decarbonisation of the electricity grid*. Fossil fuel technologies, historically used to meet the energy demand, including power plants run on coal, oil and natural gas, are being displaced by more secure energy technologies that can help meeting the carbon emissions' reduction imposed by legislations. This includes the construction and commissioning of the next generation of nuclear power stations that can deliver the base-load without the risks associated with intermittency. In addition to this, the Government is supporting the commercial scalability of emerging technologies, such as carbon capture and storage (CCS) [18], to allow a cleaner deployment of coal and fossil gas in power plants. Solar power and wind power are projected to fully develop in the near future (chapter 2), and to play an important role for the grid decarbonisation allowing substantial reductions in the reliance on fossil fuels, including gas, in the long term.

More than half of the total current energy demand is due to residential heating and relies on natural gas. Hence, the UK government is pushing the *decarbonising of the heat supply through increased electrification*. The *electrification of the transport sector* is also seen as strategic to reduce pressure on carbon emissions.

Despite the electrification of the heat supply, part of the energy demand will still rely on natural gas (as shown in chapter 2) and by 2035 imports will still supply 40-90% of GB gas demand. Therefore, effective low cost and low carbon solutions, which also take into consideration the decrease in conventional and local natural gas supply, are being explored. The UK Government has considered the *development of unconventional indigenous reserves of shale gas*. It intends to use shale gas as a bridging fuel arguing that there would be a reduction in GHG emissions during the transitional period towards renewable as gas would be used to replace coal. Therefore, shale gas is expected [19] to have an important role in the transition to a low carbon energy mix and the potential to end import dependency. In this respect, the example from the US shows that unconventional gas can transform the domestic energy market, in this case making the US self-sufficient in gas.

Again on the side of energy security, *the development of the infrastructure and the capacity for the gas and power* interconnector systems are being assessed by the Government for future planning. However, it is still unclear what proportion of the imported gas will be Liquefied Natural Gas, particularly from Qatar. The EU is pushing towards the adoption of LNG as this is believed to be a clean and highly efficient fuel [20,21].

Finally, the UK government is considering the *potential of waste as a resource* to produce energy and reduce the environmental impacts.

1.4.1 The role of waste

Municipal solid waste is considered as a renewable resource, no less than biomass or wind [22]. In the UK, the lowest cost opportunity for production of renewable waste lies with household waste [23] as the yearly average production of household waste per person amounts to around 415 kg. The UK and EU waste management strategies promote minimisation, recycling and reuse of waste, with disposal to landfill being considered as the least desirable option to prevent pollution of surface water, groundwater, soil and air, and reduce greenhouse gas emissions from the landfill site.

After recycling, there is still the potential to use almost half of the total waste produced from households for energy recovery. Therefore, thermal conversion of waste and biological treatments can be part of the answer to reduce carbon emissions, decarbonise the electricity grid and produce local renewable natural gas. A summary of the main disposal options is reported in Figure 1.4.2.

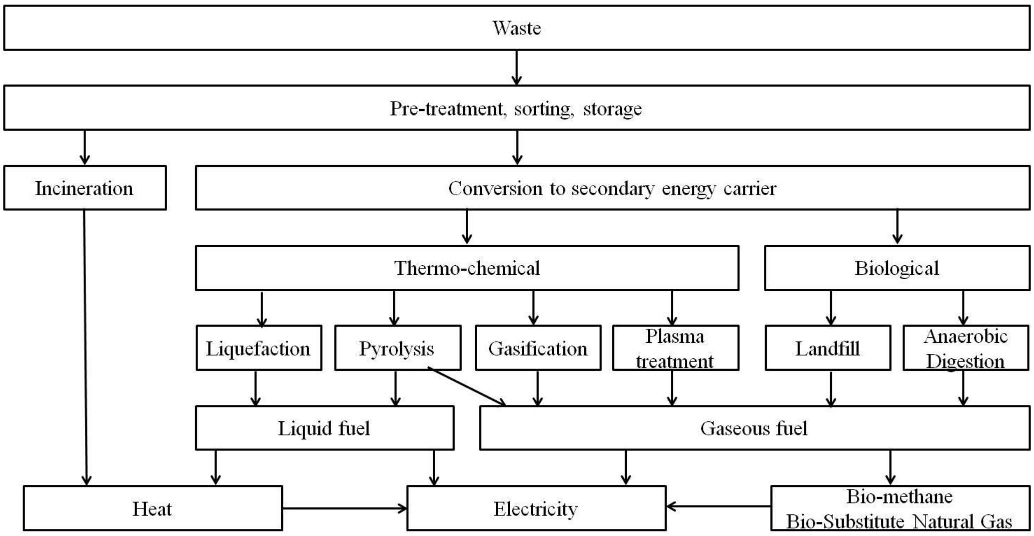

Figure 1.4.2. Residual waste disposal methods (adapted from [24]).

As explored more in details in chapter 2, the UK government is supporting the production of renewable electricity and methane from waste and biomass. This includes the production of heat and electricity from conventional direct combustion of waste in incineration plants that are becoming more technologically competitive thanks to the achievement of lower emissions and higher efficiency. At the same time, financial support is also given to processes that convert the waste into a secondary energy carrier. In particular, advanced waste-to-energy technologies (biological and thermal technologies), such as gasification for production of electricity and methane and anaerobic digestion for production of biomethane are receiving increasing attention. Various UK national competitions, including the Advanced Biofuels Demonstration Competition to support the production of UK-based advanced biofuels and the Energy Technology Institute Competition (ETI) to design a waste gasification demonstrator plant, represent the engagement to develop novel waste to energy technologies.

In this context, Advanced Plasma Power is a UK based company pioneer in the development of an advanced gasification technology able to deliver the production of electricity or Bio-Substitute Natural gas (Bio-SNG). Their technology includes the gasification of waste and the cleaning of the gas and ash fraction in a second stage of plasma converter. This allows for very high tar and contaminants conversion efficiency. This process is further analysed in chapters 2, 5 and 6.

1.5 Open questions

The UK has taken up the challenges on the energy trilemma and is pushing for big changes in the energy mix as discussed above. However, a number of open questions remain unsolved.

Despite the electrification of the grid, gas will still provide part of the UK energy demand for heat, transport and power generation (see chapter 2). The supply of natural gas from the UK and Norwegian shelf represents a proven but declining technology. The increased attention to alternative fossil natural gas resources and the aspirations to produce unconventional gas have to be balanced with the challenge to surmount technical and environmental barriers and secure public acceptance. Debated side effects of shale gas exploration include groundwater contamination, methane leakage from new wells and old wells, micro-seismic activity, air and noise pollution and visual changes to the landscape [15]. Some of these will be analysed in more details in chapter 2 and 4. It is also unclear whether UK shale gas exploitation could have the same effects on the energy markets as those reported for the US.

Liquefied natural gas also represents a possible alternative. However, its role compared to the role of shale gas needs to be analysed.

Fossil gas development of indigenous shale and imports could help meeting the energy requirements and the carbon emissions reduction at short and medium term. However, to achieve the long term increased reduction in emissions, cuts in fossil gas use and the development of renewable gas are deemed necessary.

Wind and solar energies, among others, will become the dominating renewable sources thanks to technological progress and political support. Widespread scientific literature is available to evaluate the environmental burdens and benefits of wind technologies [25–32] and photovoltaic [33–43]. However, these systems are characterized by a variable and uncertain generation pattern directly connected to meteorological conditions [44]. This is a key difference with other sources, such as conventional power plants, or energy from waste that can instead dispatch energy in a controlled manner. The development of wind and solar technologies is also reported to impose additional costs as the balance between demand and supply has to be ensured by other technologies [44,45]. Therefore, in this thesis particular emphasis is put on waste to deliver low carbon footprint and stable renewable energy.

Although landfill is the least preferred waste management option by the government, new engineered landfills, able to capture and utilized the landfill gas are also key to a more efficient waste management strategy. Both the new generation of incinerator plants and advanced conversion technologies currently receive support by the government. Uncertainties arise when old and new incineration plants are compared to advanced thermochemical processes as those are at early stage of development.

Currently, electricity is the preferred output of thermochemical conversion technologies when waste is used to produce energy. However, residual waste can also be used to generate Bio-Substitute Natural gas (Bio-SNG) in advanced waste to energy technologies, hence raising the question about the best options for recovery of energy.

Bio-methane can also be produced through biological technologies, such as Anaerobic Digestion. Importantly, renewable gas requires little or no new infrastructure to replace fossil natural gas, and can be injected into the existing pipeline network. The feasibility of biomethane production from anaerobic digestion of waste has already been demonstrated whilst the production of Bio-SNG from advanced waste gasification is currently under development. The environmental benefits of these two technologies for renewable methane production still need to be evaluated and compared.

1.6 Methodologies to balance the energy sustainability

Sustainable development is based on three aspects: environment, economic and social aspects [46]. In order to be able to fairly compare alternative choices on the basis of their sustainability, these three key aspects have to be assessed separately before they can be integrated in a comprehensive approach.

Hence, assessment methods able to reliably distinguish between comparing options are needed. In this thesis the so called ‘Life Cycle Thinking’ approach has been used. This is based on a system approach which enables quantifying the environmental, economic and social sustainability of the studied product/service, from raw material extraction, production, use and recycling or disposal.

The environmental methodology has been framed in an internationally standardised tool further analysed in chapter 3: the Life Cycle Assessment (LCA).

The economic aspects are framed in the Life Cycle Costing Methodology (LCC); this goes beyond the purely economic analysis because it takes into account the most up-to-date regulations in terms of environmental impacts to quantify the costs and benefits derived from the studied activity. The economic feasibility of the studied activity in the context of a reduced carbon economy is assessed.

Finally, the Social Life Cycle Assessment (SLCA) is the least developed and well-known method. This is used to assess the social impacts and acceptance of the studied activity, allowing for the understanding of what sort of social and socio-economic effects the targeted activity has on many diverse fields, such as employment, communities and gender opportunities.

The outcomes of each methodology, and then the integrated analysis can be used by decision makers to develop the most up-to-date legislations.

1.7 Aims and objectives of the thesis

This thesis analyses some of the challenges related to the development of the future UK energy system and adopts the environmental methodology, the LCA, to investigate the environmental impacts of key developing energy technologies currently supported by the government.

The aim of this work is to build an integrated environmental framework to assess the impacts of shale gas, LNG and waste-to-energy technologies. The methodology developed can be used by decision makers and planners to identify the most promising energy supplies, in the context of an increased focus on low carbon technologies and the requirement for a stable and secure energy supply.

The developed framework based on LCA allows making a critical assessment of the environmental performance of the technologies analysed. The UK and its challenges have been chosen as a case study for the entire analysis, but the same approach can be applied to any other country. The main questions addressed in this research are as follows:

- How has the energy mix evolved through history in the UK? What are the characteristics of the current energy mix and the projections for the next 20 years for the electricity and gas supplies? What affects the energy shifts?
- What are the future challenges for fossil energy and what renewable sources should the UK explore and develop?
    - Is shale gas a valid environmental alternative in the UK when compared to the conventional natural gas supply?
    - Is LNG a clean fuel for energy supply?
    - How does LNG compare with shale gas from an environmental perspective?

  - Electricity from waste: how do advanced thermal treatments of waste compare against conventional waste treatment processes (incineration and landfill)? What type of waste feedstock determines the lowest environmental impacts when treated in advanced thermal processes?
  - Methane from waste: how do biological and thermal processes compare?
  - Electricity from waste vs. methane from waste: which alternative prevails as the energy mix evolves?

The systems under study are expanded to consider the avoided activities related to the production of valuable products. The analysis includes transport, material, energy and water production and use and waste reprocessing, recycling and ultimately disposal. A transparent and detailed hot spot analysis of the main technologies is also reported. Despite the carbon footprint being the key environmental aspect currently dominating public concern [47–50], this thesis also considers wider environmental impacts such as acidification, water issues, toxicities, ozone depletion, etc. The adopted approach gives a temporal depth to political choices and highlights correlations between the chosen strategies.

A secondary aim of this thesis is to clearly report modelling data in a way that all models are reproducible. This point should be the basis of any scientific work but especially in the field of environmental impact assessments, many studies do not clearly report the basic assumptions on which the LCA models are built, limiting the reproducibility of the results.

This work represents the first step towards the sustainability trilemma because the framework developed in this thesis constitutes the basic structure on which future LCC and SLCA analysis can rely.

1.8 Thesis outline

This thesis consists of seven chapters. The first chapter introduces the problem of sustainable development particularly applied to the energy sector; the rationale behind the fundamental research questions addressed in this thesis is identified and the objectives of the work are stated. Chapter 2 briefly identifies the changes that have occurred through the history of the UK energy mix to modern times and then it analyses the current energy mix. The environmental legislations and the policies to ensure energy security are reviewed together with developing energy technologies. The challenges for the energy sector are fully analysed and projections of the future UK energy mix are also explored. In chapter 3 the methodology applied in this thesis is critically reviewed.

The phases which constitute the methodology are separately described and their limitations reported. The system boundaries of the entire analysis are drawn and insights on how the methodology has been applied in the thesis are given. Chapter 4 focuses on the supply of natural gas from fossil resources, investigating in particular, the environmental impacts of shale gas production for the UK situation and the life cycle of LNG supply to the UK. The two types of energy are critically compared. Sensitivity analyses are also performed for the majority of the scenarios analysed. Chapter 5 focuses on the analysis of the dual stage advanced waste to energy technology (gasification and plasma) developed by Advanced Plasma Power (APP) for the production of electricity (they sponsored this research and the author of the thesis has been working part-time for them since the beginning of the project). A hot spot analysis is reported to identify the most polluting part of the process and the environmental impacts of this technology are compared to other advanced dual stage technologies for waste treatment and conventional waste management options. Chapter 6 follows the structure of chapter 5 but concentrates on the environmental impacts of Bio-SNG production from the APP waste gasification process. Then, this technology is compared to biological processes for methane production from waste. Finally, chapter 7 sums up the results and presents the main conclusions and recommendations drawn from the entire thesis. Future research pathways are also proposed.

Chapter 2. **Energy sources and technologies: development of the UK energy mix**

This chapter firstly reports a brief historical account on the evolution of the energy mix in the UK until the present time. Then, this review focuses on the key legislations that regulate energy production and climate change and analyses the projected UK energy mix over the next 20 years. In this context, alternative energy sources and technologies, such as shale gas, Liquefied Natural gas (LNG) and energy from waste are discussed within the dual perspective of energy security and sustainability. The main conclusions of this chapter and the refined aims of the thesis are then drawn.

2.1 Energy mixes in the UK history

The evolution of energy sources and technologies depends on (and interacts with) economic, social, demographic, technological activities and resource constraints [51]. The changes to the energy mix strongly influence the quality of life and environment, in the past, but also in the future. The UK is the case study for the analysis reported in this thesis.

The medieval and modern history of energy sources and technologies in the UK is characterized by three periods [51]. In the first period, up to the end of the 18th century, human and animal power was replaced by water and wind power, whereas, wood based products for heating were displaced by coal. During the second period, from the beginning of the 19th century, to the beginning of the 20th century, power and heat were mainly sourced from coal whereas in the third period (from mid of the 20th century till now) the coal use declined and increasingly diversified sources (such as, petroleum, natural gas, nuclear, renewables) have provided heat and power.

2.1.1 First period

Power

In the 12th and 13th centuries, the British GDP and population significantly increased. This led to the rise in demand for farm and manufactured products and accordingly to the rise of power requirements for agriculture and industry (that however remained on a small scale). During this period, power was mainly produced using human and animal (oxen) strength; the demand of animal power for agriculture and other industrial activities, such as mills and transportation, rose significantly. This led to a shift in energy sources whereby oxen were largely substituted by horses. Oxen provided valuable inputs but they were quite slow whereas horses could exert double as many foot pounds per second as the oxen [52]. Despite the fact that they needed better quality food and could not work on heavy clay soils, the use of horses for power requirements increased by more than 65% from 1086 to 1574 [51]. The demand for horses in Britain grew even more during the 16th century because of the expanding trade network and the growing military pretensions.

The increasing requirements for horses increased the horse prices and this led to a subsequent shift of technology when in the late Middle Ages power started also to be sourced by falling water and wind, especially in mills [53]. These technologies had already been established in the Roman-Britain but they were only used during the late 12th century in industrial activities, especially where the product demand was sufficiently large to offset the initial investment and marginal costs. Small scale wind and water mills spread relatively rapidly until 1348 in many regions of Britain. The bubonic plague that severely hit Britain in 1348 and 1349, subsequently determined a decrease of power sourced from water and wind. Only by the end of the 15th century, did the economic activity start growing again together with energy use, determining increased requirements for wind and water power in mills of up to 50 kW capacity.

Heat

Medieval Britain was mainly an agriculture-based society and the heat requirements during this time were quite limited [54]. The heat production relied on biomass, such as wood, charcoal, peat and dung. The heat requirements started to increase during the population growth of the late middle ages in 1270 [55]; the demand exceeded the availability of local wood causing an increase its price [56]. This led to coal being increasingly used as direct heating fuel in households and industries. Many coal mines were based in North-England and Scotland and hence, in the nearby region, coal was easily accessible at a competitive price. The development of a coal trade route towards South of England meant that coal could rapidly become the main heating fuel also in other regions of Britain.

After the recession of 1357-1358, the consumption of heating fuel for households and industries remained low until the mid-15th century when the population started growing again. Wood requirements continued increasing through the 16th and 17th centuries also thanks to the expansion of the Royal Navy, the trading routes and the Great fire of 1666 in London that required a massive rebuilding. The increasing demand for wood determined a massive national wood shortage crisis between the mid-sixteenth and mid seventeenth century; as the price of coal remained lower than that of charcoal and wood, a rapid expansion in coal production followed.

During the 18th century coal continued to be the first energy source for heat production in domestic and industrial applications, including ferrous and non-ferrous metals production, construction industry and food and drink production [57]. In particular, the increasing use of coal in the iron industry was associated to the higher temperature required by the smelting iron process developed by Abraham Darby in 1709 [58]: higher temperatures could not be attained using charcoal.

2.1.2 Second period

Power and Heat

Population, agriculture and industrial productions significantly rose through the 19th century thanks to the industrial revolutions. During this period, steam became the main source of power [59], gradually substituting water as a direct power production source [51], thanks to the introduction and increasing efficiency of the steam engine [60].

The diffusion of the steam engine in the mining industry, manufacturing industry, agriculture and transport, including railways and ships, made the coal industry one of the most important sectors of the economy [53]. As in the previous century, coal continued to be the primary source for heat production in domestic [51] and industrial applications reaching its peak in 1913.

2.1.3 Third period

Power

Throughout the 20th century new energy sources and technologies rapidly developed except during the period of the First World War and the following 1930 economic depression. Cheaper prices of new energy sources and the diffusion of new technologies catalysed energy shifts towards an increased diversity.

Between the end of the 19th century and the beginning of the 20th century, coal started being used to produce gas for lighting purpose as this was less time consuming and more practical than direct use.

At the same time the electricity market started growing thanks to the development of steam turbines. Economic activities through the 20th century became electricity intensive [61] using a wide range of energy sources and technologies. Petroleum was increasingly used in transport thanks to the development of the internal combustion engine: road vehicles could now compete with rail transport.

Coal held its importance for electricity production until the 1960s despite the slow diffusion in the UK of efficient mining techniques between the two world wars and its rising prices. Since then, electricity has increasingly been produced using other sources. In the UK, in 1945 coal generated more than 95% of the electricity supply whereas in 1960 the share of coal in the electricity supply fell to 80%, the rest being provided by oil. By the 1970s, nuclear power started developing determining a further drop in the share of electricity produced from coal. In the 1990s, coal use in electricity production fell from 65% to 43% as nuclear and natural gas increased their share [51].

Heat

In 1913 the use of coal as heating fuel in industry started declining and it was replaced by petroleum. In the early 1970s also natural gas started penetrating the UK heating market. The shift was due to cost savings, and first regulations on emissions and cleaner energy sources.

During the second half of the 20th century, the domestic consumption of coal for heating declined from 42.5 million tons in 1960 to under 15 million tons in 1996, while being substituted mainly by natural gas, and nowadays also by Bio-SNG. Figure 2.1.1 shows the evolution of the fuel types used in household from 1970 to 2009 [62]. By 2010, 85% of homes in Great Britain used gas for heating, 9% used electricity, 4% used oil and over

2% used solid fuels (coal, wood etc.), LPG and district heat ('other' sources in Figure 2.1.1).

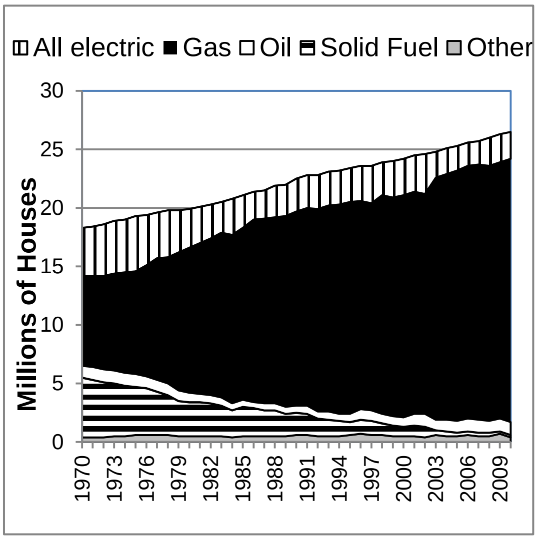

Figure 2.1.1. House fuel types between 1970 and 2010. 'Other' includes solid fuels (coal, wood etc.), LPG and district heat.

2.2 Present UK energy mix

Power

Figure 2.2.1 shows the UK electricity mix in 2014 [63]. Coal provided ~35% of the total electricity generation with the total demand for coal being 60.1 million tonnes. Gas accounted for ~23%, of the total electricity generation. Nuclear has steadily increased during the last 30 years, contributing for 20% of the total electricity production in 2014. The 20% of renewable supplies included offshore and onshore wind, hydro, solar, and also electricity from biomass and waste [64]. Defra [63] reported that almost 34% of the electricity demand was due to households, 57% to industrial and commercial, the remaining part being exported to Ireland and being losses [62].

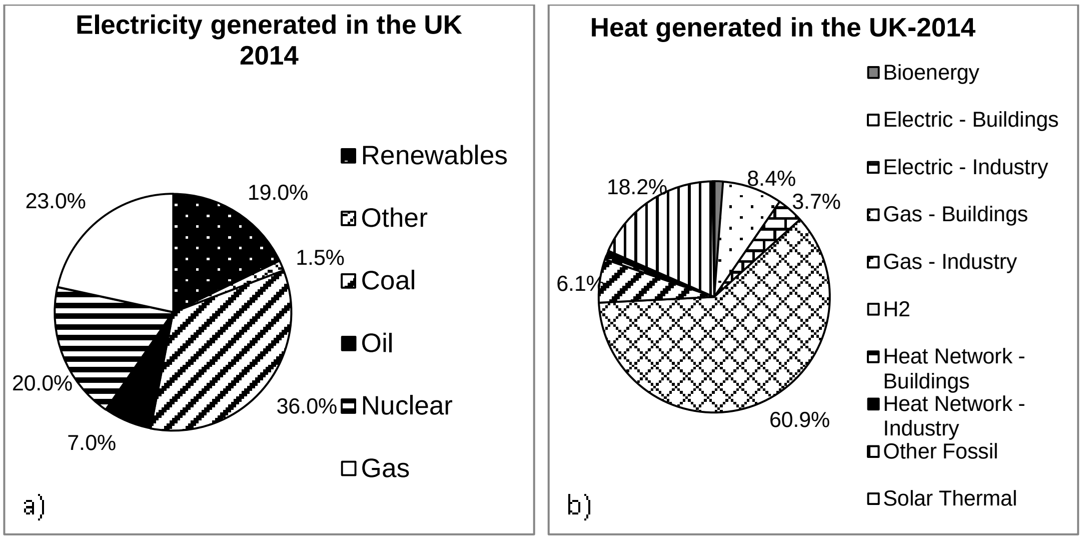

Figure 2.2.1. Energy generated in the UK in 2014 as a) electricity and b) heat.

Heat

In 2014, heat accounted for a significant proportion of the total UK energy demand (nearly half of the total energy use in the UK was for heating purpose) and more than half of the total natural gas consumed in the UK was used to provide heat to households and industry (the rest being mainly used in power stations for electricity production) [65]. In fact, the residential sector is the largest source of natural gas demand for heating [62].
As reported in Figure 2.2.1, natural gas supplied almost 60% of the total heat supply to buildings and 6% to industry; electricity supplied 11% of heat to industry and buildings. The remaining 19% of heat was supplied by other fossil fuels, such as oil and coal.
Table 2.2.1 shows the total domestic energy consumption identifying the different energy sources by end use in 1990-1991, 2000-2001 and 2013-2014. The use of natural gas for domestic space heating has kept constant since 2000; it represents now 79% of the total energy use for heating (up from the value of 75% in 1990). 2.73% of the total energy used for space heating was sourced from bio-energy in 2013-2014, a significantly higher proportion than 1990 and 2000 (only 0.8% of the energy used for space heating was sourced from bio-energy at that time).

[Thousand tonnes of oil equivalent-ktoe]	**Space heating**	**Hot water**	**Cooking**	**Lighting and Appliances**	**Total**
	1990-1991				
Solid fuel	2764.3	1390.4	14.3	0.0	4169.0
Gas	18484.1	6548.7	800.8	1.5	25835.0
Electricity	1100.8	818.9	682.8	5463.4	8066.0
Oil	2063.0	406.0	11.0	0.0	2480.0
Heat sold	0.0	0.0	0.0	0.0	0.0
Bio-energy and Waste	205.5	0.0	0.0	0.0	205.5
Total	24617.7	9164.0	1508.9	5464.9	40755.5
	2000-2001				
Solid fuel	1624.4	278.3	5.5	0.0	1908.2
Gas	24846.9	6262.7	694.4	2.5	31806.4
Electricity	1664.2	1133.0	615.6	6203.9	9616.7
Oil	2678.6	556.1	4.7	0.0	3239.4
Heat sold	44.4	0.0	0.0	0.0	44.4
Bio-energy and Waste	236.0	0.0	0.0	0.0	236.0
Total	31094.5	8230.0	1320.2	6206.5	46851.2
	2013-2014				
[Thousand tonnes of oil equivalent-ktoe]	Space heating	Water	Cooking	Lighting and Appliances	Total
Solid fuel	667.0	45.0	0.0	0.0	712.0
Gas	22865.0	6139.0	617.0	0.0	29622.0
Electricity	2148.0	653.0	490.0	6464.0	9755.0
Oil	2314.0	454.0	0.0	0.0	2769.0
Heat sold	52.0	0.0	0.0	0.0	52.0
Bio-energy and Waste	682.0	203.0	0.0	0.0	884.0
Total	28728.0	7494.0	1108.0	6464.0	43794.0

Table 2.2.1. Domestic energy consumption by end use and fuel [66].

The next paragraphs discuss the main drivers that will contribute to the development of the future energy mix in the UK.

2.3 Drivers for energy transitions

The nature of the new energy technology together with the abundance of supply, historical context, legislation support, environmental pollution, co-evolution of technologies, energy prices and consumption strongly contribute to the successful or unsuccessful uptake of the shifting energy mix [53]. New energy sources and technologies initially develop in niche markets and then, when the energy supplied by them becomes sufficiently cheap, the transition unfolds [67]. In some cases, the energy transition is created and engineered by governments [53].

On the one hand, the UK energy mix is shifting towards cleaner energy sources and technologies to comply with newer regulations for a cleaner environment; on the other hand, it is shifting towards a more affordable and more stable energy supply.

2.3.1 Climate Change: policy overview

Intergovernmental Panel on Climate Change (IPCC) has advised limiting the temperature rise well below 2°C [68] compared to pre-industrial levels to limit the threats [69] of global warming (the effects of global warming on human health are shown in Figure 2.3.1). The challenge of climate change has been addressed in a number of consecutive legislations and targets on greenhouse gas level reductions at international, European and UK level.

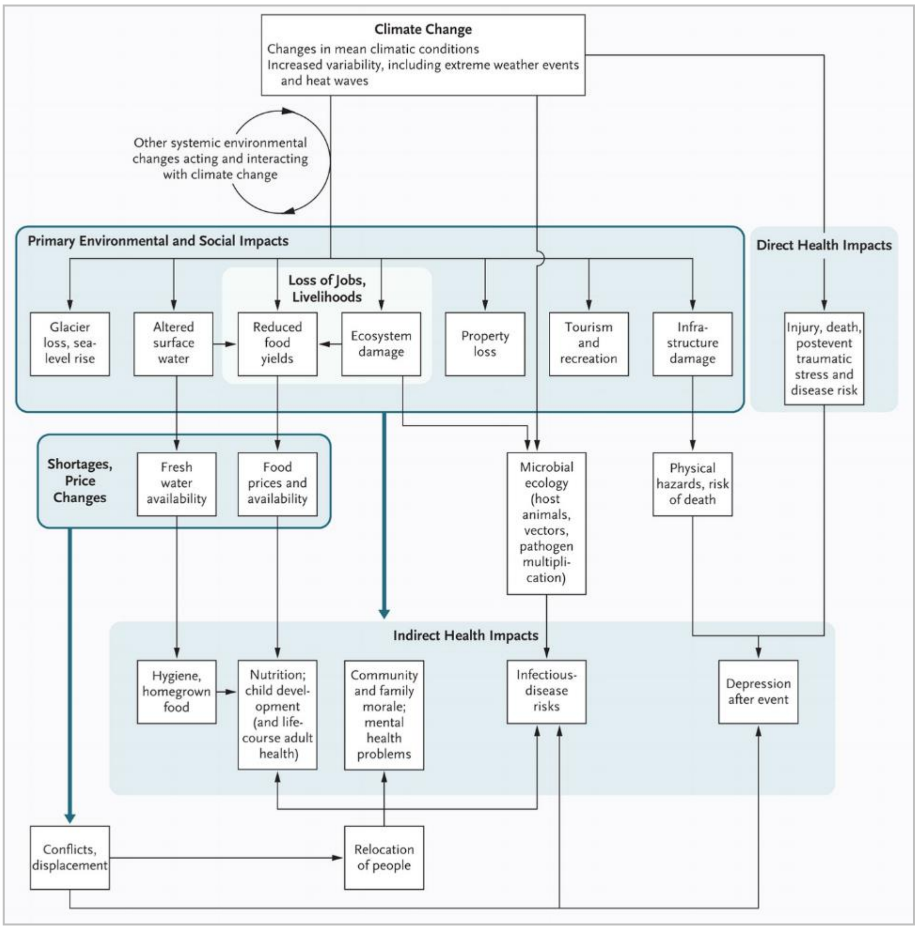

Figure 2.3.1. Pathways through which Climate Change influences human health (taken from [69]).

The legislations put in place strongly influence the evolution of the current and future make-up of the energy mix in the UK.

- Most notably at international level, the Kyoto Protocol was adopted in 1997. Different industrialised countries, including the UK, established a collective average greenhouse gas emission (including emissions of CO_2, CH_4, N_2O, etc.) reduction of 5.2% compared to the year 1990 (base year) over the first commitment period (started in 2008 and ended in 2012). To this end, the EU made a commitment to reduce greenhouse gas emissions by 8% by 2012; this target was redistributed across the different members. As part of this, the UK undertook the challenge to reduce total greenhouse gas emissions by 12.5% compared to the base year. For the UK, the base year referred to 779.9 MtCO2e (based on the 2006 UK inventory submission), thus the greenhouse gas emissions had to be below 3,412 $MtCO_2e$ over the five year period (2008-2012), or below 682.4 $MtCO_2e$ on average per year [70,71]. As shown in Figure 2.3.2, the UK met this target.
- A second commitment period, known as the Doha Amendments [72], was introduced in 2012 as an extension and amendment of the Kyoto Protocol until 2020. However, the second commitment period has yet to be formally agreed as the Amendment was not signed by the minimum number of states required for entry in force.

  Since the entry in force of the Kyoto protocol in 2005, a number of international conferences on Climate Change, including the Copenhagen Accord in 2009, the Durban Platform in 2011, and Lima in 2014, has taken place to negotiate the global greenhouse gas reduction for the period after 2020. During the last Climate Change Conference, held in Paris in 2015, a consensus between the 195 participant states was agreed [73] to reduce the carbon emissions for ‘a long-term goal of keeping the increase in global average temperature to well below 2°C above pre-industrial levels’. The agreement was deposited at the United Nations and opened for signature from the 22nd of April 2016. The agreement will enter into force provided that 55 countries, accounting for at least 55% of the global emissions sign and consent to be bound to the agreement. The EU has already prepared its intended contribution to this new agreement with a target to reduce emissions by at least 40% by 2030.
- The international agreements pushed a number of more local legislations to act against Climate Change. The energy sector was specifically addressed in the European legislations as this significantly contributes to total emissions level. In

2009, the EU set binding targets on climate and energy to be achieved by 2020. The following entered in force in 2013:

  - 20% cut in greenhouse gas emissions (from 1990 levels) [74];
  - 20% of EU energy from renewables and 10% share of renewables in the transport sector [75];
  - 20% improvement in energy efficiency for all members by 2020 [76].

To achieve the 20% target on greenhouse gas emissions, two sectors need to be regulated. i) EU emissions trading system (ETS) sectors that include large-scale facilities in the power and industry sectors, as well as the aviation sector, have to cut their emissions by 21% (compared to 2005). ii) non-ETS sectors that include housing, agriculture, waste, and transport, excluding aviation, have to cut their emissions by 10% (compared to 2005). The Department of Energy & Climate Change (DECC) reports that in the 2013 the UK was on track to achieve emissions reduction and efficiency improvement [71]. However, there is some doubt on whether the UK will meet the overall 2020 target [75] of energy consumption coming from renewable sources.

- In 2014, a new framework that builds on the 2020 climate and energy package was proposed by the European Union. Three new targets are set to be achieved by 2030 [77]:
  - At least 40% cuts in greenhouse gas emissions (from 1990 levels) [78];
  - At least 27% share for renewable energy [78];
  - At least 27% improvement in energy efficiency [79].

To achieve the 40% cut in greenhouse gas emissions, i) the EU emissions trading system (ETS) sectors would have to cut emissions by 43% (compared to 2005); ii) the non-ETS sectors would need to cut emissions by 30% (compared to 2005).

In addition to the legislations reported above, to limit atmospheric warming to below 2°C, the European Commission in its roadmap for 2050 [80] considered cost-efficient pathways to reach the reduction of 80% in greenhouse gas emissions (compared to the 1990 level) with an interim target not yet endorsed of 60% reduction by 2040.

- The UK has strengthened its commitment to tackle climate change under the Kyoto Protocol [81] with the Climate Change Act [82], which was passed in 2008. The UK government has committed to cut the carbon emissions by at least 34% by 2020 and at least 80% by 2050, compared to a 1990 baseline [83]. The Act also establishes that the Committee on Climate Change (CCC) advises the UK Parliament to set five year interim carbon budgets to help meet these targets (Table 2.3.1). The Climate Change Act reflects the UK contribution to cut the global carbon emissions of 50% by 2050 (compared to the 1990 level), which corresponds to an increase in temperature of 2°C by 2100.

	Budget 1 (2008-2012) Legislated	**Budget 2 (2013-2017) Legislated**	**Budget 3 (2018) Legislated**	**Budget 4 (2023-2027) Legislated**	**Budget 5 (2028-2032) To be legislated by June 2016**	**Climate Change Act (2050)**
Maximum total emissions during the indicated period $MtCO_2e$[1]	3018	2782	2544	1950		
Equivalent average annual emissions $MtCO_2e$	603.6	556.4	508.8	390		
Reduction below 1990 levels	22%	28%	34%	50%	57%	80%

Table 2.3.1. Carbon budgets and Climate Change Act. Adapted from [83].

The UK's progress towards meeting the carbon budgets is annually reported by the Committee on Climate Change. The last report [84] published in June 2015 indicates that the UK emissions level are currently more than 36% below the 1990 levels, an 8% decrease in emissions compared to the previous year (Figure 2.3.2). Therefore, the emissions are below the average annual level of the second carbon budget.

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[1] Like the Kyoto Protocol, the Act uses a base year which is comprised of 1990 for carbon dioxide, methane and nitrous oxide, and 1995 for fluorinated compounds. However, this base year figure differs from that used for reporting against the Kyoto Protocol in that the baseline is revised each year to incorporate revisions made to 1990 and 1995 emissions data, whereas the Kyoto Protocol base year emissions are fixed.

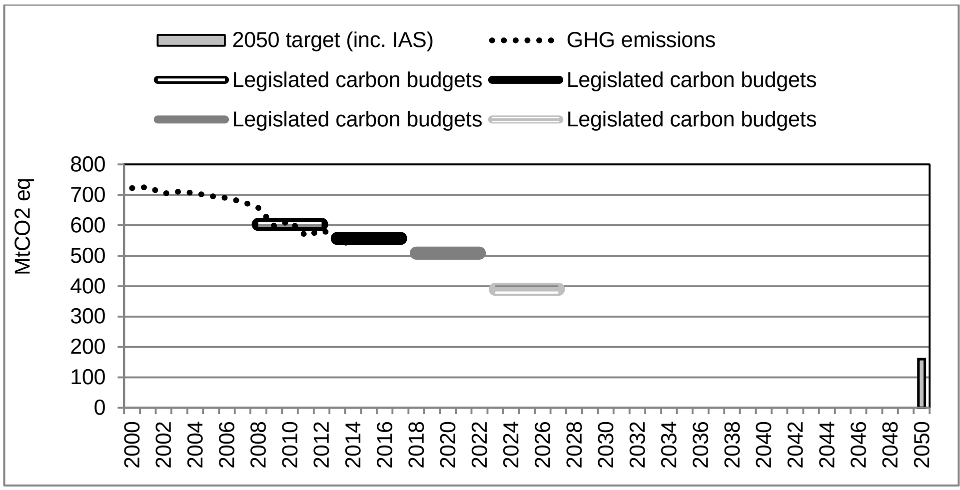


Figure 2.3.2. UK GHG emissions against legislated budgets and 2050 target [84].

2.3.2 Energy security

As identified in the White Paper on Energy published by the UK government [85], the security of energy supply is a major issue that the UK faces alongside the environmental targets. The declining reserves of gas in the UK Continental Shelf (UKCS, North Sea) [86] have pushed the UK to significantly improve the import infrastructures and to look for other indigenous gas reserves [87], as gas will continue to play a crucial role in the UK energy mix in the future [62].

The UK still partially relies on the indigenous resources of the North Sea, but imports from Norway, Belgium, the Netherlands and Qatar as Liquefied Natural Gas (LNG) are increasing. Development of new Liquefied Natural Gas (LNG) infrastructure, such as the LNG receiving terminal at South Hook is a current major project. Following the shale revolution in the US, shale gas is also being explored in the UK as a potential indigenous resource of gas to improve greater energy security [88].

Both LNG and shale gas are reported to potentially constitute a significant part of the UK gas supply within the next 20 years [62]. However, this raises the question on their environmental impact in the short and long term. This is analysed in chapter 4.

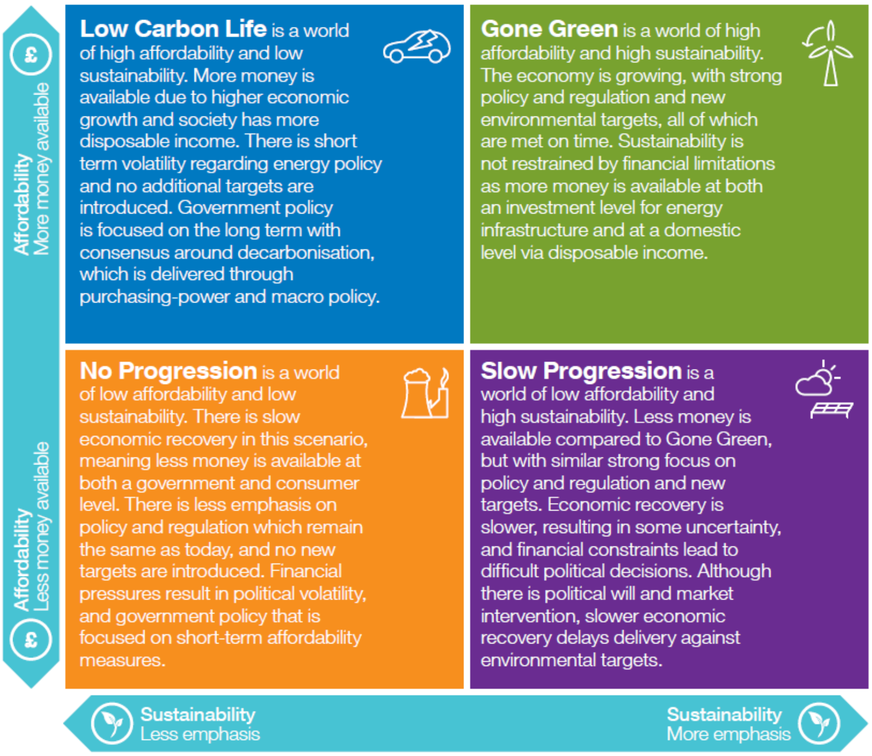
2.4 Development of future energy mixes in the UK

In the previous sections, the past UK energy mix has been analysed and the current power and natural gas mix have been discussed. The main drivers for energy changes have been explored, hence, on these premises the development of the UK energy mix is analysed for the next future.

2.4.1 Future energy scenarios

Different sources [62,89] report on future energy scenarios. However, the analysis in this thesis is based on the scenarios reported by the National Grid [62] as this reference is the most up to date and comprehensive.

The National Grid [62] analyses the possible evolution of the UK energy till 2035 and 2050 in some cases, identifying the four possible scenarios reported in Figure 2.4.1: i) Gone Green; ii) Slow Progression; iii) No Progression; iv) Low Carbon Life. 'These scenarios reflect different levels of energy sustainability, affordability and security of supply and cover the developments in electricity generation and demand, and gas supply and demand. These scenarios are used as a reference point for a range of modelling activities, including network analysis, to identify potential gas and electricity network investment requirements in the future' [62].



Low Carbon Life is a world of high affordability and low sustainability. More money is available due to higher economic growth and society has more disposable income. There is short term volatility regarding energy policy and no additional targets are introduced. Government policy is focused on the long term with consensus around decarbonisation, which is delivered through purchasing-power and macro policy.

Gone Green is a world of high affordability and high sustainability. The economy is growing, with strong policy and regulation and new environmental targets, all of which are met on time. Sustainability is not restrained by financial limitations as more money is available at both an investment level for energy infrastructure and at a domestic level via disposable income.

No Progression is a world of low affordability and low sustainability. There is slow economic recovery in this scenario, meaning less money is available at both a government and consumer level. There is less emphasis on policy and regulation which remain the same as today, and no new targets are introduced. Financial pressures result in political volatility, and government policy that is focused on short-term affordability measures.

Slow Progression is a world of low affordability and high sustainability. Less money is available compared to Gone Green, but with similar strong focus on policy and regulation and new targets. Economic recovery is slower, resulting in some uncertainty, and financial constraints lead to difficult political decisions. Although there is political will and market intervention, slower economic recovery delays delivery against environmental targets.

Figure 2.4.1. UK future energy scenarios [62].

i) Gone Green is a 'future where more money is available, with strong policy and regulation and new environmental targets. The economy is growing, and environmental sustainability is not restrained by financial limitations as more money is available at both an investment level for energy infrastructure and at a domestic level via disposable income' [62].

ii) Slow Progression is a 'future where less money is available compared to Gone Green, but with similar strong policy and regulation and new targets.

Economic recovery is slower in this scenario than in Gone Green and Low Carbon Life, resulting in investor uncertainty. Financial constraints lead to difficult political decisions' [62].

iii) No Progression is a 'future where there is less money available and less emphasis on sustainability. There is slower economic recovery in this scenario, meaning less money is available at both a government and consumer level. Government policy and regulation remains the same as today, and no new targets are introduced' [62].

iv) Low Carbon Life is a 'future where more money is available and there is less emphasis on sustainability. There is higher economic growth. Society has more disposable income which results in the higher uptake of electric vehicles, and more renewable generation at a local level. Government policy is focused on the long term' [62].

2.4.2 Power demand

The total power demand includes the total amount of generation required for residential, commercial and industrials and also losses and exports to Ireland, see Figure 2.4.2. The residential power demand is reported [62] to account for almost a third of the total demand whereas the commercial and industrial demand is for the remaining two thirds.

In the Low Carbon Life scenario, both industrial and commercial and residential demands increase thanks to the assumption of higher customer affordability that determines more appliances, electric vehicles and light use.

In the Gone Green scenario, the residential demand decreases till 2025 because of a higher energy efficiency and replacement of old devices. After 2025, the residential power demands increase because of the increased power requirements for household heating. On the other hand, the industrial demand increases thanks to the assumed economic growth. The total trend, reported in Figure 2.4.2, follows the trend of the residential demand.

The residential demand in the Slow Progression scenario initially decreases as well, thanks to higher energy efficiency, and then keeps constant, whereas the industrial power demand decreases due to a reduced economic growth. The total trend, reported in Figure 2.4.2, almost follows the trend of the residential demand.

Finally, the total power demand in the No Progression scenario follows a similar trend to the Slow Progression trend but it is lower.

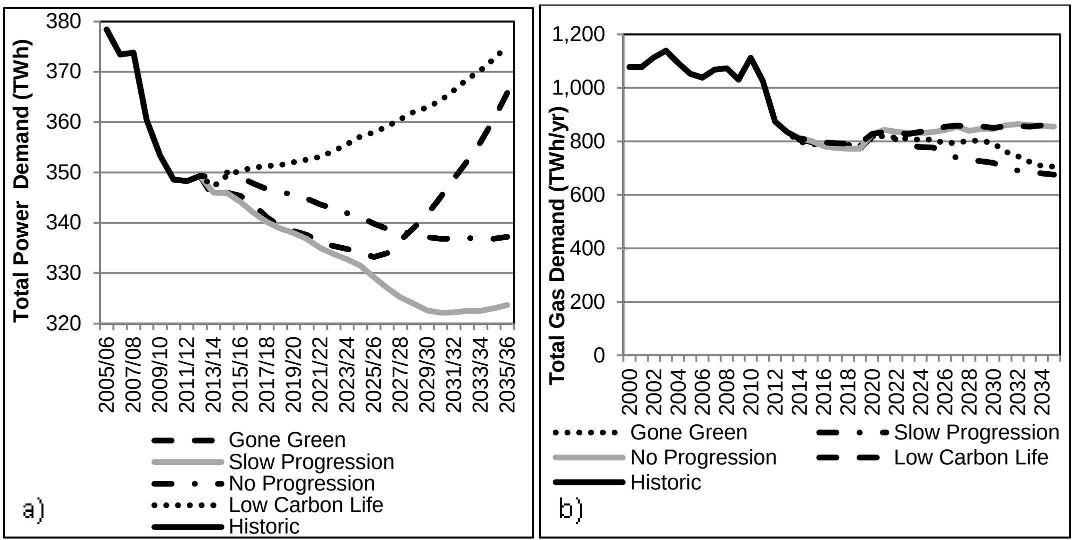


Figure 2.4.2. UK total annual a) power and b) natural gas demand [62].

2.4.3 Power supply

Figure 2.4.3 shows the electricity generation mixes and how the electricity sources may change in the future according to the four scenarios analysed by the National Grid [62].

Between 2013/2014 and 2018/2019 traditional sources and technologies still largely supply the bulk of the energy generation with renewable energy increasing. All four scenarios are quite similar within this time frame.

Between 2019/2020 and 2025/2026, the environmental legislation and targets are expected to influence the electricity mix; increasing volumes of renewable electricity and electricity from natural gas are projected to be supplied as old fossil fuel power plant are assumed to shut down.

The periods between 2026/2027 and 2035/2036 see an increase in nuclear, natural gas and renewables, such as biomass and wind; standard coal fired stations significantly decrease their production but the use of CCS for coal and natural gas fired station increases, depending on the scenario analysed.

In the Gone Green scenario, the supply mix shifts from traditional technologies and sources to a supply mix with increased use of renewables and low carbon technologies, including also use of wind, solar and biomass; hence, generation of electricity using coal and oil declines. New power stations are assumed to be fed with gas and this determines the increased electricity generation from gas. This scenario also sees an increased use of biomass that substitutes coal.

The Slow Progression scenario is very similar to the Gone Green scenario but renewables are deployed in a slightly smaller amount and at a slower rate.

In the No Progression scenario, cheaper energy technologies (traditional and renewables) dominate the electricity generation (renewables are deployed at a smaller rate than the Gone Green and Slow Progression scenarios). Natural gas-fired generation substantially increases from 2020 due to a reduced deployment of renewables and shutting down of coal-fired generators. Nuclear power generation is projected to remain stable from 2030 onward. CCS is not included in the electricity mix due to the less favourable economic background.

In the Low Carbon Life scenario, distributed generation grows fast with the deployment of mainly solar and wind. Old coal fired stations are replaced during the 2020s by natural gas fired station. From the 2030s, thanks to the favourable economic background and the strong de-carbonisation strategy, nuclear and CCS technologies are more widely used.

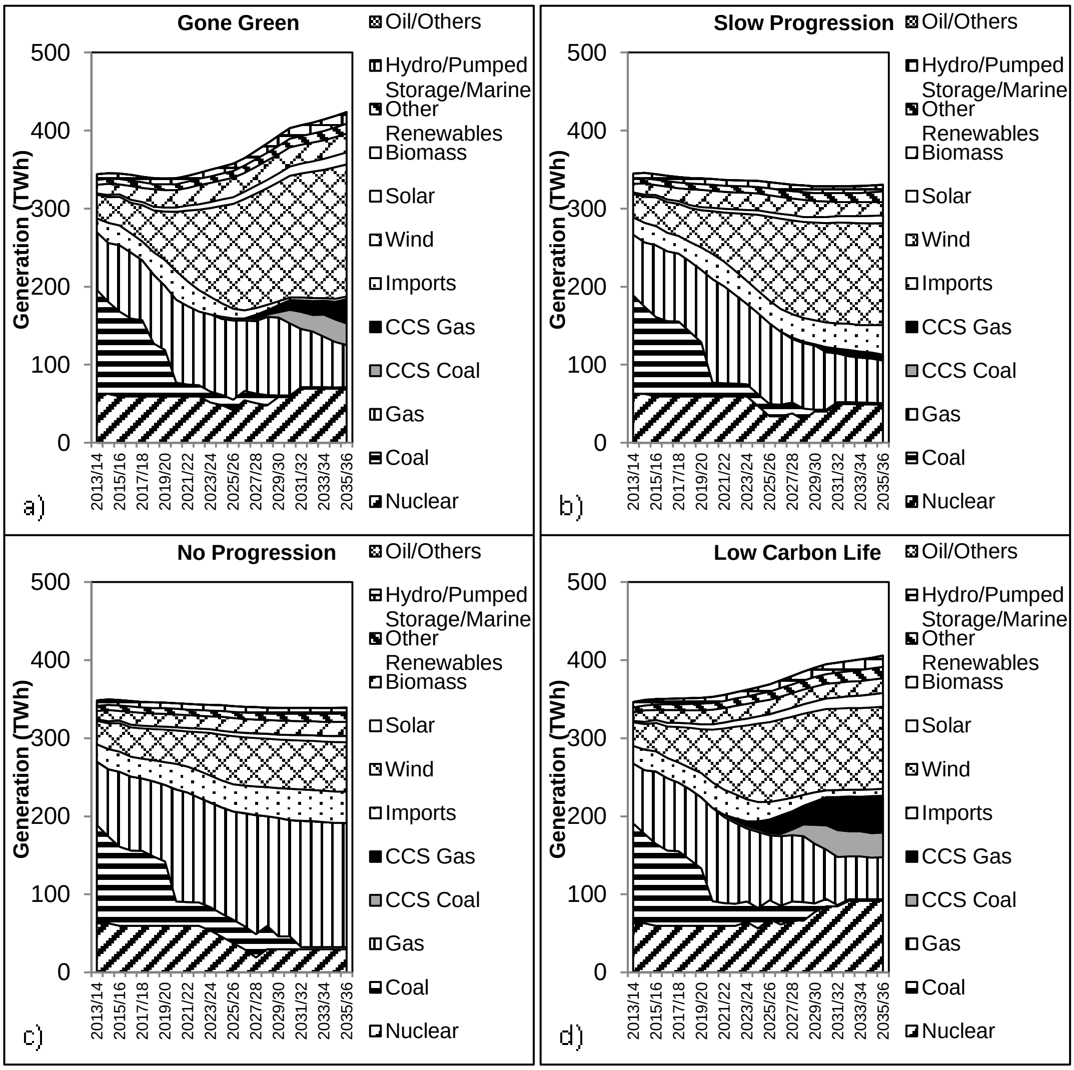


Figure 2.4.3. Future power supply in the UK according to four scenarios: a) Gone Green; b) Slow Progression; c) No Progression; d) Low Carbon Life [62].

Longer but less certain are the prevision of the UK electricity mix till 2050. This sees a substantial expansion of the nuclear production (up to 45-50% of the total electricity generation) together with higher production of electricity from gas, both CCS and no-CCS, depending on the scenario considered.

2.4.4 Natural gas demand

The natural gas demand/supply is analysed separately because of the significant projected uptake of new technologies, such as shale gas production and biomethane production from waste, that are still at the development stage.

Four main categories make up the UK gas demand [62]: 1) residential demand (the highest as it accounts for almost one third of the total demand); 2) industrial and commercial demand (accounting for a quarter of the total UK demand); 3) power generation demand (accounting for a quarter of the total UK demand); 4) export (that is only one sixth of the total demand).

Predictions of future total natural gas demand are reported in Figure 2.4.2. The Gone Green and the Slow Progression scenarios are quite similar as also the No Progression and the Low Carbon Life scenarios.

The residential demand in the Gone Green scenario is predicted to decline from 2030 thanks to the increased use of electricity for residential heating. The Slow Progression scenario follows the same trend although it is built according to a slower economic growth.

In the other scenarios, natural gas boilers are assumed to heat households, hence the residential demand does not significantly decline. The Low Carbon Life and the No Progression scenarios show the highest total gas demand and this is due to the increased natural gas demand for residential heating, power generation and exports to Ireland.

2.4.5 Natural gas supply

Figure 2.4.4 shows predictions of the future mix of the gas supply in the UK.

In the Gone Green scenario, the supplies of biomethane and shale gas are the highest. Both technologies are not fully developed yet in the UK but are expected to significantly expand in future and to contribute substantially to the total gas supply. The overall gas imports, including those from Norway, are moderate when compared to the other scenarios.

In the Slow Progression scenario, the total gas demand is shown to be quite low but imports are expected to be higher as the production of UK Continental Shelf (UKCS), shale and biomethane decreases. Gas imports from Norway and the continent as well as imports of LNG increase mainly during the second half of the 2020s, representing up to 70-75% of the total supply.

The gas demand in the No Progression scenario is the highest; natural gas is mainly used for heating but also for electricity production as low electricity carbon technologies are not expected to develop (due to the reduced economic growth). Low economic development determines also a reduced deployment of the UKCS, shale gas and biomethane; hence, gas is mainly imported from Norway and the continent as well as LNG (imports reach over 90% of the total supply).

In the Low Carbon Life scenario, the UKCS and shale gas are fully exploited and this decreases the amount of imports (LNG and imports from the continent and Norway).

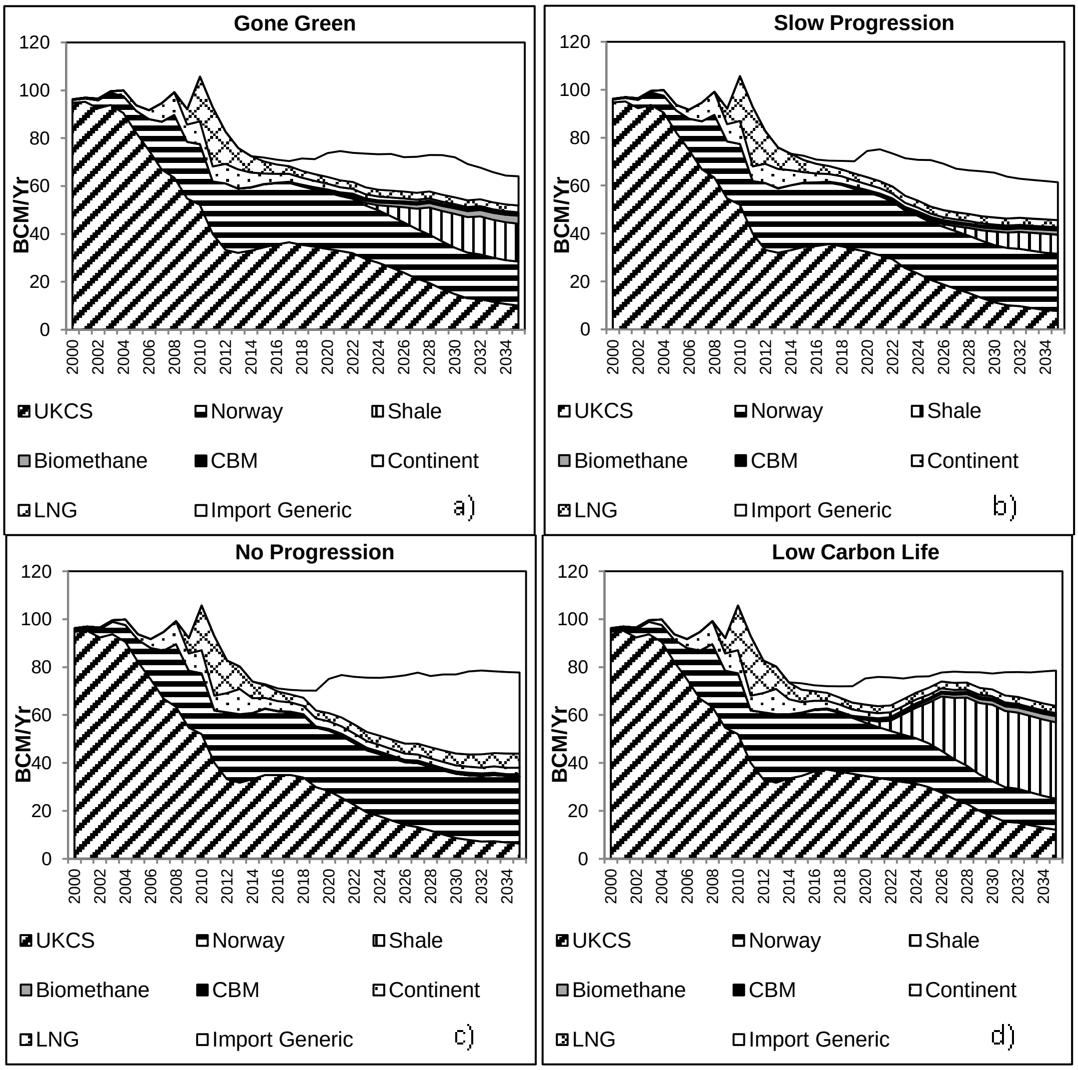


Figure 2.4.4. Future natural gas supply in the UK according to four scenarios: a) Gone Green; b) Slow Progression; c) No Progression; d) Low Carbon Life [62].

The main future sources of gas development are separately analysed in Figure 2.4.5.

UK Continental Shelf (UKCS) and Norwegian gas (Figure 2.4.5)

After quite a few years of declining, the UK continental shelf production has seen a slight increase in 2013-2014. This increase in production is expected to continue for a couple of years in all scenarios, before a stabilization of the production and then a decline. The decline is less strong for the Gone Green and Low Carbon Life scenarios.

The Norwegian imports of natural gas decrease in all scenarios and this, together with the decrease of the production from the UKCS, is the driver for increased production from alternative sources, included biomethane, LNG and shale.

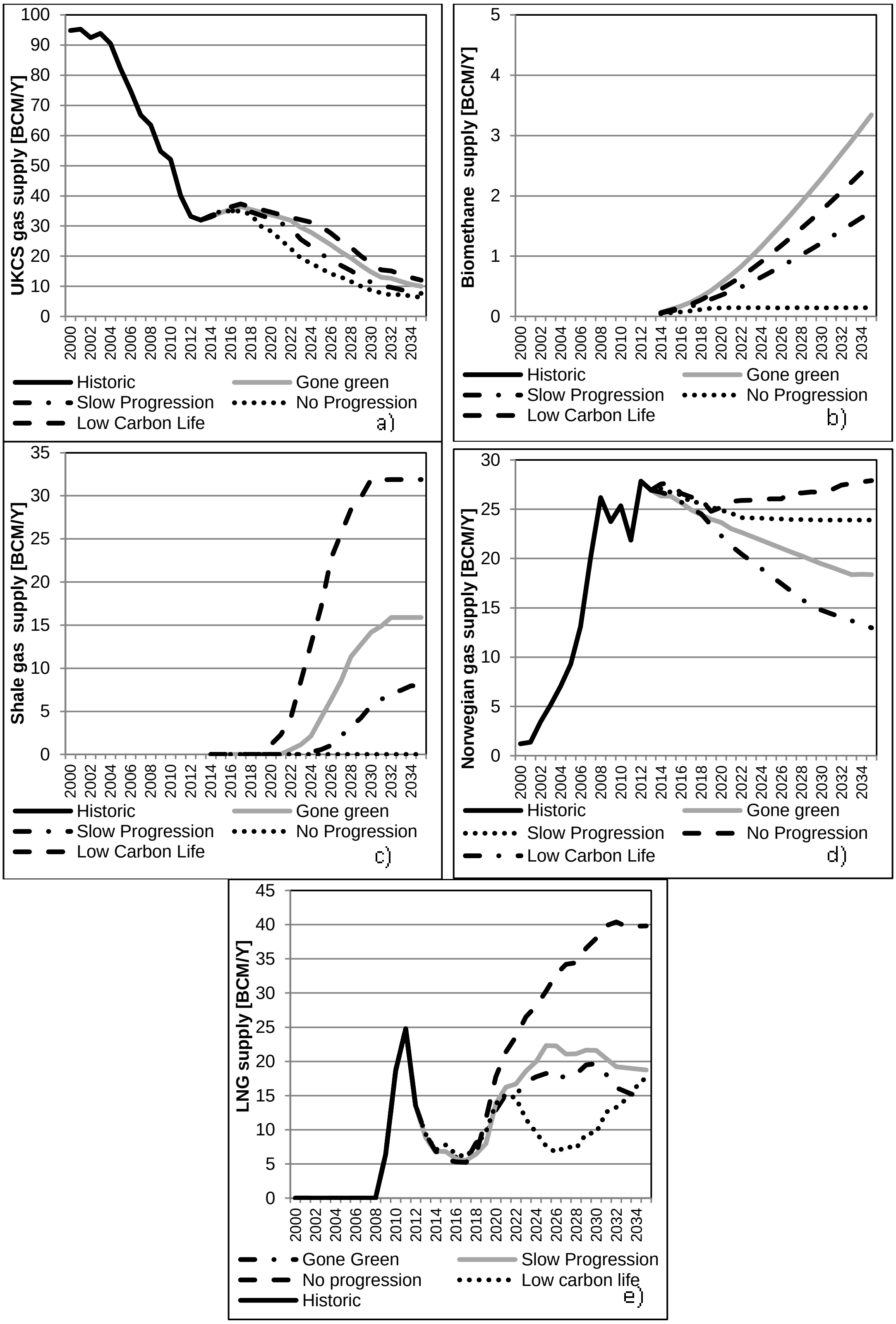


Figure 2.4.5. a) UKCS supplies. b) Biomethane supplies. c) Shale gas supplies. d) Norwegian gas supplies. e) LNG supplies. (Taken from National Grid estimates [62])

Biomethane (Figure 2.4.5)

The production of bio-methane is financially supported by the UK government within the Renewable Heat Incentives scheme as it contributes meeting the targets on renewable energy production.

The development of the bio-methane production for grid injection started in 2014 and will keep stable until 2018; depending on the scenarios, it will then significantly rise or keep constant until 2035.

The Gone Green scenario shows the highest production in biomethane thanks to an increased focus on sustainability supported by economic growth. In the Slow Progression scenario, the development is slower as the economic growth is but the government still supports sustainability. In the Low Carbon Life scenario there is less focus on sustainability but higher economic growth, whereas in No Progression both economic growth and focus on sustainability are low.

Shale gas (Figure 2.4.5)

The shale gas boom in the US has catalysed high interest in shale resources also in other parts of the world, including the UK. The UK owns some shale gas reserves; so far commercial shale gas production has not started, but exploration and testing are taking place.

A wide range of possibilities are analysed by the National Grid: in the No Progression scenario, shale gas does not successfully develop whereas it reaches the highest production in the Low Carbon Life scenario (as discussed in chapter 1, shale gas is seen as a clean energy fuel able to help meeting energy security and sustainability at short term). In the other two scenarios production of shale is foreseen and, in both cases, it is higher than the biomethane production.

LNG (Figure 2.4.5)

Under the Qatargas project UK has secured LNG supplies from Qatar, hence the volume of LNG in the gas grid mix is expected to increase [90]. However, the National Grid [62] does not attempt to make well defined projections on LNG and continental gas imports in the future scenarios on gas supply but refers to generic imports which ‘could be any mixture of the two’. Progressive Energy [91] has advised that the generic imports of natural gas will completely be supplied by LNG. Following this assumption, the data reported by the National Grid [62] have been re-elaborated and the results are reported in Figure 2.4.5.

The historic curve for the LNG supply shows a decline after the peak of 2011. This is due to the increased price of LNG, driven by increased demand in other countries (particularly in Japan, which continues to move away from nuclear energy), and the increase in pipeline imports into the UK, from Belgium and Norway, in 2013 [92].

From 2017, an increase of the LNG supply to the UK is shown in all scenarios with the highest values shown for the No Progression scenario. In this case, the projected economic recession does not allow for the development of low carbon options. The trend of the LNG imports is the same for all four scenarios until almost 2020; after 2020, the lowest LNG imports are shown for the scenarios with increased focus on sustainability.

2.4.6 Heat supply

The heat demand is not discussed explicitly as it is included in the power and natural gas demand.

The projection of the heat supply in the UK follows the trends reported in Figure 2.4.6. The UK National Grid [62] has reported the projections until 2050. A significant change is shown for the Gone Green, Low Carbon Life and Slow Progression scenarios as a result of the increasing uptake of the renewable technologies in order to meet the carbon targets. In these scenarios, direct use of natural gas for household heating is replaced by electric heating using low carbon intense electricity (chapter 1). Also the use of other fossil fuels decreases and the direct use of biomass for heating develops mainly in the Gone Green and Slow Progression scenarios. In the No Progression scenario, the amount of natural gas used for heating is constant because of a reduced economic growth and lower importance put on sustainability; at the same time the supply of electric heating does not increase.

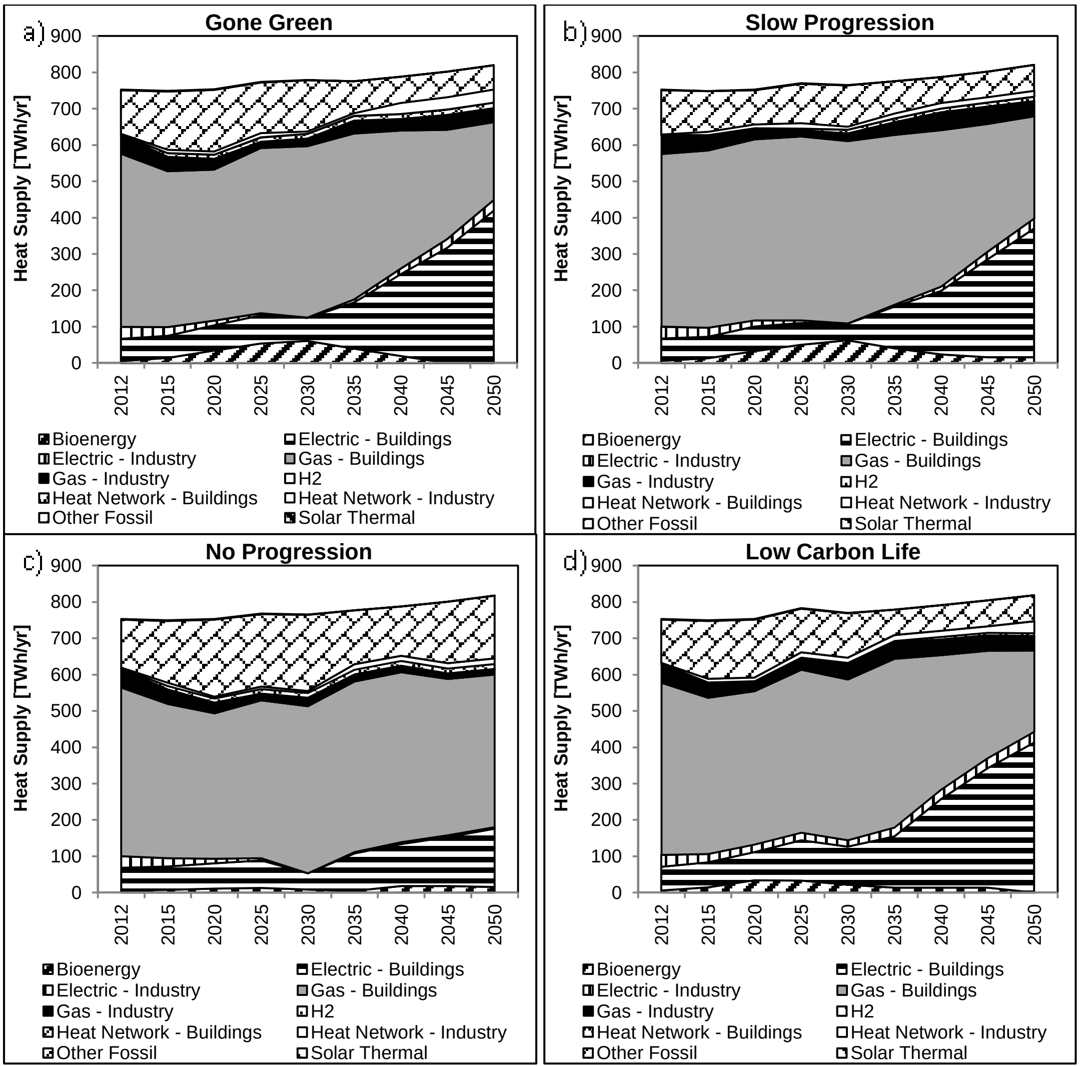


Figure 2.4.6. Future heat supply in the UK according to four scenarios [62]: a) Gone Green; b) Slow Progression; c) No Progression; d) Low Carbon Life.

2.5 Fossil natural gas: conventional and unconventional sources

Fossil fuels were formed from organic debris through millions of years. Decaying organic matter built up on the ground over times; it was compressed and buried by sediments and mud piling on the top of it and transformed in rocks. High temperature, pressure and time, transformed the organic matter trapped into the source rock into fossil fuels. The nature of the organic debris and the conditions differentiated the fossil fuels into natural gas, petroleum or coal. Higher pressure and temperature are possibly responsible for the preferential formation of natural gas over other fossil fuels [93].

Natural gas, produced in this way, is generally known as thermogenic methane and it is distinguished by the biogenic methane that is produced by living bacteria transforming organic matter present in the source rock. Once formed, natural gas tends to migrate upwards from the source rock, through pores and cracks, sometimes reaching the surface. Some migrating gas can be trapped and collected into geological porous formations capped by impermeable rocks. Hence, if the formation is large enough, a reservoir is formed.

The reservoirs are not continuous spaces under the ground, but are formed by a porous rock. Porosity and permeability (the measure of the rock's ability to allow gas flowing) determine the quality of the rock. If the reservoir rock is highly porous a huge amount of gas can be contained; if the pores are interconnected together, the reservoir rock has a high permeability. This is the point for the differentiation of conventional or unconventional gas.

Conventional natural gas is usually trapped in highly porous and permeable formations, (10^{-9} Darcy in unconventional shale formation versus 10^{-2} Darcy in conventional sandstones [94,95]) such as sandstones, where grains of rock are locked together and there is a lot of space between the particles [96,97]. In this case, the gas can be easily extracted using standard techniques. Conventional gas reservoirs can be either associated or non-associated. In the associated reservoirs, gas and petroleum are found at the same time, whereas the non-associated reservoirs do not contain oil.

The term 'unconventional gas' identifies three main types of natural gas sources [98]: shale gas, tight gas and coalbed methane. Coalbed methane is trapped in coal deposits and it is mostly adsorbed on the surface of the coal. Tight gas is usually trapped in the void space of a reservoir with a very low porosity and permeability; shale gas is trapped in a very low permeability source rock [99] (from which the gas did not migrate) and it is either adsorbed on the surface of the minerals and the organic matter or it fills the void porosity spaces [100,101].

There is no significant variation in the composition of unconventional and conventional natural gas. It is a combustible mixture of hydrocarbon gases [102], including primarily methane, ethane, propane, butane, and pentane. The non-hydrocarbon gases, contained in variable proportion, include N_2, CO_2, He, H_2S, water vapour and other sulfur compounds.

To extract conventional natural gas, first the reservoir rock is localized and the correct place to drill is identified, then vertical drilling takes place. If the drilled well comes in contact with the reservoir but it does not produce a marketable amount of gas, the production is ended and the well abandoned. Otherwise, if the well produces enough natural gas it is completed. The well hole is cased and cemented, and the well heads and pipelines are put in place in order to gather natural gas towards the processing plant. After well completion the production of natural gas begins. During the life of the well, if the production rate decreases, stimulation techniques or lifting equipment can improve the production rate. Workovers and unloading are also performed; workovers involve periodical cleaning, replacement of production pipes and stimulation techniques, whereas the unloading operations involve the clearing of the liquids that built up and may block the gas flow through the well.

Natural gas, when extracted from the reservoir, is not suitable for distribution and it needs to be treated in processing plants before storage and distribution to the consumers. Water vapour, solids, CO_2, sulfur compounds, Hg can cause corrosion and liquid slag formation and are therefore removed from natural gas before distribution. In particular, if the natural gas is sour (the concentration of H_2S in the gas is higher than 4 ppm at standard condition) the excess H_2S must be separated from the gas through a process called “sweetening” to avoid corrosion. Higher hydrocarbons are also separated as they are a valuable by-product and can be sold separately. Cleaned natural gas is then transported in several ways, such as in pipelines, as Liquefied Natural Gas (LNG) or as compressed gas, etc[103].

The transport of LNG is rapidly developing because it permits the transport of natural gas over longer distances that cannot be covered by pipeline distribution. Furthermore, thanks to its reduced volume, LNG can easily be stored for peak demand periods without the construction of additional pipelines [104].

For shale gas, similar extraction techniques and processing are used, but the process of hydraulic fracturing is required to increase the mobility of the gas in the source rock and improve productivity [105]. Tight gas is usually produced using a combination of hydraulic fracturing and acidizing techniques. The main focus of this thesis is on shale gas and the hydraulic fracturing process.

2.5.1 Shale gas

The first shale well was drilled in 1821 in Fredonia (New York, USA) with the gas used for town lighting [106]. However, the low productivity of shale wells delayed large scale development of shale gas until the 1970s and 1980s when the development of new technologies made the production of shale gas cost-effective [107]. During the 2000s, shale gas exploitation has seen a rapid growth: shale gas accounted only for 1.6 % of total US gas production in 2000, but by 2013 it had increased to 34 %. It is expected that nearly half of US production will be supplied by shale gas by 2035 [108].

The remarkable US growth of shale gas production and the associated decrease in US gas prices have catalysed an increasing interest in the exploration of possible shale gas resources in other areas of the world and some countries are considering the development of their own resources. The International Energy Agency has estimated that several European countries hold significant recoverable resources of shale gas [109]. None of them has currently started commercial production of shale gas, but exploration is taking place in the UK, Poland (where numerous wells were drilled), Germany, Romania, Denmark and Hungary [110].

At the same time, controversies have arisen over whether European shale gas exploitation could parallel that in the US [111,112]. Major concerns have been voiced about different European conditions compared to the US: different geology, higher population density, different laws governing land ownership and lack of relevant drilling expertise and infrastructure.

Hence, the picture of a possible new global market that includes the production of shale gas in different parts of the world is not clear. A potential shift of emissions between fossil and renewable energy sectors has also to be identified. Only limited information on the potential implications between shale gas development, global warming, policy and economics are available [113]. Concerns about the environmental burdens of drilling and production of shale gas coupled with a strong nuclear lobby have pushed some countries, such as France, to ban exploration and trials. In other countries, such as the UK, where exploration and trials have started, speculation on the risks and advantages of shale gas local production have broadly developed [114,115]. It is clear that the global economic debate on shale gas development needs to be rooted on solid knowledge about its environmental impacts and it is of major importance to identify the possible sources of pollution. A rigorous assessment in the European context requires contextualizing studies on shale gas extraction to EU conditions (policies, geological shale formation, technologies used, etc.) as done in chapter 4.

2.5.1.1 The shale gas extraction process and its environmental challenges

As previously explained, geophysical locations and the extraction process differentiate conventional and unconventional natural gas [116]. Conventional gas is located in permeable rocks and is easily extracted after drilling because it has been expelled from source rock and has migrated into adjacent reservoir rocks [117]. By contrast, unconventional gas is trapped in insufficiently permeable formations, such as shale, tight sands and coal beds; for shale gas horizontal drilling and hydraulic fracturing of the rock are essential to release the gas. The term “hydraulic fracturing” is often wrongly used to refer to the entire process of shale gas extraction and production which in reality involves the following operations: site exploration and preparation, road and well pad construction, vertical and then horizontal drilling, well casing, perforation, hydraulic fracturing, completion, production, abandonment and reclamation of the site [118] (Figure 2.5.1).

Strata containing shale gas can be very thin so efficient extraction is achieved through a combination of vertical and horizontal drilling: vertical to reach the shale formation to a depth of around 1500 m then directional drilling to follow the shale formation for a horizontal section of at least 1500 m [14]. Horizontal drilling provides additional contact between the wellbore and the source rock [119]. Usually 6-8 wellbores are drilled per well pad [120]. Steel casing pipes are installed in the borehole and cemented to the surrounding rock formation. If properly done, this operation should prevent leakage of natural gas through the well bore during the production phase.

The casing of the horizontal well is then perforated and hydraulic fracturing is carried out to provide fractures in the shale to allow the gas to escape. Fracturing fluids are pumped into the well at high pressure to create fractures in the rock from the horizontal bore [120] that can extend a few hundred metres into the rock. The fracturing fluid is composed by almost 99% of water and proppants, with a blend of different chemicals; the proppants (usually silica sand) holds the rock fractures open while the chemicals enhance the fracturing fluid properties [118]. Chemicals include: acids, biocides to prevent organism growth that might cause the fractures to clog, corrosion and scale inhibitor and guar gum to change the fluid rheology and friction reducers to improve the fluid flow [118].

Flowback water is the water produced from the well immediately after the pressure of fracturing fluids is released and before gas production commences; it must be collected and disposed of safely to avoid environmental contamination. The amount and composition of the flowback water are not the same as those of the fracturing fluids injected. The ratio of flowback fluid to fracturing fluid volume (flow back ratio) varies between different plays and is a characteristic of the shale play itself. The flowback ratio is usually 20%-80% but for some plays in the US it is reported to be even higher than 100% [121]. The flowback water contains part of the chemicals injected with the fracturing fluids and also substances naturally present in the reservoir: salt, radioactive materials, hydrocarbon, metals, etc. The volume of flowback water reduces over time and its composition converges towards that of brine naturally present in conventional and unconventional formations.

The process of well completion, following construction and drilling of the well and preceding gas production, includes preparation of the borehole, installation of pipes, escape of gas to clear the debris and, for shale plays, also the flowback period just described. The completion emissions associated with shale gas are different from those associated with conventional natural gas, because of the hydraulic fracturing and flowback phases. During the life of a well, other gas emissions may occur during workovers and unloadings. Following production, shale gas is processed and passed into a main transport pipeline.

Two characteristics are often reported about shale plays: the initial production rate and the production decline rate. The initial production rate is highly variable between different wells [122] but it is reported to decrease very rapidly; for example, Speight [123] estimates that half the production of a shale well occurs within five years while O'Sullivan *et al.* [122] report that shale well output tends to drop by 60% or more over the first year of production. However, productivity and its decline are more moderate during the following years of well life. For these reasons, shale wells are commonly re-fractured to restore gas production. The amount of gas expected from a well during its entire life is defined as the estimated ultimate recovery (EUR).

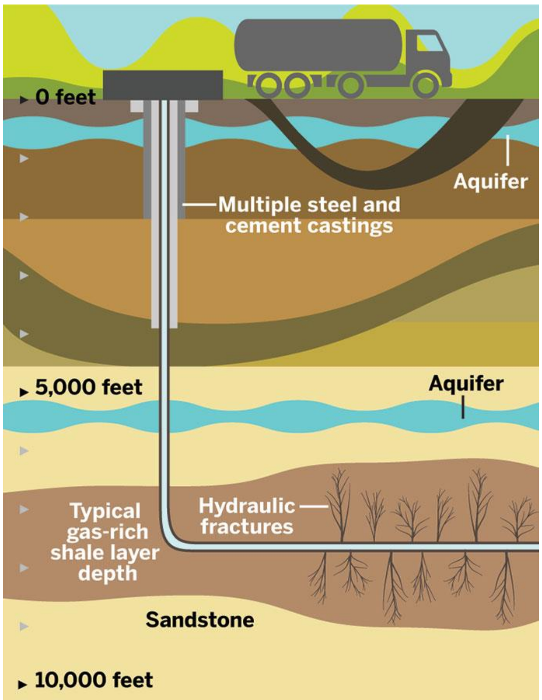


Figure 2.5.1. Schematic of the shale gas extraction process [124]. Not drawn to scale.

Many criticisms about the pollution caused by this extraction process have been the reason of shale gas banning in some countries. For example, the problem of fresh water contamination due to shale gas production was firstly raised in the film documentary 'Gasland' of Josh Fox. It reported on the phenomena of burning water due to contamination of water aquifers with methane and it highlighted critical aspects about the direct disposal of flowback water to fresh water course or in evaporation pits. Heavy environmental pollution are due to the contamination of fresh water with the chemicals used for hydraulic fracturing [125] and the chemicals and radioactive elements found in flow back water. The major problems are associated with improper operational practice of waste water disposal and well casing [126]. Critics were also raised in the US because the composition of fracturing fluids and waste water, at the beginning of the shale extraction development, was not disclosed by industries on the basis of commercial confidentiality. In addition, the threats of shale gas extraction on water scarcity and contamination are well documented and reviewed [127,128].

However, since the beginning of shale exploration in the US, new policy rules about the disposal of flowback water have been introduced in some US states to ban direct flowback disposal to fresh water. Possible alternative include i) flow back recycling [128]; ii) flowback injection in deep well of Class II [118,129]; iii) disposal to industrial waste treatment plant [130]. If shale gas production develops in Europe, direct disposal to water and disposal to injection wells will be banned and full details about the chemicals used in fracking fluids will have to be publicly available [131].

Another point of debate regards the potential emissions (the volume of natural gas that may leak, i.e. the potential emissions, not the actual emissions) due to well completion and workover that are higher in the case of shale gas extraction because of the hydraulic fracturing phase and well maintenance. The handling method of the potential emissions during completion and workovers can have a dramatic impact on the carbon footprint of the shale gas production process [122]. Venting of natural gas emissions would significantly increase the carbon footprint of the process; alternative more challenging handling methods such as flaring or capture and injection in the grid can decrease the potential warming of the process.

For example, reduced emissions completions (RECs) – also known as reduced flaring completions or green completions – is a term used to describe a practice that captures gas produced during well completions and well workovers following hydraulic fracturing. Portable equipment is brought on site to separate the gas from the solids and liquids produced during the high-rate flowback, and produce gas that can be delivered into the sales pipeline [132]. Flaring is another option to handle the potential emissions but it only eliminates methane and other hydrocarbons contained in the natural gas and it does not avoid the emission of other polluting compounds (such as SO_x, NO_x, PM and CO).

Chapter 4 analyses the environmental impacts of possible shale gas exploitation in the UK and addresses some of the challenges reported above.

2.5.2 Liquefied Natural Gas

High costs, limited local demand and practical difficulties may limit the supply of natural gas via pipelines within the region of production. The supply of natural gas over long distances at low cost has become feasible during the 1980s when Sliepcevich pioneered and managed the research, development and implementation of the first commercial process for liquefaction and liquefied natural gas (LNG) ocean transport. Natural gas can be liquefied, stored and transported at a temperature of -161˚C and at atmospheric pressure. The volume of LNG is less than 0.2% of the gas volume so that it can be more easily shipped or stored for use during high demand periods [133]. Use of LNG is expected to grow, to the point where it could overtake the supply of natural gas through pipes [20] also because it has been promoted as a “clean fuel” able to play a major role in meeting global energy demands [20,21].

The first shipment of LNG using a purpose built tanker was delivered to the UK from Algeria in 1964, with a few shipments delivered in the following years. However, in 1965 the natural gas reserves in the North Sea were discovered, delaying further the development of LNG infrastructure in the UK [134]. More than 40 years on, production of natural gas on the UK Continental Shelf of the North Sea is declining; attention has therefore returned to LNG import facilities and infrastructure developments, such as the Europe’s largest LNG receiving terminal at South Hook in South West Wales [62,87]. Other countries increasing their LNG imports include Japan, South Korea, Portugal, Italy and Taiwan.

In the prospect of energy security, one of the principal sources of the increasing imports of LNG to the UK is Qatar, thanks to the Qatargas II project, agreed in 2009 with Qatar’s national gas company, Qatargas. This project included the development of 30 offshore wells and three new platforms in Qatar’s North Field. A total of 45 new LNG tankers have been built for the new projects developed by Qatargas, including the Q_{flex} and Q_{max} types [135]. In 2013, Centrica (the owner of British gas) signed a contract with Qatargas to import 3 million tonnes of LNG, which will provide 13% of the UK gas demand from now until 2018 [136].

2.5.2.1 The Liquefied Natural Gas supply chain

Liquefied natural gas is natural gas that has been converted into liquid after being pre-processed to eliminate impurities, such as water, associated liquids and acids. The heavier liquids and condensate are extracted to be sold separately or used as refrigerant later in the cooling process. If present, nitrogen and helium are also removed. CO_2 is chemically or physically absorbed in solvents and removed to avoid frozen deposits that could cause clogging in the downstream liquefaction equipment. H_2S is also absorbed to meet product specifications, avoid blockages and prevent damage to process equipment. The purified gas is finally dried before liquefaction in chillers. Different liquefaction technologies are described in literature [133]. LNG is then stored and loaded into purpose designed tankers to be shipped to the destination port.

The vapours of greenhouse gases created by the ambient heat input, while maintaining constant pressure in the storage vessel, are called “boil-off” [137] and depend on many factors including weather, sea states, shipboard operations, ambient temperature, etc. The boil off rate ranges between 0.15% and 0.25% per day, during loading of the tanker, and can reach values as high as 150% of the normal rate immediately after loading, when the cargo tank and insulation cool down [138,139]. These operations are critical for their contribution to global warming.

Conventional tankers, using steam-turbine propulsion systems, could run on a range of different fuels. Boil-off gas produced from the cargo tanks can be compressed and burned in the boilers in place of some of the normal fuel oil, at the expense of LNG loss during transport [140]. However, in the new fleet used for Qatari LNG exports, the boil-off gas is re-liquefied and returned to the cargo tanks to manage the pressure and to minimise losses during transport, albeit at the expense of power consumption for liquefaction.

The slow-speed diesel engines, used instead of the conventional steam turbines in the Qatari fleet, are reported to have higher fuel efficiency and lower emissions [138]; this new fleet has a cargo capacity over 50% higher than conventional carriers- from 135000 m^3 [141] of the old tankers, up to 266000 m^3 for the new tankers [138].

After transport, at the destination port, the LNG is unloaded, stored and vaporised when needed.

These critical aspects of the LNG supply chain are analysed in more details in chapter 4 where a full environmental analysis of LNG supply to the UK is tackled.

2.6 The waste debate

The UK Government is looking for alternative energy sources that can contribute to a secure energy supply and carbon emission reduction. The increased focus on renewable resources and climate change has altered the perception of waste [142]: the United Kingdom is moving towards a 'reduced waste economy' [143]. This means moving towards a society where waste resources are fully valued, financially and environmentally. Residual waste can still be recovered as energy or by-products and therefore, landfill should be the last resort for waste disposal [143]. In fact, the UK Government is determined to increase energy from waste generation because i) energy from waste is a valuable domestic source of energy that contributes to energy security; ii) waste is a low-carbon energy source that can contribute to the UK's renewable energy targets; iii) unlike wind and solar technologies, energy from waste has the advantage of being a non-intermittent energy source.

2.6.1 Waste arisings

Both developed and developing countries faces the major challenges of managing increasing volumes of waste. This represents a significant problem at local and governmental level mainly because of huge investments needed for infrastructure and technology development and institutional and political oppositions [144].

Defra [145] reported the most up-to-date statistics on waste production in the UK. As shown in Table 2.6.1, the 200 million tonnes of total waste produced in 2012 were very heterogeneous, the main waste material being mineral wastes (34% of the total waste material produced in the UK).

	UK	
Waste material	**Tonnage [thousand tonnes]**	**Proportion of total**
Metallic wastes	6,060	3.0%
Glass wastes	2,250	1.1%
Paper & cardboard wastes	3,659	1.8%
Plastic wastes	3,199	1.6%
Wood wastes	2,306	1.2%
Vegetal wastes	6,602	3.3%
Household & similar wastes	26,446	13.2%
Mineral wastes	69,205	34.6%
Soils	41,625	20.8%
Dredging spoils	14,721	7.4%
Other wastes	23,948	12.0%
All wastes	200,020	100.0%

Table 2.6.1. Waste generation split by waste material in the UK in 2012 [145].

Table 2.6.2 reports the waste generation split by economic activity in 2012. Hazardous waste represented 5.9 million tonnes of the total amount while the largest contribution was from construction/demolition activities that produced about 50% of waste among all different activities.

[thousand tonnes]	**Commercial and industrial (C&I)**	**Construction**	**Households**	**Other**	**Total**
UK	47,567	100,230	27,506	24,716	200,020
of which hazardous	3,173	1,057	1,306	395	5,931

Table 2.6.2. Waste generation split by economic activity, UK, 2012 [145].

The definition of Municipal Solid Waste (MSW) includes all local authority collected waste and waste from the commercial and industrial sector which is of a similar nature to local authority collected waste. MSW and Commercial and Industrial Waste are of particular interest because they are considered as renewable feedstock valuable for energy production. Hence, they contribute to meet the renewable energy targets. In 2012, commercial and industrial activities generated about 24% of the total amount of UK waste, whereas households were responsible for the production of 14% of the total waste. As shown in Table 2.6.3, waste arising from households in the UK has increased by 3.3% between 2013 and 2014 but has decreased by 0.6 % since 2010.

Year	Measure	UK	England	NI	Scotland	Wales
2010	Arising	26,954	22,131	829	2,649	1,344
	Recycled	10,879	9,112	315	861	591
	Recycling rate	40.4%	41.2%	38.0%	32.5%	44.0%
2011	Arising	26,793	22,170	810	2,484	1,329
	Recycled	11,496	9,596	327	922	651
	Recycling rate	42.9%	43.3%	40.4%	37.1%	49.0%
2012	Arising	26,428	21,956	783	2,383	1,306
	Recycled	11,603	9,684	326	912	681
	Recycling rate	43.9%	44.1%	41.7%	38.3%	52.1%
2013	Arising	25,929	21,564	781	2,311	1,274
	Recycled	11,445	9,523	335	916	671
	Recycling rate	44.1%	44.2%	42.9%	39.6%	52.6%
2014	Arising	26,797	22,355	808	2,349	1,285
	Recycled	12,044	10,025	352	962	705
	Recycling rate	44.9%	44.8%	43.6%	41.0%	54.8%

Table 2.6.3. Waste arising from households and recycled in ‘000 tonnes and recycling rate from households in the UK, England, Northern Ireland, Scotland, and Wales between 2010 and 2014 [146].

Figure 2.6.1 shows a typical composition of the UK municipal solid waste. More than 35% of municipal waste is usually composed of organics mainly including food waste and garden waste. Paper and cardboard account for 23%, followed by plastics (10%), glass (6.6%) and metals (4.3%).

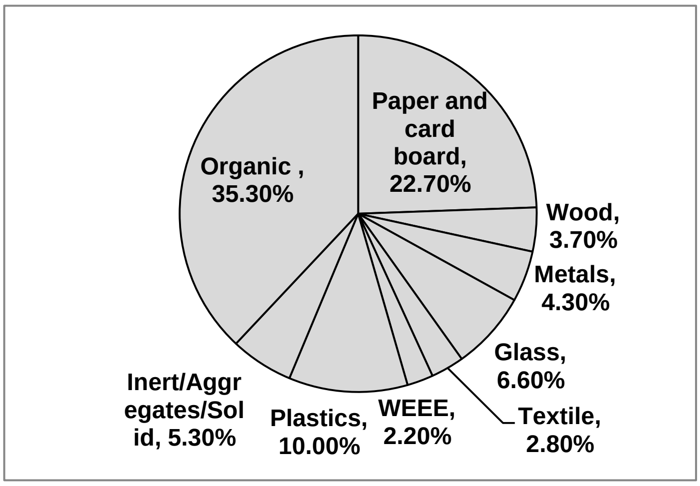


Figure 2.6.1. UK average municipal solid waste composition.

2.6.2 Waste policies

Municipal waste arisings and disposal in the UK are controlled by a number of key legislations:

- The EU Waste Framework Directives adopted in 2008 [147]. Under this Directive the UK and other EU Member States must recycle 50 % of household waste by

2020. Furthermore, the Article 4 of the Directive sets out five steps for dealing with waste, ranked according to the environmental impacts and identified as the 'waste hierarchy' (Figure 2.6.2).

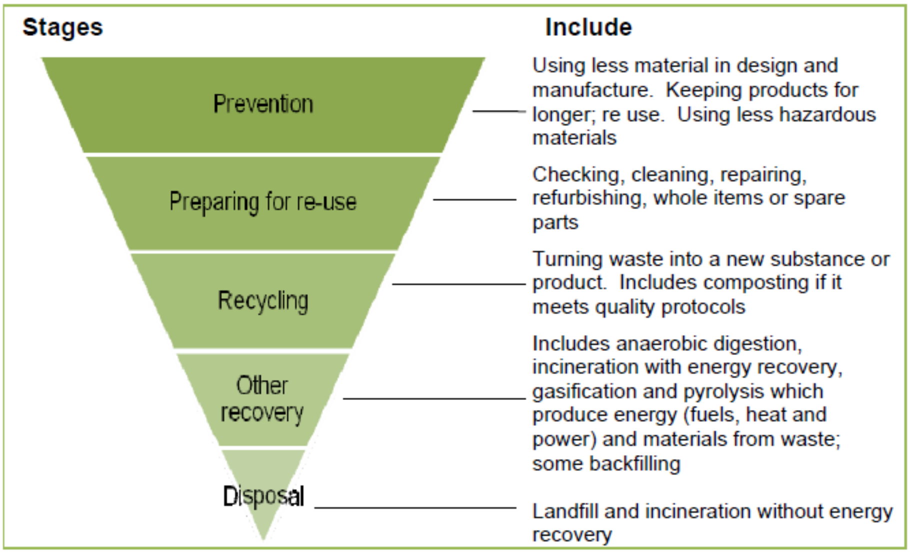


Figure 2.6.2. Waste hierarchy [148].

The base of the reverse pyramid indicates that the best option is the reduction of the amount of waste produced; then, re-use and recycling are recognized as a means to reduce the exploitation of primary resources and thus to reduce pollution created by mining and ore processing [149]; finally, the point of the pyramid indicates that disposal without valuable substances recovery should be the last option when dealing with waste management.

However, when waste is treated in alternative technologies and a deviation from the waste hierarchy [148] is applied, the EU Commission explicitly reports that the Life Cycle Thinking approach and in particular the Life Cycle Assessment (LCA) should be used to assess the environmental burdens of the developing alternatives [150]. Therefore, extensive LCA work is needed to assess the environmental performance of renewable energy production from waste, including thermal and biological technologies.

- The Landfill Directive (1999/31/EC) [151] aims at preventing and reducing the effects of landfilling waste on surface water, groundwater, soil, air, and on human health. According to this Directive, by 2010, biodegradable municipal waste going to landfills must be reduced by 30% of the total amount (by weight) of biodegradable municipal waste produced in 1995; 35% by 2013 and 50% by 2020.

- The Waste Incineration Directive [152] specifically limits the emissions to air, water and soil produced by thermal processes treating waste.

In order to meet the targets on energy and waste, the UK is financially supporting technologies that transform renewable sources into energy. For the development of renewable energy three points are considered fundamental [153]: i) financial support to the uptake of novel more energy efficient technologies; ii) unblocking barriers to delivery renewable energy and iii) invest in innovation for low-carbon technologies and development of emerging technologies. In accordance with these, the UK government introduced the renewable obligations in 2002 (RO) and the Feed-in tariffs in 2010 for electricity generation, the Renewable Heat Incentives (RHI) in 2011 for heat production and the Renewable Transport Fuel Obligation (RTFO) in 2007 for road transport fuel sales as financial incentives to all renewable sources to meet the renewable energy targets.

As reported in the Waste Framework Directives [147] (waste hierarchy), European governments, including the UK, are pushing towards more sustainable waste disposal options to alleviate the problem of waste management and to integrate it within the framework of energy supply. Therefore, the production of energy from waste, such as electricity, renewable methane and bio-fuels is eligible for financial support within the renewable schemes mentioned above to actively promote growth in this sector [154].

This has recently catalysed an increase in the bio-energy production from waste as reported in Table 2.6.4. Since 2009 the total electricity produced from waste has more than doubled and at the same time, the total production of electricity from all sources has decreased. In addition to this, the production of energy from waste is reported [155] to have a significant role in the renewable energy sector itself (up to 37% of the total renewable energy produced in 2014 was coming from waste, see Table 2.6.4).

[GWh]	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**
Landfill gas	4,918	5031	5085	5145	5160	5045
Sewage Sludge Digestion	603	697	764	719	761	845
Energy From Bio-degradable Waste Combustion	1509	1530	1503	1774	1649	1950
Co-firing with fossil fuels	1625	2332	2964	1783	309	133
Animal Biomass (Includes electricity from poultry litter combustion, and meat & bone combustion)	637	627	615	643	628	614
Anaerobic Digestion (Includes electricity from farm waste digestion and other AD)	43	111	273	501	722	1009
Plant Biomass (Includes electricity from straw and energy crops)	1379	1293	1749	4083	8929	13105
Waste (Non-biodegradable part of municipal solid waste plus waste tyres, hospital waste, and general industrial waste)	868	987	1085	1429	1481	1951
Total Electricity generated from waste	11582	12908	14038	16077	19640	24653
Total renewable energy generated[2]	26112	26770	35614	42553	54758	66605
Total electricity generated from all sources.	342011	347846	332461	328270	324725	300823

Table 2.6.4. Electricity generated from waste [156].

In the UK, public investments are supporting the design, installation and operation of advanced waste-to-energy technologies to achieve high recovery, efficiency and flexibility. Examples of this support includes i) the UK national competition (2013) to design efficient, economical and commercially viable gasification demonstrator plants supported by the Energy Technology Institute (ETI), and ii) funding from the Department for Transport (DFT) to develop and build a plant for production of renewable methane from waste to power heavy goods vehicles. These have promoted the development of a number of multi-stage advanced thermochemical processes including fast pyrolysis with combustion and gasification. One example is the process developed by Advanced Plasma Power (APP) which is at the centre of this thesis and described in detail in the following sections.

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[2] It includes: onshore and offshore wind, wave and tidal, solar photo-voltaics, hydro and energy from waste.

The role and impact of advanced thermal technologies for electricity production are fully explored in chapter 5 whereas the advanced thermal and biological waste treatment technologies for renewable methane production are considered in chapter 6.

2.6.3 Waste disposal

According to the latest waste statistics released by Defra[145] in 2012, almost half of the total 186.2 million tonnes of total waste that entered final treatment in the UK were recovered. The majority of this (77.5Mt) was ‘Recovery except backfilling’, with ‘Incineration’ and ‘Energy recovery’ making the smallest contributions (Table 2.6.5). However, more than 25% of waste was still landfilled, an improvement compared to 2008 when 53% of the total waste produced in the UK was sent to a landfill [157].

		Energy recovery	**Incineration**	**Recovery other than energy recovery - Except backfilling**	**Recovery other than energy recovery - Backfilling**	**Deposit onto or into land (landfill)**	**Land treatment and release into water bodies**	**Total**
UK	Tonnage [thousand tonnes]	1,585	6,102	77,467	14,114	48,512	38,383	**186,163**
	Proportion of total	0.9%	3.3%	41.6%	7.6%	26.1%	20.6%	**100.0%**

Table 2.6.5. Waste entering final treatment, split by final treatment method, UK 2012 [145].

Thanks to the Landfill Directive, the amount of Municipal Waste sent to landfill decreased from the 35.6 million tonnes in 1995 to 18 million tonnes in 2013 (Table 2.6.6). The biodegradable waste sent to landfill in 2013 was 26% of the 1995 baseline value; hence, in 2010 and 2013 the UK met the targets set by the Directive.

Year	**Measure [thousand tonnes]**	**UK**	**England**	**NI**	**Scotland**	**Wales**
1995	Municipal Waste to Landfill					
	of which BMW to Landfill	35,688	29,030	1,225	3,595	1,837
2010	Municipal Waste to Landfill	24,807	20,298	893	2,296	1,319
	of which BMW to Landfill	12,904	10,339	558	1,406	600
2011	Municipal Waste to Landfill	22,432	18,421	734	2,113	1,164
	of which BMW to Landfill	11,644	9,360	464	1,282	538
2012	Municipal Waste to Landfill	19,733	16,187	622	1,902	1,023
	of which BMW to Landfill	10,215	8,129	394	1,170	522
2013	Municipal Waste to Landfill	17,990	14,780	472	1,784	954
	of which BMW to Landfill	9,219	7,347	299	1,076	497

Table 2.6.6. Municipal Waste and Biodegradable waste to Landfill, UK and country split, 1995, 2010-13 [145].

As shown in Table 2.6.3, the UK also improved the recycling rate for the waste arising from households. It achieved a recycling rate of 44.9 % in 2014 compared to 40.4 % in 2010. The highest recycling rate in 2014 is shown for Wales. Conversely, England was the region to produce the highest amount of households. The recycling, reuse and composting percentages for England are shown in Figure 2.6.3.

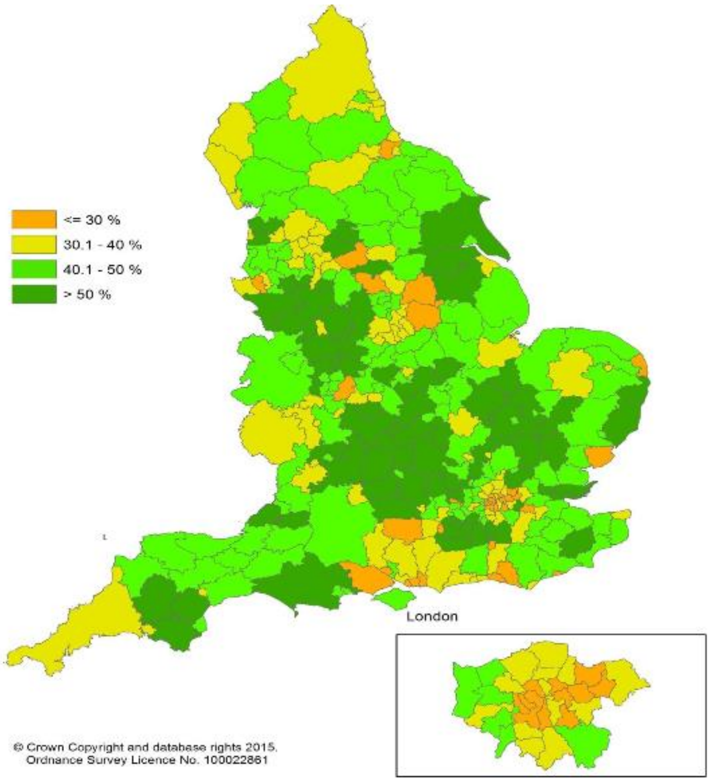

Figure 2.6.3. Percentage of household waste sent for recycling, reuse or composting, England, 2014/15 [146].

In England, in 2014-2015, for the first time the amount of household waste sent to incineration with energy recovery exceeded the amount of household waste disposed to landfill (30% and 25% of the total local authority waste, respectively). The amount of waste sent to incineration with energy recovery increased by 25% in 2014/15 compared to 2013/14 with a more than three-fold increase since 2000/01 [158] (Figure 2.6.4).

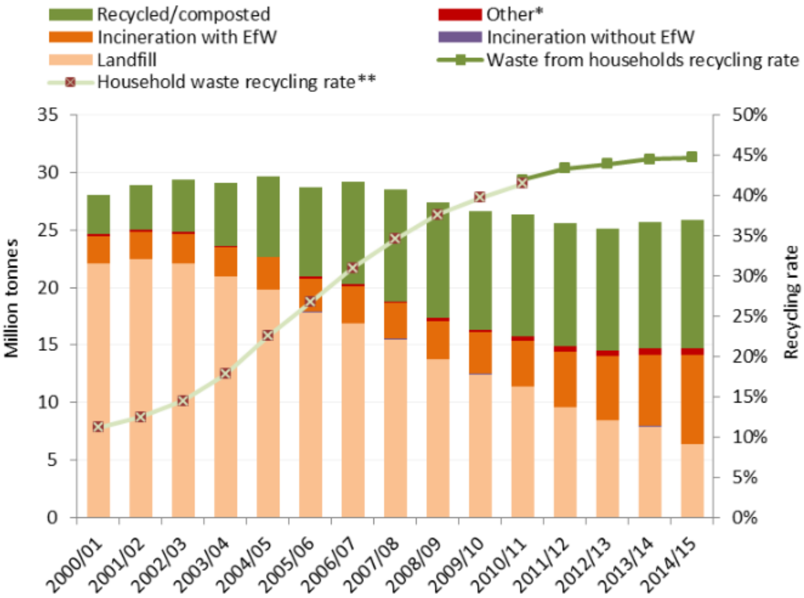

Figure 2.6.4. Management of all Local Authority collected waste and recycling rates in England, 2000/01-2014/15 [158].[3]

Until recently, incineration has been the main alternative to landfill for the treatment of municipal solid waste [159–161]. However, local authorities, pushed by public environmental concerns and fierce opposition to incineration plants, started considering at other thermochemical treatment options produce energy, including pyrolysis, gasification and plasma arc technologies. Hence, nowadays the management of waste is a complex integrated system that includes the separation of different streams, recycling, conventional and advanced thermal and biological treatments.

2.7 Waste treatment technologies

It is clear that there is a huge potential to use waste as renewable feedstock to help meeting the targets on renewable energy and climate change while addressing at the same time energy security. The most widespread technologies for waste treatment and disposal fall into two categories: biochemical (anaerobic digestion, composting and landfill) and thermochemical conversions (gasification, pyrolysis and incineration). In the following sections the main disposal treatments are reviewed.

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[3] Notes: Incineration with energy recovery/without energy recovery includes incineration bottom ash (IBA) and metals from IBA.
* Other includes waste treated/disposed through other unspecified methods, process and moisture loss.

2.7.1 Bio-chemical conversion

Landfills

Landfilling is the most common and ancient waste disposal method throughout the world also because it can accept a wide range of waste. In some countries, the majority of municipal waste is still buried in landfill. The term landfill is used to refer to a wide range of facilities ranging from open dumps to highly engineered facilities as bioreactor landfills, flushing-bioreactor landfills and semi-aerobic landfills [162]. In landfills, waste material degrades and yields the landfill gas (almost 50% methane), the leachate and the residual material which has not degraded. Depending on the type of landfill, the greenhouse gas emissions can reach the atmosphere and the leachate, if not correctly handled can cause risk to soil and water. Therefore, this disposal option is directly addressed by the Landfill Directive.

In a conventional landfill measures to enhance the waste degradation are not taken but measures are implemented to collect and manage the leachate and gas generated. For the leachate, these measures usually include a collection system and treatment prior to discharge to surface water bodies. In addition, to reduce greenhouse gas emissions of conventional landfill, top soil cover is used for mitigation and the landfill gas is collected and flared. Conversely, modern engineered landfills have a range of landfill gas utilization and control systems that reduce the emissions of methane and include recovery of energy. Active measures such as leachate recirculation, water addition and air injection, to enhance the waste degradation process and be more efficient, are usually adopted.

Anaerobic digestion (AD)

In the UK, the amount of food waste sent for composting (including anaerobic digestion) has more than doubled since 2010 and in 2013 it was 273 thousand tonnes, a 19 per cent increase compared to the previous year [163]. Thanks to this technology, in 2014, the UK produced 2.6 billion cubic metres (bcm) of biogas. With the growth of anaerobic digestion and advanced thermal technologies that convert waste to Bio-SNG, renewable gas production could reach 7 bcm per year by 2025 [23]. In fact, bio-chemical conversions are expected to develop even further and, together with thermo-chemical conversions are deemed necessary to meet the increased bio-methane supply of the future energy mix.

Anaerobic digestion can treat a wide range of organic waste (such as animal manures, agricultural crops, agri-food processing residues, food residues, the organic fraction of household waste, organic fractions of industrial wastes and sewage sludge, except feedstock with a high content of lignin that cannot be anaerobically degraded) to convert them into energy [164] and substitute fossil energy. This technology has developed mainly in rural areas where it is currently used to treat waste at the production site [165].

Six operations are essential for the AD process: i) pre-treatment; ii) anaerobic digestion; iii) water and acid compounds removal; iv) upgrading of the biogas; v) disposal/utilisation of the digestate.

When AD is applied to waste, pre-treatment is always required: non-biodegradable materials are removed, waste is shredded and maceration and sterilization (if required for the specific substrate treated) take place [166].

Anaerobic digestion naturally occurs in the absence of oxygen when the methanogens bacteria digest the organic substrate to produce the biogas and the digestate; it can occur under controlled conditions in chemical reactors but also spontaneously, for example, at landfill sites. The yield in biogas production during digestion is dependent on the composition of the feedstock and it decreases with an increase in impurities. The two main products of AD are the biogas and the slurry (the digestate) consisting of what is left of the treated substrate. The composition of the biogas produced depends on the feedstock but it usually contains CO_2, CH_4, N_2, H_2S and other contaminants including NH_3, particles and oxygen [167]. Typical biogas composition is reported in Table 2.7.1. For reference, landfill gas and North Sea gas are also shown.

Components	**Landfill gas**	**Biogas from AD**	**North Sea natural gas**
CH_4 [% vol]	36-65	53-70	87
Higher hydrocarbons [% vol]	0	0	12
H_2 [% vol]	0-3	0	0
CO_2 [% vol]	15-50	30-47	1.2
N_2 [% vol]	5-40	0.2	0.3
O_2 [% vol]	0-5	0	0
H_2O [% vol]			
H_2S [ppm]	0-100	0-10000	1-2
NH_3 [ppm]	5	<100	0

Table 2.7.1. Composition of gas from different sources [168].

If the feedstock used is highly pure and contains low levels of inert or unwanted materials, the digestate (used as it is produced into the digester, or separated in liquor and fibre) can be used as organic fertilizer for substitution of chemical fertilizer [169]. In this case, significant nutrient lost can increase the environmental impact of AD as further analysed in chapter 6.

The energy recovered from biogas is either used on the production site or sent to the national energy network. Biogas can be directly used to produce heat and power or cleaned and upgraded to produce bio-methane. This is usually injected into the grid as grid quality methane. In the cleaning process, H_2S, water and contaminants are removed to avoid corrosion to the following appliances. The most common method for H_2S removal from the crude biogas is through the reaction of H_2S with metal oxides [170,171] whereas water is usually adsorbed on silica gel. In the upgrading process, the calorific content of the gas is also increased to meet the grid specification [172] by CO_2 removal. CO_2 can be removed using membranes, Pressure Swing Adsorption (PSA), physical or chemical absorption and also cryogenic separation [173]. These technologies have been described and reviewed in section 2.7.2.1.2 and in Table 2.7.4. Critical losses of methane usually occur in the digester and upgrading operations.

AD can be characterised according to the following parameters [174].

i) Dry/wet digestion

The process is defined wet when the moisture content of the digester is more than 90%, whereas dry when it is less than 75% [174]. Wet AD has higher operational simplicity and also feedstock with higher solid content can be handled. However, particular care must be given to the pre-treatment to limit the content of the heaviest fraction. During digestion, this could sink and accumulate at the bottom of the reactor causing damages [166]. Conversely, the dry process requires a less intense pre-treatment because the system has higher tolerance to impurities, but the moving equipment of the pre-treatment stage is more complex as a result of the higher viscosity of the feedstock [166].

ii) Thermophilic/mesophilic digestion

AD can be maintained at mesophilic (35–37 ˚C) or thermophilic conditions (55–60 ˚C); the optimum temperature depends on the feedstock and digester reactor [166]. A larger diversity of bacteria can live at mesophilic conditions; the bacteria population is more resistant and hence, this type of digestion is considered more stable [175]. Conversely, the digestion rate is faster at thermophilic conditions but this type of AD requires more energy to be run [175].

iii) One-phase/two-phase digestion.

The production of biogas takes place in two distinct chemical stages, acidification and methanogenesis that can be performed in one or two separate reactors. However, the latter configuration is not widely used because of technical and economic limitations [174]. AD is usually performed in a single stage and one reactor.

Chapter 6 reports on the environmental impacts of the AD and it fully analyses the critical aspects of methane emissions and digestate utilization. Comparison with thermochemical processes is also proposed.

2.7.2 Thermo-chemical conversion

Conventional and advanced thermal technologies for waste-to-energy facilities, such as combustion, gasification and pyrolysis, respectively, are spreading in order to recover valuables from waste. Those technologies are reported to have a number of advantages, such as, i) land saving over landfills thanks to a reduction of waste volume [176]; ii) reduced level of air emissions, including greenhouse gases [177–179] when compared to landfill and anaerobic digestion; iii) exploitation of rich, renewable energy streams and recycling of valuable materials including metals.

The characteristics of the three main thermal processes for waste treatment, combustion, gasification and pyrolysis, are reported in Table 2.7.2. Energy production from thermal processing of waste always involve three sections [180]: syngas/flue gas production, syngas/flue gas utilization and energy recovery, syngas/flue gas cleaning (the sequence of the last two sections can also be inverted).

	Combustion	**Gasification**	**Pyrolysis**
Aim of the process	Maximize waste conversion to high temperature flue gases, mainly CO_2 and H_2O	Maximize waste conversion to high heating value fuel gases, mainly CO, H_2 and CH_4	Maximize thermal decomposition of solid waste to gases and condensed phases
Operating conditions			
Reaction environment	Oxidizing (oxidant amount larger than that required by stoichiometric combustion)	Reducing (oxidant amount lower than that required by stoichiometric combustion)	Absence of any oxidant
Reactant gas	Air	Air, pure oxygen, oxygen enriched air, steam	None
Temperature	Between 850 C and 1200 C	Between 550–900 C (in air gasification) and 1000–1600 C	Between 500 C and 800 C
Pressure	Generally atmospheric	Generally atmospheric	Slightly over-pressure
Process output			
Produced gases	CO_2, H_2O	CO, H_2, CO_2, H_2O, CH_4	CO, H_2, CH_4 and other hydrocarbons
Pollutants	SO_2, NO_x, HCl, PCDD/F, particulate	H_2S, HCl, COS, NH_3, HCN, tar, alkali, particulate	H_2S, HCl, NH_3, HCN, tar, particulate
Ash	Bottom ash can be treated to recover ferrous (iron, steel) and non-ferrous metals (such as aluminium, copper and zinc) and inert materials (to be utilized as a sustainable building material). Air Pollution Control residues are generally treated and disposed as industrial waste	As for combustion process. Bottom ash are often produced as vitreous slag that can be utilized as backfilling material for road construction	Often having not negligible carbon content. Treated and disposed as industrial special waste
Gas cleaning			
	Flue gas treated in air pollution control units to meet the emission limits and then sent to the stack	Syngas cleaning to meet the standards of chemicals production processes or those of high efficiency energy conversion devices	Syngas cleaning to meet the standards of chemicals production processes or those of high efficiency energy conversion devices

Table 2.7.2. Key characteristics of thermal processes for waste treatment [181].

Among the three thermal treatments, gasification and pyrolysis are not new concepts. Although pyrolysis and gasification have been used extensively in the past to produce charcoal, coke or other fuels, it is only recently that these two technologies have received increasing attention due to their higher recycling rates, lower emissions, higher energy efficiencies, lower costs, smaller footprints and reduced visual impact [177]. In particular, fluidized beds are considered to be one of the most effective technologies for gasification or pyrolysis due to their high process flexibility [182]. Even so, the majority of existing energy-from-waste plants are grate-fired boilers (i.e. incinerators) [183].

The three main processes for thermal conversion are here reviewed.

Incineration

The oldest waste-to-energy technology is the incineration process. This is a well established process with over 900 plants spread worldwide and numerous reviews of this technology are available [179,184,185]. Depending on the incineration technology used, the waste can be directly burnt without pre-treatment or valuable material can be sorted out and refused derived fuels burnt. The furnace can be any of a number of different technologies such as moving grates, rotary kilns or combustion cones. The mass burn moving grate technology [186] is the most widely used for waste combustion. Waste is combusted at a temperature of about 800-1000 ˚C to produce a hot flue combusted gas used for energy generation [187]. Bottom ash is also produced; this needs to be treated before disposal or use [188]. Heat is recovered from the hot gases, usually to produce electricity or CHP and then flue gases are cleaned and emitted to environment. Restriction on emissions includes NO_x, acid gases, heavy metals and dioxins and furans; in particular, in the UK, the emissions need to comply with the WID directive [152].

As previously stated, incinerators have long raised concerns for emissions of dioxins, furans and heavy metals, all representing serious threats to human health. Novel generations of incinerators have achieved more controlled emissions that, associated with higher energy efficiency and sophisticated gas clean-up methods, greatly reduce the environmental impacts. This is analysed more in depth in chapter 5.

Pyrolysis

Pyrolysis and gasification are considered advanced technologies for waste treatment. In the pyrolysis process, the feedstock is thermally degraded in absence of oxygen using an external heat source. A wide range of reactors is used for the pyrolysis process, including fixed-bed, rotary kilns and fluidized bed. Process temperature can vary between 300 and 900 ˚C [189] but commonly 550 ˚C are used for MSW pyrolysis [190–192].

The products of pyrolysis depend on the operational conditions of the process and feedstock. As shown in Figure 2.7.1, they include liquid hydrocarbons (that are mainly produced when the feedstock has a high level of plastics), light gases, char and solid residues (containing carbon and inert material).

The solid residue usually undertakes a secondary treatment that can include quenching, combustion, gasification [193], separation of metals and stabilization. The gases are cleaned before utilization, for example, using scrubbing oil [194,195], filtering, scrubbers, etc. whereas the product oils are usually catalytically cracked [196]. Contaminants of pyrolysis products (gas and oils) include HCl, NH_3, SO_2, S, Cl, Br K, S, P, Ca, Zn and Cr [191].

When compared to incineration, the pyrolysis process determines lower production of NO_x and SO_2 as a consequence of the inert atmosphere and a better quality solid residue [197,198]; corrosion is also limited and dioxins formation is reduced [198].

A well-developed literature is available on pyrolysis of industrial waste including, for example, tyres [199,200], plastics [201,202] and sewage sludge [203]. Commercial pyrolysis processes are often associated with gasification or combustion to achieve higher process flexibility [198]; they mainly accept already pre-treated waste and pre-treatment includes screening, shredding and drying and/or production of refuse derive fuel (RDF) [204,205].

Conversely, pyrolysis of MSW is in its development stage and the number of commercial pyrolysis plants treating municipal waste in the world is very limited [189]. Nevertheless, research on MSW pyrolysis is ongoing at demonstration and pilot size [206,207].

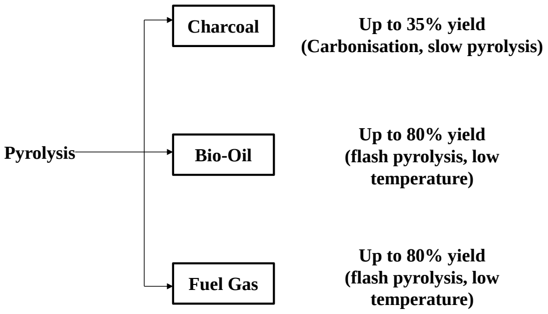


Figure 2.7.1. Products from pyrolysis [187].

Gasification

The technology of gasification is not a novel concept as coal gasification has extensively been used in a wide range of industrial applications, including town gas and synthesis gas production [208].

Waste gasification is rapidly spreading thanks to the advantages over conventional combustion technology of waste that between others, include: i) production of syngas suitable for many diverse applications; ii) higher energy efficiency when power is directly produced from the combustible gas; iii) high plant flexibility that allows a wide range of waste mass and composition to be treated; iv) reduced production of dioxins and furans and lower footprint [177]; v) higher recycling rates.

Before gasification, the waste is pre-treated (valuable materials are sorted out; the waste is shredded and dried; in some cases refused derive fuels are also produced and treated in the gasification technologies). Then, it is transformed into a combustible gas that can be used directly or indirectly for electricity production and also for chemical synthesis.

The waste is partially-oxidized into a synthesis gas mainly formed of CO, H_2, CH_4, CO_2 and water vapour [209]. The partial oxidation can be carried out using air, oxygen-enriched air, pure oxygen, steam or also plasma. Depending on the oxidizing medium, syngas with different compositions and heating values is produced [210]. The produced gas includes undesirable contaminants, such as tar, metals, particulate, etc. The tars are one of the most significant problems for industrial gasification processes.

The gasification involves a number of exothermic and endothermic reactions and steps [211] that occur at a temperature higher than 600˚C, depending on the reactor type and waste characteristics [181]. First, during devolatisation (thermal decomposition or pyrolysis) occurring at a temperature up to 700˚C in the absence of oxygen, thermal cracking reactions determine the production of volatile gases, mainly H_2, CO, CO_2 CH_4, hydrocarbon gases, tar, char and water vapour [212]. After pyrolysis, some of the tars and hydrocarbons in the vapours are thermally cracked and char is gasified by steam and CO_2 to produce carbon monoxide and hydrogen. The reducing environment with the absence of a completely oxidizing atmosphere strongly limits the mechanics of dioxins formation [213]. A char/ash solid residue including also non-combustible material is produced together with the gas and they must be treated before disposal or use.

In auto-thermal gasification the heat required from the previous reactions is provided by the partial combustion of part of the feedstock with a controlled amount of air, oxygen, or oxygen enriched air [214]. In allo-thermal gasification heat is provided by external sources such as, heated bed materials, plasma torch or heat provided by separate combustion reactions. For example, when the steam is the only oxidizing agent, no exothermic reactions occur and hence, heat must be provided externally.

Table 2.7.3 identifies different gasification technologies according to different criteria, such as the way of supply heat, pressure, temperature, reactor design, etc.

Criteria	**Types**
Heat supply	Directly heated (auto-thermal) gasifiers
	Indirectly heated (allo-thermal) gasifiers
Pressure	Atmospheric gasifiers
	Pressurized gasifiers
Gasification agent	Air gasifiers
	Oxygen enriched-air gasifiers
	Oxygen gasifiers
	Steam gasifiers
	Plasma gasifiers
Reactor design	Fixed bed gasifiers: - Updraft; -Downdraft
	Fluidized bed gasifiers: - Bubbling fluidized bed; -Circulating fluidized bed; -Internally circulating fluidized bed
	Entrained flow gasifiers
	Rotary kiln gasifiers
	Moving grate gasifiers
	Plasma gasifier
Temperature	Low-temperature gasifiers (typically below 900 C)
	High-temperature gasifiers (typically above 1200 C)
Bottom ash status	Dry bottom ash gasifiers
	Vitrified slag gasifiers
Energy recovery	Heat gasifiers (dirty syngas is post-combusted in a recovery boiler with heat/electricity production via steam turbine cycle)
	Power gasifiers (syngas is first cooled and cleaned and clean syngas is then burned in an internal combustion engine or a gas turbine)

Table 2.7.3. Types of waste gasifiers [181].

Among all gasification reactor designs, fluidised bed reactors are reported to be the most promising for a number of factors [215,216], including the enhanced mixing of reactants and almost constant temperature, high flexibility to treat different types of feedstock, and the possibility to use different fluidizing agents [217].

Fluidized bed gasifiers are usually operated in the auto-thermal configuration and at lower temperature, below 900 °C, to prevent ash and bed material melting, agglomeration and sintering [218]. Consequently, the produced syngas contains high levels of tar and other condensable hydrocarbons that can limit the application of this technology [177]. Furthermore, the energy conversion efficiency of the entire process can also be limited by the high carbon content and organic pollutants of the ash that need extensive reprocessing before use or disposal.

The use of plasma represents a valid option to improve the quality of the syngas produced (a completely decomposed tar free syngas) and convert the solid materials into a vitrified stable material. For example [219], a residence time of 4 seconds at temperatures from 900 to 1150 ˚C determines the thermal conversion of 97% of tars in a fluidized bed. Hence, the plasma overcomes the problems of syngas contamination that has been 'the major obstacle in deploying the gas engines and turbines necessary to achieve higher electrical generating efficiencies' [220].

Thanks to these properties, the technology of plasma gasification is used for the treatment of hazardous waste [221–223] and is also spreading for the treatment of MSW [224–226]. However, despite the fact that plasma technologies have been proven economically and technically feasible for a large range of hazardous waste, the same is not valid for the treatment of MSW as a result of the power requirements for plasma vitrification [227].

Plasma-gasifiers usually treat waste in a single stage with a plasma torch located at the bottom of the gasifier but more recently, dual stage gasifiers and plasma are also developing as reported in the following sections.

2.7.2.1 The Gasplasma process

Advanced Plasma Power (APP) has pioneered the development of a highly flexible two-stage thermal process, 'the Gasplasma process', which can treat a wide range of organic and inorganic wastes, including Municipal Solid Waste and Refuse Derived Fuel (RDF), to produce renewable energy, steam and a vitrified product. The technology comprises a fluid bed gasifier working with steam and oxygen coupled to a plasma converter. The crude syngas and ash from the gasifier are converted into the plasma converter that acts as a cleaner for the production of a high quality syngas. This is suitable for electricity production in a gas engine or for chemical conversions [228,229] to methane, hydrogen or higher hydrocarbons. The plasma cleaning is able to produce a low impurity, high energy syngas with high carbon conversion efficiencies [229]. One of the main advantages of a two-stage gasification-plasma process over a more traditional thermo-chemical treatment, such as a single stage gasification plant, is that it achieves better control on tar abatement in the syngas. Tars are undesirable because of various problems associated with condensation, formation of aerosols and polymerization to more complex structures, which may damage process equipment as well as end-use devices (e.g. gas engines and fuel cells). In the two-stage gasification and plasma process the tars are almost completely converted into H_2 and CO [218].

APP is currently commercializing the technology for production of electricity from syngas and is developing the pilot and semi-commercial plant for Bio-Substitute natural gas (Bio-SNG) production from syngas. If electricity is produced four main sections of the process are identified: solid fuel preparation unit; syngas generator unit; syngas refining unit; and power production unit. Conversely, when Bio-SNG is produced five main sections are identified (the first three sections are the same whether electricity or Bio-SNG is produced): solid fuel preparation unit; syngas generator unit; syngas refining unit; Bio-SNG production; and Bio-SNG upgrading and energy recovery. Each section of the commercial processes (~ 155ktpa of MSW) is analysed in the following paragraphs.

In 2007, APP built a demonstration plant of reduced capacity (100 kg/h of RDF input and 400 kW NCV of syngas produced) in Swindon (see Figure 2.7.2). It was designed for the electricity production from syngas and it includes the syngas generator unit, syngas refining unit and power production unit. Since 2008, APP has been running this plant using a variety of feedstocks and has demonstrated the production of a syngas of consistent quality (the gross heating value of syngas is in the range of 9–13 MJ/kg, and is widely suitable for electric power generation [177]). In 2013 the company has taken part in a national competition supported by the Energy Institute (ETI) for the development of high efficiency gasification process treating MSW at commercial scale.

Figure 2.7.2. Pilot plant of the Gasplasma process developed in Swindon for electricity production.

In 2013, APP alongside with its partners National Grid, Progressive Energy and Carbotech, won funding from both Ofgem's network Innovation Competition and from the European BESTF-ERANET programme to build a pilot plant (~35 kW NCV syngas input and ~15 kW NCV of Bio-SNG produced) that demonstrates renewable gas production by thermal gasification of mixed waste [230]. This plant is currently under commissioning and trials (see Figure 2.7.3). In addition to this, after a national competition, APP together with its consortium formed by the National Grid, Progressive Energy and CNG Service, was awarded government funding to develop a semi-commercial plant (~4.4 MW NCV of RDF input and 2.9 MW NCV of Bio-SNG output) that will produce compressed Bio-SNG from waste using the gasification and plasma technology to power heavy goods vehicles [231].

Figure 2.7.3. Pilot Plant of the Gasplasma process developed in Swindon for Bio-SNG production.

Solid fuel preparation unit

In the solid fuel preparation unit, the received waste is pre-treated and transformed to a refuse derived fuel (RDF) by shredding and drying. No pelletisation is required but if required by the fuel type, ferrous and non-ferrous metals are recovered by mechanical sorting, to be reprocessed for final sale as recycled metals. The steam required by the drier is internally supplied by the waste heat boiler of the syngas refining unit which recovers the high thermal energy content of the syngas at the plasma's exit. The moisture content of the feedstock is reduced from 40% to 12%.

Syngas generator unit

In the syngas generator unit, the waste is thermally treated to produce a high temperature syngas. Figure 2.7.4 represents the core of the Gasplasma technology. RDF is thermally decomposed in a bubbling fluidised bed gasifier and transformed into a raw syngas. Oxygen and steam are used as oxidising agents. Their flows are controlled to maintain the bed temperature (850 ˚C) and the required syngas quality, and an inert gas (nitrogen) is supplied to the gasifier as a purge. The high heat transfer and reaction rates, obtained in the fluid bed system thanks to an intense contacting between the gas and the solid phases, ensure an efficient gasification of the waste fuel. Two main streams are distinguished going from the gasifier to the plasma converter: raw syngas and ash. Primary components of the raw syngas are CO, H_2, CO_2 and H_2O. However, this stream produced in the gasifier still contains entrained ash particles, unconverted char and residual tars which preclude a direct use for energy production [228]. Hence, further processing is required in the plasma converter. This unit produces a high purity syngas as a result of the cracking of the tars exposed to the high plasma temperature and reactive environment: the tars are almost completely converted into H_2 and CO, resulting in high syngas yield, few by-products and nearly 100% carbon conversion [218]. The gas within the converter swirls towards the bottom of the reactor thanks to the cyclonic geometry and particles entrained in the gas together with the ash coming from the gasifier melt in a slag at the base of the converter. This slag is continuously separated from the gas stream to produce a vitrified material, the so called ‘Plasmarok’ product. This is a stabilised product which can directly be used as an aggregate material in road construction, without further reprocessing.

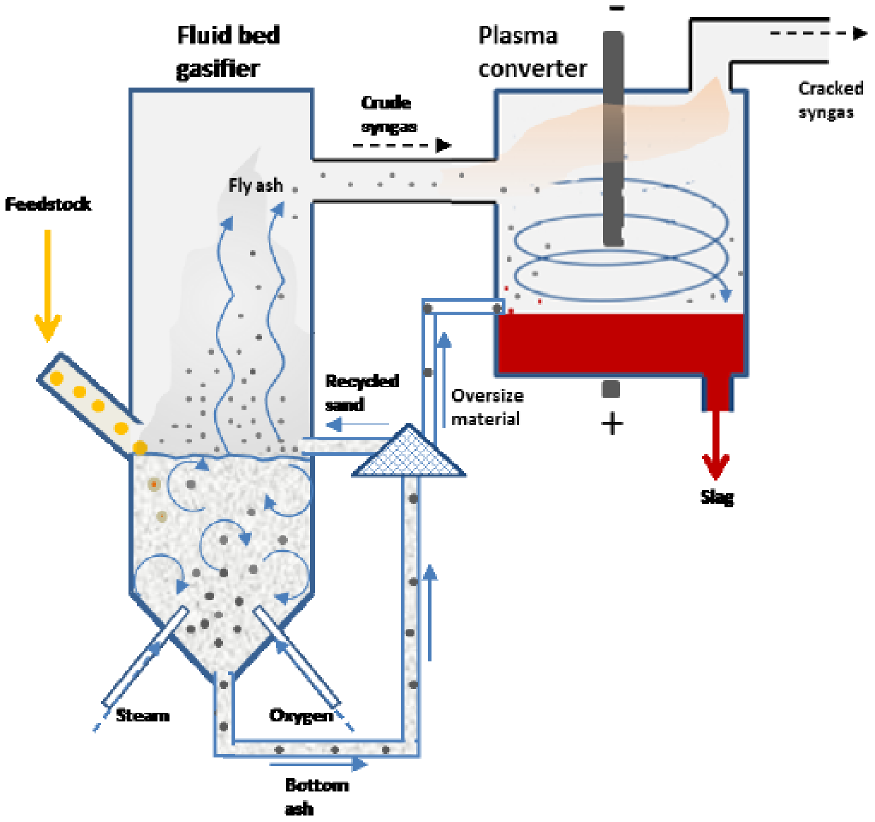


Figure 2.7.4. Schematic of core units of the Gasplasma technology [232].

To summarise, the major benefits of this technology are:

- High flexibility in the range of material that can be treated thanks to both the fluidised bed system (high heat and mass transfer) and the plasma system (high cleaning efficiency);

- Limited energy requirement of the pre-treatment section as the fluidised bed and gasification system can treat coarsely shredded and dried waste with no pelletisation required.
- No need for fossil fuel co - gasification;
- High carbon conversion efficiencies and low gas volume obtained thanks to the oxy-steam gasification (the nitrogen composition of the syngas depends only on the nitrogen composition of the waste);
- Independent control of the gasification and plasma as these are two separate units;
- Unlike common incineration plants that produce bottom ash which must be stabilised before use or disposal, the two-stage gasification and plasma process produces a vitrified and stabilized product (Plasmarok) that can be directly used for road construction;
- High particulate recovery in the vitrified slag that decreases the cleaning pressure of the downstream units in the syngas refining section.
- When compared to simple gasification and pyrolysis technologies, the syngas generated has a low level of contaminants and is rich in hydrogen. Therefore, it is suitable for both electricity generation and bio-fuel production with high energy efficiency;
- Low use of water and chemicals.

Syngas refining unit

In the syngas refining section the syngas is cooled and cleaned. The steam required by the dryer of the pre-treatment which reduces the moisture content of the feedstock from 40% to 12% is internally supplied by the waste heat boiler. This recovers the high thermal energy content of the syngas at the exit of the plasma unit. Then, the finest ash which is still contained in the syngas is collected and removed in the dry filter where air pollution control (APC) residues are produced. Further cooling is then achieved in the quench and water scrubbing systems (such as acid and alkali scrubbers) that are used to remove contaminant compounds, i.e. phenol, sulphur dioxide, hydrogen sulphide and ammonia. The syngas produced at this stage is suitable for both electricity production and chemical conversion as highlighted in the following paragraphs.

2.7.2.1.1 The Gasplasma process for electricity production

Power production unit

If the syngas is used to produce electricity, the last section of the process a gas engine and cleaning of the flue gas. The steam produced by cooling the flue gas section is fed to a steam turbine to produce additional power. The catalytic reactor cleans the flue gas leaving the gas engine, decreasing the amount of carbon monoxide and nitrogen oxides emitted to the atmosphere. The exhaust gas is finally released to the atmosphere through a stack at almost 200 ˚C.

Figure 2.7.5 reports the process layout of the advanced gasification and plasma process when electricity is produced.

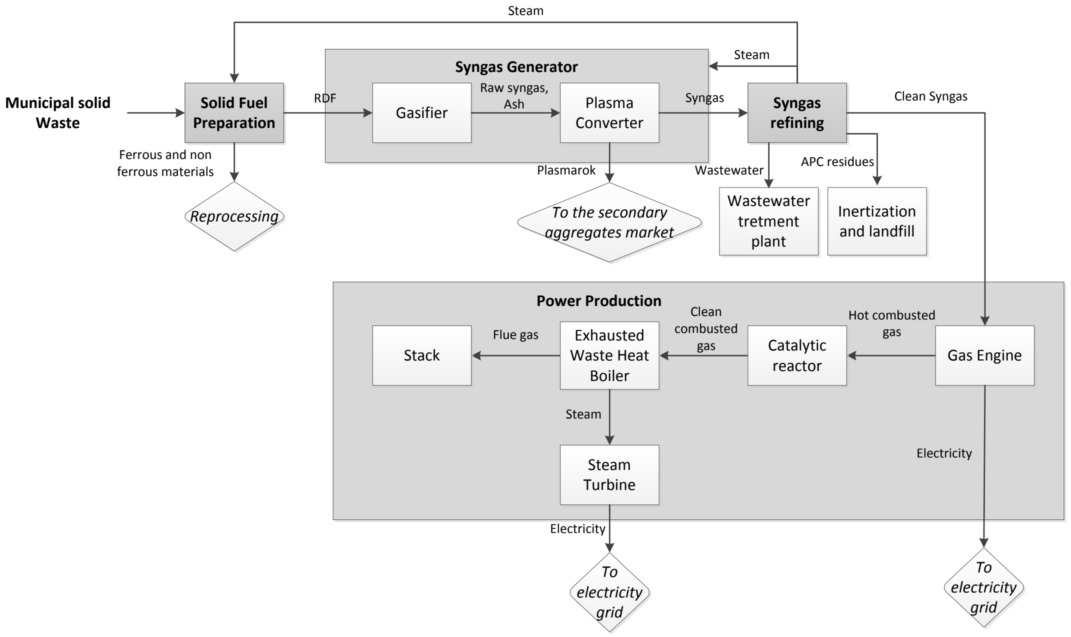


Figure 2.7.5. High level schematic of the two-stage gasification and plasma process (G-Pl). The four main sections of the process are highlighted in bold.

2.7.2.1.2 The Gasplasma process for Bio-Substitute Natural Gas production

As the clean syngas produced from the refining section complies with the composition limits required for chemical conversion (its composition avoids catalyst poisoning [233,234]), it can also be used for Bio-SNG production rather than being directly burnt for electricity generation. The chemical reactions used to produce methane from syngas (including water gas shift, methanation and up-grading) are already well established, but those have not yet been industrially proved for the production of methane from waste. Some literature is available on the production of Bio-SNG from thermal gasification of biomass, for example wood [235–239], but no literature is available on the production of Bio-SNG via thermal gasification of waste.

As previously reported, APP is developing pilot and semi commercial plants for the development of this technology [240]. The following paragraphs and Figure 2.7.6 identify the unit operations required to produce Bio-SNG from syngas in the APP process.

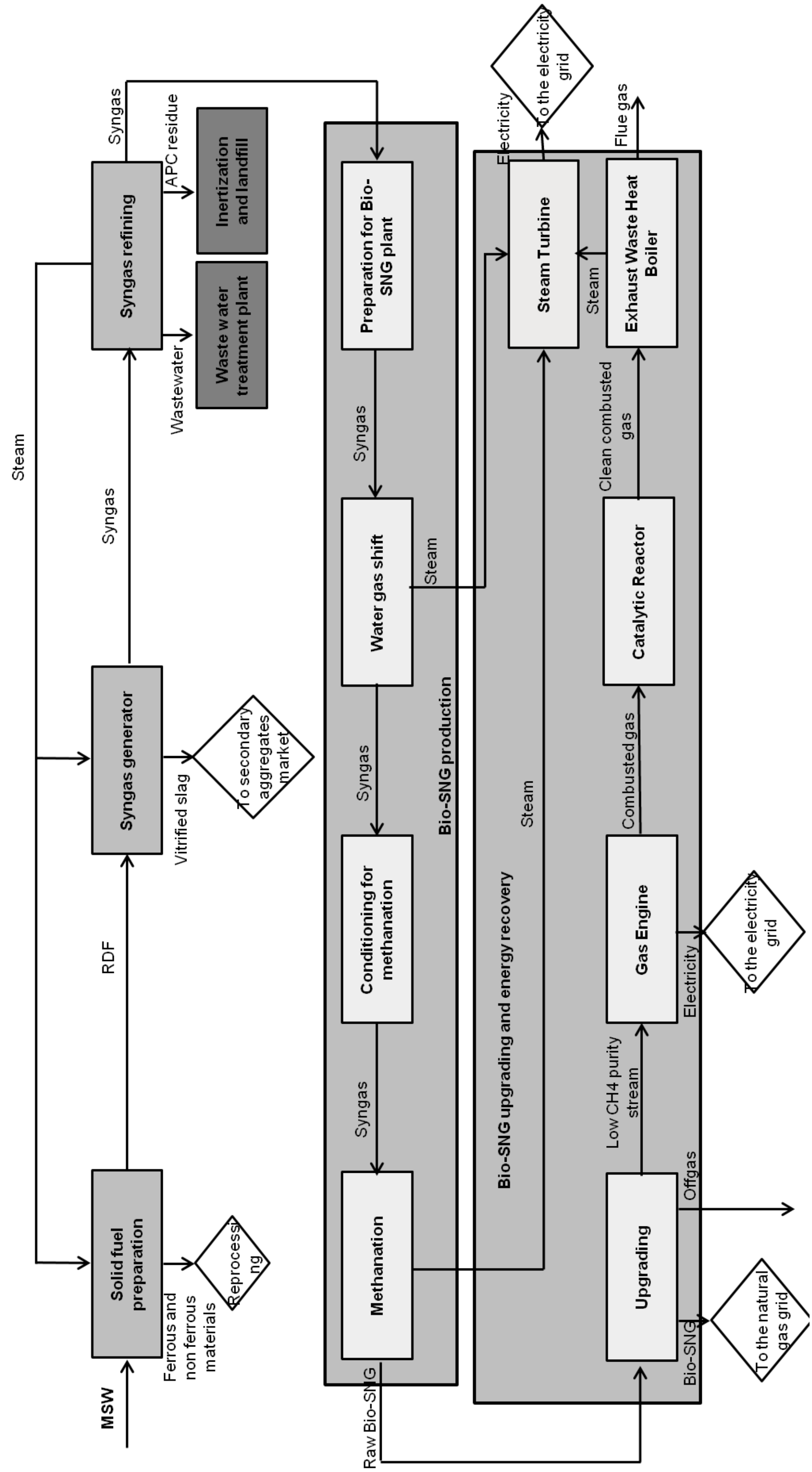


Figure 2.7.6. High level schematic of the advanced dual stage gasification a plasma process for Bio-SNG production.

Bio-SNG production

After the solid fuel preparation, syngas generator and syngas refining, the cleaned syngas undergoes catalytic reactions to increase its methane content and a raw Bio-SNG is produced. The syngas produced in the refining section is cleaned in an active carbon guard bed to remove the remaining contaminants, such as, NH_3, HCl, HBr, HCN and condensable hydrocarbons to <0.1ppmv. The syngas is heated and compressed before entering the water gas shift unit where the H_2/CO ratio is increased using saturated steam ($H_2O + CO = H_2 + CO_2$). A ZnO based guard bed is then used to achieve final polishing of the gas to reduce sulphur and chloride impurities to below the 20ppb in order to minimise the poisoning of the methanator reactor catalyst. After the guard bed, a gas compression system compresses the gas and a heat exchanger adjusts the temperature of the gas to 380°C required by the methanator reactor. The CH_4 content of the gas is then increased in the methanator unit converting CO and CO_2 with hydrogen ($CO + 3\ H_2 = CH_4 + H_2O$ and $CO_2 + 4\ H_2 = CH_4 + 2\ H_2O$).

Bio-SNG upgrading and energy recovery

The raw Bio-SNG produced in the previous section still contains a significant quantity of CO_2. Therefore, before gas injection or utilisation as compressed natural gas in heavy vehicles for example, the gas needs to be upgraded. A number of upgrading technologies are available, including Pressure Swing Adsorber (PSA), chemical or physical absorption and cryogenic separation:

- Cryogenic separation. In the cryogenic method CO_2 is condensed through a series of compression, cooling and expansion steps and it is separated from the gas. The process allows the production of a pure CO_2 liquid that can be stored or sequestered. The cryogenic process, however, is extremely energy intensive. The energy required to keep the system cool (often using liquid nitrogen) makes the current process cost ineffective and therefore, this process has not been considered for the upgrading of the Bio-SNG.
- Physical absorption. The contaminants are absorbed in the solvent according to Henry’s law. The regeneration in physical absorption is usually accomplished by pressure reduction.
- Chemical adsorption. The contaminants are chemical absorbed in the solvent. The equilibrium line in chemical absorption is bowed sharply during saturation of the chemically active solvent component. After saturation of the chemical capacity of the solution, only weak physical absorption is possible, which results

in a steep linear equilibrium curve. Regeneration in chemical absorption processes is almost always accomplished by re-boiling.

- PSAs are columns filled with a special porous material that selectively separate the desired component from the raw gas. At high pressure carbon dioxide and other undesirable molecules, such as N_2, H_2, CO and O_2 are adsorbed on the granular solid by reversible bonds. At lower pressure the regeneration of the catalyst is possible thanks to the desorption of the gas previously captured. A PSA process is cyclic in nature. In a multicolumn process, the gas mixture is fed into the first adsorption column where it is purified and during this time the remaining columns undergo regeneration.

A summary of the advantages and disadvantages of the main technologies is reported in Table 2.7.4.

	Advantage	Disadvantage
Physical absorption	Lower heat of regeneration than chemical absorption (physical solvents can easily be regenerated to low residual loadings by pressure reduction and mild reboiling)	Physical solvents have a high capacity for absorbing water-The raw gas has to be dried
	Higher net CO_2 transfer capacity at high partial pressures	Physical absorption processes usually operate at low temperatures (i.e. Selexol process operates at 5˚ C)
	Offers multipollutant control	At low partial pressure the absorption capacity of the physical solvent is lower than the absorption capacity of the chemical solvent
		Equipment corrosion occurs in the presence of O_2.
		Methane can also be absorbed with CO_2 in the solvent
		H_2S should be preferably removed before CO_2 absorption to avoid absorption in the solvent-This can increase energy requirement for regeneration when absorption is done with Selexol and cause problems when absorption is done with water (When physical absorption is done with water, H_2S is absorbed as well but this is only partially desorbed with CO_2. When regenerating with air, H_2S can oxidise with oxygen to elemental sulphur in the desorption column. This can cause problems with plugging and fouling)
		When physical absorption is done with water, the upgraded gas is saturated with water and needs to be dried
Chemical absorption	Applicable to low-CO_2 partial pressures	Process consumes considerable amount of energy for regeneration (solvents require high energy input for reboiling to achieve low loading of CO_2 and then high purity in the clean stream gas).
	Removal rates of up to 98% and product purity >99 vol% can be achieved	Solvent degradation and equipment corrosion occur in the presence of O_2.
	Fewer trays than physical absorption are generally required for chemical absorption	Fly ash, SOx and NOx in the gas stream combine with the amine to form non-regenerable, heat-stable salts. The levels of SOx should be lower than 0.001% to avoid salt formation with the amine solvent. An electrostatic precipitator to decrease particle content of the gas may be required
	Solvent is selective for CO_2 and no methane is removed	The high temperature (at least 100 °C, higher than the maximum ideal temperature for MEA at 45 °C) associated with flue gases can degrade solvents and lower the solubility of CO_2.
	Operates at atmospheric pressure	Chemical leakage
	High purity of the separated CO_2 stream	

Table 2.7.4. Advantages and disadvantages of the main CO_2 separation process considered by APP to upgrade the raw Bio-SNG.

Adsorption	Can handle also low concentration of CO_2 (0.04%-1.5%)	The sorbents' ability is usually based on pore size. When CO_2 is the target to be selectively adsorbed, gases smaller than CO_2 can also penetrate the pores. N_2 is the gas that most commonly fills up pore space in sorbents.
	Impurity such as Nox, SOx do not cause problems to the adsorption process	Adsorption is slow: the residence time for maximum adsorption depends on the sorbent, but 20 min is a reasonable estimate for typical adsorbing material. When dealing with large volumes of flue gas, as in a power plant, this can be too slow to be practical.
	Equipment must withstand small pressure change	Water needs to be pre- separated to prevent damages to the adsorption process
		H_2S is irreversibly adsorbed-the technique has high security against H_2S but usually H_2S is pre-separated in a cleaning step not to destroy the adsorption material
		The material in the adsorption column can form dust that can plug the valves. Activated carbon creates this problem more often than zeolites

(continued). Table 2.7.4. Advantages and disadvantages of the main CO2 separation process considered by APP to upgrade the raw Bio-SNG.

Thanks to an extensive review process of the technical and cost characteristics of the different separation systems of CO_2, to which the Author has directly contributed while at APP, the company is currently developing the process using a PSA system.

The raw Bio-SNG is upgraded in two Pressure Swing Adsorption systems (PSA). The top gas of the first PSA constitutes 97% w/w clean Bio-SNG product compliant with the regulation limit for injection into the gas grid [172]. The tail gas of the first PSA unit undertakes a second separation in order to obtain a top gas containing about 80% mol/mol of methane and a high purity CO_2 stream that is vented to the atmosphere after flaring. The low methane purity stream can be used for electricity production. The steam recovered throughout the Bio-SNG production section is used in a steam turbine to produce electricity. The net energy conversion efficiency of the process (i.e. energy content of SNG fuel and power as a percentage of the energy content of the RDF) is calculated by the APP to be between 55-60%.

2.8 Conclusions

This chapter highlighted the challenges that current societies, and in particular the UK, are facing in order to contribute to the reduction of environmental threats derived from climate change and at the same time to ensure a stable and reliable energy supply. The plan developed by the UK government to face these challenges includes the development of the fossil component of the energy mix, especially gas, and strong emphasis on renewable sources. In this context, shale gas and LNG are seen as bridging fuels that will help to meet short-term emission targets for power generation, heating and transport while restructuring the electricity mix towards renewables. This thesis focuses on these two fossil supplies and explores their environmental burdens.

At the same time, renewable energy production from waste is seen as key to meet the national and international targets on carbon emissions and renewables. The UK government is supporting the production of renewable electricity and methane from advanced waste treatment technologies. In this context, Advanced Plasma Power has developed an advanced gasification technology able to deliver the production of electricity or Bio-Substitute Natural gas (Bio-SNG), which is also the focus of this thesis.

In light of the above, the main research aims addressed in this thesis are here refined.

- Indigenous shale gas is reported to have the potential to provide the UK with a greater energy security. However, fierce opposition for possible environmental threats of shale gas exploitation have arisen. This thesis analyses the environmental burdens of shale gas production in the UK, including water pollution, air pollution and land impacts. This thesis provides the analysis of different process options and explores the comparison with conventional gas supply.
- Major international joint ventures have seen the UK engaged in increasing the amount of imported LNG. The LNG is advocated as a clean fuel, able to reduce the environmental impacts of the transport and energy sectors. This thesis challenges this view and reports on the life cycle environmental burdens of the LNG production and transport from Qatar to the UK.
- Public investments in the UK support the research and development of advanced waste to energy technologies. Possible environmental benefits of advanced waste processes are unclear due to their early development stage. This thesis proposes a life cycle framework able to compare the environmental burdens of conventional and advanced waste technologies producing electricity.
- Methane production from waste is also financially supported by the government and is projected to fully develop in the near future. However, trade-offs between electricity and methane production from waste are unclear. This thesis covers this gap and analyses biological and thermal processes for methane or electricity production from waste and identifies the condition in which the legislations should sustain either one or the other energy type.

Chapter 3. **Life cycle assessment: general methodology and applied framework**

In this chapter the methodology of Life Cycle Assessment (LCA) is firstly reviewed; the theory and the modelling principles, on which the methodology is based, are explained. Then, the framework developed in this study and analysed through the LCA methodology is reported.

3.1 A general methodology for environmental issues: the Life Cycle Assessment methodology

Concerns about the environment have become primary points of discussion for policymakers, public administrators, businesses and individuals. Environmental considerations are often integrated into challenging decisions regarding our society [241]. The application of the life cycle approach and the Life Cycle Assessment (LCA) are now increasingly being required by EU legislation [242]: Life Cycle Assessment can help decision makers to assess the environmental burdens associated with a product or a system [243–246]. This tool systematically analyses the entire life cycle of valuable products from raw material extraction to the product final disposal, including manufacturing, transport, use, re-use, maintenance and recycling; it describes and assesses all flows to and from nature, from a ‘cradle to grave’ perspective [247,248]. The LCA can be applied to many diverse systems and fields including decision making (e.g., product design and development) and communication (e.g., asserting environmental product claims) [249]. Life Cycle Assessment considers environmental concerns about natural environment, fossil resources, human health and threats to the ecosystem [250,251].

The European Commission [150] refers to LCA as the best framework for assessing the potential environmental impacts of products currently available. In brief, LCA practitioners build a life-cycle model of the product of interest and the inputs and outputs of each process in the model are quantified and calculated as either resource use or emissions in the product life cycle (emission to soil, water and air). These inputs and outputs are interpreted as potential impacts to the environment.

Following the international standardised method [250], the LCA methodology establishes a rigorous approach for the analysis of the environmental burden of a product or service. LCA consists of four very distinct phases, as reported in Figure 3.1.1.

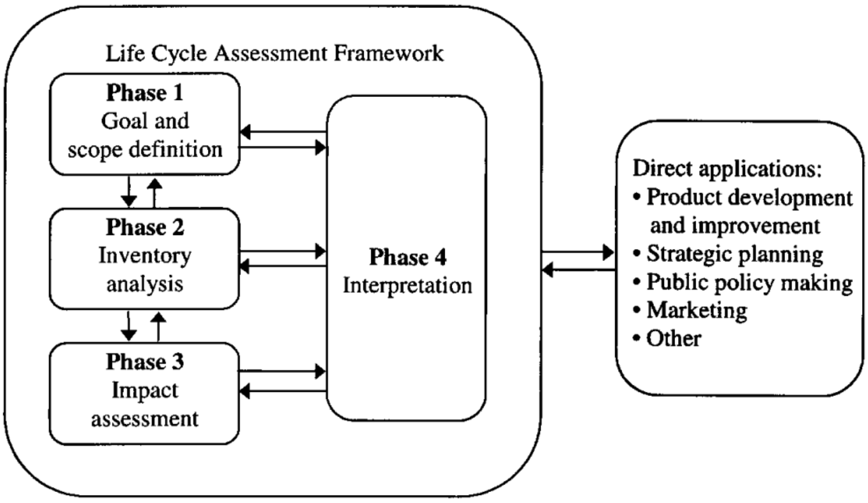


Figure 3.1.1 Phases in Life Cycle Assessment [252].

These LCA phases are:

1. Goal and scope definition.
2. Inventory analysis.
3. Impact assessment.
4. Interpretation.

A LCA study is an iterative process because earlier phases may be revisited in light of the results of later phases.

3.1.1 LCA: consequential or attributional?

Depending on the purpose of the study, two methods of LCA can be distinguished: attributional and consequential [244]. These two approaches answer different questions, and a failure to distinguish between them can result in the wrong method being applied or misinterpretation of the results [253].

In the attributional assessment, the system under analysis is assumed to be existing and the average impact of the product is calculated. Attributional LCA focuses on the burdens due to relevant physical flows to and from the studied system and analyses the impact of the technologies used to produce valuable goods/services [254]. It allows the comparison between direct impacts of different products and it is a tool to identify the most polluting part of the process. Attributional LCA aims to answer to the question 'What environmental impact can be associated with this product?' [255]

Consequential LCA provides information about the consequences of changes in relevant flows in response to possible decisions [254,256]. Consequential studies model the environmental effects due to the decision of changing one or more variable of the process. This type of studies aims to answer to the question 'what would happen if...'.

It has been widely shown in the literature how the two different methods can lead to differences in the results [257,258]. Attributional LCAs are generally based on stoichiometric relationships between inputs and outputs, and the results may be produced with known levels of accuracy and precision, whereas consequential LCAs depending on suppositions, provide a lower level of accuracy [253]. Therefore, some [259,260] argue that consequential LCA should be used for decision-making but not when the uncertainties in the consequential modelling outweigh the insights gained from it. Others [261] agree with the previous authors and add that it is also important to increase the understanding of the product chain and to identify the processes and relations to be improved. Conversely, some authors [262,263] report that attributional and consequential LCA can be indiscriminately applied for modelling of past, current and future systems.

These two types of LCA lead usually to different methodological approaches regarding the type of data used in the LCA model. 'Average data' represent the average environmental burdens for producing a unit of the good and/or service in the system; whereas 'marginal data' represent the effect of a small change in the output of goods and/or services on the environmental burdens of the system [254]. Marginal data represent the production technology affected by the induced change to the system [264].

Depending on the question to be answered in the study, either one or the other type of data can be used [265]. As an attributional LCA aims to quantify the environmental load of a specific system, average data are usually preferred; conversely, in a consequential approach marginal data are mostly used [266].

Some studies cannot be strictly categorized as attributional or consequential [267] : this is also the case for the study presented in this thesis. This work is an attributional analysis with a system expansion approach (or also consequential approach) because it analyses the environmental impacts of different scenarios without including possible variations to the valuable products, but allocation is performed using the method of system expansion that is usually typical of consequential analysis (see the following paragraphs). Average data have usually been used in this thesis. However, it is reported [268] that to increase the robustness of the environmental assessments, relevant sensitivity analyses, including the effects of selecting different marginal technologies should be carried out. In particular, the decision to use marginal data can be significant for the modelling of electricity production [269]. Hence, sensitivity analysis using a consequential approach has been used for the development of the analysis reported in this thesis.

3.2 Goal and scope definition

In the goal and scope definition, the purpose of the study has unambiguously to be defined and also the following points should be addressed [252] i) why LCA has to be carried out; ii) what political or technical decision will depend on the results of the study; iii) what are the system boundaries for the study iv) what is the basis for comparison between different alternatives. The scope of the study has to be defined in relation to how the results are to be used and the necessary data and information needed in the following phases have also to be identified [270]. In this phase, the decision of the inclusion of the hot spot analysis is taken: this is used to define the unit operations within the analysed process that mostly contribute to the total environmental burdens [271].

3.2.1 Functional unit

LCA studies are commonly performed to compare alternative ways of delivering some functions. The basis for comparison, common between all alternatives, is termed the functional unit of the study. The results of the entire analysis are strongly related to the choice of this quantity [270] as will also be shown in this work.

3.2.2 Foreground and background

Many authors report how the subdivision of the system into a foreground and background is useful for the definition of the system boundary and can guide the choice of the type of data to use in the model [252,259,272]. The foreground and background processes of a generic system are exemplified in Figure 3.2.1.

The concepts of foreground and background systems were developed in 1999 by SETAC [273]. Decision changes on the system analysed affect both foreground and background. The foreground system comprises the processes whose selection or mode of operation is directly affected by changes of the decision on the study. The background system consists of all other modelled processes influenced by measures taken in the foreground system; it includes the economic activities which exchange materials and energy with the foreground. It is reported [259] that a sufficient but not necessary condition for a process or a group of processes to be in the background is that the exchange with the foreground takes place through a homogeneous market (marketplace in which the products traded are very similar although there may be some minor differences in design).

The distinction between foreground and background does not imply any distinction between the importance of the burden related to those systems. The environmental impact of anyone of the two can be the largest.

It is advised [252] to use a different kind of data set in the inventory analysis for the foreground and the background. If available, primary data of actual processes under study should be used to analyse the foreground. Conversely, average industry data, taken from a reliable database, or marginal data, can be used to describe the background system if primary data are not available.

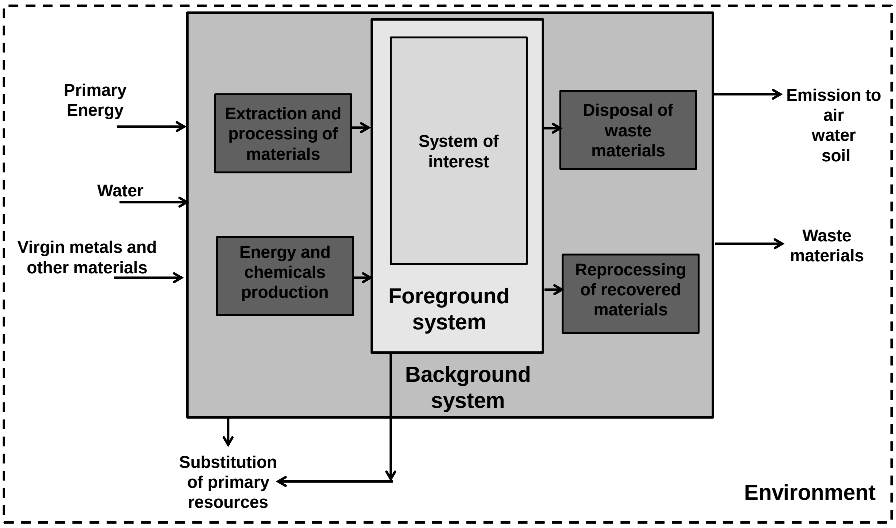


Figure 3.2.1. Foreground and Background of a generic system.

3.3 Life cycle inventory

This phase involves the searching and the collection of an enormous amount of data.

The studied process is modelled in a schematic flowchart [255] whose level of detail depends on the purpose of the study and on the points reported in the goal and scope definition. Material and energy balances of the studied process are compiled ignoring environmentally irrelevant flows but focusing the attention also on the trace emissions of high environmental impact: species, such as chlorinated ones, dioxins, furans, might not be relevant for the mass balance but have a dominant contribution to the toxic effect of the process [252].

Inputs to the system boundary are quantified and traced back to the primary materials extracted from earth; outputs from the system boundaries are also identified and quantified. The collected data are then scaled according to the chosen functional unit.

3.4 Impact assessment

The impact assessment phase is defined [270] as a quantitative and/or qualitative process that identifies, characterises and assesses the potential impacts of the environmental interventions listed in the inventory analysis. Mass and energy flows are translated into environmental impact, such as global warming, acidification, etc. The impact assessment consists of different phases [251], as here specified:

- Classification. The inputs and outputs previously collected are qualitatively sorted according to the type of environmental impact they contribute to and assigned to a specific impact category such as resource depletion, global warming, acidification, etc. Certain species can be assigned to more than one category.
- Characterization. The interventions are translated into an environmental burden based on a single unit of measure specific for each impact category. The environmental impacts are quantified into category indicators, using publicly available equivalency factors obtained by modelling cause-effect chains [255]. These equivalency factors indicate how much a substance contributes to a category indicator (such as the Global Warming Potential, the Acidification Potential, etc.) compared to a reference substance and this value depends on the characterization method used. Characterization methods are based on physico-chemical mechanisms that link the compound to its environmental disruption. The characterization phase takes into account both the environmental magnitude and the potency of polluting compounds [274]; the magnitude is accounted when considering the mass flow of the input/output species whereas the potency is accounted when considering the equivalency factors. The quantification method to calculate the impact related to a general category x is summarized in equation (3.4.1):

$$x = \sum_{i=1}^{n} e_i \cdot m_i \tag{3.4.1}$$

Where x is the environmental burden according to the general impact category, n is the total number of mass species contributing to this category, i is the species, e is the equivalency factor of species i and m is the mass of species i.

- Normalization, grouping and weighting are not compulsory. In the normalization step the results of the characterization phase are normalized referring to a reference value. In doing this, all the impact categories become dimensionless and the impact to which the studied process is contributing most can be identified. Grouping includes sorting and ranking of the indicators. The weighting phase aims at converting and aggregating indicator results across impact categories and this results in a single score (such as in the single score of the ready-made assessment methods of the Ecoindicator'99, the EPS, etc.) [252]. Further information is reported in Bauman *et al.* [255].

The impact categories, the equivalence factors, the indicators and also the factors used for normalization and weighting depend on the impact assessment methods used. The most important [275] for Europe are CML (Institute of environmental science, Universiteit Leiden), ILCD (the international reference life cycle data system), Recipe, etc. whereas for North America they are Traci, Bees, etc.

3.4.1 Environmental indicators

A distinction between mid-point and end-point indicators is widely reported [254,255,276]. Endpoint indicators are defined at the level of the areas of protection; these are of easier public interpretation as they address socio-economic issues, but have higher uncertainty. Conversely, mid-point indicators are defined in a point in the cause-effect chain (environmental mechanism) of a particular impact category, prior to the endpoint, at which characterization factors can be calculated to reflect the relative importance of an emission or extraction in a Life Cycle Inventory. For example [277], as shown in Figure 3.4.1, Global Warming impacts involve a series of steps, starting with the release of greenhouse gases, and ending with impacts on humans and ecosystems. There is a point where the greenhouse gases have an effect on the radiative forcing. Greenhouse gas emissions have a pathway that is different before that point, but identical beyond that point. Therefore, the radiative forcing provides a suitable indicator for the midpoint impact category of Global Warming.

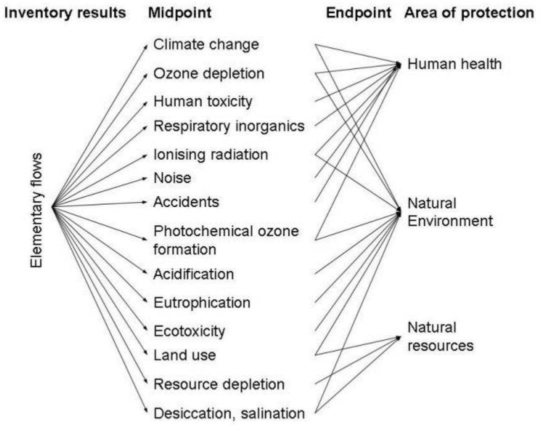


Figure 3.4.1. Characterisation modelling at midpoint and endpoint levels [277].

In this study, mid-point indicators are analysed: these show lower uncertainties even though they are more difficult to interpret because they often have an abstract meaning [278]. For studies related to the European condition, the ILCD or the CML assessment methods are the most appropriate. Both methodologies are very similar except for the 'particulate matter' indicator that is not reported in the CML methodology (this latter indicator is not relevant for the purpose of this study). In this study the CML methodology Method Characterization factors, version 4.5 (April 2015) which is based on the ISO standards [250] is used as this is the oldest and most robust. The environmental indicators used are described according to the CML impact assessment method [279].

The *Global Warming Potential (GWP)* characterizes and calculates the impact of greenhouse gases based on the extent to which these gases enhance the radiative forcing. GWP values for specific gases, developed by the Intergovernmental Panel on Climate Change (IPCC), express the cumulative radiative forcing of an emission over a given time period in terms of the quantity of carbon dioxide giving the same effect [280,281].

The *Acidification Potential (AP)* quantifies the impact of acid substances and precursors such as SO_2, NO_x and HCl. Rain, fog and snow trap the atmospheric pollutants and cause fish mortality, leakage of toxic metals from soil and rocks and damage to forests and to buildings and monuments.

The *Abiotic Depletion (ADP)* addresses the problem of the diminishing pool of resources, focusing on the depletion of non-living resources such as iron ore, crude oil, etc. The abiotic depletion is usually measured in MJ when the deployment of energy sources is assessed. Otherwise, when the depletion of virgin metal is assessed, it is measured in Sb eq.

The *Eutrophication Potential (EP)* includes all pollutants that promote microbiological growth leading to oxygen consumption, such as "algal blooms". Nitrogen and phosphorus are the two main nutrients implicated in eutrophication: they can cause shifts of species composition and increased biological productivity.

The *Photochemical Ozone Creation Potential (POCP)* quantifies the potential to create tropospheric ozone, expressed in equivalents to ethene as the reference species.

The *Ozone Layer Depletion Potential (ODP)* quantifies the thinning of the stratospheric ozone. Chlorinated and bromated substances increase the rate of ozone destruction.

The *Toxicity* impacts include many types of indicators causing damages to different environments based on both the inherent toxicity of a compound and potential exposure. Toxicity categories indicate the toxicological impacts of pollutants emitted to the environment, such as neurological damage, carcinogenic, mutagenic, etc. For this reason, toxicity indicators can be divided in *Human (HTP)* and Eco-toxicity (that is fresh *water (FAETP)*, *marine (MAETP)* and *terrestrial (TETP)*). FAETP and MAETP assess the toxic effects of polluting compounds to water life, while TETP is related to land based ecosystems.

Those impact categories which are considered most significant for the purpose of this thesis [282] are summarized in Table 3.4.1.

Impact categories	Impact Indicator	Acronym	Characterisation model	Units
Climate change	Global warming potential	GWP	CML 2001 baseline [283] (Apr. 2015)	kg CO_2eq
Acidification	Acidification potential	AP	CML 2001 baseline [284] (Apr. 2015)	kg SO_2eq
Resources depletion (fossil)	Abiotic depletion	ADP	CML 2001 baseline [279] (Apr. 2015)	MJ
Eutrophication	Eutrophication potential	EP	CML 2001 baseline [284] (Apr. 2015)	kg phosphate eq
Photochemical ozone formation	Photochemical ozone creation potential	POCP	CML 2001 baseline [285] (Apr. 2015)	kg ethane eq
Ecotoxicity (freshwater)	Fresh water aquatic ecotoxicity potential	FAETP	USEtox model [286] (Apr. 2015)	kg DCB[1] eq
Ecotoxicity (terrestrial)	Terrestric ecotoxicity potential	TETP	USEtox model [286]	kg DCB eq
Human toxicity	Human toxicity potential	HTP	USEtox model [286]	kg DCB eq
Ozone depletion	Ozone layer depletion potential	ODP	CML 2001 baseline [287] (Apr. 2015)	kg R11[2] eq

Table 3.4.1. Impact categories and indicators used in this study.

3.4.2 System expansion or allocation

One problem that arises in LCA studies is how to allocate the environmental burdens when the studied process is multifunctional. A multifunctional process is a process/activity that at the same time achieves more than one function [288]. It is difficult to decide how to partition the environmental interventions between the multiple functions.

The problem of allocation is widely debated in literature [259,289–294]. It is recommended [255] avoiding allocation based on physical and economical relationships but the expansion of the system boundaries should be performed [288] instead.

The following examples explain how the system boundaries can be expanded to avoid other allocation methods [278,295]. The first example refers to the assessment of the environmental burdens of a single multifunctional process whereas the second example shows the comparison of two processes.

As reported in Figure 3.4.2, the system under analysis ‘process A’ delivers two valuable products/services, product 1 and product 2; the environmental burdens of the entire system are to be allocated between them. For doing this, a parallel process (Figure 3.4.2), ‘process B’, which delivers only one of the two functions delivered by process A, is considered. The increased production of product 2 from process A reduces the demand of product 2 from process B. In the system expansion methodology the two systems are lumped together, as shown in Figure 3.4.2. Hence, the environmental impacts associated with product 2 are assumed to be the same as those produced from system B and the remaining burdens are allocated to product 1.

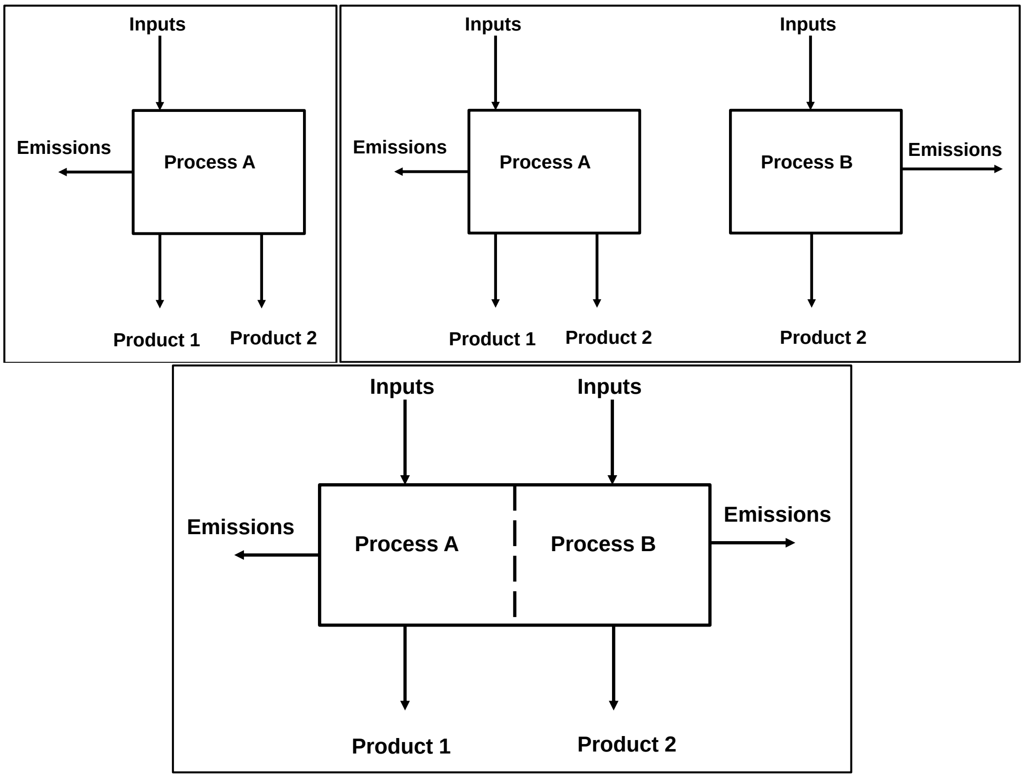


Figure 3.4.2. System expansion methodology, single multi-functional process.

In the second example, Figure 3.4.3, system I produces products A and B and system II produces only product C. When the aim of the study is to compare product A to product C, the method of system expansion can be applied in two ways. The system boundaries are expanded so that an alternative system III for producing B is included in the analysis, Figure 3.4.3. The new two comparing systems have now both become two multifunctional systems; system I produces A+B and system II produces B+C and the two can be compared. The same can be done using an equivalent approach, Figure 3.4.3: the burdens arising from system I are subtracted by the burdens arising from system III. In this case, system I produces only A and is directly comparable to system II producing B. The environmental burdens of the main system analysed are decreased by the ‘avoided burdens’ and hence the total burdens can result in being negative. The latter approach is known as the ‘avoided burden method’; it is widely used for example, in the field of waste management. Valuable energy and materials produced substitute for activities included in the background system.

This method determines a complete and accurate model but the system itself and the boundaries are complicated. In addition to this, to apply this methodology, system III must represent a realistic alternative process available on the market.

When allocation cannot be avoided alternative method including mass, energy or exergy content, volume and molecular mass can be used for allocating the burdens of a multifunctional system.

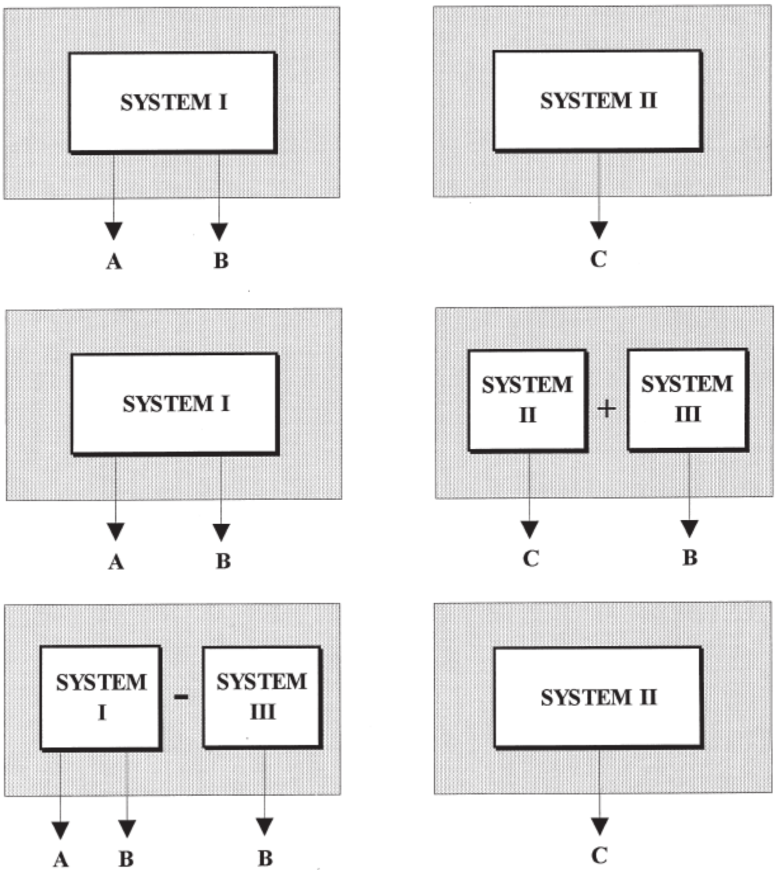


Figure 3.4.3. System expansion methodology, comparison of two systems [295].

3.4.3 Direct, indirect and avoided burdens

Many authors [47,252,272] identify three different groups of environmental burdens associated to a system:

1. Direct contributions are directly linked to the activities due to the process under study and arise from the foreground system;
2. Indirect contributions arise from the background, from the supply chains of materials and energy provided to the foreground;
3. Avoided burdens (see paragraph 3.4.3) are associated with the activities displaced by the production of valuable materials.

3.4.4 Biotic carbon-biogenic emissions

The biotic carbon concept is widely used in the methodology of LCA when energy production from bio sources is analysed [296,297]. This relates to the carbon emitted as CO_2 from a process dealing with biodegradable material, such as thermal processes treating waste (all or part of the waste treated usually derives from bio-degradable materials including paper, textiles, food and garden waste).

The uptake of carbon dioxide from the atmosphere during the photosynthesis process is characteristic of plant biomass. Biotic carbon was originally removed from the atmosphere via photosynthesis, and under natural conditions, it would eventually cycle back to the atmosphere as CO_2 as a result of degradation processes [298].

The combustion of biomass and bio-degradable materials causes the emission of part of the carbon (mainly as CO_2) that the biomass contributed to absorb during its growth, Figure 3.4.4 [299]. Hence, non-fossil CO_2 (that relates through the mass balance to the biotic carbon content of the input material) is considered by the Intergovernmental Panel on Climate Change (IPCC) to be part of the natural carbon balance, and therefore not a contributor to the atmospheric concentrations of CO_2.

The methodology to account for biogenic emissions is a widely debated subject in literature. According to IPCC the biogenic emission from biomass combustion should not be included in the count for GWP only when biomass is sustainably produced. If biomass is harvested at unsustainable rates, that means faster than annual re-growth, also the biotic carbon should be considered for the calculation of the global warming potential. The same is also stated in other works [300]: biomass often needs many years to mature, therefore, by the time it grows up, the amount of carbon released by the plants replacing fossil fuels with bio-energy results in a net carbon emission to environment. Also, if the biomass is not harvested and left in forest, it is true that it will eventually degrade and enter the atmosphere, but this degradation can however be quite slow, and the time frame has to be extended to several centuries before all biotic materials have been degraded [301].

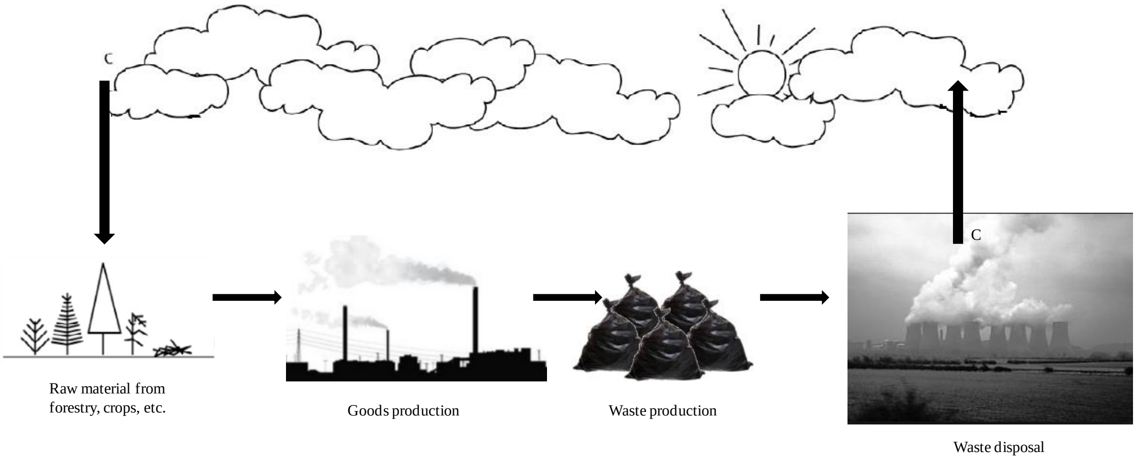


Figure 3.4.4. Biogenic carbon cycle.

Many authors consider appropriate to exclude biogenic CO_2 from the total GWP and they do so in their works [162,296,300,302–305]. On the other hand, it is reported [301] that it might be inappropriate to exclude the biogenic carbon from the GWP when analysing processes dealing with biogenic carbon in different ways; in some cases this practice can lead to erroneous results. For example, when incineration and landfill of wood are compared, 100% of the carbon contained in the wood is emitted as CO_2 in the incineration process whereas in landfill only 70% of this carbon is emitted as CO_2. The remaining 30% is trapped in the landfill soil. Thus, there is a difference between incineration and landfill because in the first case all carbon is emitted whereas in the second case some of the carbon is trapped. If the biotic carbon is entirely considered neutral this difference is disregarded and this can lead to mistakes.

In LCA studies on waste management systems, the so called ‘zero burden approach’ is often used [306,307]: the boundary of the waste life-cycle study starts from the moment when the material becomes waste, continues through the treatment processes until the material ceases to be waste and becomes an emission into air, soil or water, inert material in a landfill, or a useful product. When this methodology is applied, there is no need to look at the biotic carbon content of waste and biogenic emissions: the system boundaries do not include the life of the waste before it became a waste and hence, its bio-degradable content has not to be accounted.

Furthermore, when the aim of the study is to compare alternative processes treating the same input material (such as different thermal waste management options, all treating the same waste), the exclusion of the biogenic emissions from the total GWP does not alter the relative performances of all processes as the biogenic emissions count the same for all scenarios.

3.5 Interpretation

The last phase of a LCA analysis is often referred to as interpretation; it is the process of assessing results in order to draw conclusions [255]. The findings of the inventory and assessment phases are analysed in conjunction with what was reported in the goal and scope definition. The results should not only be used to state the status quo but also to identify possible environmental improvements.

3.6 Limitation of the LCA methodology

Many authors report on the limitations of the LCA methodology [301,308,309]. An artificial simplification of the real word with some neglected fluxes arising from human activities [309] and with some imperfect modelling determine the following main limitations:

- The focus on a global scale. LCA does not allow the identification of the level of pollutants and their effects in the local area where those have been emitted.
- Steady-state modelling. LCA does not include varying of temporal characteristics.
- Linear modelling. In LCA all flows and burdens are linearly scaled according to the functional unit without considering plant scale, economic and market effects.
- Inputs to and outputs from the product system. LCA mainly focuses on impacts that are related to raw material extraction, emissions and land use. Some type of environmental burdens, such as noise and smell, do not fit into the indicator schemes.
- Time pattern of impacts. LCA focuses on regular occurring environmental interventions but usually disregards irregular emissions (such as emissions that cannot be described on a yearly basis).

3.7 Examples

The LCA methodology is exemplified in this section according to the work of Thomassesn *et al.* [258]. It is demonstrated how to perform an ALCA on a mass and economic allocation and a CLCA with system expansion of an average conventional milk production in The Netherlands. The chosen functional unit is ‘1 kg of fat and protein corrected milk leaving the farm gate’. Table 3.7.1 shows key inventory data of the average conventional milk production system based on data of 286 conventional farms in The Netherlands from 2003. The only two outputs from the farm are milk and animals (mostly bull calves and milking cows).

Characteristic	Unit	Value
Grassland	ha	29.9
Arable land	ha	8.6
Milking cows	Amount	63
Heifers	Amount	25
Breeding calves	Amount	21
Electricity use	kWh	25690
Diesel use	l	4780
Natural gas use	m^3	1430
Milk production	kg/cow	7630
Fat content	%	4.42
Protein content	%	3.49
Pesticides	kg/ha	0.25
Concentrates	kg/cow	2160
Attributional		
90 DVE =The intestine digestible protein content of the concentrates based on the Dutch DVE system	Tonnes	85
120 DVE =The intestine digestible protein content of the concentrates based on the Dutch DVE system	Tonnes	43
180 DVE =The intestine digestible protein content of the concentrates based on the Dutch DVE system	Tonnes	7
Consequential		
Soybean meal	Tonnes dry matter	71
Spring barley	Tonnes dry matter	64
Purchased artificial fertiliser	kg N/farm	5750
Exported animals	kg N/farm	650

Table 3.7.1. Description of the main characteristics of the average conventional milk production system in The Netherlands for the year 2003. Taken from [258].

The boundary of the system and the operations analysed are identified in Figure 3.7.1 and Figure 3.7.2 according to the two different LCA approaches.

Figure 3.7.1 shows the ALCA flowchart of the system based on average historical data. An electricity mix for The Netherlands is used. Purchased concentrates are related to three groups of concentrates with different protein and energy contents. Each group of concentrates have a different composition (see also Table 3.7.1). The life cycle inventory of each ingredient includes cultivation, processing and transport.

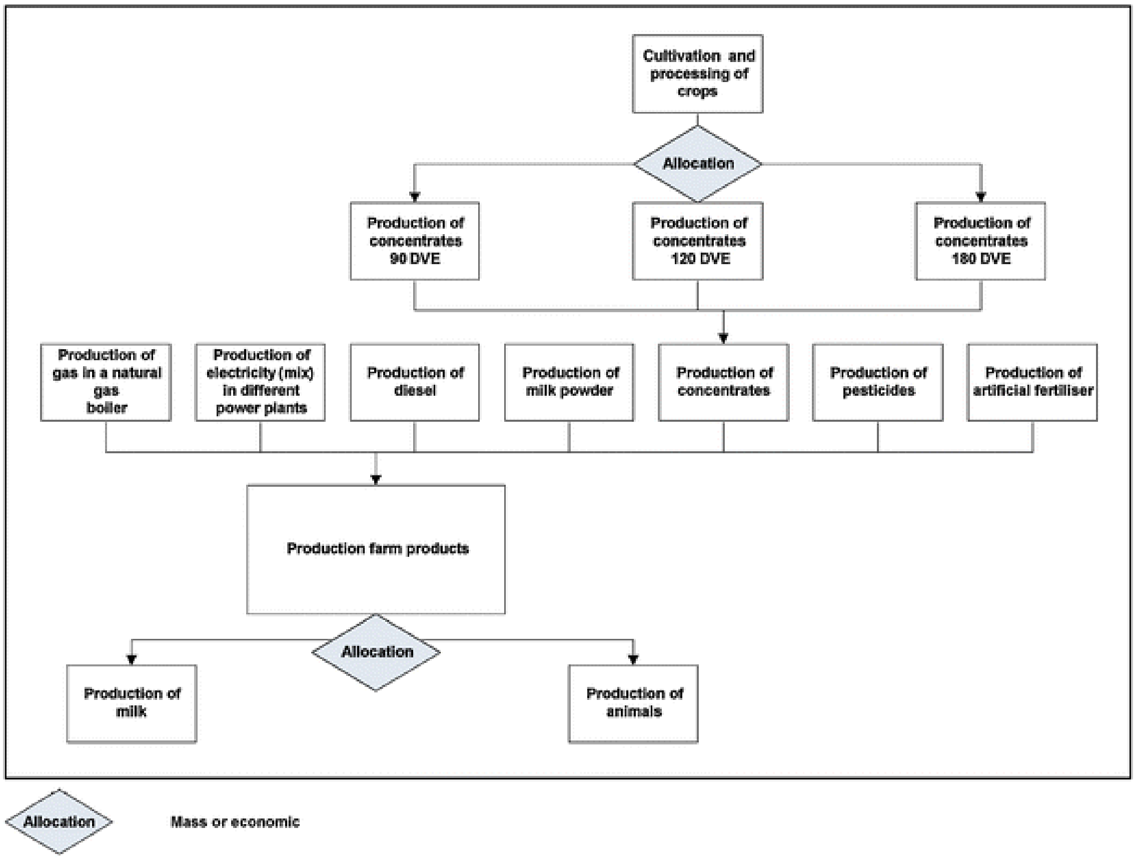


Figure 3.7.1. Flowchart for the attributional LCA of conventional milk production with allocation. Taken from [258].

Figure 3.7.1 also indicates where the allocation of co-products is encountered. Allocation problems occur when concentrate ingredients are part of a multi-functional process and when dividing the environmental burden between milk and animals. Mass allocation is applied by computing the share in quantity of a product. Economic allocation is applied by computing the share in revenues of a product by taking into account quantity and economic value of the products. Table 3.7.2 shows the mass and economic allocation factors used by Thomassesn *et al.* [258].

Product	Mass Allocation (%)	Economic Allocation (%)
Milk	96	92
Beet puòp	20	15
Molasses	10	5
Maize gluten meal	2.5	8
Maize gluten meal (pairie gold)	2.5	10
Palm kernel meal	11	3
Rape seed meal	56	38
Soy hulls	3	1
Tapioca	22	27
Triticale grain	60	71
Wheat grain	61	85
Wheat hulls	17	9

Table 3.7.2. Overview of allocation factors within attributional LCA. Taken from [258].

Figure 3.7.2 shows the CLCA flowchart of the system based on marginal data. CLCA reflects the possible future environmental impact from a change in demand of the product under study (milk). The size of change in demand is an increase in milk production, which needs at least one more dairy farm. For the CLCA marginal data are used.

For the marginal input of electricity, the question to be asked is: what kind of electricity plant will satisfy the increased electricity demand due to the increased production of milk in The Netherlands? On the base of actual data obtained by the energy sector, Thomassesn *et al.* [258] identified the next power plant in The Netherlands to be a natural gas power plant.

In the case of feed, the question to be asked is: which feed ingredient will meet the increased protein demand of cows as a result of the increased milk production? Taking into account the market trend, production volume and price, Thomassesn *et al.* [258] identified the soybean as the marginal fodder protein.

Thomassesn *et al.* [258] expanded the boundaries of the system investigated to include the alternative production of exported functions with an avoided burden method. Figure 3.7.2 shows the avoided products when the chosen increase in milk production (at least one more dairy farm is needed) occurs. For example, soybean meal has the co-product soybean oil. Therefore, increased demand for soybean meal leads to increased production of soybean oil, which substitutes palm oil, as Figure 3.7.2 shows. However, when less palm oil is produced, also less palm kernel meal is produced. To compensate for this ‘missing’ palm kernel meal, more soybean meal should be produced. Both soybean meal and palm kernel meal are used as feed for livestock. According to Figure 3.7.2, the avoided production of palm kernel meal is compensated by both soybean meal and spring barley production. This is because the substitution ratio is based on both energy and protein content of the meal, and as the protein and energy content differs between palm kernel meal and soybean meal, part of the palm kernel meal is substituted by spring barley.

The milk system is also expanded because milk is associated with the co-product of beef. When identifying the avoided burden of meat from dairy cows, the question to be asked is: what will not be purchased by retailers/supermarkets when more meat from dairy cows is provided? Thomassesn *et al.* [258] assumed that meat both from calves and dairy cows substituted beef and pork.

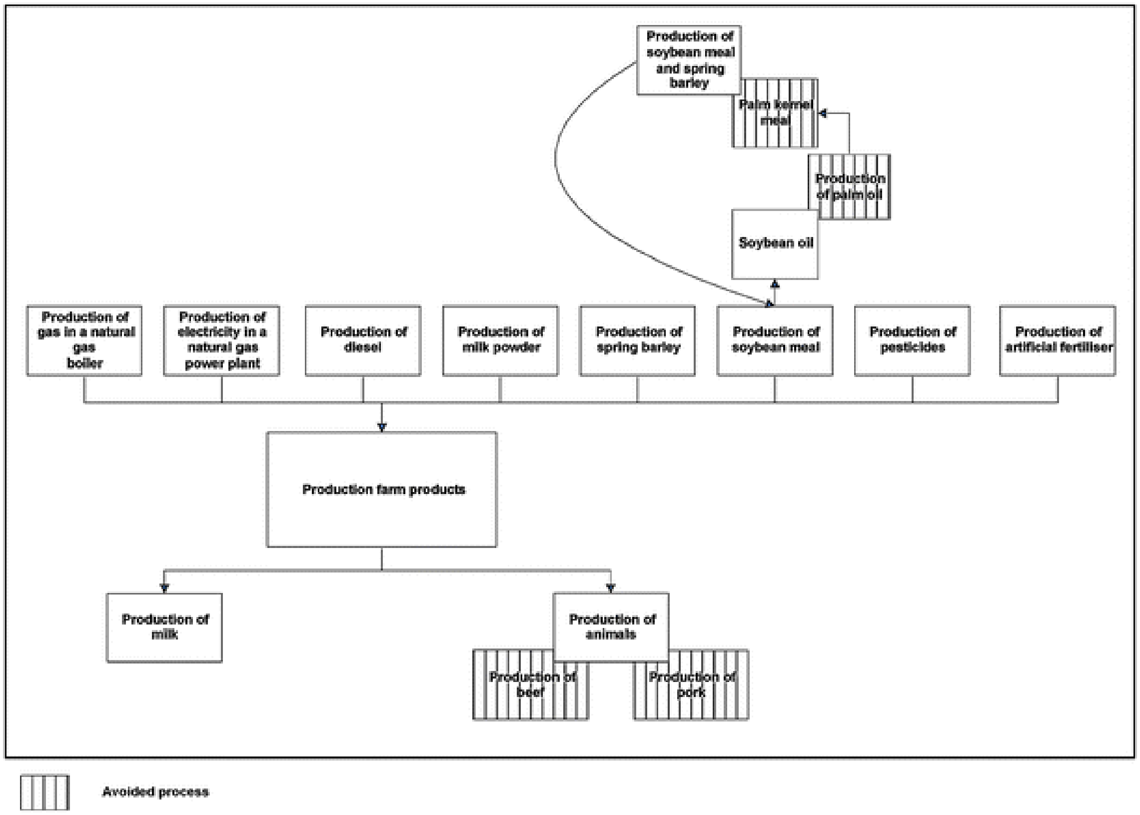


Figure 3.7.2. Flowchart for the consequential LCA of conventional milk production with system expansion. Taken from [258].

Table 3.7.3 shows the results of the average conventional milk production system using ALCA and mass and economic allocation, besides CLCA and system expansion. Table 3.7.3 shows that when using mass or economic allocation within ALCA, total environmental burdens change slightly. Furthermore, the energy use computed by CLCA is only 35−45% of the energy use found by ALCA. Acidification computed by CLCA is around 40% of acidification by ALCA, climate change 55−60%, eutrophication 65−70%. These lower values of CLCA are mainly caused by the subtraction of avoided burdens of identified alternative products. Difference in feed type within ALCA (three concentrates with different compositions) and CLCA (spring barley and soybean meal) is the main cause of the lower energy use. Both avoided beef production and difference in feed type within ALCA and CLCA caused lower acidification, eutrophication and climate change.

Impact category	Unit/kg of functional unit	Attributional mass allocation	Attributional economic allocation	Consequential system expansion
Fossil energy use	MJ	5.77	6.91	2.55
Eutrophication	g NO3-eq	163	170	113
Acidification	g SO2-eq	10.9	11.2	4.78
Climate Change	gCO2-eq	1560	1610	901

Table 3.7.3. Characterized results of the average conventional milk production system using attributional LCA and mass, economic allocation, besides consequential LCA and system expansion. Taken from [258].

3.8 System boundary

In light of the above analysis of the LCA methodology and of the objectives reported in chapters 1 and 2, Figure 3.8.1 summarises the system boundaries of the work reported in this thesis. Energy technologies, developing in the future UK energy mix are analysed under a life cycle perspective. Future energy mixes in the UK include the novel and unconventional supply of fossil natural gas: LNG and shale gas production, processing and distribution are analysed and compared to the conventional current supply of natural gas to the UK. Then, waste-to-energy technologies for electricity production are studied; conventional and advanced technologies, including plasma gasification, pyrolysis, incineration and landfill are compared. A detailed hot spot analysis is performed for the Gasplasma advanced dual stage thermal process treating a wide range of solid waste. The focus is moved to processes producing methane from renewable. The production of bio-methane/Bio-SNG from waste treatment processes is studied; the hot spot analysis of the Gasplasma advanced dual stage thermal processes is performed and then this is compared to conventional biological processes for the production of biomethane, such as anaerobic digestion.

The UK represents the base case for the entire analysis; hence, the model is based on country specific data. The system boundaries always include indirect and direct activities; the allocation of the environmental burdens in multi-functional systems is performed according to the method of system expansion and avoided burdens are also included in the assessment. The LCA models are based on both modelling and experimental results and also on literature data, as further specified in the following chapters. The environmental indicators are chosen and considered appropriate for each system analysed and sensitivity analysis are performed on key parameters.

3.8.1 GaBi 6 sustainability software

Almost 30 software packages are currently available to perform LCA. Many of them are specific for a certain application field and others have been developed internally by industrial organizations. They mainly differ in modelling approach and database. In this study Gabi software 6 [310] has been used; it contains databases developed by Thinkstep and incorporates industry organizations' databases and also regional and national databases. Mass and energy balance of the studied process can also be provided by the user and are used by the software to calculate the environmental impacts of the product's life cycle. The software uses a flowsheet approach.

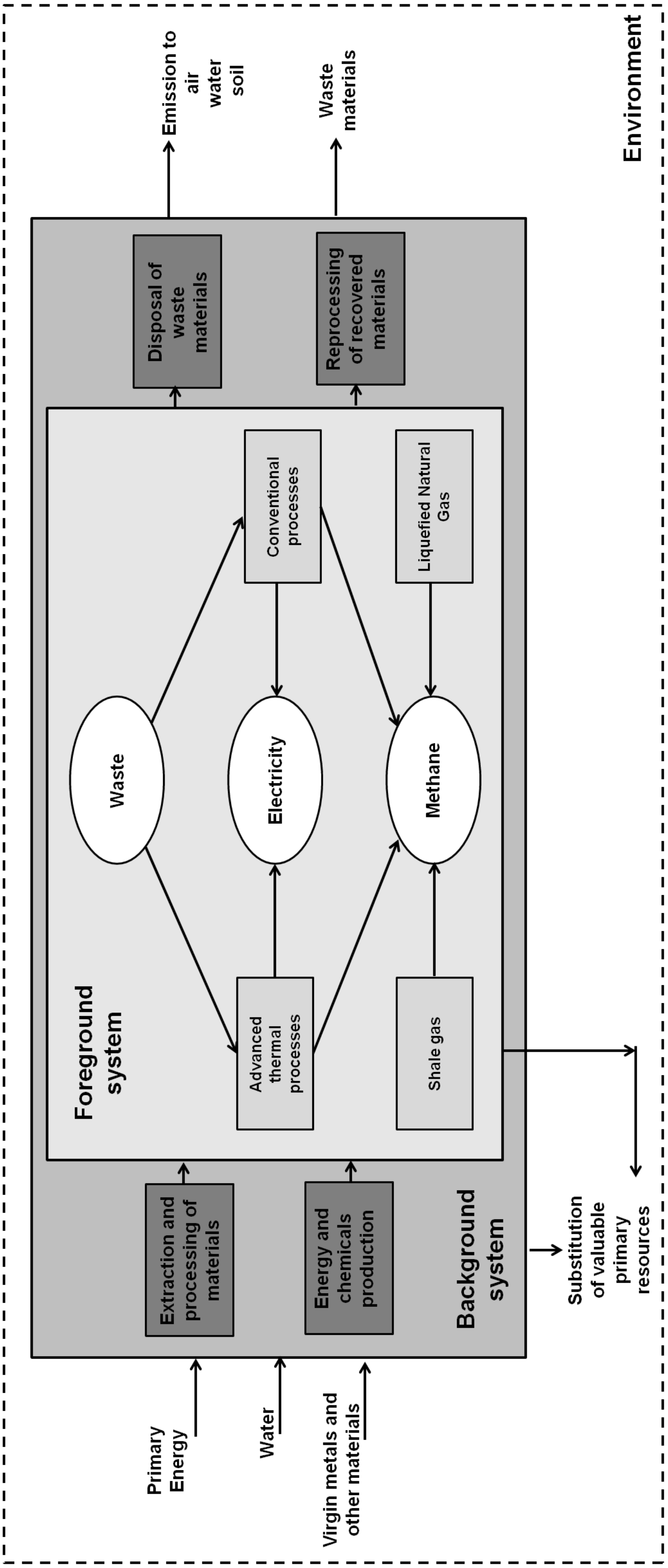


Figure 3.8.1. Summary of the system boundary of the entire study.

Chapter 4. **Methane from fossil resources: shale gas and LNG**

This chapter focuses on the supply of natural gas from fossil resources. Firstly, it investigates the environmental impacts of shale gas production for the UK perspective and then, it explores the life cycle of LNG supply to the UK. Finally, the two types of energy are critically compared.

The content of this chapter was partially published in:

Carla Tagliaferri, Roland Clift, Paola Lettieri, Chris Chapman, Shale gas: a life cycle perspective for UK production, Int. J. of LCA, revisions submitted.

Carla Tagliaferri, Roland Clift, Paola Lettieri, Chris Chapman, Liquefied natural gas for the UK: a life cycle assessment, Int. J. of LCA, under review.

Carla Tagliaferri, Paola Lettieri, Chris Chapman, Life Cycle Assessment of Shale Gas in the UK, Energy Procedia, 75 (2015) 2706 – 2712.

4.1 Introduction

As reported in chapter 2, both LNG and shale gas will possibly fully develop in the near future and constitute a significant part of the UK gas supply within the next 20 years [62]. Are those alternative technologies viable for the environment? Which option should be preferred?

Countless opinions and scientific studies often in opposition have rapidly developed for shale gas [311,312]. DECC [313] summarised the background geological knowledge and methodology which has enabled a preliminary in-place gas resource calculation to be undertaken for UK shale gas play across central Britain. In the meanwhile, Centrica, the owner of British gas, has signed a deal with the national gas company in Qatar, Qatargas, to import 3 million tonnes of LNG that will be enough to supply 13% of the total UK gas demand until 2018 [136].

Literature is lacking of a complete and up-to-date assessment regarding the total environmental burdens of the LNG supply chain and shale gas production from extraction to distribution to the final consumer in the UK. Conversely, a relatively wide spread literature is available on the analysis of the carbon footprint of LNG imported to Japan [314,315], the US [139,316–318] and Central Europe [319–324]. Furthermore, many studies have analysed the shale gas production in the US [130,325] but very few refer to the EU. Further robust and reliable environmental studies are necessary to answer the questions previously reported.

4.2 Shale gas: a life cycle perspective for UK production

4.2.1 Environmental impacts of shale gas

The rapid spread of shale gas in the US in 2010 led to a sharp increase in research activity focussing on the carbon footprint of this new energy source, with US research groups leading the work in this field [129,130,326,327].

Howarth *et al.* [326] were among the first authors to estimate the global warming potential of US shale; they made the highly contested observation [325,328–331] that shale gas may deliver an even higher carbon footprint than coal; this was, however, based on high estimates for fugitive emissions, using a high GWP (GWP at 20 year time horizon) for methane and comparing the results per MJ of energy in the fuel as opposed to kWh of electricity generated, thereby ignoring the higher average efficiency of gas compared to coal-fired power plants.

Conversely, other studies [129,130,325,332–335] reported that the emissions due to shale gas production and use do not significantly differ from those of conventional gas and are significantly lower than the emissions due to electricity production from coal. For example, Hultman *et al.* [330] estimated the GHG impact of shale gas to be 11% higher than that of conventional gas production but only 56% that of coal. More recent studies evaluated the greenhouse gas emissions from shale gas on the basis of experimental measurements at well sites [336–338], and examined [113] how the lower prices of natural gas due to shale gas development in the US might affect the national greenhouse gas emissions.

Some studies on the environmental impact of shale gas extraction have also been conducted in countries other than the US, that might develop their own shale gas reserves (such as in China [339,340]). In the UK, some [120,341] have calculated the carbon footprint of shale gas extraction, however, their inventory data rely mainly on US estimates.

The main focus of all the studies previously mentioned is the estimation of the emissions and the associated carbon footprint of shale gas production and use. Very few studies explore also other impacts, such as the water life cycle of US shale gas extraction [128,342]; in the UK, Stamford and Azapagic [343] and Cooper *et al.* [14] are the only ones so far who have analysed different impact indicators (i.e., depletion of energy sources, acidification potential etc.) in addition to the carbon footprint. Stamford and Azapagic [343] and Cooper *et al.* [14] compared the environmental burdens of the electricity production from shale gas to the electricity production from other sources including coal, nuclear, wind and solar, hydro and biomass. They considered the role that shale gas may play in affecting the impacts of electricity generation in the UK. This perspective refers to a particular use of natural gas (electricity production) which currently represents only 23% of the total electricity generated in the UK. Conversely, more than 67% of the total heat generated for the residential and commercial sector is currently produced using natural gas [63]. More than half of the total natural gas consumed in the UK is used to provide heat to households and industry. Although the decarbonisation of the electricity grid mix supported by the UK government [344] will determine an increase in the gas share in the electricity mix in the short term [16], natural gas is and will continue to be the main fossil source used for residential and commercial heating purposes at least until 2030-2033 [16].

Within this context, this work specifically adopts a natural gas production perspective and does not consider the final use of natural gas for electricity generation as done by Stamford and Azapagic [343] and Cooper *et al.* [14]. This perspective has been chosen to reflect the current and future major use of natural gas in the UK. This study reports a comprehensive attributional life cycle assessment and hot spot analysis of shale gas production and distribution in the UK, considering the current EU exploration and development of shale gas. The environmental burdens of shale gas production in the UK are also compared to the current supply of natural gas mix. The main contribution of this work regards the analysis of water consumption, degradation and use of shale gas production within the UK context. This work also considers a broad range of environmental impacts, including acidification potential, abiotic depletion fossil, toxicities, etc., hence providing a further reference point against previous works.

The LCA model is based on the analysis of literature data from more than 60 publicly available sources. The robustness of the model is checked through sensitivity analysis on key parameters. The system boundary excludes the use of natural gas, as this would be identical whether shale gas or conventional gas is used.

4.2.2 Modelling assumptions and system boundary

The modelling approach and the system boundary are shown in Figure 4.2.1 and the main inventory data are reported in Table 4.2.1 (more details on data source and assumptions of the operations analysed are reported in the Annex). The entire life cycle of shale gas production process has been considered in the modelling approach. This includes the indirect activities of energy, chemicals and water production and recovery and final disposal of waste material identified in the background of Figure 4.2.1. The background system exchanges energy and material with the foreground system. This includes the entire supply chain of shale gas production, processing and distribution to the final consumer at low pressure. Avoided burdens have also been considered for the production of valuable hydrocarbon by-products other than natural gas.

The following stages are considered:

1. *Well site exploration and investigation.*
2. *Well pad and road preparation and construction.*
3. *Well drilling.* Production of materials needed for drilling; transport of materials; energy required during drilling and emissions from machinery; emissions during drilling; casing and cementing; disposal of drilling wastes; horizontal drilling.
4. *Hydraulic fracturing of the well.* Production and transport of water, chemicals and sand needed for fracturing; energy used during the hydraulic fracturing and emissions from machinery; disposal of wastes.
5. *Well Completion.* Energy and materials required; disposal of flowback and produced water from the well; emissions of natural gas during well completion, workovers, unloadings; re-fracturing.
6. *Production.* Processing and cleaning.
7. *Pipe construction and transmission.*
8. *Post production phase.* Decommissioning, leakage due to decommissioning, plugging and removing of equipment.

As widely reported in literature [130,333,343], it is assumed that extraction, processing and distribution of shale gas involve exactly the same processes as onshore extraction of conventional gas. Operations associated with hydraulic fracturing are instead considered specific for shale gas. Therefore, two models have been built: the first (identified as the common operations model) accounts for the extraction of conventional gas and includes all the common processes between conventional and unconventional extraction: gas field exploration, natural gas production, purification, long distance transport and regional distribution. The second model (identified as the hydraulic fracturing model) includes all the processes specific to shale gas: horizontal drilling, fracking of the shale rocks, flowback disposal and handling of emissions associated with hydraulic fracturing. The emissions in the hydraulic fracturing model represent the difference in emissions between production of shale gas and conventional gas.

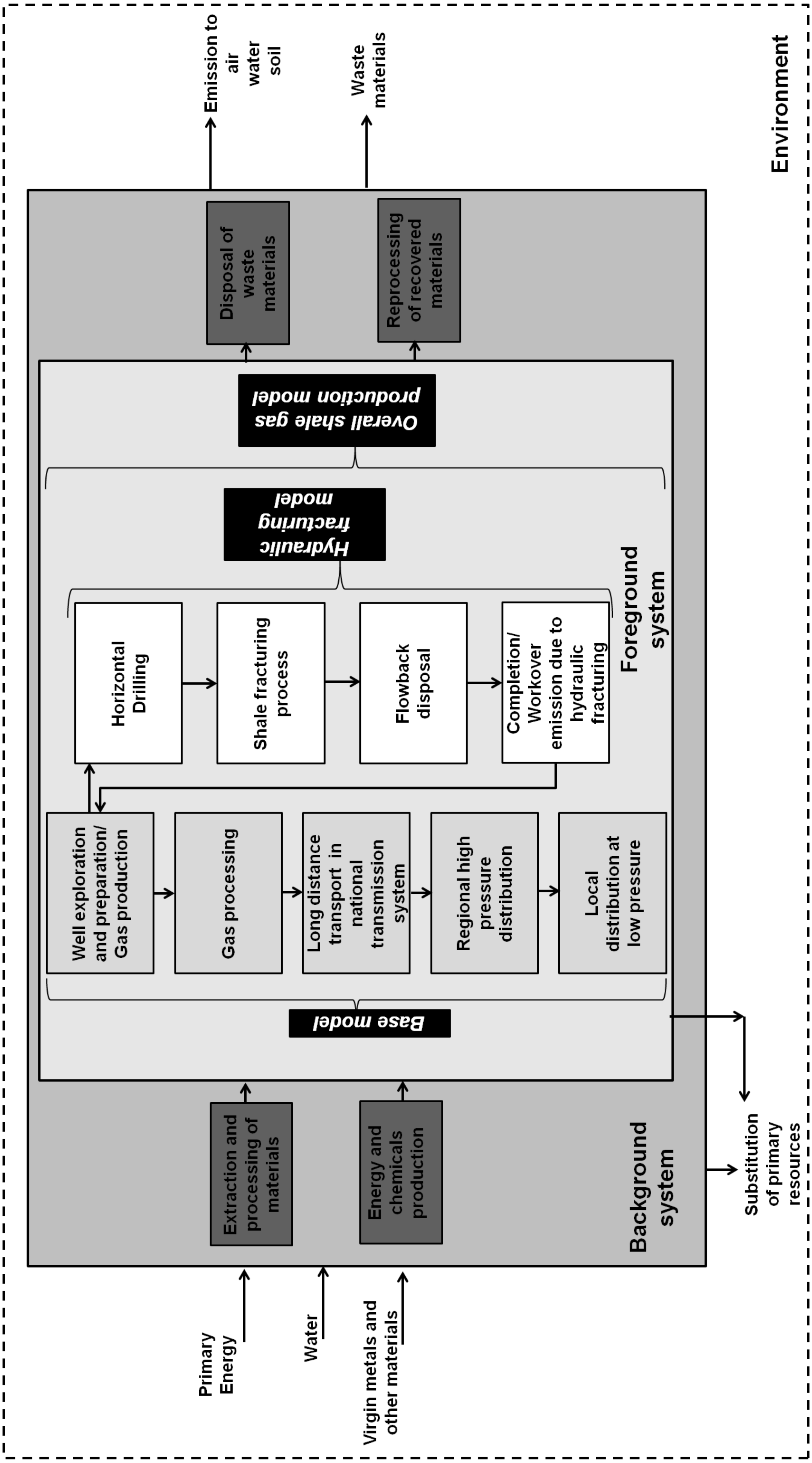


Figure 4.2.1. System boundary.

The two models are integrated and hence the total environmental burdens of shale gas extraction, processing, transport and distribution are calculated. The model for conventional onshore gas - common operations model - relies on data from Ecoinvent database v 3.1 [345], whereas the hydraulic fracturing model is based on estimates for the UK taken from literature (for key data and modelling see Table 4.2.1 and the Appendix). Very few inventory data, such as the composition of flowback water for the UK, were not publicly available in literature because of the early stage development of the shale gas exploitation in the UK. Hence, for these cases, values for the US were used in the model, as further specified in the appendix. All indirect and avoided burdens considered in the LCA model are UK-specific. Goods transport is also included in the system boundaries.

The shale gas burdens are compared to the current UK gas grid mix as modelled in GaBi database [310]. However, this dataset does not include the production of pipeline for onshore distribution. Therefore, the model has been modified to account for this according to the data reported in the Ecoinvent database [345].

The functional unit of this work is the delivery of 1MJ LHV of natural gas to the final consumer at low pressure (< 7 bar and > 0.75 mbar gauge). All results are reported according to the functional unit.

Further modelling assumptions and inventory data are reported in the Appendix.

Water for hydraulic fracturing [129,130,325,333–335,343,346]	m^3	5.46E-06
Sand for fracturing (silica, quarts sand) [129,325,334,335,341,343]	kg	7.25E-04
Additives of fracking fluids [129]		
Acid: Hydrochloric acid or muriatic acid	kg	1.61E-05
Friction reducer: Petroleum distillate	kg	4.02E-06
Surfactant: Isopropanol	kg	4.02E-06
Cly stabilizer/controler: Potassium chloride	kg	2.68E-06
Geling agent; Guar gum or hydroxyethyl cellulose	kg	2.68E-06
Scale inhibitor: Ethylene glycol	kg	2.01E-06
PH Adjusting agent: Sodium bicarbonate and sodium potassium hydroxide	kg	5.36E-07
Breaker: Ammonium persulfate	kg	5.36E-07
Crosslinker: Borate salts	kg	5.36E-07
Iron control: Citric acid	kg	2.01E-07
Bactericide/biocide: Glutaraldehyde	kg	5.36E-08
Corrosion Inhibitor: Formamide	kg	5.36E-08
Flowback disposed to industrial treatment	kg	1.55E-03
Energy requirements for the freeze-thaw evaporation process [130,310]	kWh	1.96E-05
Energy requirements for pumping the hydraulic fracturing fluids in the well [130]		
Diesel	kg	2.26E-05
Emissions for pumping the hydraulic fracturing fluids in the well [347,348]		
CO_2	kg	7.19E-05
SO_2	kg	7.29E-08
NO_x	kg	1.01E-06
PM	kg	8.39E-08
CO	kg	2.18E-07
NMVOC	kg	3.17E-09
Materials used for horizontal drilling [121,325,343,349]		
Steel	kg	3.68E-05
Portland cement	kg	5.57E-05
Gilsonite (asphaltite,)	kg	-2.10E-06
Diesel fuel	kg	4.29E-05
Bentonite	kg	1.03E-05
Soda Ash	kg	1.72E-07
Gelex	kg	1.25E-09
Polypac	kg	3.24E-07
Xanthum Gum	kg	1.64E-07
Water throughput	kg	1.80E-07
Emission due to horizontal drilling [347,348]		
CO_2	kg	1.36E-04
SO_2	kg	1.38E-07
NO_x	kg	1.91E-06
PM	kg	1.59E-07
CO	kg	4.13E-07
NMVOC	kg	6.01E-09

Table 4.2.1. Key inventory data used in the hydraulic fracturing model. Values are reported per functional unit for S.0.

Potential emission due to well completion and workover allocated to the hydraulic fracturing model-those emissions have been further modified to account for REC [337,338,349,350]		
CH_4	g CH_4	5.40E-02
CO_2	g CO_2	5.18E-03
C_2H_6	g C_2H_6	3.54E-03
C_3H_8	g C_3H_8	1.73E-03
N_2	g N_2	7.70E-03

(Continued) Table 4.2.1. Key inventory data used in the hydraulic fracturing model. Values are reported per functional unit for S.0.

4.2.2.1 Scenarios

The process of shale gas extraction using hydraulic fracturing is still in the development stage in terms of both technology and regulation in the UK. Industry data is therefore rarely publicly disclosed and field measurements are lacking so that the limited inventory data available are widely contested [334,335]. A sensitivity analysis on key parameters is therefore important. In this work it is performed according to the literature, as here reported.

Some authors [122,129,325,333] report that the amount of emissions and the emissions handling method are the most important parameters influencing the uncertainty in the carbon footprint of shale gas. Conversely, others [335] have concluded that the Estimated Ultimate Recovery (EUR) is the parameter that most influences the results.

Few authors have critically reviewed the potential risks that shale gas operation and mainly flowback disposal pose to the water source [108] and water life cycle, hence analysing different key parameters such as flowback ratio, flowback recycled fraction, the amount of water used according to different shale plays [128,351] and wastewater composition [352].

In this work, the sensitivity analysis explores 18 scenarios (S.) and 5 key parameters as reported in Table 4.2.2 and summarised here.

S.0 (base scenario). This represents the best option regarding the emission handling method and flowback water disposal. Emissions are assumed to be completely captured and gathered into the pipeline [333]. Flowback is assumed to be completely disposed through adequate industrial treatment and the fraction of flowback is assumed to be 25% as largely reported in literature [353]. The estimate ultimate recovery is 85 million m^3 [341,343].

Scenarios 1-3 explore different flowback fractions. All the assumptions are the same as in the base scenario except for the flowback fraction that varies between 25% and 150% as also reported in literature [121,349].

Scenarios 4-9 analyse different flowback disposal methods including 100% direct disposal to environment, 100% recycling, and 100% disposal to Class II wells. Combinations of the different methods are also explored. Avoided burdens are allocated to the flowback recycling as this avoids the exploitation of new resources for the fracturing of new wells.

Scenarios 10-12 analyse different handling methods of the emissions from completion and workover due to the process of hydraulic fracturing. 100% flaring and 100% venting are analysed, as well as 50% flaring and 50% capturing. Although scenario 11 (100% of the emissions are vented) does not represent a feasible option according to the UK regulation, it has nevertheless been explored to identify the potential threats of complete venting.

Scenarios 13-14 explore a 15% increase/decrease in the amount of completion/workover emissions due to the hydraulic fracturing process, in virtue of the current debate on this in the literature [129,325,326,328,354,355].

Scenario 15-16 investigate an increase/decrease of EUR according to the values reported by DECC [341].

Scenario 17 studies the effect of goods transport distances on the environmental burden of shale gas. In this scenario, all transport distances are doubled.

a)		**Processed gas recovery**	**Flowback fraction**	**Fraction of flowback recycled**	**Fraction of flowback disposed to Class II well**	**Fraction of flowback directly disposed to environment**	**Fraction of flowback disposed to proper industrial treatment**	**Fraction of emission sent to flaring**	**Fraction of emission sent to REC**	**Completion emissions associated with hydraulic fracturing**	**Workover emissions associated with hydraulic fracturing**
	Unit	10^7 m3/well	%	%	%	%	%	%	%	10^8 g CH4 per well	10^7 g CH4 per well
	S.0	7.31	25	0	0	0	100	0	100	1.61	3.23
Flowback fraction	**S.1**	7.31	50	0	0	0	100	0	100	1.61	3.23
	S.2	7.31	90	0	0	0	100	0	100	1.61	3.23
	S.3	7.31	150	0	0	0	100	0	100	1.61	3.23
Disposal method	**S.4**	7.31	25	100	0	0	0	0	100	1.61	3.23
	S.5	7.31	25	0	100	0	0	0	100	1.61	3.23
	S.6	7.31	25	0	0	100	0	0	100	1.61	3.23
	S.7	7.31	25	50	0	0	50	0	100	1.61	3.23
	S.8	7.31	25	50	50	0	0	0	100	1.61	3.23
	S.9	7.31	25	0	50	0	50	0	100	1.61	3.23
Handling of emissions	**S.10**	7.31	25	0	0	0	100	100	0	1.61	3.23
	S.11	7.31	25	0	0	0	100	0	0	1.61	3.23
	S.12	7.31	25	0	0	0	100	50	50	1.61	3.23
Emissions	**S.13**	7.31	25	0	0	0	100	0	100	1.85	3.72
	S.14	7.31	25	0	0	0	100	0	100	1.37	2.75
EUR	**S.15**	4.90	25	0	0	0	100	0	100	1.61	3.23
	S.16	12.0	25	0	0	0	100	0	100	1.61	3.23
Transport	**S.17**	7.31	25	0	0	0	100	0	100	1.61	3.23

Table 4.2.2. Key inventory data of the a) scenarios analysed and b) transport distances.

b)	**Transport distances**								
	Fracturing freshwater from surface water to well site - truck	**Fracturing freshwater from municipal water plant to well site - pipe**	**Sand used for fracturing fluids from quarry site to well site - truck**	**Chemicals for fracturing fluids from industrial production site to well site - truck**	**Diesel for fracturing operations from refinery to well site - truck**	**Flowback to be recycled from well site to other shale gas wells - truck**	**Flowback water to be injected in class II wells from well site to class II wells - truck**	**Flowback water to be directly disposed to environment from well site to disposal site - truck**	**Flowback water from well site to proper industrial waste water treatment plant – truck**
	km	km	km	km	km	km	km	km	km
S.0-S.16	10	15	30	50	50	10	250	10	50
S.17	20	30	60	100	100	20	500	20	100

(Continued) Table 4.2.2. Key inventory data of the a) scenarios analysed and b) transport distances.

4.2.3 Water modelling principles

The environmental concerns related to water scarcity due to population growth and economic development have been growing during the last decade [356]. This justifies a water-related approach in life cycle inventories and assessments in particular processes that determine an increased load on water resources, such as shale gas extraction. For the shale gas production, the impact related to fresh water use is analysed separately in this thesis, according to the standard ISO 14046 [357].

A definition of the terminology used in the water modelling principles is reported in equations (4.2)-(4.4) [358] and in Figure 4.2.2. Water use is the measured amount of water input into a product system or process (this usually is the total water withdrawn from the environment). Fresh water use is further differentiated in consumptive water use and degradative water use.

The freshwater consumption includes all fresh water losses on a watershed level which are caused by evaporation, release of fresh water into the sea (as fresh water is a limited natural resource), etc. The water consumption identifies the water losses associated with water use.

Degradative water use identifies the use of water that determines quality degradation and pollution. When the polluted water is released again to watershed then this use of water does not have to be considered consumptive.

$$\text{Fresh water use} = \text{consumptive use} + \text{degradative use} \tag{4.2}$$

$$\text{Degradative use of fresh water} = \\ \text{freshwater released back to water shed with possible alteration in quality} \tag{4.3}$$

$$\text{Fresh water consumption} = \text{ freshwater lost to the watershed} \quad (4.4)$$

Surface (lake and river) and groundwater are usually considered in the environmental assessments, rain water is excluded.

Some studies that analyse the problem of water [351], focus only on the LCI. However, degradative water use defines the amount of water whose changes can only be assessed by specific impact categories of LCA.

In this thesis, the two levels of analysis are joined and focus is on both polluting impacts to fresh water (assessed by FAETP, MAETP and EP) and the quatification of water use.

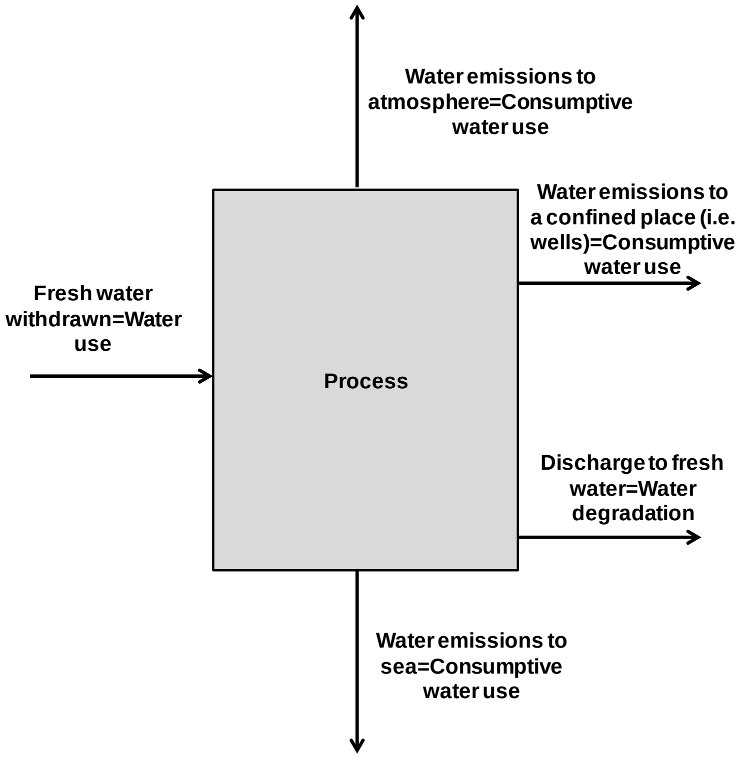


Figure 4.2.2. Water modelling principles.

4.2.4 Results

Water use (Figure 4.2.3).

The hydraulic fracturing process is the main contributor to the freshwater use of the overall shale gas model (it determines between 95% and 87% of the total water use for S.15 and 16, respectively, see Figure 4.2.3a); the results are highly dependent on the EUR as shown for S.15 and S.16. S.3 and S.15 show the absolute highest water use because of the highest flowback ratio (and therefore the highest amount of fresh water needed to treat the flowback) and lowest EUR, respectively. S.4 and S.16 determine the lowest absolute water use because of the flowback recycling (that avoids new fresh water withdrawal) and the highest EUR, respectively.

As shown in the hot spot analysis of the hydraulic fracturing model (Figure 4.2.3b), the main contributor to the water use is the fracking of shale formation and, depending on the scenario, also the flowback disposal process. No parameter influences the water use of the shale fracturing process except for the EUR. Conversely, the flowback ratio and the flowback handling method influence the water use of the flowback disposal process (as shown for S.1-S.9).

The fracturing process and the flowback disposal are further analysed in Figure 4.2.3c and Figure 4.2.3d. The hot spot analysis of the fracturing process (Figure 4.2.3c) shows that the water use is due to the excavation and processing of the sand used in the fracturing liquids (60%), to the withdrawal of fresh water used for cracking the rocks (23%) and to the production of fracturing chemicals (17%). Frack sand must be of uniform size and shape and to achieve this, a deep processing is needed [359]. The processing plants wash, dry, sort, and store the sand and waste water is produced. This explains the indirect water use associated with the process of sand mining and processing (contributing for 60% of the total water use of fracturing operations). On the other hand, the water used to produce diesel for transport is negligible.

According to the disposal of flowback fluids, shown in Figure 4.2.3d, direct disposal to the environment and disposal to class II wells determine the minimum use of fresh water as no further treatment of waste water is needed (S.5-S.6). S.0-S3 show that increasing the flowback ratio, the water used to treat and dispose of it increases whereas all other scenarios show a constant amount of water used to treat flowback water.

A further analysis on the comparison with the UK grid mix is reported in the discussion section.

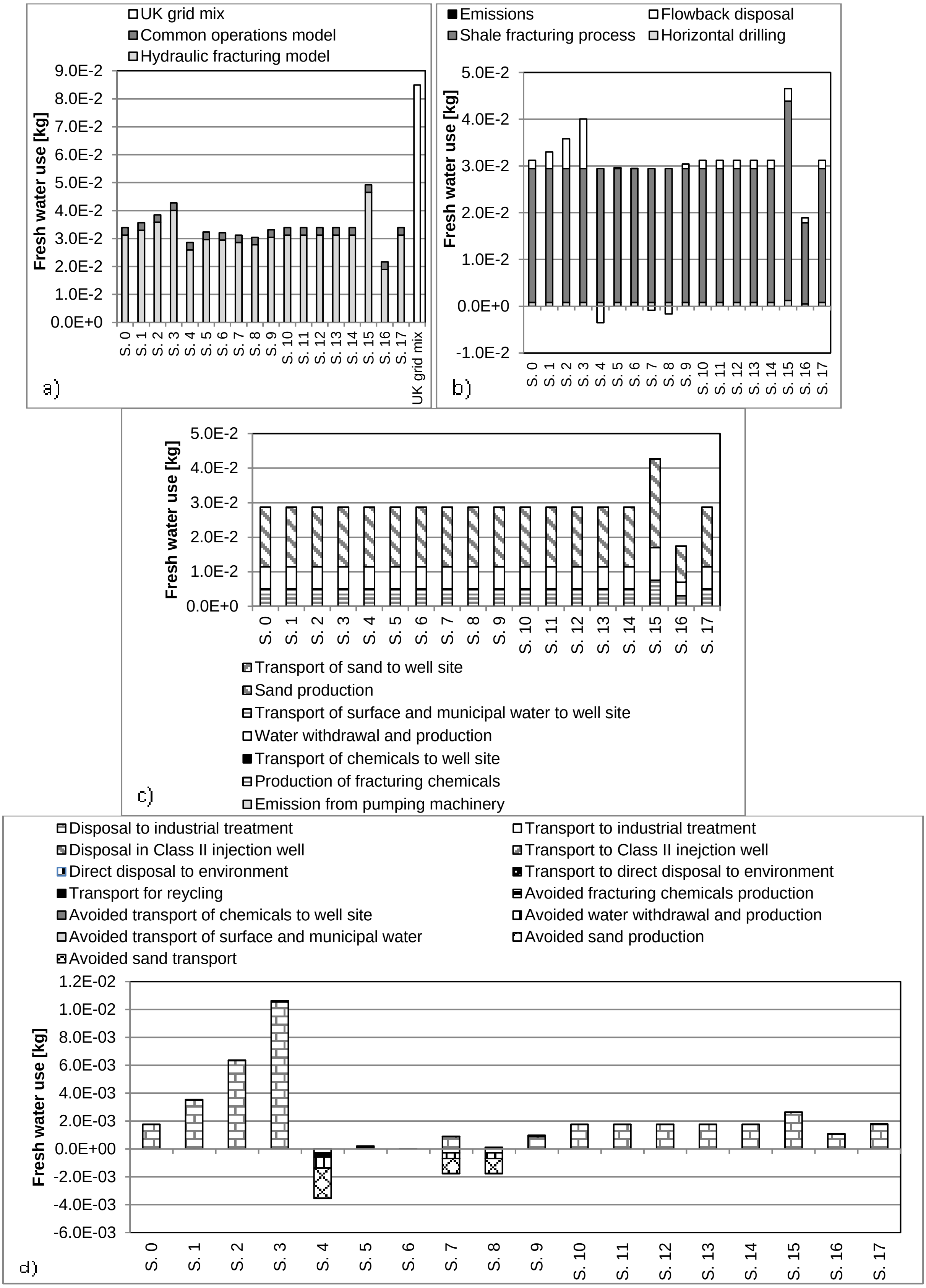


Figure 4.2.3. a) Water use associated with the overall shale gas model. b) Hot spot analysis: water use associated with the hydraulic fracturing model. c) Hot spot analysis: water use associated with the hydraulic fracturing of shale formations. d) Hot spot analysis: water use associated with flowback disposal.

Consumptive water use

The water consumed in shale gas extraction is (except for scenario S.3) due to the water consumption associated with the hydraulic fracturing model for more than 50% (Figure 4.2.4a). Conversely, for S.3 the water consumption is mainly due to the processes included in the common operations model. In this case, the water consumption of the entire hydraulic fracturing model is lower because the amount of water ultimately released to the environment (degradate water) is higher as the flowback ratio is higher than 100%. As shown for S.15 ($4.9*10^7$ m^3/well of gas recovered in this scenario) and for S.16 ($12*10^7$ m^3/well of gas recovered in this scenario), the EUR is a key parameter for the assessment of the water consumption as it determines a change in the results of 8% (Figure 4.2.4a).

Rock fracturing and flowback disposal are the two main contributors to water consumption as shown in the hot spot analysis of the hydraulic fracturing model (Figure 4.2.4b). For some scenarios, the water consumption of the flowback disposal is negative because this unit operation implies a net release of lower quality water to watersheds.

Regarding the injection of fracturing fluids into the shale formation (shale fracturing process in Figure 4.2.4b), the consumption of water is 99% due to the withdrawal of fresh water used in the fracturing fluids. In this case the production of sand and chemicals are negligible.

As shown in the flowback disposal hot spot analysis of Figure 4.2.4c, the water consumed to treat the flowback is offset by the degradate water released to the environment for S.0-S.3, S.6 and S.9-S.17. S.5 is the only scenario that does not show any negative water consumption in the hot spot analysis of the hydraulic fracturing model because water is injected into wells and therefore it is not released to its withdrawal watershed. For S.4, S.7 and S.8 the negative values of the water consumption are not due to a net release of water to the environment but to the avoided burden allocated to the recycling of flowback.

The EUR and the flowback ratio are again key parameters for the results of the hot spot analysis of shale gas.

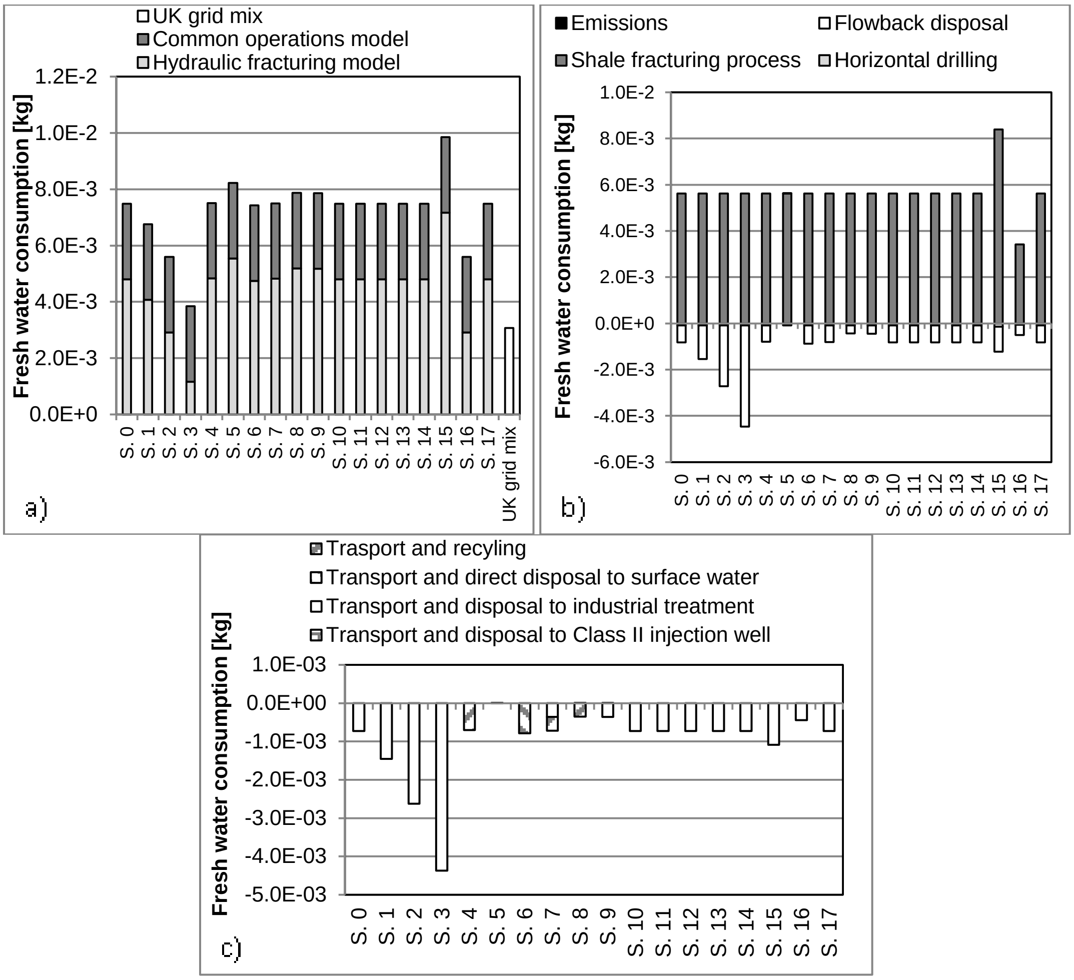


Figure 4.2.4. a) Water consumption associated with the overall shale gas model. b) Hot spot analysis: water consumption associated with the hydraulic fracturing model. c) Hot spot analysis: water consumption associated with flowback disposal.

Degradative water use

Water degradation represents the difference between water use and consumption and it is reported in the Appendix.

The degradative effects associated with the water life cycle of shale gas have been quantified using the *Fresh water aquatic ecotoxicity potential.* As shown in Figure 4.2.5a, the fresh water aquatic ecotoxicity of shale gas extraction is mainly associated with the common operations model; the impact of the hydraulic fracturing model is negligible except for scenario 6 where it represents 26% of the total burden. The flowback disposal method determines the higher FAETP of S.6 because the flowback is assumed to be completely discharged to rivers without any further treatment. High content of solids, radioactive elements and polluting chemicals determine the FAETP associated with the water discharge. All other disposal methods do not significantly contribute to the FAETP. In the UK, direct discharge of flowback to fresh water and injection into wells are not permitted but this analysis assesses the potential impact of illegal disposal or unwanted spills of flowback water. The FAETP of the common operations model is 50% attributable to onshore gas extraction and processing, and 34% to the production of pipes used in low pressure distribution network (Figure 4.2.5c). The supply of UK natural gas mix causes a significantly lower impact.

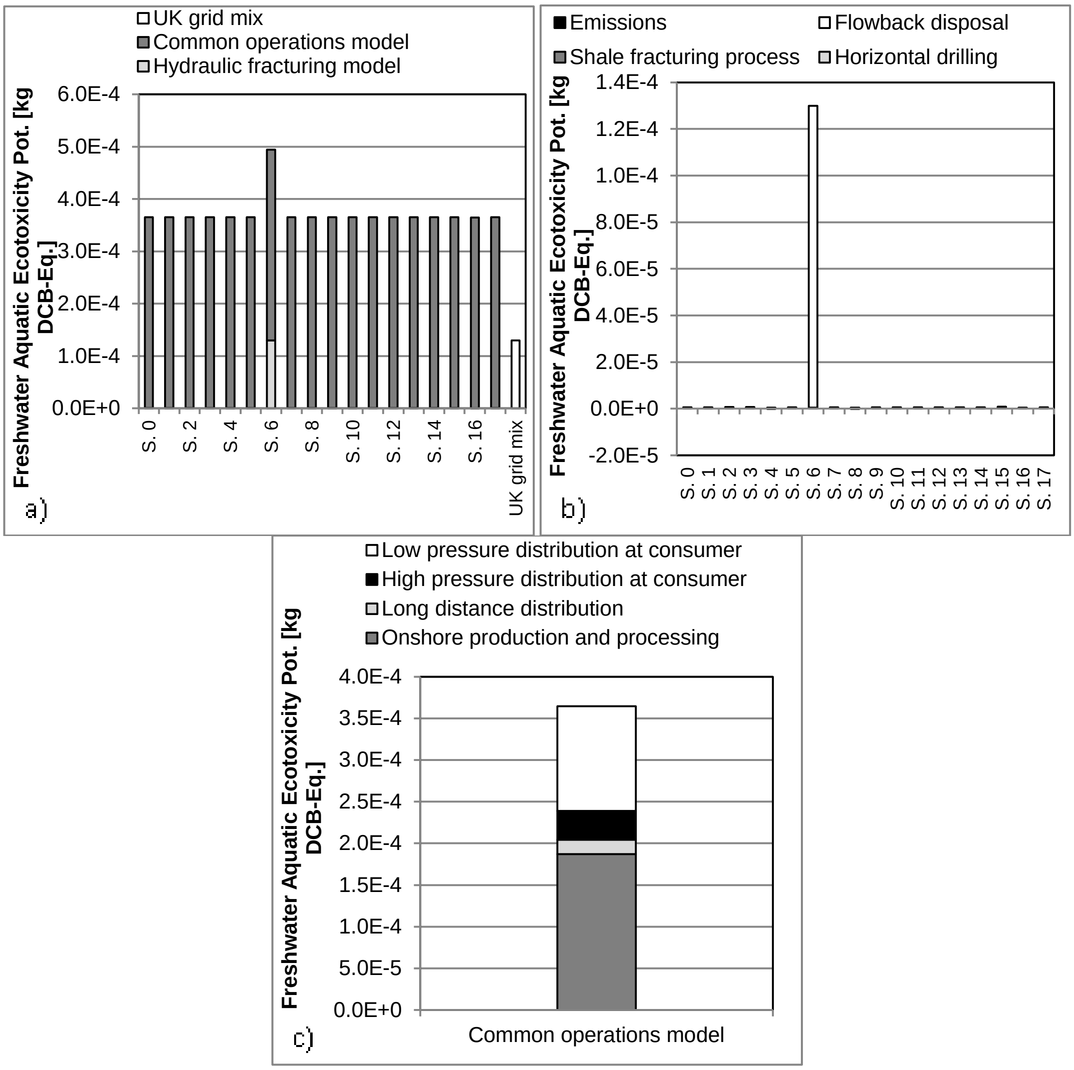


Figure 4.2.5. a) FAETP of the overall shale gas model. b) FAETP hot spot analysis of the hydraulic fracturing model. c) FAETP of the common operations model.

Comparison with UK natural gas grid mix-Water impacts

The analysis of the water life cycle of shale gas has been strengthened comparing it with the water use of conventional natural gas currently supplied to the UK. According to Figure 4.2.3a, the UK mix model for natural gas causes a higher degradative water use. The natural gas datasets reported in GaBi database [310] show a high variability in the water use results according to the different country specifications. This difference is based on the electricity used during the production phase. The amount of the electricity requirement is not the driver of the water use variability, whereas the different country specific electricity mix determines it. The countries that base their national electricity consumption on hydropower (such as Norway (NO)) show an elevated water use in the results. For the UK natural gas mix, two drivers dominate 75% of the results: 1. the water use included in the UK electricity grid mix which is used for gas regional distribution; 2. the Norwegian electricity grid mix (identified as a data set with a very high amount of water use) used to produce Norwegian natural gas imported to the UK, on the base of the UK gas import mix. The higher value of the natural gas mix shown in Figure 4.2.3a is due to the water use and degradation associated with the UK natural gas imports and in particular to the imports of Norwegian natural gas. Norwegian gas is produced using NO electricity mix that shows high water degradation due to the hydropower share of the electricity mix. As previously mentioned, degradative use takes place when the water used remains in the same watershed but the quality has been altered. Hydropower dams alter water flows and this causes water degradation due to change in water temperature and consequently in biological and chemical composition (amount of dissolved oxygen, nutrients and dissolved solid) [360].

Global warming potential (GWP)

The emissions associated only with the fracturing model (the emissions due to completion and workover for conventional extraction are included in the common operations processes) determine between 3% and 12% of the total GWP of the overall shale gas model (Figure 4.2.6a). The peak value of 12% is shown for S.11 where the emissions from completion and workovers are assumed to be vented. The use of green devices (S.0) does not seem to significantly improve the GWP of shale gas extraction when compared to the GWP of S.10 where emissions are assumed to be completely flared. It should also be highlighted that the analysis does not include the production and assembly of Reduce Emission Completion (REC) devices as no inventory was available. If the latter were included in the assessment, the use of REC devices may show a different trend in the results. The difference between the GWP of scenario S.10 (where emissions are completely flared) and the GWP of scenario S.12 (where emissions are half flared half captured) is negligible. A change of +-15% in the amount of emissions does not have a significant impact on the overall results.

As shown in Figure 4.2.6c, except for S.11, the main contributors to the GWP of the hydraulic fracturing model are horizontal drilling (more than 50% of the GWP of hydraulic fracturing model) and fracturing of shale rocks (around 23% of the GWP of hydraulic fracturing model). The remaining burdens are due to the emissions associated with completion and workover; for all scenarios, 83% of those emissions are due to the phase of completion (Figure 4.2.6c and f).

The amount of diesel used for horizontal drilling is almost twice the amount of diesel used to inject the fracking fluids in the rocks [130,349]. Consequently, direct emissions from machinery during horizontal drilling have a higher environmental impact accounting for 50% of the total impact of this unit operation. Indirect GWP of horizontal drilling (Figure 4.2.6d) is mainly due to the production of the steel welded pipes for casing the drilling hole (31% of the total GWP of horizontal drilling) and to the cement production required for drilling mud (17% of the total GWP of horizontal drilling). Direct emissions from pumping machinery during the hydraulic fracturing also determine the main GWP contribution to the fracturing process (Figure 4.2.6e). Slips of gas from onshore gas production and processing and from distribution determine the GWP of the common operations model reported in Figure 4.2.6b.

The values found for the GWP are in line with the range reported by other authors [129,325]. The difference between shale gas and UK grid mix gas is due to lower emissions during the offshore extraction activities and different EUR.

However, it is worth pointing out that the GWP reported in this work does not include unwanted slippage of methane through the well casing during production. It is assumed that the well has been properly installed and therefore no gas escapes through a faulty casing into shallow aquifers and then into the atmosphere (the evidence of this faulty possibility is reported in literature [361]). Usually to prevent leaks, during well installation, cement is pumped into the space between the pipes and the surrounding rocks but if the cement has gaps, gas can bubble up. The frequency of well construction problems in US is reported to be between 3.4% and 6% [362]. If shale gas extraction develops in the UK, it is assumed that the casing will be properly installed and checked and therefore the frequency of well construction problems should be lower than that reported in the US; for this reason in this analysis, unwanted fugitive methane from well casing was not taken into account.

In the current analysis 2 re-fracturing jobs have been consider throughout the life of the well. Tavassoli *et al.* [363] reported that the EUR of a shale well could increase of 30% with up to 4 optimised re-fracturing jobs. Therefore, in the best case, additional re-fracturing could increase the EUR from 7.31 10^7 m^3/well to 9.5 $*10^7$ m^3/well which would not significantly affect the GWP and the other environmental results. In fact, the lower environmental impact of offshore wells supplying gas to the grid mix is usually associated to the production of 10^9 m^3/well as opposed to the 10^7 m^3/well in the case of shale wells.

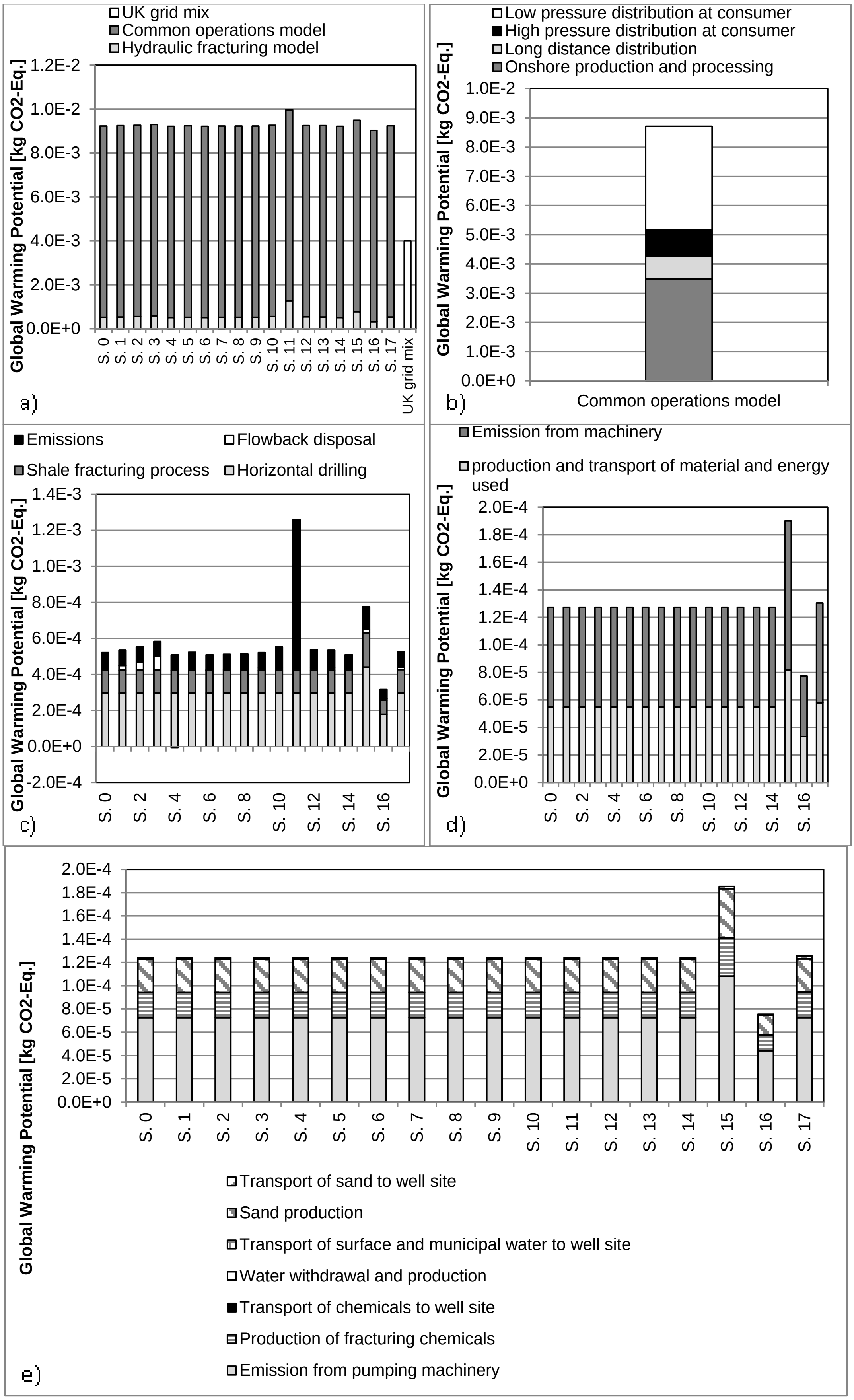
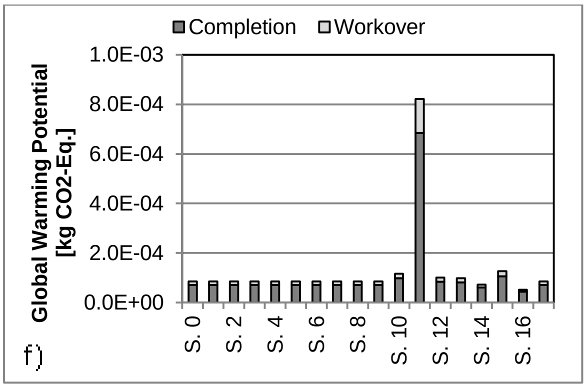


Figure 4.2.6. GWP of the a) overall shale gas model; b) common operations model; c) hydraulic fracturing model; d) horizontal drilling; e) fracturing of shale rocks; f) emissions due to completion and workovers.



(Continued) Figure 4.2.6. GWP of the a) overall shale gas model; b) common operations model; c) hydraulic fracturing model; d) horizontal drilling; e) fracturing of shale rocks; f) emissions due to completion and workovers.

Abiotic depletion potential (ADP)

The model of hydraulic fracturing determines a negligible ADP when compared to the common operations model, (Figure 4.2.7a) because the main depletion of fossil resources is due to gas exploitation (Figure 4.2.7b). The negligible ADP remaining shown in Figure 4.2.7b is associated with the energy required for distribution and pipe installation. The ADP associated with the hydraulic fracturing model is two orders of magnitude lower than that associated with the common operations model. No sensible variation is shown for the different parameters; the ADP of shale gas is comparable to the ADP of conventional UK grid mix supply as also already reported in literature [343].

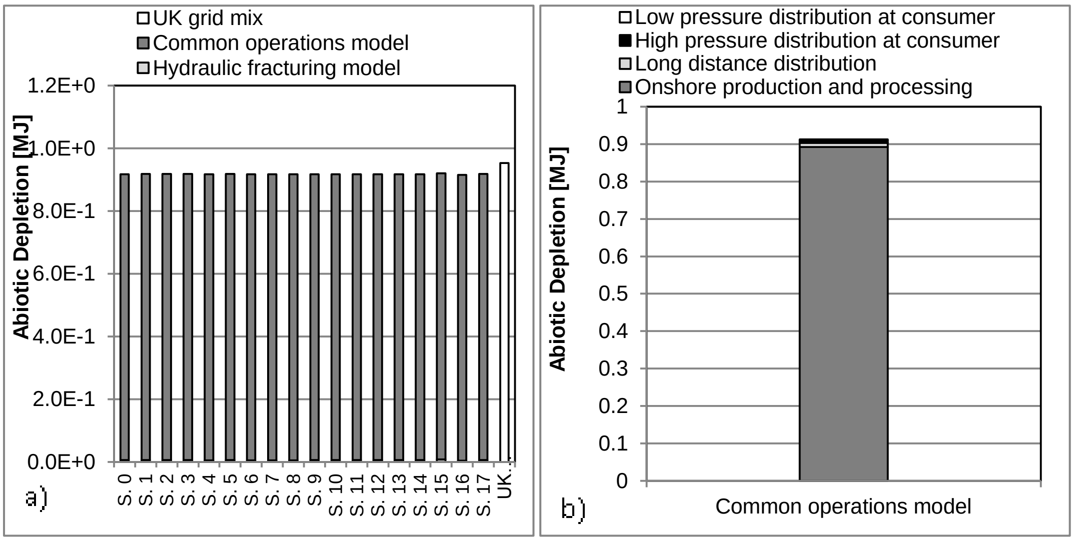


Figure 4.2.7. ADP of the a) overall shale gas model; b) common operations model.

Acidification potential (AP)

The burden of the hydraulic fracturing model represents between 1.5 and 5% of the total environmental burden of overall shale gas model (Figure 4.2.8a). More than 90% of the

AP is due again to the conventional extraction of the common operations model, specifically to emissions due to drilling, leakage and processing (Figure 4.2.8b); no sensible variation is shown for the different parameters. Shale gas is likely to have an impact about 7 times higher than the impact of UK conventional gas mix because of lower emissions during the offshore extraction phase [310] and higher EUR (Figure 4.2.8).

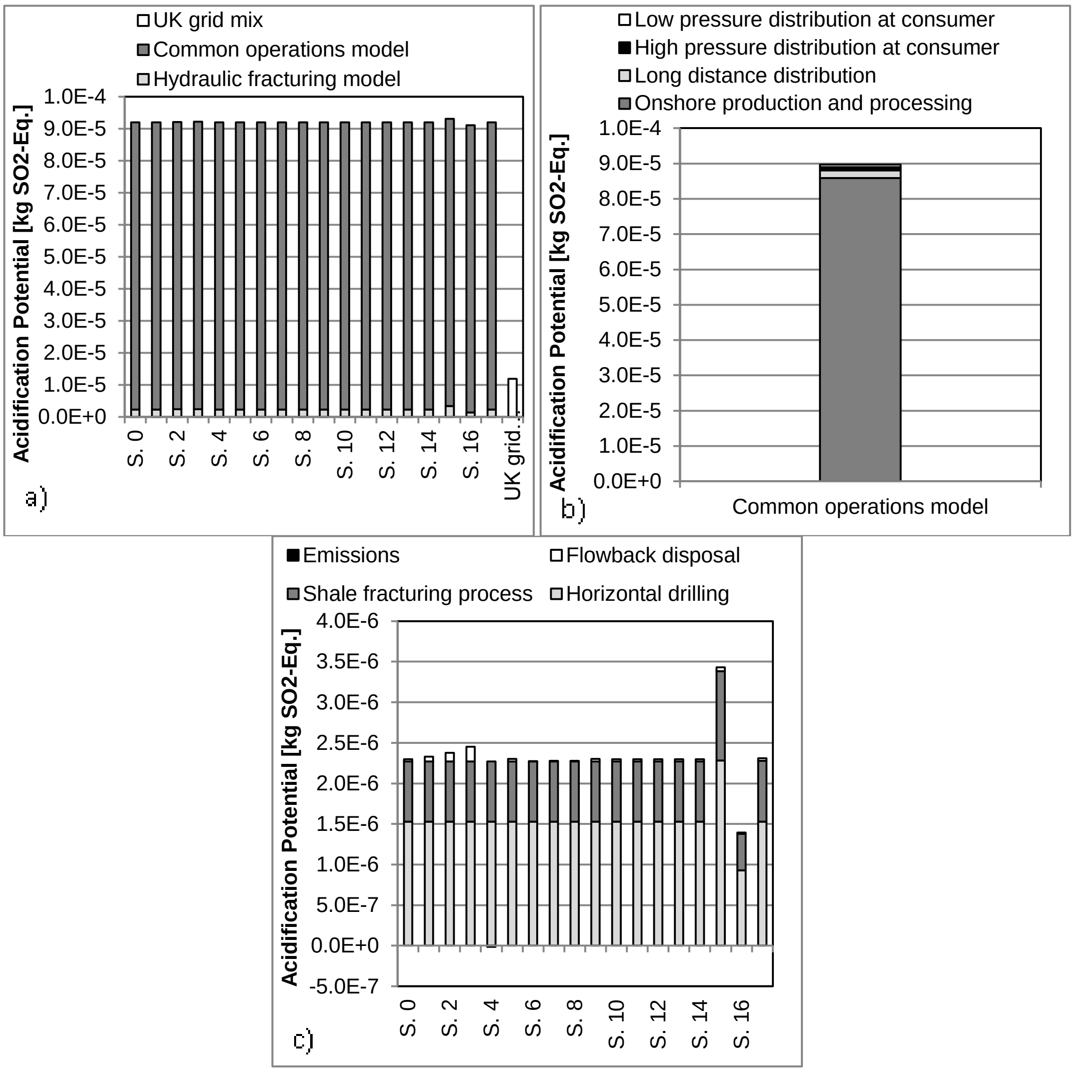


Figure 4.2.8. AP of the a) overall shale gas model; b) common operations model; c) hydraulic fracturing model.

Eutrophication potential (EP)

The hydraulic fracturing model significantly contributes to the total burden of overall shale gas model (between 9 and 20% of the total EP of the shale gas model is due to hydraulic fracturing, see Figure 4.2.9a). The major variability of the results is seen in S.15 and S.16 according to a change in EUR. 57% of the EP of the common operations model is due to the emissions associated with the production phase; 20% of the EP of the common operations model is due to the low pressure distribution by pipes (pipe construction is the process that mainly contributes to the distribution process) (see

Figure 4.2.9b). The hot spot analysis of the hydraulic fracturing model (see Figure 4.2.9c) shows that horizontal drilling is the main contributor to this process contributing more than 50% of the total (93% of which is due to direct emissions from drilling machinery). Furthermore, the direct emissions from pumping machinery contribute 90% to the burden of the fracturing process. In S.6 the increased EP of the hydraulic fracturing model (due to the flowback disposal operations) is due to the direct discharge of waste water to rivers. The comparison with conventional UK grid mix shows similar results to AP.

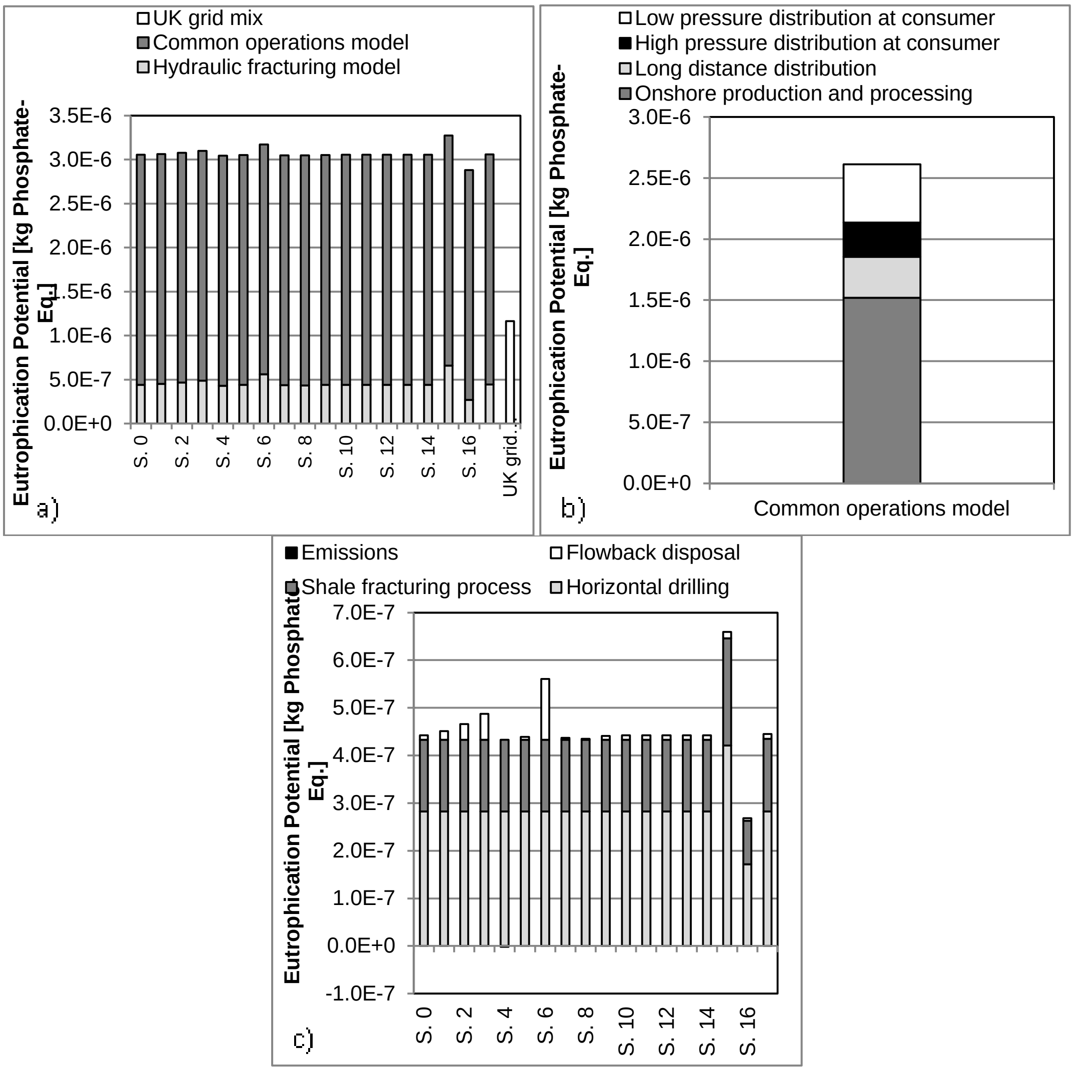


Figure 4.2.9. EP of the a) overall shale gas model; b) common operations model; c) hydraulic fracturing model.

Human toxicity potential (HTP)

As shown in Figure 4.2.10a, the hydraulic fracturing model constantly determines 4.5% of the HTP of the overall shale gas model except for S.6 (40%), S.15 (6%) and S.16 (2,5%); the EUR does not significantly influence the results. As already reported for the FAETP, the increased toxicity of S6 is due to the assumption of flowback discharge to rivers (Figure 4.2.10c). The HTP associated with the common operations model is due to pipeline production for low pressure distribution (49%) and to onshore well drilling and gas production (35%) as shown in Figure 4.2.10b. Conventional gas shows a lower HTP than shale gas.

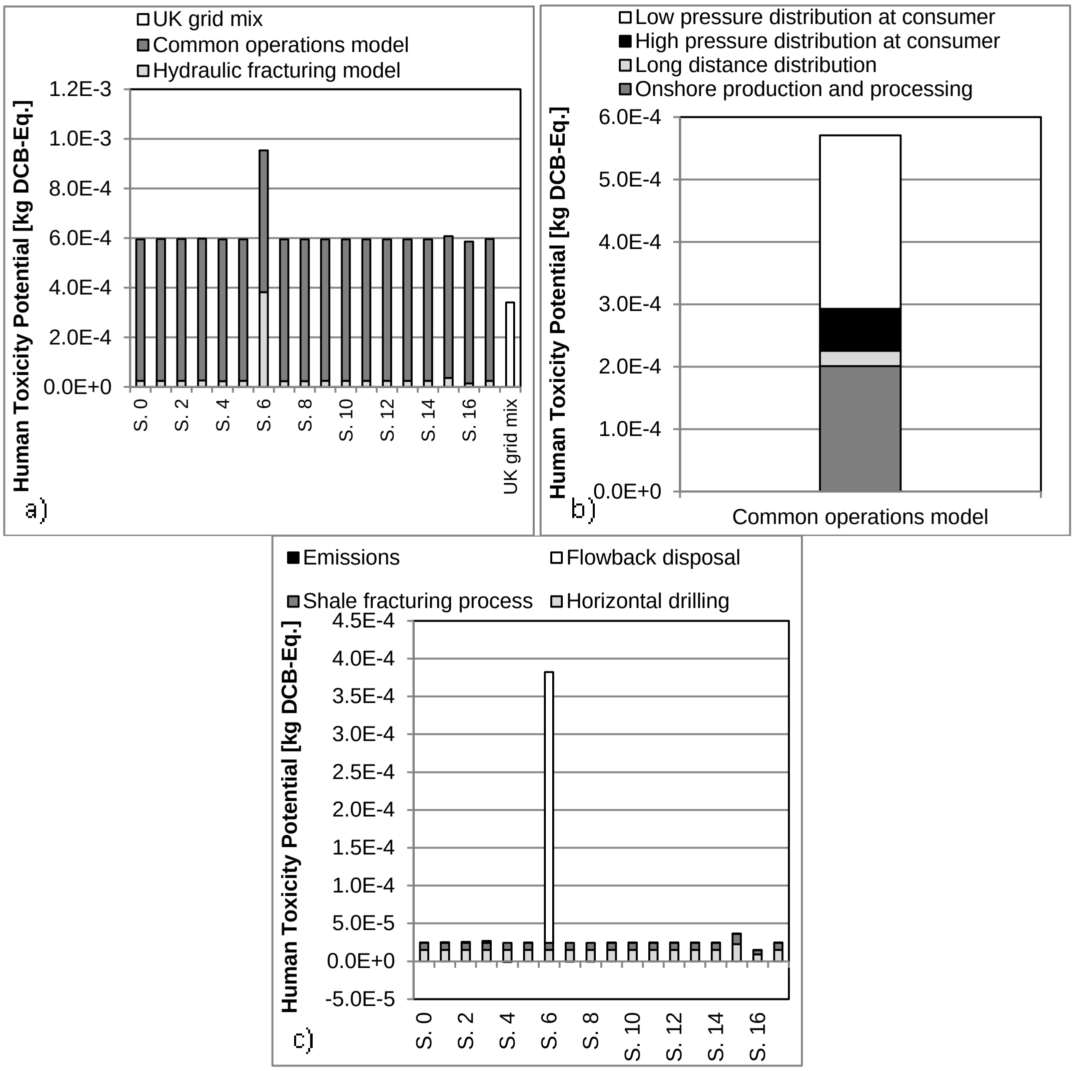


Figure 4.2.10. HTP of the a) overall shale gas model; b) common operations model; c) hydraulic fracturing model.

<u>Ozone layer depletion potential (ODP)</u>

The ODP of the hydraulic fracturing model is negligible when compared to the ODP of the overall shale gas model (Figure 4.2.11a). The main contribution is due to the common operations model and in particular to the pipe construction for long distance and high pressure transport (Figure 4.2.11b). No sensible variation is shown for the different parameters.

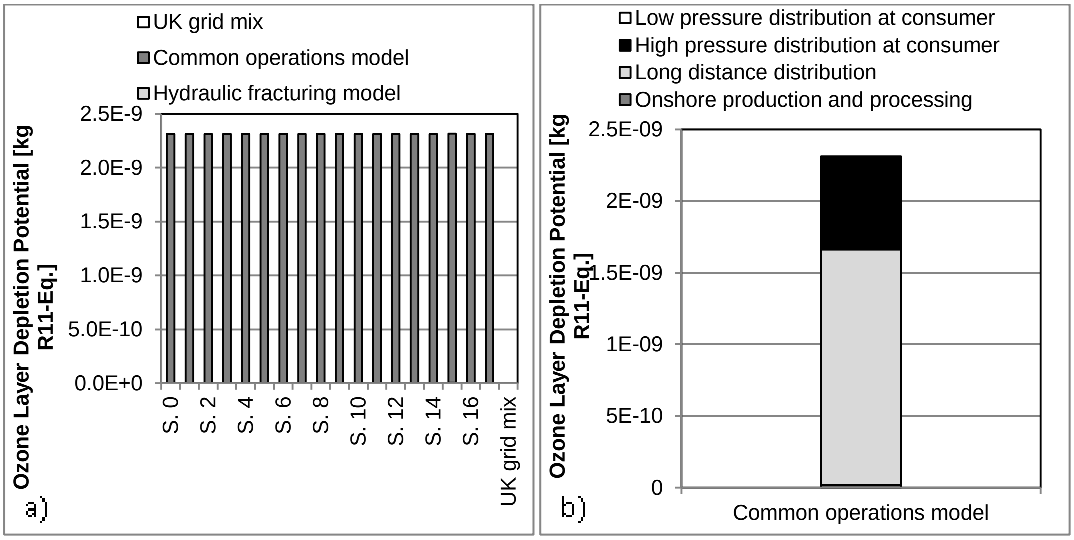


Figure 4.2.11. ODP of the a) overall shale gas model; b) common operations model.

Photochemical ozone creation potential (POCP)

The POCP of the hydraulic fracturing model is negligible as well (Figure 4.2.12a), but in this case the POCP of S.11 reaches 4% of the total POCP of the overall shale gas model. Emissions of higher hydrocarbons to the atmosphere (C_2H_6 and C_3H_8 constituting part of the natural gas) increase the POCP value of the common operations model and of the hydraulic fracturing model in the case of S.11 (Figure 4.2.12a). The UK grid gas shows lower ODP and POCP compared to shale gas and this is due to lower emissions during the offshore extraction activities and higher EUR.

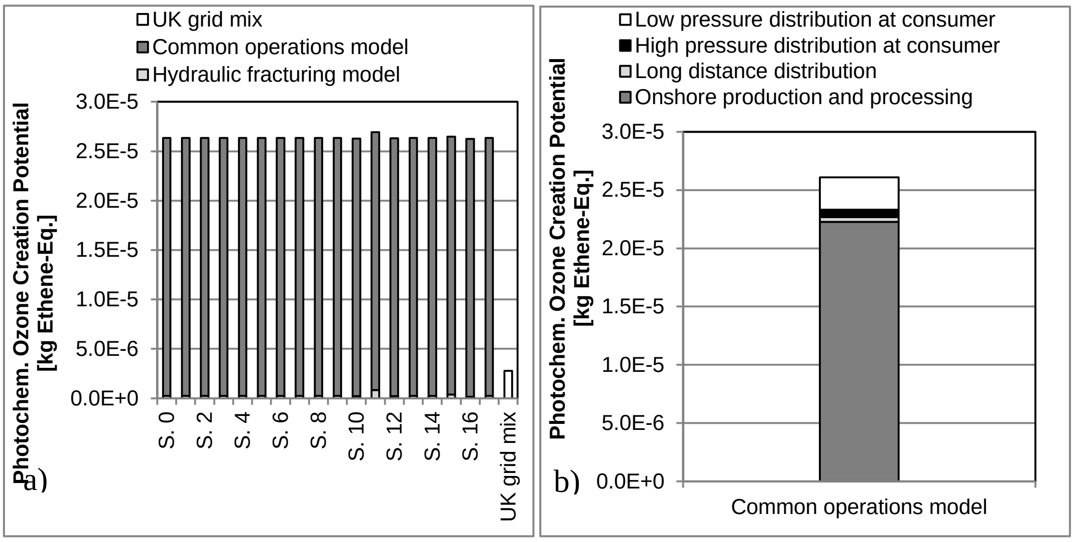


Figure 4.2.12. POCP of the a) overall shale gas model; b) common operations model.

Terrestric ecotoxicity potential (TETP)

The hydraulic fracturing model determines 2-3% of the total TETP of the overall shale gas model (Figure 4.2.13a). Onshore drilling, extraction and processing are the main contributors to the common operations model (Figure 4.2.13b). The UK gas grid mix shows a lower TETP compared to shale gas (Figure 4.2.13a).

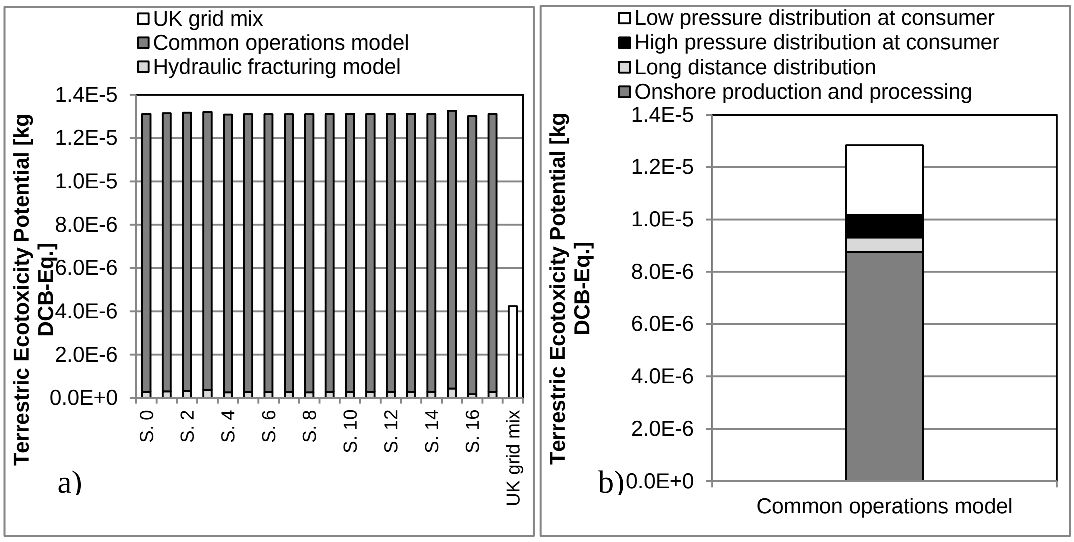


Figure 4.2.13. TEPT of the a) overall shale gas model; b) common operations model

Overall, for all the indicators analysed, transport of water, chemicals and sand does not show any strong impact on the total environmental burden of shale gas extraction as scenario 17 does not significantly change the trend of the results. Also the production of chemicals does not strongly influence the results.

4.2.5 Discussion

This discussion analyses the most critical operations of shale gas production particularly looking at the water life cycle, and shows the comparison of the results presented in this study with the results reported in literature.

The previous analysis has shown that the flowback ratio and the flowback handling method are key elements for the water use results. Hence, this aspect is further discussed according to Figure 4.2.14 that reports the life cycle of water when the flowback ratio is lower than 100%. Each of the disposal methods previously analysed - i) disposal to class II wells, ii) direct disposal to fresh water, iii) recycling iv) disposal to industrial facilities- can determine either a consumptive or degradative use of water.

1. Disposal through well injection always determines a consumptive use of water (independently from the flowback ratio) as the fresh water withdrawn from the environment before hydraulic fracturing is not released again to watersheds.
2. The direct disposal of flowback to rivers might determine either a degradative use or a consumptive use of water depending on the flowback ratio. If the flowback ratio is lower than 100% then the direct disposal to rivers and lakes

always determines a consumptive use of water. In this case, the amount of water withdrawn from environment prior to hydraulic fracturing is not completely consumed; part of it is released again to rivers but in a lower grade (degradative use of water) during flowback disposal. When the flowback ratio is higher than 100%, then the use of water is not consumptive but only degradative.

3. For disposal to proper industrial treatment the same comments as direct disposal to the environment apply.
4. Conversely, for flowback recycling, the use of water is consumptive when the flowback ratio is lower than 100%. In this case, the amount of water consumed is decreased by the amount of fresh water that is not withdrawn to hydraulically fracture another well thanks to water recycling.

A summary of the overall results is reported in Table 4.2.3.

Spillage of flowback water can negatively affect the toxicity indicators because of the composition of this water. However, if sensible solutions of flow disposal are adopted, this study showed that the operations associated with hydraulic fracturing do not substantially increase the environmental burdens of the shale gas production, as quantified by the indicators of the CML methodology.

Nevertheless, the production and distribution of the UK natural gas mix determines lower impacts for almost all the indicators. The current UK gas supply mainly comes from offshore resources and production rates for offshore wells tend to be high. This is because the natural gas reservoir must be large enough to justify the capital outlay for the completion of a well and the construction of an offshore drilling platform [364]. Therefore, higher productivity of offshore wells than shale gas wells ($9.4*10^8$ m^3 for UK offshore wells [310] and 10^7 m^3 of gas for shale wells) determine lower environmental impacts.

Furthermore, shale gas production does strongly impact the water resource use when compared to the conventional production – this also depends on the electricity mix used to fulfil the energy requirements of the production processes, as shown, for example, for the natural gas imported from Norway.

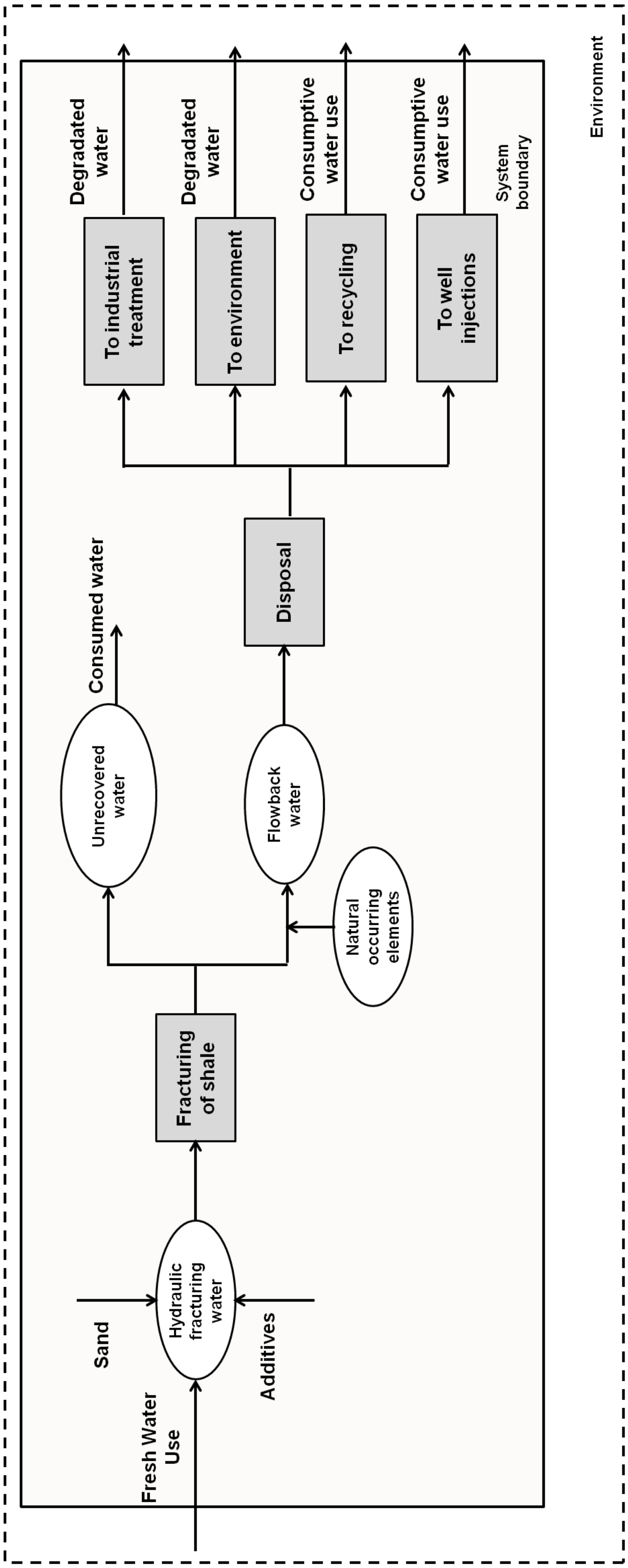


Figure 4.2.14. Life cycle of water when the flowback ratio is lower than 100%.

	Hot spot analysis of the shale gas production process		**Comparison of shale gas production with the production of the UK natural gas grid mix**
	Common operations	**Hydraulic fracturing**	**(shale gas-conventional gas)/conventional gas**
Abiotic Depletion [%]	9.94E+01	5.85E-01	-3.91E+00
Acidification Potential [%]	9.75E+01	2.50E+00	6.32E+02
Eutrophication Potential [%]	8.55E+01	1.45E+01	1.62E+02
Freshwater Aquatic Ecotoxicity Pot. [%]	9.98E+01	1.55E-01	1.82E+02
Global Warming Potential [%]	9.44E+01	5.64E+00	1.31E+02
Human Toxicity Potential [%]	9.59E+01	4.14E+00	7.50E+01
Ozone Layer Depletion Potential [%]	9.99E+01	1.34E-01	2.27E+04
Photochem. Ozone Creation Potential [%]	9.90E+01	9.59E-01	8.08E+02
Terrestric Ecotoxicity Potential [%]	9.78E+01	2.24E+00	2.10E+02
Water use [%]	7.92E+00	9.21E+01	-6.01E+01
Water consumption [%]	3.58E+01	6.42E+01	1.43E+02
Water degradation [%]	0.00E+00	1.00E+02	-6.78E+01

Table 4.2.3. Summary of the results. For each environmental indicator, the table indicates which are the operations that mainly contribute to the environmental impact of shale gas production. Hydraulic fracturing includes horizontal drilling, shale rock fracturing, flowback disposal and emissions associated to hydraulic fracturing whereas the common operations refers to gas field exploration, natural gas production, purification, long distance transport and regional distribution. In the last two columns, the table summarises whether shale gas or the UK grid mix causes the highest environmental impacts. Red represents higher values, green represents lower values.

The results shown for S.0 were compared to the results of the ‘central case’ reported in Cooper *et al.*[14] and Stamford *et al.* [343]. We considered the functional unit of 1 MJ of gas delivered to the final consumer for the three studies and excluded the use phase. Figure 4.2.15a) shows the variations obtained between our results and the literature and also directly compares the results of the two previous studies. A good agreement among all three studies is shown for the ADP fossil, the AP, the GWP, the ODP and the POCP. In particular, a variation of less than 5% is shown for the GWP. However, the results regarding the toxicity impacts and the EP significantly differ from the values reported in previous studies. This can be explained looking at the following factors.

i) Different sources of data for the processes that affect the toxicities impacts (such as the waste's disposal processes, production of drilling material, etc.).

ii) Cooper *et al.* [14] and Stamford *et al.* [343] already showed a significant variation in their results for these impacts when comparing worst, central and best case scenarios. Therefore, we compared the toxicities and the EP of S.0 to the best cases of Cooper *et al.* [14] and Stamford *et al.* [343] (see Figure 4.2.15b). A lower variation is shown.

iii) The HTP, the FAETP and the TETP are the least robust impact categories and this could affect the results.

Overall, our study confirms the central case results obtained in literature for the ADP fossil, the AP, the GWP, the ODP and the POCP; whereas the best case scenarios are confirmed for the remaining impact categories.

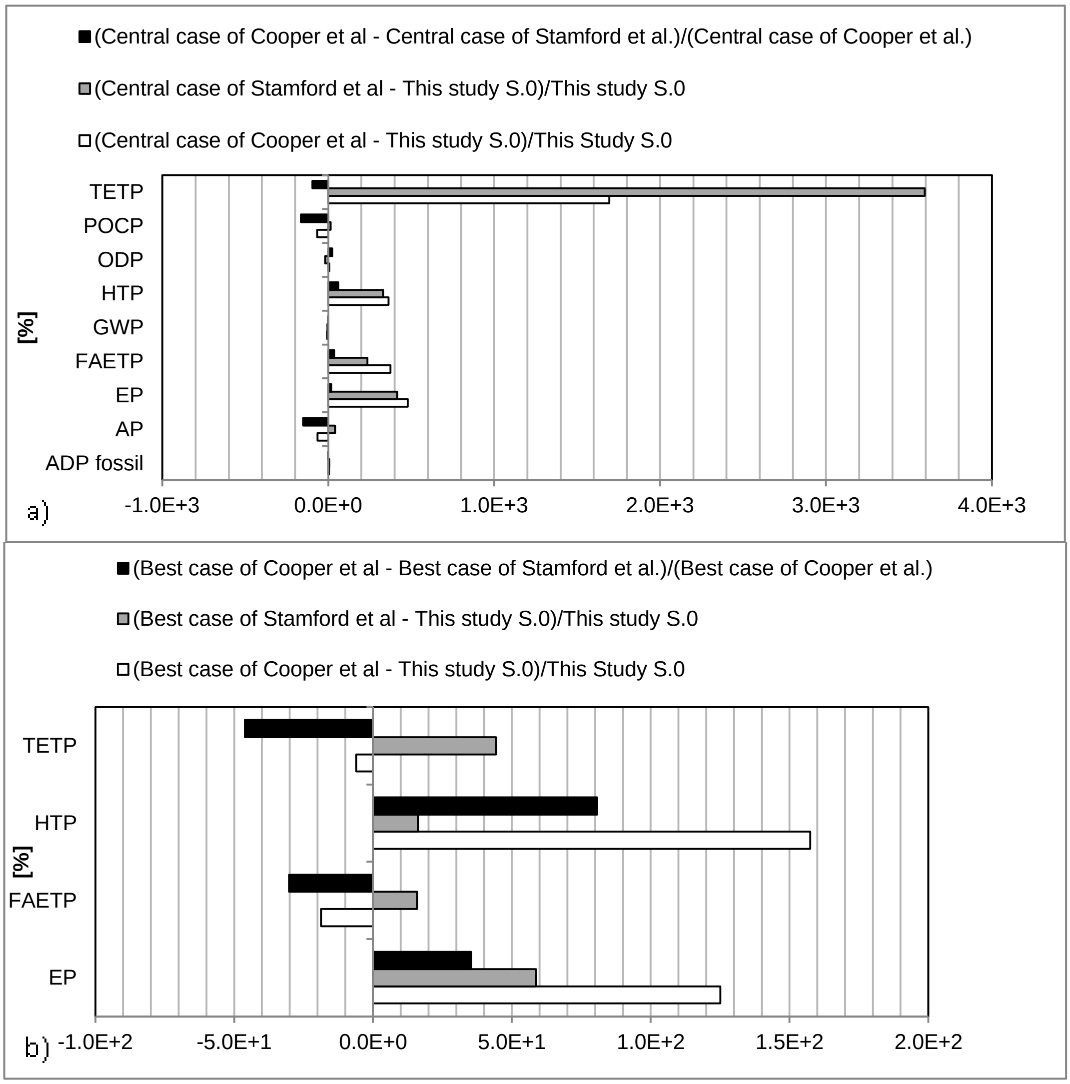


Figure 4.2.15. Comparison with literature. a) The results of S.0 are compared to the normal cases of Cooper *et al.* [14] and Stamford *et al.* [343]; b) the results of S.0 are compared to the best cases of Cooper *et al.* [14] and Stamford *et al.* [343].

4.2.6 Conclusions

Shale gas in Europe is in its early stage of exploration and research, and is pushed by the promising development it had in the US where hydraulic fracturing is already a well-known technology. In the UK, exploration and trials of UK shale gas reserves have just started but commercial production has not begun yet. This work analysed the environmental impacts of UK shale gas exploration, production and transmission at low pressure to the consumer. A sensitivity analysis was performed on EUR, fraction of flowback, emission handling methods and amount of emissions, flowback disposal method and transport distance. Particular focus was put on the water impacts; water use, water degradation and water consumption were explored and environmental impacts, including acidification potential, abiotic depletion fossil, toxicities, etc. were also considered, hence providing a further reference point against previous works.

The water impacts of shale gas significantly depend on the procedures adopted during gas production. Direct disposal of the waste water, produced during the fracking operations, into fresh water is banned by law in the UK together with injection into wells. This work analysed what the possible threats are of unwanted spills of flowback water when compared to the environmental impacts of conventional gas supply to the UK. Improper waste water management substantially increases all the toxicity impacts. This means that water and human life can be exposed to unnecessary threats, even double those caused by industrial waste water treatment.

The water degradation of the conventional natural gas supply to the UK was shown to be even higher than that of shale gas. Conversely, the water used for the shale fracturing process significantly increases water consumption when compared to the water consumption of the UK gas mix. Hence, legislations should support the recycling of flowback water as this solution allows a reduction of the total water consumption associated with shale gas production. In particular, advanced researches to improve the efficiency of the recycling process should be strongly taken into consideration.

The EUR of the well was shown to have the greatest impact on the results as well as flowback ratio and flowback disposal method. This is explained knowing that the energy and materials used for one well are the same whether the well is going to have high or low productivity. Therefore, the impacts associated with a low productive well are higher. This is valid also for the comparison between shale gas and UK grid mix gas; currently, the UK gas supply mainly comes from offshore platforms and a higher productivity of offshore wells (compared to onshore wells and shale gas) determine the higher environmental impact of shale gas. The environmental impacts of shale gas should mainly be considered according to the low productivity of shale wells that force the drilling and exploitation of a high number of wells. This is the drive of higher environmental impacts.

The emissions from drilling and pumping machineries associated with the operations specifically required for hydraulic fracturing significantly contribute to the total EP of the shale gas production. Conversely, the operations involved in hydraulic fracturing have shown a minor impact on the ADP fossil, AP, ODP, TETP and POCP. For the GWP, the handling methods of the emissions associated with the hydraulic fracturing influence the results only when emissions are vented.

The results of this study were compared to the results of previous studies and a good agreement was shown.

The analysis is limited by the early development of the shale gas exploitation in the EU. Available data are usually scarce and not robust. Further development and trials of UK shale gas extraction together with field data publicly released may help to overcome this limitation in the future.

4.3 Liquefied natural gas for the UK: a life cycle assessment

4.3.1 Environmental impacts of LNG

Natural gas and LNG have been advocated to reduce GHG emissions because they have lower carbon intensity than other fossil fuels, such as coal and oil [134,365]. However, the LNG processing and transportation emissions are reported to be even greater than those associated with coal [316]. Once vaporised, LNG has the same environmental profile as piped natural gas but the processes of liquefaction and tanker transport need to be taken into account when assessing its overall environmental performance. No complete and up-to-date environmental assessment of the LNG supply chain, from extraction to distribution to the final end-user, is available in the open literature for the UK.

A number of scientific studies have analysed the carbon footprint of LNG production and use in specific geographical contexts and against alternative energy supplies, including, for example, coal [139], compressed natural gas [319] or also heavy fuel [320] and shale gas [343].

A number of analyses have addressed the LNG supply to Japan: two studies [314,315] considered five exporting countries and both concluded that on average, the liquefaction process alone determines more than 70% of the total LNG upstream production footprint. These findings are in contrast with the results reported for the US.

LNG supply to the US has been considered mainly from Trinidad and Tobago [139,316,317]. Previous studies pointed out that the LNG imported to the US causes more GHG emissions compared to the production of domestic natural gas (from 30% [139] to 85% [317]) and that the LNG processing and transportation emissions represents almost 50% of the total supply chain. In addition, fugitive emissions from LNG transport were found to be a source of increased GHG emissions that can even dominate the whole life cycle impact [318].

European imports of LNG have been studied, but in this case the scientific literature is less homogenous and a cross comparison is more difficult. Some studies are difficult to interpret as a result of little information reported on the base assumptions used to calculate the impact of the LNG supply chain and also on the hot spot analysis. Referring to the EU, Lopez *et al.* [319] reported on the greenhouse gas emissions of two different engines using three different fuels, including compressed natural gas from LNG in Spain. In opposition to what was found for the US, Arteconi *et al.* [320] concluded for the EU that the upstream emissions for diesel and LNG for use in heavy-duty vehicles were almost identical. As later highlighted in the present study, the source and transportation distance are key factors for the total environmental burden of the LNG supply chain: some authors [321,322] analysed the upstream (emissions before the gas use) LNG emissions for different sending (West Africa or North Africa) and receiving ports (North EU or South EU) and showed that higher transportation distance can even double the GHG emissions. Conversely, others [323] focused the analysis on emissions due to the venting practice at well fields in Nigeria: the GHG emissions related to the transportation of LNG from Nigeria to Portugal can increase by 48% when considering a scenario with a higher rate of venting emissions at well field. The conclusions for the LNG imports to the US were confirmed by Korre *et al.* [324] who analysed the import of LNG to the UK from Qatar and found that almost 50% of GHG emissions were due to natural gas processing after extraction, liquefaction, LNG shipping and operations at the LNG receiving terminal. The GHG emissions from the offshore platform and pipeline transportation were found not to be significant. Stamford *et al.* [343] presented the first and only study so far covering a full range of environmental impacts for the LNG supply. They analysed the LNG import to the UK from Qatar and Algeria; however, as the study does not state the modelling principles, technologies analysed nor the system boundaries used, its significance is somewhat difficult to analyse.

The present study was undertaken to quantify the environmental burdens of LNG production, transport and distribution. This study is specific to the transport of LNG from Qatar to the UK within the Qatargas project II, and is aimed at revealing the most polluting operations in the entire life cycle. A broad range of environmental indicators are considered [366]. The modelling principles used and the technologies assumed for LNG liquefaction and transport are detailed in the following section.

4.3.2 Modelling assumptions and system boundary

4.3.2.1 LNG supply chain

The functional unit used in this study is natural gas with a gross CV of 1 MJ delivered to the end-user at grid pressure, i.e., below 7 bar. Figure 4.3.1shows the operations in the supply chain included in the system boundary. The purpose of this study is to calculate the environmental burdens of the production of natural gas as delivered to the final user; hence, the use phase is not included; in particular, the emissions from combustion are not considered.

The gas considered in this study is extracted from Qatar's North Dome Gas-Condensate field located about 80 km NE of Qatar's mainland. From here, it is sent to the Industrial city of Ras Laffan through wet subsea pipelines, where it is processed and liquefied by chilling. The natural gas extracted in Qatar usually has a high concentration of SO_x, so that desulphurization is essential [315]. Once at the facilities onshore, condensable components, sulphur compounds and CO_2 are removed [135] (the sweetening process modelled is described in the appendix). Of the gas reaching the liquefaction plant, 8.8% is used to meet parasitic energy demand, primarily for liquefaction [139,314,316]. Then, the LNG is loaded onto the purpose-built tankers. This assessment is based on gas transported from Ras Laffan to South Hook, through the Suez Canal, using state-of-the-art Q_{max} tankers, each carrying 110,000 tonnes of LNG, powered by slow-speed diesel engines fuelled by heavy fuel oil. Burdens associated with fuel use for propulsion cover the outward journey with the payload of LNG and the return journey under ballast conditions. The tankers have an average cruise speed of 19.5 knots [139,316,324]. The distance between Ras Laffan and South Hook Terminal is 11821 km [367]; the voyage lasts less than 13 days. The conservative estimate of 14 days of journey time was used in the model to account for eventual delays and for the waiting time at the entry of Suez Canal before the scheduled convoy time.

The boil off rate of the new tankers is 0.14% of the cargo volume per day [138] and this vapour is re-liquefied. The electric power requirement for reliquefaction, 6 MW, is provided by auxiliary diesel generators. The associated emissions are based on the most recent emission standards for marine transportation, commonly referred to as Tier II and introduced in 2008 in the Annex VI of the IMO 1997 [368]. Power requirements for re-liquefaction on the return journey at ballast condition are considered negligible and not included in the inventory.

The LNG is transported to and unloaded at the South Hook LNG terminal at Milford Haven in South Wales. Vaporization facilities are the last step the LNG must go through before going into the pipeline system. On arrival at the LNG terminal, the LNG is pumped ashore into insulated tanks, where it is stored at approximately -160 ˚C and atmospheric pressure [369]. When needed for injection into the National Transmission System, the LNG is pumped to heated vaporisers [370]. Operating the vaporisers uses typically 3% of the gas being vaporised [139,371].

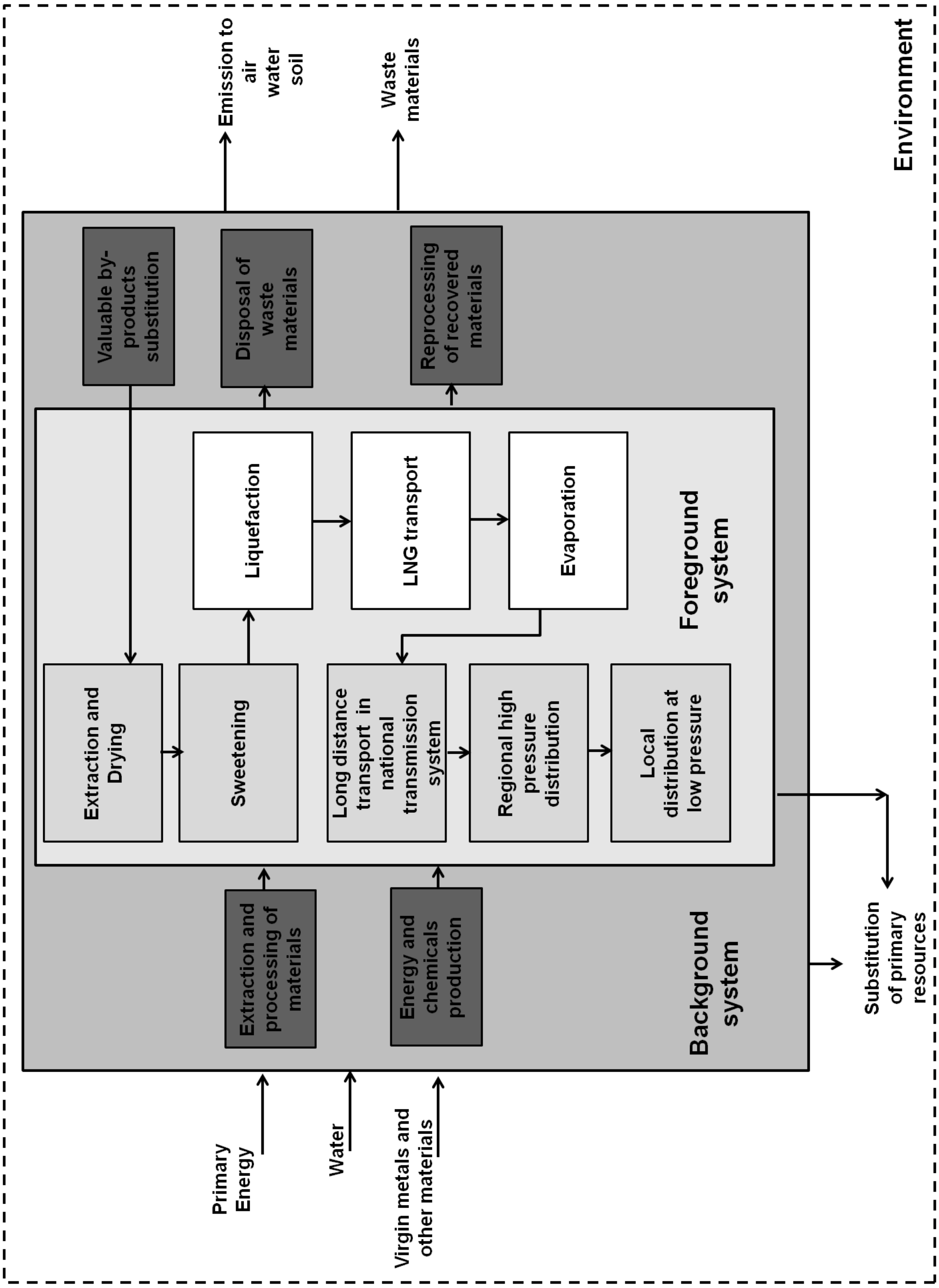


Figure 4.3.1. System boundary.

The assessment undertaken in this study covers direct burdens from all of the operations mentioned, the indirect burdens from the "upstream" supply of materials and energy, and the avoided burdens allocated to any valuable by-products obtained during gas extraction (including, for example, condensables). The key inventory data of inputs and emissions are summarised in Table 4.3.1 and Table 4.3.2, respectively; details are given in the Appendix. The inventory is mainly based on literature data, primarily the

Ecoinvent database [345] updated according to more recent data and modified for the specific case assessed here.

4.3.2.2 Scenarios

To assess the robustness of the results, a sensitivity analysis has been performed on the following parameters:

1. Emissions due to the propulsion diesel engine;
2. Tanker volume;
3. Shipping distance;
4. Energy requirements for the liquefaction process;
5. Energy requirements for the vaporization process;
6. Fuel used for propulsion;
7. Acid gases and sulphur removal.

The scenarios explored are summarised in Table 4.3.1:

S.0. Base scenario, using the assumptions reported above.

S.1-S.2. These scenarios explore different emission levels from the propulsion engines of the tanker: respectively 5/7 and 9/7 times the values in the base case were considered to account for higher and lower fuel and off gas cleaning efficiency.

S.3-S.4. The capacity of the tanker is changed from 210000 m^3 to 266000 m^3, to represent the minimum and the maximum capacity of the new tanker ships (Q_{flex} and Q_{max}, see chapter 2).

S.5-S.6. Different transport distances are considered: 15000 km in S5 and 5000 km in S6, compared with 11,821 km in the base case, to represent the maximum and minimum distances to Europe from Qatar: 5000 km is the average distance between Qatar and the Northern end of the Suez Canal, whilst 15000 is the average distance from Qatar through the Suez canal to the most northerly European countries (including Norway and Lithuania).

S.7-S.8 and S.9-S.10. Changes in the liquefaction and vaporization energy requirements are explored.

S.11. The tanker main propellers and auxiliaries are assumed to be powered by marine gas oil instead of diesel oil.

*S.12.*The sweetening process is not included in this scenario, representing a possible case in which the natural gas extracted is sweet and does not require acid gases removal.

		Lower emission than steam turbine	**Tanker volume**	**Shipping distance**	**Days of navigation**	**Days of navigation**	**Liquefaction energy requirements**	**Regasification energy requirements**	**Propulsion fuel**	**Gas composition** Sour- sweetening included?
	Unit	%	10^3 m^3	km	days	days	%	%	HFO	Y
	S.0	-30	263	11281	13.01	14	8.8	3	HFO	Y
Emission level	**S.1**	-50	263	11281	13.01	14	8.8	3	HFO	Y
	S.2	-10	263	11281	13.01	14	8.8	3	HFO	Y
Tanker Volume	**S.3**	-30	210	11281	13.01	14	8.8	3	HFO	Y
	S.4	-30	266	11281	13.01	14	8.8	3	HFO	Y
Shipping distance	**S.5**	-30	263	15000	17.30	18	8.8	3	HFO	Y
	S.6	-30	263	5000	5.77	6	8.8	3	HFO	Y
Liquefaction system	**S.7**	-30	263	11281	13.01	14	15.0	3	HFO	Y
	S.8	-30	263	11281	13.01	14	5.0	3	HFO	Y
Regasification system	**S.9**	-30	263	11281	13.01	14	8.8	5	HFO	Y
	S.10	-30	263	11281	13.01	14	8.8	1	HFO	Y
Propulsion and reliquefaction fuel	**S.11**	-30	263	11281	13.01	14	8.8	3	MGO	Y
Gas composition	**S.12**	-30	263	11281	13.01	14	8.8	3	HFO	N

Table 4.3.1. Scenarios analysed and key input data for the delivery of 1 MJ of natural gas.

		Extraction and drying	Sweetening	Natural gas liquefaction p	Transport of LNG	LNG evaporation	Natural gas distribution at long distance pipeline	Natural gas distribution at high pressure to the consumer	Natural gas distribution at low pressure to the consumer	Total
CO_2	10^{-3} kg	2.3	5.9	5.5	6.8	2.4	0.66	0.41	0.18	24
CO	10^{-6} kg	6.3	14	8.6	6.4	3.8	0.87	0.62	1.3	42
H_2S	10^{-9} kg	2.8	1.3	2.3	9.9	2.2	0.27	4.2	1.0	24
N_2O	10^{-9} kg	39	9.0	3.6	120	32	10	6.2	2.6	220
S_2O	10^{-6} kg	1.4	140	13	3.6	0.43	0.90	0.30	0.41	160
Group NMVOC to air	10^{-6} kg	5.8	110	12	1.4	0.14	0.73	2.8	11	140

Table 4.3.2. Key output emissions reported for the delivery of 1 MJ of natural gas (base scenario is considered).

4.3.3 Results and discussion

The environmental burdens of the base scenario were normalised according to the normalisation factors of the CML, IPCC, ReCiPe (region equivalents, EU25+3, year 2000) [310] method and the results are reported in Table 4.3.3. The normalised results show that the most significant impacts are GWP, HTP, FAETP, ADP, POCP, and AP. As better highlighted later in the result discussion, both AP and POCP are strongly influenced by the sulphur content of the treated gas. Conversely, the burdens associated with GWP, FAETP and HTP are more uniformly distributed among the operations included in the system boundaries; and finally, the ADP is driven by the depletion of the fossil resource of natural gas.

1 MJ of natural gas	**UK LNG**
Abiotic Depletion Potential elements (ADP)	$6.58\ 10^{-16}$
Abiotic Depletion Fossil (ADP)	$3.12\ 10^{-14}$
Acidification Potential (AP)	$1.32\ 10^{-14}$
Eutrophication Potential (EP)	$5.75\ 10^{-16}$
Fresh Water Aquatic Eco-toxicity Potential (FAETP)	$4.56\ 10^{-15}$
Global Warming Potential (GWP)	$5.51\ 10^{-15}$
Human Toxicity Potential (HTP)	$2.71\ 10^{-15}$
Ozone Layer Depletion Potential (ODP)	$2.46\ 10^{-16}$
Photochemical Ozone Creation Potential (POCP)	$3.46\ 10^{-14}$
Terrestric Eco-toxicity Potential (TETP)	$2.68\ 10^{-16}$

Table 4.3.3. Normalized results. The normalisation is done based on CML, IPCC, ReCiPe (region equivalents), EU25+3, year 2000 [310].

Global Warming Potential (GWP) (see Figure 4.3.2e)

All parameters analysed determine a variation in the results. The major variations from the base case are shown for S.6 and S.7 and S.12. The GWP of the extraction and drying and the sweetening processes are the same for all scenarios. The same is also valid for the GWP of the liquefaction process except for S.7 and S.8. Doubling or halving the energy requirement compared to the base scenario, significantly changes the GWP of the liquefaction process- the GWP of the liquefaction process is $9.4*10^{-3}$ kg of CO_2 eq. for S.7 and $3.4*10^{-3}$ kg of CO_2 eq. for S.8 compared to $5.71*10^{-3}$ kg of CO_2 eq. of S.0. The GWP of the LNG transport is mainly due to the direct emissions from the diesel engine and shipping distance (the direct activities determine almost 87% of the GWP associated with transport in S.0). The processes of liquefaction, LNG transport and evaporation determine more than 50% of the total GWP. The GWP of the evaporation process almost doubles between S.9 ($1.5*10^{-3}$ kg of CO_2 eq.) and S.10 ($3.7\ *10^{-3}$ kg of CO_2 eq.), as a result of different energy requirements. For all scenarios, almost 18% of the total GWP is due to the distribution of the evaporated gas.

The total GWP calculated in this study for the upstream processing (including production and processing, liquefaction, transport and evaporation but not distribution) equals 0.0174 kg of CO_2 eq. and this value is perfectly in line with values already reported in literature ~ 0.016-0.018 kg of CO_2 eq [314,317,321,323,372].

Fresh Water Aquatic Eco-toxicity Potential (FAETP) (see Figure 4.3.2d).

The variation of the results from the base scenario, except for S.5, S.6 and S.12 is negligible as it is always lower than 1%. The shipping distance is the parameter that determines a substantial variation of the results in S.5 and S.6 (+10% and -16% from the base scenario for S.5 and S.6, respectively). About 18% of the total FAETP is due to the emission to fresh water occurring during the extraction and drying process. The burdens allocated to the sweetening process and to the liquefaction process are mainly due to the indirect activities linked to the production of the processing plants- for S.0, 88% and 93% of the total FAETP of sweetening and liquefaction, respectively, are due to the production of the processing plants. Transport of LNG always determines 30% of the total FAETP except for S.5 and S.6 (35% and 16%, respectively). The burden due to the transport of LNG is equally distributed between outward and inward journey and this is due to the operation and maintenance of the sending ports (this represents 64% of the total burden due to transport for S.0). The evaporation process determines between 10 and 15% of the total FAETP. About 13% of the FAETP is due to the distribution of vaporised natural gas to the final consumer. Pipeline construction and installation determine the main burden allocated to the distribution.

Human Toxicity Potential (HTP) (see Figure 4.3.2f)

Negligible variation of the results is shown for the HTP. The variation from the base scenario is always less than 8% except for S.5 (+12% from the base scenario). Similarly to the FAETP, the extraction of gas determines around 25% of the total HTP. The sweetening and liquefaction always cause between 10 and 19% of the total HTP. The impact of LNG transport is always around 22% of the total HTP except for S.5 (27%) and S.6 (11%). The impact due to the LNG transport is allocated to the indirect activities, such as fuel, chemicals and material production, port maintenance, etc. and these account for almost 78% of it. A significant contribution to the HTP (around 28%) is due to the distribution of the vaporised gas (mainly the construction, installation and maintenance of pipes).

Ozone Layer Depletion Potential (ODP) (see Figure 4.3.2g)

ODP is the only indicator that is almost completely due to the distribution at long distance pipeline and low pressure to the final consumer. Negligible variation in the results is shown. Pipelines production, installation and maintenance determine 99% of this burden.

Abiotic Depletion Potential (ADP) (see Figure 4.3.2a)

This indicator does not show a high variability in the results; it does not strongly depend on the parameters analysed in the sensitivity analysis. S.12 shows a lower ADP because the sweetening is not considered and hence the total energy requirements for processing are lower. S.12 shows a variation from the base scenario lower than 5%. For all the scenarios more than 95% of the ADP is due to the depletion of natural gas associated with gas extraction. As in this particular study the depletion of fossil resources is under study, the normalised ADP of Table 4.3.3 shows the highest contribution compared to all other indicators.

Acidification Potential (AP) (see Figure 4.3.2b)

For all the scenarios analysed, except for S.12, more than 75% of the total AP is due to the sweetening of the natural gas before liquefaction. This value increases to 83% for S.6 as a result of the lower environmental burden associated to transport (shorter transport distance is assumed for this scenario). In S.0, the emissions due to the use of sour gas for energy requirements in a gas turbines during the removal of S, determine 86% of the AP of the total sweetening process. The burden of the liquefaction process is due to the indirect activities associated with the production of its energy requirements (that is the production and processing of the natural gas used within the liquefaction process). The LNG transport contributes between 3.1 and 9% (for S.6 and S.5, respectively) to the total AP thanks to the strict limits that regulate the sulphur content of the fuel oils. Except for S.12, the sensitivity analysis shows a negligible variation in the results- the min and max value of the AP are 2.13^{-4} kg of SO2 eq. and 2.33^{-4} kg of SO2 eq., respectively for S.6 and S.7, corresponding to a -3.8% and +5.1% variation from the base scenario. When the extracted gas is assumed to be sweet (S.12), a deep pre-processing including sulphur and acid gas removal is not included in the assessment and hence, the AP dramatically decreases. In this case, LNG transport (the direct emissions to environment) and liquefaction are major contributions to AP.

Photochemical Ozone Creation Potential (POCP) (see Figure 4.3.2h)

This indicator is driven by the direct emissions associated with the gas sweetening. The burden allocated to the liquefaction process is associated with the indirect emission due to the processing of the natural gas used for energy requirements. The burden allocated to the evaporation process is negligible. Negligible variation of the results is shown for the sensitivity analysis, except for S.12. For all scenarios, except for S.12, about 11-

12% of the total POCP is due to the extraction and distribution of the vaporised natural gas.

Eutrophication Potential (EP) (see Figure 4.3.2c)

S.5 shows the highest EP because of the longer transport distance, whereas the lowest burden is shown for S.6 because of the lower transport distance (+13% and -23% from the base scenario, respectively). Transport always determines between 25% and 49% of the total EP (lowest value of 25 % is reported for S.5). The burden for transport is prevalently allocated to the direct emissions from diesel engines used for propulsion and auxiliaries (that includes the re-liquefaction systems). For all scenarios, except S.11, the sweetening process always determines about 10% of the total EP and in particular this is due to the direct emission due to the burning of sour gas used as energy source for the process. The extraction and drying of gas determines up to 23% (for S.6) of the total EP. Conversely, the evaporation process determines between 7.5 and 15% of the total EP (for S.10 and S.6, respectively). This is due to the indirect emissions allocated to the processing of the natural gas used for the energy requirements (58% of the total evaporation EP) and to the production and maintenance of the evaporation plant (35% of the total evaporation EP). The distribution of the vaporised LNG in the UK causes around 10% of the total EP.

Terrestric Eco-toxicity Potential (TETP) (see Figure 4.3.2i)

The variation of the results for this indicator is negligible. Around 35% is due to the extraction process. The sweetening contributes 15% to the total TETP, except in S.12. About 15% of the TETP is due to the transport and in particular half of this burden is due to direct emissions from the engines (for S.5 and S.6 this value changes into 18 and 6%, respectively, according to different transport distances). The evaporation process causes about 10-13% of the total TETP; in this case the burden is mainly due to indirect activities (including plant production and maintenance). The remaining TETP is due to the production, installation and maintenance of the pipes used to deliver the vaporised natural gas to the final consumer.

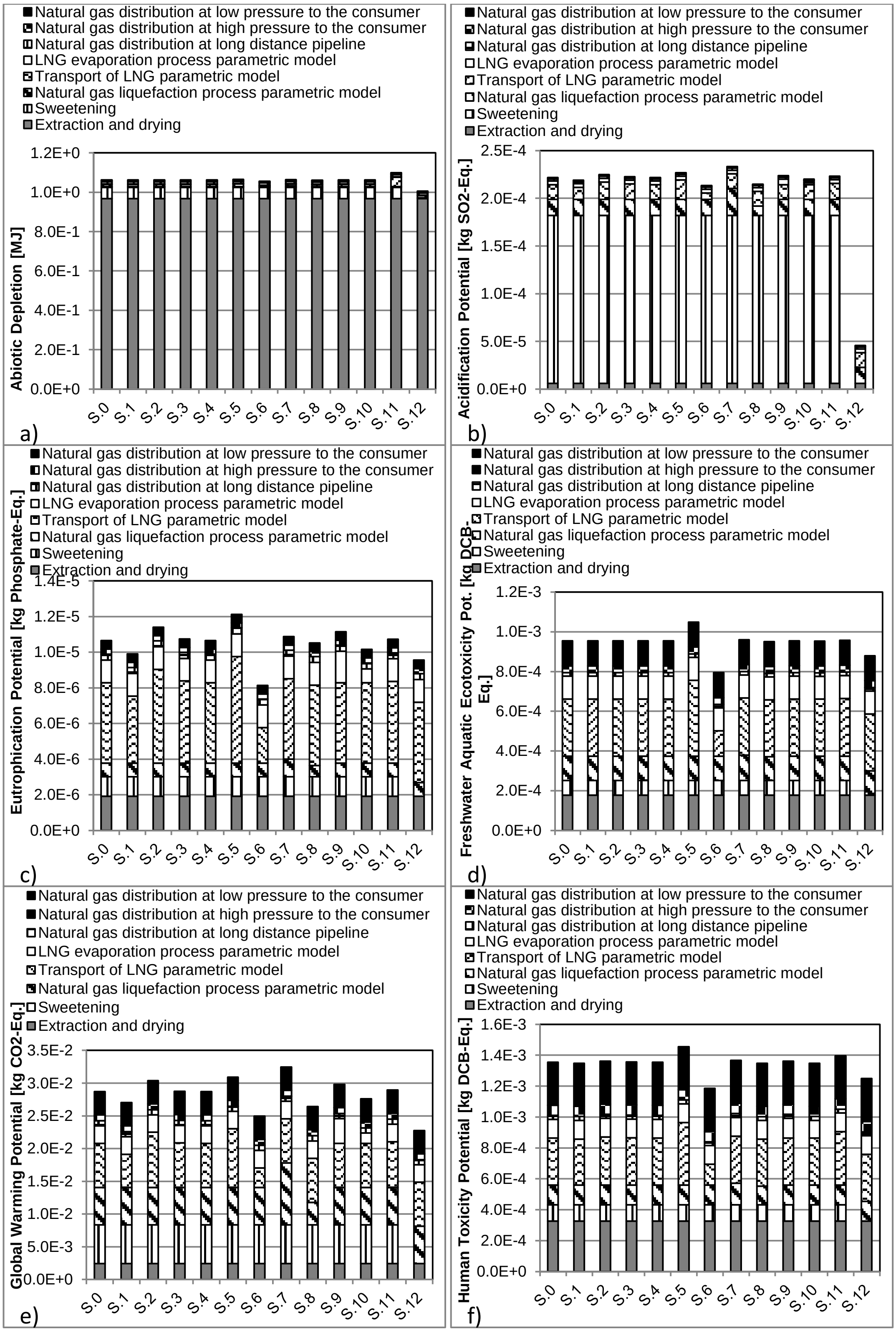
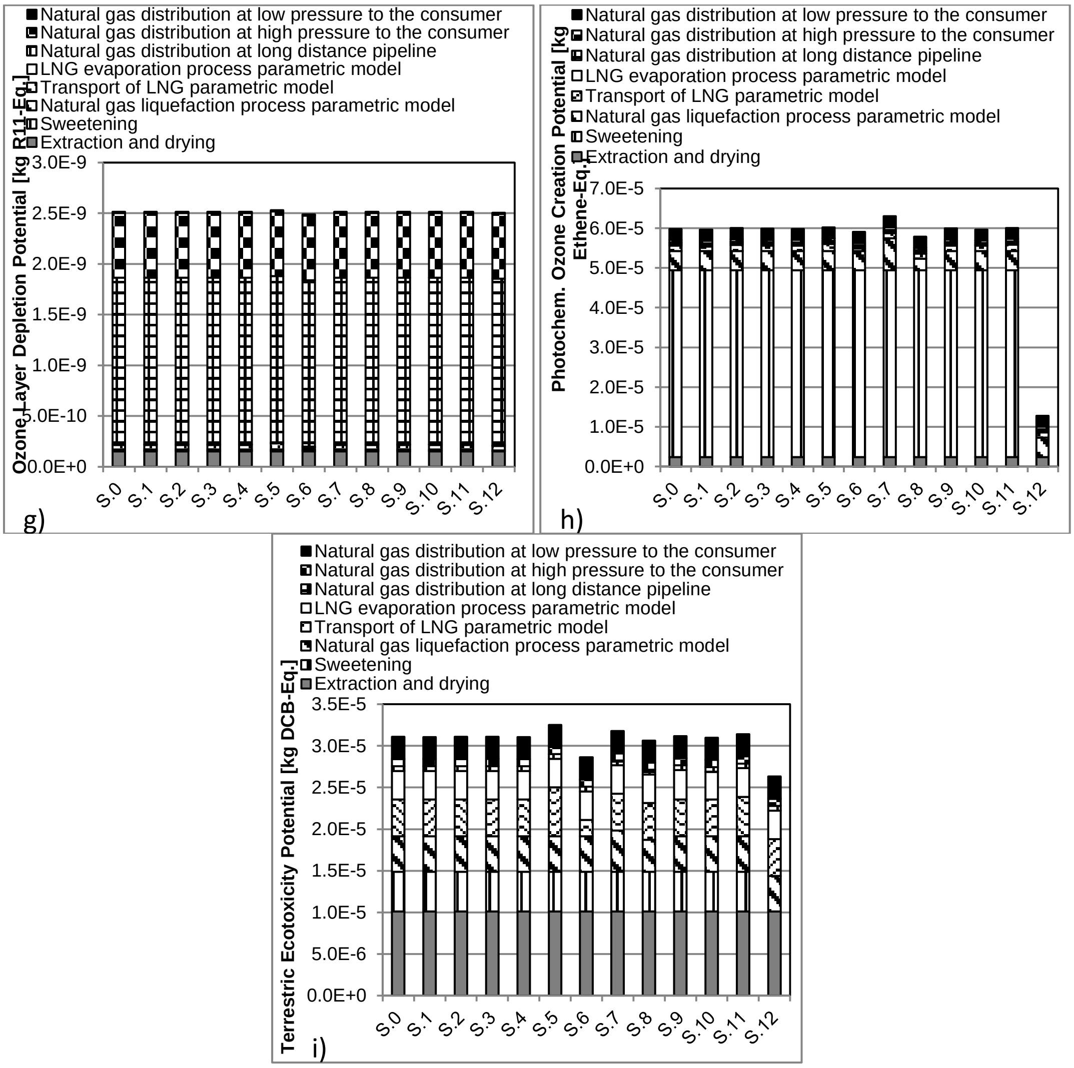


Figure 4.3.2. Life cycle assessment of the LNG supply chain to the final consumer. a) ADP; b) AP; c) EP; d) FAETP; e) GWP; f) HTP; g) ODP; h) POCP; i) TETP.



(Continued) Figure 4.3.2. Life cycle assessment of the LNG supply chain to the final consumer. a) ADP; b) AP; c) EP; d) FAETP; e) GWP; f) HTP; g) ODP; h) POCP; i) TETP.

Table 4.3.4 shows the direct emissions due to the shipping of LNG according to the delivery of 1 Nm^3 of LNG at the receiving terminal for the base scenario. The previous discussion on the different environmental impacts accounted for the use of re-liquefaction systems; Table 4.3.4 shows the emissions included in the base model and the potential emissions in case re-liquefaction is not used. If the re-liquefaction system is not included, the emissions of CO_2, CO, NOx are reduced by a nugatory amount (see Table 4.3.4) as a result of the lower amount of energy required by the auxiliaries (i.e. by the lower amount of fuel burnt in the auxiliary engines). Conversely, the potential methane emissions significantly increase because of the boiled off gas. In this case, the boil-off gas needs to be used as fuel to reduce the GWP. However, given that the new diesel engines of the newer tanker ships do not run using methane but use HFO, in this case the re-liquefaction system is a necessary requirement in order to limit the GWP and not emit flared vaporised gas into the atmosphere.

10^{-6} kg of emission during the outward journey/Nm^3 of delivered gas	With reliquefaction system	No reliquefaction system
Carbon dioxide	125,707	118,070
Carbon monoxide	90.7	83
Methane	11.8	11.8 + 14,034 (of regasified gas)
Nitrogen oxides	519	402
Nitrous oxide	2.07	2.07
NMVOC	12.4	2.07

Table 4.3.4. Direct emission due to LNG shipping including and excluding the on-board reliquefaction systems.

Furthermore, the effect of possible fugitive methane emissions along the supply chain has been analysed. These emissions can be critical for operations such as extraction, liquefaction, storage before transport, transport itself and evaporation. On the basis of 1 MJ of gas delivered to the final consumer as functional unit, when 1% of the total gas delivered is vented as methane emissions leakage throughout the entire supply chain, the GWP increases by 15% compared to the GWP for the base scenario. The GWP increases by 78% compared to the base scenario when 5% of the delivered gas is considered to be lost as vented emissions. The GWP increase ranges between 2% and 9% if a flaring system is considered to abate methane leakage. This analysis confirms the extremely dangerous effect of fugitive emissions on the total GWP of the LNG supply chain.

Except for ADP, for all the scenarios and the indicators analysed, the extraction and drying of natural gas shows limited impact on the total environmental burden of the LNG supply chain. This is due to the natural gas pre-processing, the liquefaction, the LNG transport and the LNG vaporization that determine a substantial contribution to the total environmental burden of the LNG production and distribution.

The GWP obtained in this study is compared to the GWP reported in other literature studies, as shown in Figure 4.3.2 (the base case has been used for comparison with literature data). As far as possible, data for the GWP on the LNG upstream production have been collected according to three categories- production and processing, liquefaction and evaporation- from literature [314,315,317,319–323,372]. The results reported in Figure 4.3.2 are strictly correlated to the assumptions used at the basis of each analysis. However, the GWP obtained in this study is the same order of magnitude as the results previously reported. Production and processing are shown to determine a significant contribution to the GWP although not the main one. In Safei *et al.* [323] the contribution of production and processing is the highest as a result of the assumption of flaring emissions at LNG fields in Nigeria and leasing of fuel emissions. For all the studies analysed, the liquefaction process determines a relatively high contribution to the total upstream life cycle of LNG. Conversely, the evaporation process is not the main cause of GWP in the entire life cycle. As also highlighted in the sensitivity analysis of this study, the shipping distance of the LNG is a key factor for the emissions associated with transport. Studies considering longer transport distance, for example from Middle East to North of Europe [322], show a higher GWP.

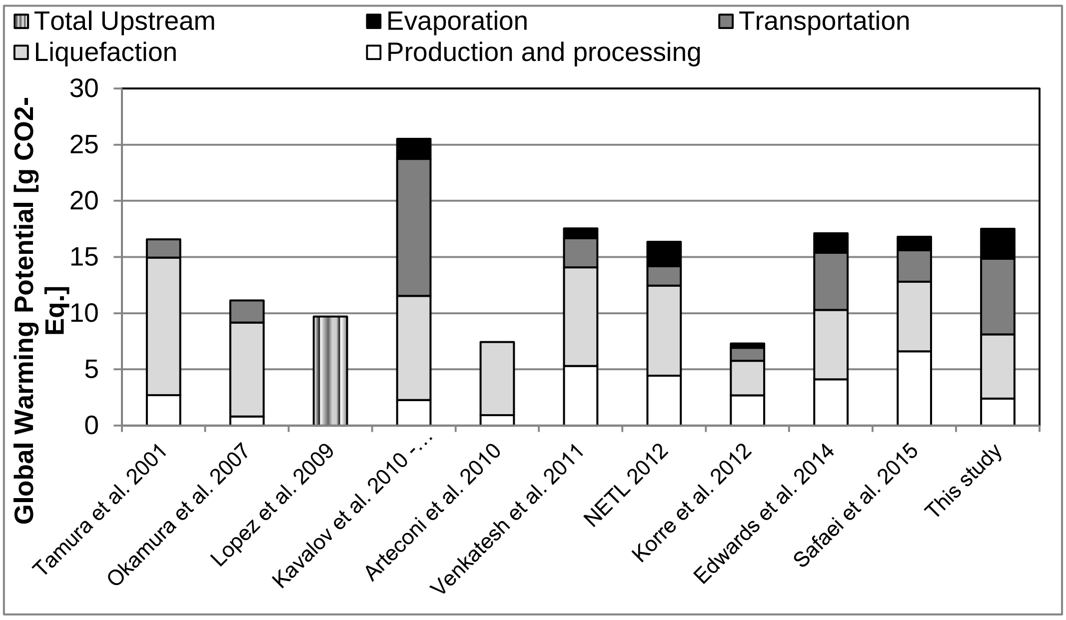


Figure 4.3.3. GWP: LNG upstream life cycle according to different studies.

The present analysis which has also been expanded to other environmental impacts than the GWP shows that both direct and indirect burdens specific for the LNG supply (that are processing, liquefaction, transport and evaporation) cannot be considered negligible when looking at the supply of natural gas for future energy scenarios.

The parameters that mainly influence the results are the shipping distance, the tanker volume, the sweetening of the gas and eventual methane leakage. A change in the tanker volume means a change in the amount of gas transported and hence, a higher/lower environmental impact per MJ of gas - the higher energy requirements of a bigger tank are offset by the higher amount of LNG transported. Conversely, a longer or smaller shipping distance significantly changes the amount of energy and emissions during transport. As far as the use of bigger engineered tanks to reduce the environmental impact is a costly but achievable solution, the burden due to the distance of LNG shipping to the UK from Qatar cannot be changed. Unless other types of transport, such as gas piping, become a technically feasible and least polluting option than LNG shipping for very long distances, LNG shipping seems to be a compromise solution for natural gas imports from long distances.

However, the impacts of the distribution of the vaporised gas to the final consumer at low pressure cannot be considered negligible and are mainly due to the construction, installation and maintenance of the pipes and also to the energy requirements and emissions from piping.

The environmental burden of the LNG production and transport needs to be clearly and fairly allocated between exporting and importing countries [373]. Some may claim that the emissions due to extraction, processing and transport have to be allocated to the exporting countries, hence alleviating the greenhouse gas emission of the importing countries in order to meet the strong regulating limits, or, the other way around. It has been be suggested that the allocation of the burdens due to activities spread through different countries should take into account not only the country specific regulations to meet greenhouse gas emissions but also their global effects [373].

4.3.4 Conclusions

LNG is expected to be an increasing supply of energy for the UK while the national reserves of the continental shelf are diminishing. Qatar is increasing the export of LNG thanks to world-wide ventures and to the improvement of LNG technologies. As a case study, the environmental impacts of the LNG supply to the UK within the new project Qatargas II were analysed. New tanker ships and facilities were assumed to be used in the analysis. The entire life cycle of the LNG supply chain, from the gas extraction to the distribution to the consumer, has been included in the assessment. The main findings of this study highlighted how the operations specifically associated with LNG, that include liquefaction, transport and vaporization, significantly influence the environmental impact of the total supply chain and hence they cannot be considered negligible in a complete environmental assessment. The sensitivity analysis has explored the influence of some key parameters, such as energy requirements of the liquefaction and vaporisation process, fuel for propulsion, days of navigation (that is shipping distance), tanker volume and sweetening process (specifically required to process natural gas extracted in Qatar as this is reported [315] to have a particularly high concentration of SO_x). The last three parameters determine the main variation in the results. The case study here reported highlights how i) long distance LNG transport and

ii) natural gas processing including sweetening, liquefaction and vaporisation, are the key aspects that alter the total environmental burdens. Fugitive emissions that occur during LNG loading, transport, and unloading must strictly be avoided to reduce the impact on global warming.

4.4 LNG-shale gas: a life cycle assessment comparison for the UK

The aim of this section is to compare the life cycle assessment of the production of shale gas in the UK (see section 4.2) to the supply of LNG from Qatar (see section 4.3). The environmental burdens of these two technologies are compared to the environmental burdens of the production and delivery of the conventional natural gas grid mix in the UK [310]. The first part of the results is reported for to the delivery of 1MJ of gas to the final consumer at low pressure (< 7 bar and > 0.75 mbar gauge). In the second part of the analysis, natural gas is assumed to be used for household heating and hence burnt in a boiler. The boiler, running at atmospheric pressure, is condensing, non-modulating (<100 kW). The use phase is assumed to be the same for all three technologies analysed. In this case, the results are reported for the use of 1 MJ of natural gas (extraction processing and distribution are also included).

4.4.1 Environmental burdens of LNG, shale gas and grid mix

Table 4.4.1 shows the environmental burdens of the systems analysed in this study according to the delivery of 1 MJ of gas to the final consumer; the use phase is not included in this case. The first column reports the environmental burdens associated with the UK gas mix in 2015. The second column of Table 4.4.1 refers to the burdens of shale gas production in the UK (S.0 as reported in section 4.2). The third column reports the burdens of LNG imports of natural from Qatar to the UK (S.0 as reported in section 4.3).

Values for 1 MJ of natural gas delivered to the final consumer	**UK natural gas grid mix supply to the final consumer**	**UK shale gas supply to the final consumer**	**UK LNG supply to the final consumer**
Abiotic Depletion [MJ]	0.955	0.918	1.06
Acidification Potential [10^{-6} kg SO2-Eq.]	12.6	92.0	222.0
Eutrophication Potential [10^{-6} kg Phosphate-Eq.]	1.16	3.06	10.6
Freshwater Aquatic Ecotoxicity Pot. [10^{-6} kg DCB-Eq.]	129	365	954
Global Warming Potential [10^{-3} kg CO2-Eq.]	4.00	9.23	28.7
Human Toxicity Potential [10^{-3} kg DCB-Eq.]	0.34	0.596	1.35
Ozone Layer Depletion Potential [10^{-9} kg R11-Eq.]	0.0102	2.31	2.51
Photochem. Ozone Creation Potential [10^{-6} kg Ethene-Eq.]	2.90	26.3	59.9
Terrestric Ecotoxicity Potential [10^{-6} kg DCB-Eq.]	4.24	13.1	31.1

Table 4.4.1. Environmental burdens of natural gas extraction, processing and supply according to different sources and technologies. Results are reported for 1 MJ of delivered gas.

All the indicators, except the ADP show the same trend: the UK natural gas mix determines the lowest environmental burden whereas the LNG supply shows the highest impacts. For all the indicators, the results differ in one order of magnitude. The UK grid mix represents the best option as a result of the higher gas recovery from offshore wells that decrease the environmental burden calculated per MJ of delivered gas. The delivered LNG is also produced at offshore wells; those usually have a higher recovery than onshore wells, therefore the total environmental burden should be similar to the burdens of the UK gas grid. Conversely, the highest burdens of the LNG are due to the gas processing, ship transport and vaporization process; the environmental burdens of these processes are added to the burdens of offshore extraction and inland distribution thus increasing the total environmental burdens of the LNG supply. The shale gas supply sits between the two options. It determines higher burdens than the UK grid mix supply, because it is extracted from onshore wells with particularly low recovery; however, it determines a lower impact than LNG because no additional processing (liquefaction and vaporization) and ship transport are required.

The results for the ADP are similar and of the same order of magnitude for all three systems analysed (see ADP in Table 4.4.1). ADP fossil calculates the depletion of fossil resources: the extraction of natural gas and hence the depletion of 1 MJ of fossil resources (1 MJ, as the results are reported for this functional unit) determines the main contribution to this indicator. The ADP value for the LNG is slightly higher than the other two values (11% and 15% higher than the ADP of natural gas grid mix and shale gas, respectively); the energy requirements of the deep processing required by the LNG supply chain increase the abiotic depletion of the LNG model.

The environmental burdens of the three systems analysed are also compared to the average European impact for each indicator: Figure 4.4.1 reports the normalised results. The processes analysed influence mostly the averaged ADP, AP and POCP. The assumption of sour gas extraction for the LNG system specifically influences the results of the AP and POCP. Sour gas is assumed to be extracted only for the LNG system because the natural gas extracted in Qatar usually has a higher concentration of sulphur than that from other gas fields, hence the desulphurization process is specifically required [315]. However, a sensitivity analysis has been performed: in Figure 4.4.2 the normalised results are reported assuming that the raw gas is sweet for all three systems analysed. In this case, the AP and POCP are the only indicators according to which the shale gas is shown to determine a higher environmental impact than LNG. Conversely, the results of the other indicators do not significantly shift. In both cases, the depletion of fossil resources determines the main contribution to the total averaged impacts.

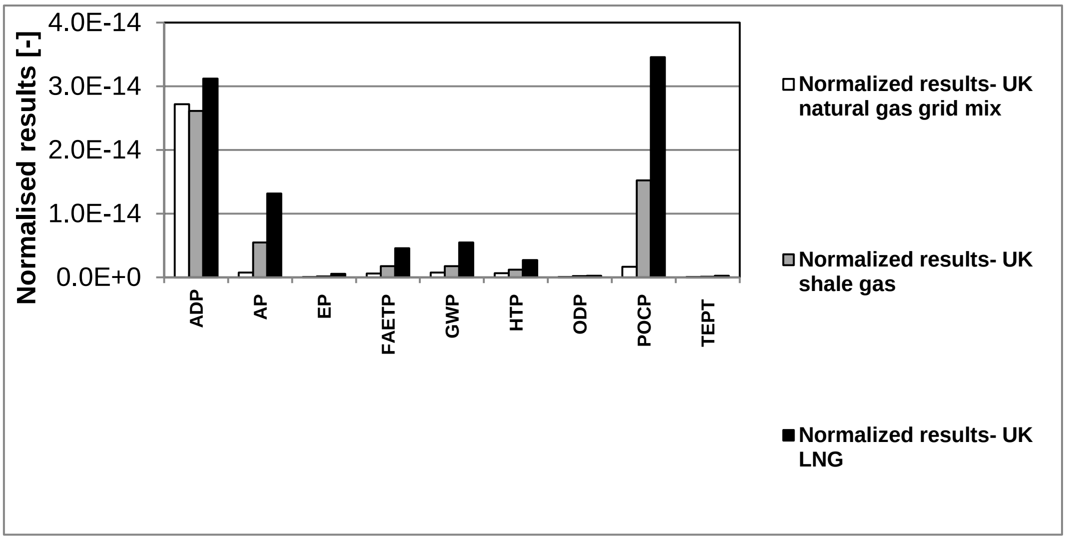


Figure 4.4.1. Normalized results for natural gas extraction, processing and supply according to different sources and technologies. Results are reported for 1 MJ of delivered gas and the processing of sour gas for LNG and sweet gas for the other options.

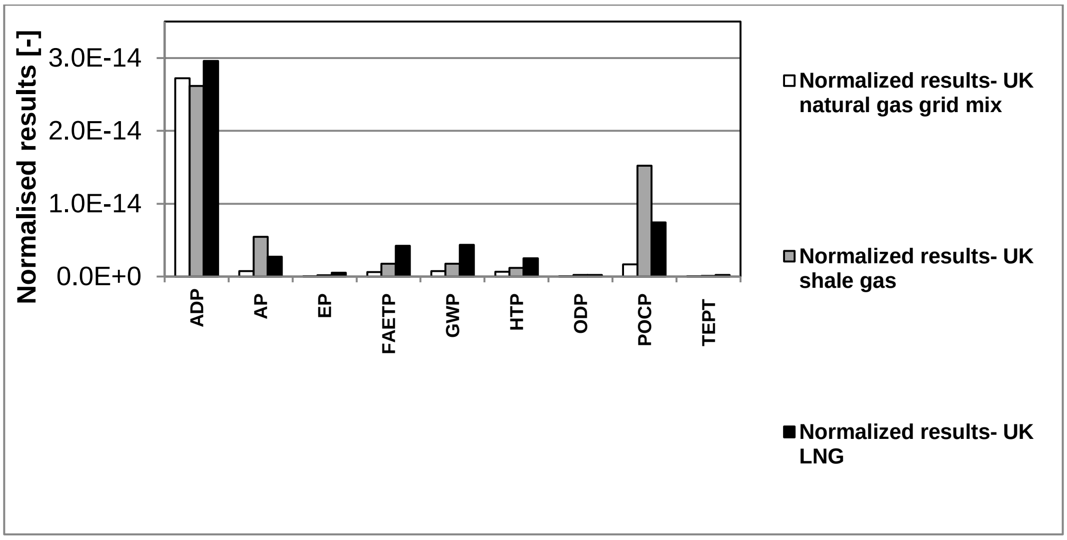


Figure 4.4.2. Normalized results for natural gas extraction, processing and supply according to different sources and technologies. Results are reported for 1 MJ of delivered gas and the processing of sweet gas for all three options.

The analysis has been further developed to include the use phase in the results to discuss the effect of the upstream emissions on the total supply chain of natural gas delivery. Table 4.4.2 reports the results for the use of 1 MJ of gas in a household gas boiler for the three systems analysed, including the entire life cycle from extraction to the use. As the use phase is the same for all three options, the environmental burden of each scenario is increased by the same amount and hence, the same ranking as Table 4.4.1 is shown also for Table 4.4.2. For each indicator, the contribution of the use phase to the total environmental burdens (that include production, processing, distribution and use) is also specified. For the GWP, HTP and TETP, the use phase determines more than 60% of the total environmental burden. The same is not valid for the remaining indicators; the contribution of the use phase is less significant for the ADP, AP, ODP and POCP. For all the indicators, except for the ADP, the contribution to the total environmental burden of the use phase is always lower for the LNG than that of the shale gas. This is due to the higher environmental burden associated with the production and distribution of 1 MJ of LNG. The claim that the use phase determines the highest environmental impact and hence that the production of natural gas can be neglected is true only for some indicators but not for all scenarios. For example, for the GWP of the UK grid mix and shale gas, respectively, the use phase is 93% and 86% of the total burden, but the same is not true for the LNG (the use phase in this case determines only 66% of the total GWP). Conversely, for all three options analysed, the total HTP is substantially due to the use phase.

		UK natural gas grid mix	**UK shale gas**	**UK LNG**
Abiotic Depletion	[MJ]	0.978	0.940	1.09
	% [use phase/total]	2.30	2.39	2.07
Acidification Potential	[10^{-6} kg SO2-Equiv.]	25.2	105	235
	% [use phase/total]	50.10	12.07	5.38
Eutrophication Potential	[10^{-6} kg Phosphate-Equiv.]	4.17	6.06	13.7
	% [use phase/total]	72.08	49.60	22.03
Freshwater Aquatic Ecotoxicity Pot.	[10^{-3} kg DCB-Equiv.]	0.523	0.758	1.35
	% [use phase/total]	75.23	51.85	29.20
Global Warming Potential	[10^{-3} kg CO2-Equiv.]	62.0	67.2	86.7
	% [use phase/total]	93.56	86.27	66.90
Human Toxicity Potential	[10^{-3} kg DCB-Equiv.]	8.35	8.61	9.37
	% [use phase/total]	95.92	93.08	85.54
Ozone Layer Depletion Potential	[10^{-9} kg R11-Equiv.]	0.0276	2.33	2.53
	% [use phase/total]	63.22	0.75	0.69
Photochem. Ozone Creation Potential	[10^{-6} kg Ethene-Equiv.]	4.92	28.4	61.9
	% [use phase/total]	41.04	7.12	3.26
Terrestric Ecotoxicity Potential	[10^{-6} kg DCB-Equiv.]	35.7	44.6	62.5
	% [use phase/total]	88.14	70.57	50.32

Table 4.4.2. Environmental burdens for natural gas use in boiler, delivery, processing and extraction according to different sources and technologies. The contribution of the use phase to the total impact is also reported. Results are reported for 1 MJ of delivered gas and the processing of sour gas for LNG and sweet gas for the other options.

Finally, Figure 4.4.3 reports the normalised results including also the use phase. When the use phase is included (and the same raw gas composition is assumed for the three systems), the GWP, HTP and ADP cause the main contribution to the total averaged European impacts.

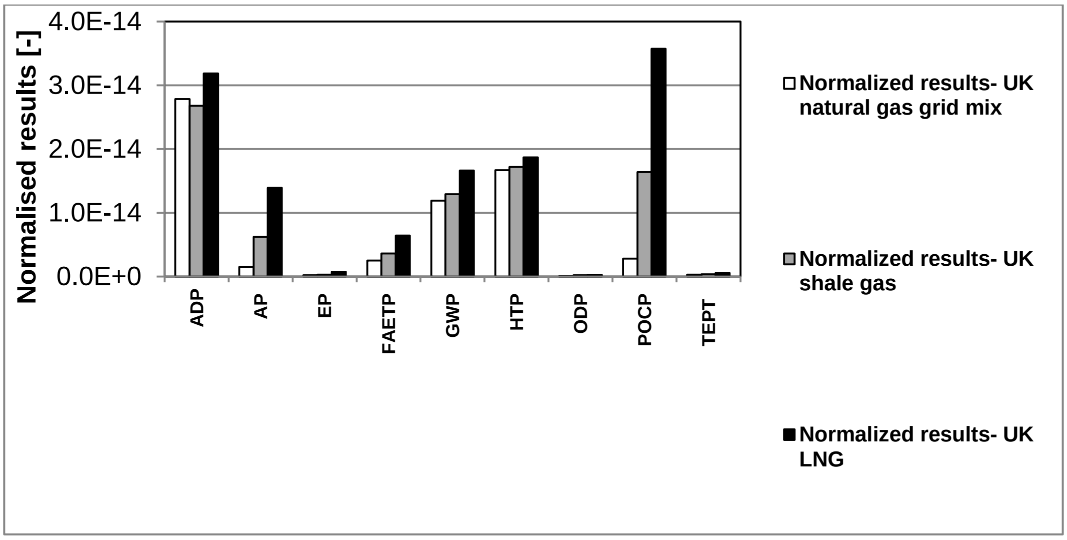


Figure 4.4.3. Normalized results for natural gas extraction, processing, delivery and use in boiler according to different sources and technologies. Results are reported for 1 MJ of delivered gas and the processing of sour gas for LNG and sweet gas for the other options.

4.4.2 Where should the supply of fossil natural gas in the UK be heading?

The environmental burdens of two fossil supplies of natural gas to the UK, shale gas and LNG, have been analysed. The life cycle of the production and distribution of 1 MJ of shale gas and LNG, including extraction, processing, transport and distribution to the final consumer has been modelled. Firstly, the environmental burdens of shale gas and LNG supply have been compared to the present supply of natural gas to the UK and then, the use of natural gas in a household boiler has also been considered.

In order to identify the major environmental contributions, the results have been normalised according to average European impacts. The GWP has been shown not to be the first concern according to the normalised results when the use phase is not included in the results. When looking at different strategies for energy supply, developing legislations shall keep the greenhouse gas emissions in high focus but shall also broaden the environmental spectrum to other environmental burdens in order to consider those determining higher impacts to the global averages. The AP and POCP, significantly depend on the quality of the extracted gas. Sour gas and associated emissions of sulphur contaminants during the sweetening process cause increased pressure on the averaged European burdens. Improved technologies for sulphur removal and emissions control are suggested to decrease the environmental burden of acid substances.

The total environment impacts from cradle to grave have also been analysed; the upstream processes of extraction, processing, transport and distribution have been added to the impacts associated with the use of 1 MJ of natural gas in a household boiler. The use phase does not contribute uniformly to all indicators. For example, it determines the main impact on the total GWP of the shale gas and UK natural gas mix (up to 85%) whereas for the LNG, the upstream processes cause a higher contribution to the total GWP (they cause 40% of the total impact). A different trend is shown for all other indicators.

Only when the results include the use phase, does the GWP show an increased impact on the total averaged European results and it thus becomes one of the main indicators on which to focus. However, also in this case other indicators should be addressed when looking at the environmental impacts of different natural gas supply sources and technologies.

4.5 General conclusions

The potential of fossil natural gas supply to the UK was explored in this chapter. The environmental impacts of the entire supply chain of shale gas and LNG were analysed. The environmental impacts of natural gas supply to the UK are dictated by well's total recovery, processing of the raw gas, and transport; they are significantly reduced when high recovery wells are exploited, light processing is required and transport is reduced. Therefore, the supply of natural gas from close wells with high gas recovery is always to be preferred to the import of natural gas requiring deep processing.

Chapter 5. **Electricity from waste: advanced and conventional technologies**

This chapter begins with the analysis of the dual stage (gasification and plasma) advanced thermal technology developed by APP for the treatment of waste and production of electricity. In the second section of the chapter, this technology is compared to alternative advanced dual stage technologies and to conventional technologies for electricity production from waste.

For each section the modelling principles, the assumption and the inventory on which the LCA models are based are reported. Then, the environmental burdens are calculated in order to identify the most polluting sections of the processes analysed and the process that determines the lowest environmental impacts.

The technology of gasification and plasma for waste conversion has been developed by APP who is sponsoring this project. The company has taken part to a national competition supported by the Energy Institute (ETI) to identify the most suitable gasification technology for alternative energy production. The results reported in this chapter have been included in a wider analysis performed by the company on advanced gasification technologies to compete for the ETI project.

Part of the technical knowledge required to develop this LCA analysis on advanced waste treatment options has been developed whilst the Author was working part-time as process engineer at Advanced Plasma Power (APP). Mass and energy balance required for the development of this chapter have been generated using Aspen simulations complemented by experimental data.

The content of this chapter was partially published in:

Evangelisti, S., Tagliaferri, C., Clift, R., Lettieri, P., Taylor, R., Chapman, C., 2015. Integrated gasification and plasma cleaning for waste treatment: A life cycle perspective. Waste Manag. 43, 485–96. doi:10.1016/j.wasman.2015.05.037

Evangelisti, S., Tagliaferri, C., Clift, R., Lettieri, P., Taylor, R., Chapman, C., 2015. Life cycle assessment of conventional and two-stage advanced energy-from-waste technologies for municipal solid waste treatment. J. Clean. Prod. 100, 212–223. doi:10.1016/j.jclepro.2015.03.062

The comparison of waste to electricity technologies reported in Evangelisti *et al.* [374] has been expanded in this chapter as a high temperature gasification and combustion process has also been considered.

5.1 Introduction

Life cycle assessment (LCA) is a tool that can be used to compare waste management technologies and to evaluate their environmental performances, allowing decision makers to be correctly informed [375]. Since the early 1990s, LCA based models have been applied to the assessment of waste management systems [376]. Therefore, there is extensive information in the scientific literature on the environmental impact of waste management options for electricity or CHP production; all processes from collection to material recovery and electricity generation [176,267,377,378] are usually accounted.

In general, these studies are difficult to generalize because waste characteristics and technology choices are highly dependent on the local situation and some of them even show opposite results [49]. Studies from the Northern European countries [375,379,380] usually assess the environmental competitiveness of advanced technologies and compare them to the local, widespread, conventional technology of incineration that is often used for district heating [381]. Increasing population and developing industry has meant that developing countries are also becoming aware of the environmental threats of unregulated waste dumping. In this case, many LCA works focus on conventional waste treatment methods, such as landfill, and reconsider these methods as feasible options to limit the uncontrolled waste dumping [49,382–384].

It is reported that in the field of waste management an integrated system approach is essential as the sub-systems are interrelated. Therefore, many studies try to approach the problem of waste management as a whole and analyse integrated systems for the entire society including a combination of technologies and waste types for a particular area [306,385–391]. Conversely, some other studies analyse the environmental impact of a particular technology [47,174,392,393], or waste stream [394–396] and compare it with alternative options. In this case, the technology analysed is usually in the developing stage and further assessment is required before commercialization.

Relatively few studies have been published on the LCA of advanced thermal treatments other than incineration. Khoo *et al.* [397] assessed eight different advanced treatment technologies but the analysis was not consistent in terms of feedstock treated by each plant (from MSW to tyres). Zaman *et al.* [398] analysed a pyrolysis gasification process for MSW, revealing better environmental performance compared with conventional incineration. Inventory data for the process were based on a plant in the UK although the study was based in Sweden. Pressley *et al.* [399] published a study on gasification of MSW with a Fischer-Tropsch process for production of liquid transport fuel from the resultant gas but the production and consumption of chemicals used were not included in the system. Al-Salem *et al.* [400] analysed a low-temperature pyrolysis process for waste treatment. However, the study considered plastic residue only. Finally, Arena *et al.* [401,402] presented a comparative attributional LCA of a moving grate combustor and a vertical gas shaft gasifier coupled with direct melting, a technology mostly used in Japan.

The majority of the studies mentioned above are comparative LCA where the advanced thermal treatment is evaluated against more traditional technologies, rather than pure attributional LCA studies which give full understanding of a specific technology [397,398,400]. As noted by Astrup *et al.* [377], very few of the existing LCA studies on waste-to-energy technologies provide sufficient description of the technologies investigated and the key assumptions of the LCA; as a consequence, the applicability of inventory data and LCA results provided by the majority of the existing studies are often limited [377].

The first part of this chapter evaluates the life cycle environmental impacts of the two-stage thermochemical process developed by APP, i.e. a gasification-plasma process (G-Pl), assessing different waste composition and heating values. Several environmental impact categories are analysed and a hot spot analysis is performed. A scenario analysis on some key processes later specified is also presented.

In the second part of this chapter, the environmental performances of four two-stage thermochemical processes for the treatment of MSW are assessed: a gasification-plasma process (G-Pl), fast pyrolysis with combustion (FP-C), high temperature gasification with direct melting and combustion (GM-C) and gasification with secondary high temperature oxidation of the syngas (G-SC). The processes are compared against conventional waste treatment technologies of the same scale, specifically incineration with energy recovery and landfill with electricity recovery from the landfill gas.

Overall the study is intended to be performed ensuring transparency in the methodological choices and robustness of the results and recommendations are provided. The following questions are addressed in this chapter:

- Which is the environmental impact of an advanced dual stage gasification-plasma technology when treating different types of waste for the production of electricity?
- In terms of the lowest environmental impact, which is the best technology for waste treatment and electricity production among different advanced and conventional alternatives?

5.2 Integrated gasification and plasma cleaning for electricity production

5.2.1 Goal and scope definition: functional unit and system boundary

The aim of this study is to perform an attributional life cycle assessment with system expansion of a two stage gasification and plasma process for waste treatment, assessing different feedstock compositions (i.e., ultimate analysis and heating value). The Functional unit used is 1 kg of waste as received at the plant, because the primary function of the studied process is to treat waste. Results based on the secondary function of the plant, i.e., electricity production, are shown in Section 5.4. Figure 5.2.1 shows the boundary of the system considered in this work.

Following conventional practices [403] secondary data for the indirect and avoided burdens are taken as the averages for the background system, while primary data are used for the Foreground operations. The latter are UK site-specific wherever possible and based on experimental and modelling results from the pilot plant developed at APP. Otherwise, average data from the literature and specific datasets are used.

Neither transportation of waste from the generation point to the plant nor the generation of the waste are considered in the system. In this study carbon dioxide emissions from biogenic carbon are included in the estimates for the Global Warming Potential (GWP) because the assessment is based on existing waste streams with defined carbon content so that the production of the materials in the waste does not enter the analysis (the uptake of CO_2 from the atmosphere by the once living bio-degradable part of waste is outside the system boundary). Therefore, the total carbon content of the waste is considered, with no distinction between biogenic and non-biogenic carbon in the baseline. A further analysis is presented in Section 5.2.3.4 where the results of the global warming potential excluding biogenic carbon are shown.

The valuable outputs considered in the system expansion are: the electricity generated by the process; the vitrified product referred to as Plasmarok generated by the plasma converter and the metals recovered from the waste pre-treatment process. The electricity produced by the thermal treatment of the waste is assumed to substitute the electricity from the UK grid [310]. This is reflected in the environmental impact associated with the production of 1 MJ of electricity from the grid, and in particular in the carbon footprint which is equal to 0.155 kg of CO_2 eq (including biogenic carbon) for the UK [310].

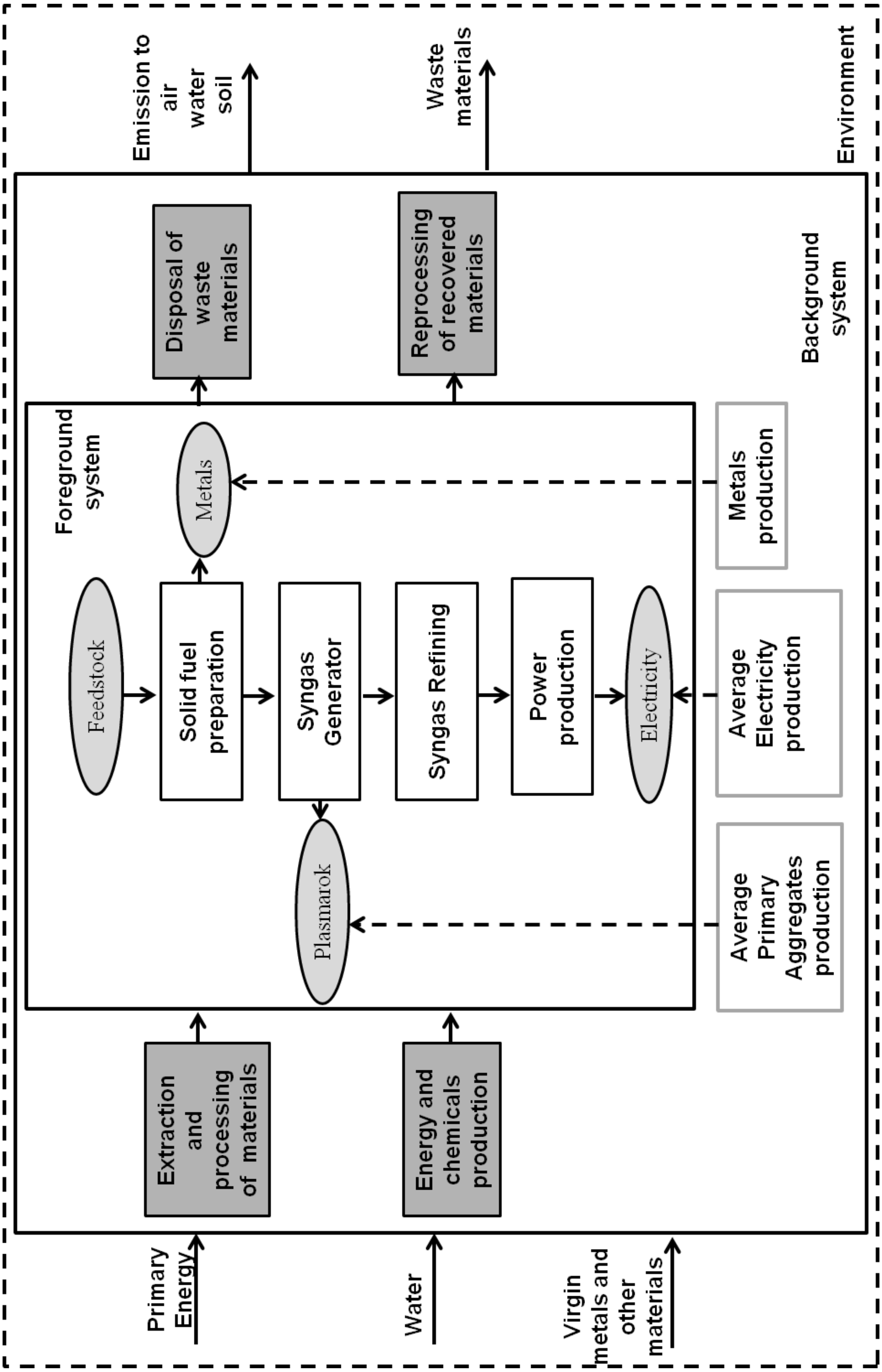


Figure 5.2.1. System boundary.

5.2.2 Life cycle inventory

The mass and energy balances of the two-stage thermo-chemical process analysed is based on a set of input developed at the company Advanced Plasma Power.

The two-stage gasification and plasma process (G-Pl) can treat different waste feedstocks to produce electricity, steam and a vitrified product. The core of the system is the two stage process treating the waste by gasification followed by plasma cleaning to produce a low impurity, high energy syngas with high carbon conversion efficiencies [229]. Four main sections are identified: solid fuel preparation unit; syngas generator unit; syngas refining unit; and power production unit (see chapter 2 for more details). Figure 5.2.2 shows the main flows between the units as well as the allocation of the avoided burdens. The received waste is usually pre-treated. Ferrous and non-ferrous metals are recovered in the solid fuel preparation section. However, if the plant is treating biomass or refuse-derived fuel only drying is required without metals' recovery before entering the gasification unit.

Unlike common incineration plants that produce bottom ash, which must be stabilised before use or disposal, a two-stage gasification and plasma process produces a vitrified and stabilized product (Plasmarok).

Steam is then recovered in the syngas refining section during the cooling of the syngas after the plasma treatment and it is assumed to be re-used within the process; no export of steam is accounted for.

Electricity is generated using a gas engine and a steam turbine (additional 0.5 MWe of electricity production) and is exported to the grid. In this study, the electrical efficiency of the gas engine is taken from the manufacturer's specification as 39-41% [229,404] and the oxidising agent is air. The steam produced by flue gas cooling is fed to the steam turbine of the power production unit.

A more detailed description of the dual stage process is reported in chapter 2.

All energy and chemical consumptions have been taken into account to calculate the indirect environmental burdens. Based on literature [405], stabilisation of the APC residue produced in the syngas refining section is also considered according to a physico-chemical treatment with acidic wastes. The energy required for this process is 0.6 litres of diesel and 13 kWhe (46.8 MJ) per tonne of APC residue [406]. In this model it is assumed that effluents from the quench and scrubber units are treated in standard chemical waste water treatment plants. The European Waste Incineration Directive - which includes gasification plant such as the two-stage G-Pl - defines the acceptable amounts of polluting species in aqueous effluents that can be discharged without further treatment to the public sewer system [152,407]. However, these limits can be further constrained by local discharge limits embedded in Discharge Consents.

The energy required for the start-up of the process has been considered negligible as it contributes less than 0.1% to the total energy requirement. An inventory table of the process analysed is available in the Appendix.

The impact of building the plant has not been considered in this assessment, as standard practice in LCA studies of waste-to-energy technologies. The environmental impacts of building the plants are usually considered negligible compared to the total environmental impacts of the processes [208,378,408].

The economy of scale on the environmental impacts of waste-to-energy plants is usually visible only through an increase in the energy efficiency of the plants. The environmental impacts of larger plants with constant efficiencies are not affected by the bigger scale. This is due to the limitation of the linear scalability in the LCA methodology. Therefore, size effects of the analysed plants have not been considered.

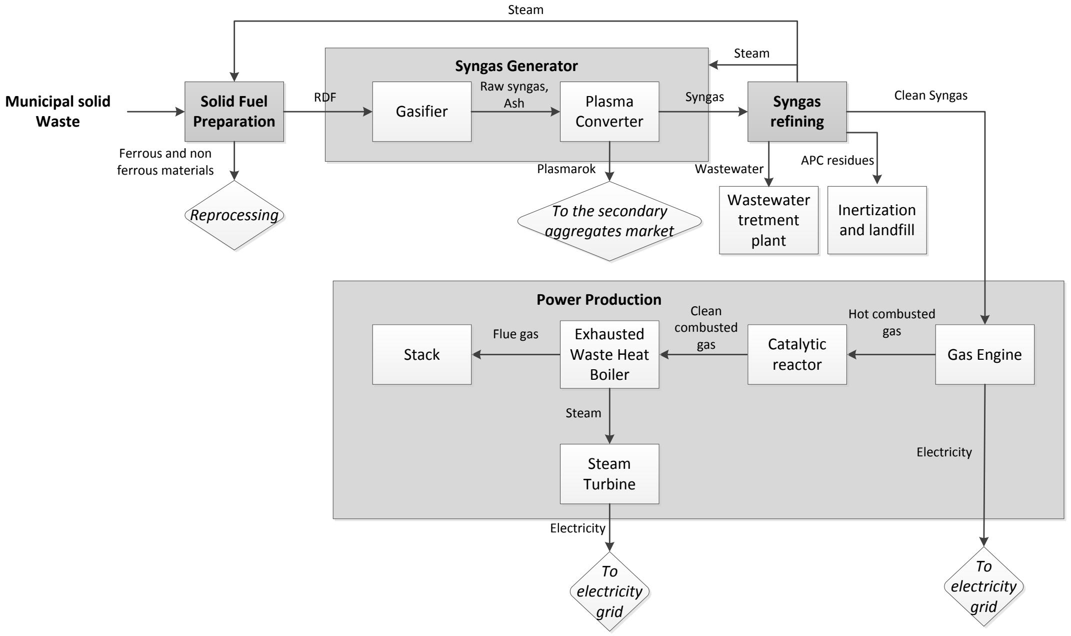


Figure 5.2.2. High level diagram of the two-stage gasification and plasma process (G-Pl). The four main sections of the process are highlighted in bold. System expansion is represented by a rhombus.

5.2.2.1 Feedstock composition

Seven different waste streams treated in the G-Pl process are analysed in this study:

- MSW1, which reflects the average MSW in the UK;
- MSW2, with no organic fines but with a higher amount of plastics;
- MSW3, with higher amount of inert material and wood waste. MSW2 and MSW3 represents two alternative municipal waste streams in the UK [409];

- Solid refuse fuel (SRF), which is a standardized solid fuel prepared from non-hazardous waste meeting specific requirements in terms of thermal value, chlorine content and mercury content [189];
- Refuse derived fuel (RDF), similar to the SRF but with no strict composition range;
- Biomass (wood chips);
- Commercial and industrial waste (C&I).

Table 5.2.1 reports the composition of the feedstocks assumed in this study and their ultimate analysis, which derives from samples of feedstocks collected in different part of the UK [409]. Individual LCA models were built for each feedstock. All require pre-treatment (i.e. separation of metals, shredding and drying), except for RDF (which requires separation of metals and drying, but not shredding), biomass (which require drying only) and SRF, which requires drying only as it is assumed to arrive at the plant already pre-treated. Table 5.2.2 shows the quantities of valuable products for the different waste streams considered the amount of oxygen required in the gasifier and plasma and the total waste throughputs.

Waste Fractions (% as received)	*MSW1*	*MSW2*	*MSW3*	*C&I*	*Biomass*	*RDF/SRF*
Paper and card board	22.7	10	22.4	34.5		58
Wood	3.7	2	7	11.5	100	5
Metals	4.3	7	7	4.1		
Glass	6.6	6	3.5	1.8		
Textile	2.8	7	0	0.5		15
WEEE	2.2	5	3.5	4.9		20
Plastics	10	35	14	24.7		
Inert/Aggregates/Solid	5.3	22	28	13.9		
Organic Fines	35.3	0	3.5	1.9		
Miscellaneous	7.1	6	11.1	2.2		2
Ultimate analysis						
Ash content (%)	12.5	9.2	11.3	11.4	7.5	0.4
% C (Dry Ash Free basis (DAF)) fossil	17.9	34.7	26.4	26.6	0	18.9
% C (DAF) biogenic	31.8	14.8	39.6	24.5	53.7	32.3
% H (DAF)	6.1	6.1	4	3.5	7	6.2
% O (DAF)	42	43.2	29.4	44.3	37	42.2
% N (DAF)	1.2	0.8	0.4	0.7	0.9	0.2
% S (DAF)	0.3	0.2	0.1	0.2	0.6	0.1
% Cl (DAF)	0.7	0.2	0.1	0.2	0.8	0.1

Table 5.2.1. Feedstocks composition, as received basis and ultimate analysis (provided by the company APP).

	unit	MSW1	MSW2	MSW3	C&I	Biomass	RDF	SRF
LHV	**MJ/kg**	9	13	14.7	11	16.9	16	16
Total Net electricity produced	**kWh/kg waste as input**	0.84	0.88	1.43	1.02	1.49	0.89	1.52
Total Plasmarok produced	**kg plasmarok/ kg waste as input**	0.079	0.030	0.050	0.072	0.003	0.089	0.065
Total ferrous material recovered	**kg/kg waste as input**	0.029	0.047	0.047	0.028	0	0.012	0
Total non-ferrous material recovered	**kg/kg waste as input**	0.01	0.016	0.016	0.008	0	0.038	0
Total oxygen demand at gasifier	**kg/kg waste as input**	0.201	0.226	0.312	0.268	0.436	0.238	0.342
Total oxygen demand at plasma	**kg/kg waste as input**	0.035	0.041	0.051	0.04	0.069	0.032	0.062
Total waste input	**ktpa**[4]	259	244	140	216	122	247	135

Table 5.2.2. Key parameters for the seven waste streams analysed per kg of waste received at the plant.

5.2.2.2 Metal and Plasmarok recoveries and oxygen production process

The assumptions made in the baseline model concerning recycling of recovered metals, oxygen production technology and the use of Plasmarok are summarised in Table 5.2.3. There is no general consensus in the waste management sector on which specific process should be offset by the recovery of metals from waste and therefore how the avoided burdens should be evaluated. This depends on several factors: the quality of the collected waste, the collection method, the separation method used at the waste treatment facility and the reprocessing route used to treat the recovered metal. For this reason a scenario analysis based on the substituted process for ferrous/non-ferrous production and the substitution ratio was performed. The recycling rate is defined as

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[4] kilo tonnes per year for 20MWe net electrical output

'the ratio between the amount of metal scrap ready for use in conventional metal production processes and the metals recovered from the waste'; i.e., the efficiency of reprocessing of metals recovered from the waste into metal scrap ready for use in conventional metal production. The approach used to estimate the avoided burdens is that described by Thinkstep as the 'value of scrap approach' [310]: per unit quantity of material recycled, the avoided burdens are those arising from primary production minus the burdens from the reprocessing process. Moreover, a substitution ratio which represents the amount of recycled materials supposed to replace a defined quantity of virgin materials with the same quality is also used [410]. According to the ILCD methodology, this value is based on the average market mix of primary and secondary material for both ferrous and non-ferrous material [310]. If the metal recovery included only virgin materials substitution the comparison among the processes analysed would have been strengthened.

The assumptions adopted in the baseline scenario for the substituted processes for metal production and substitution ratios are:

- ferrous material is assumed to be substituted at a 1 to 1 rate, thus no changes occur in the inherent proprieties of the recycled material [411]. Recovered ferrous material is assumed to be recycled by electric furnace processing, as reported in the GaBi database [310] and taken from the Worldsteel LCA Methodology report [412].
- non-ferrous material is assumed to be substituted at a 1 to 1 rate [411]. The recovered aluminium is assumed to be recycled by clean scrap melting and casting, as reported in the GaBi database [310] and taken from the Environmental profile report for the Aluminium Industry [413].

Plasmarok production is assumed to avoid the production of crushed rock for the primary aggregate industry on a mass basis, as suggested by industrial reports [414] and based on the process burdens reported in literature [415].

Finally, the oxygen supplied to the process is assumed to be produced through cryogenic separation of air; this is likely to be the process used for a 20MWe plant although a pressure swing adsorption (PSA) technology would be used at a smaller scale, such as a demonstration plant. An average UK cryogenic process is considered, based on GaBi database.

Scenarios	*Process for Ferrous materials*	*Process for non-ferrous materials*	*Process for Oxygen production*	*Process for primary aggregates production*
Baseline Two-stage gasification and plasma process	Steel plate production process, substitution ratio:1:1	Aluminium clean scrap melting and casting process, substitution ratio 1:0.99	UK based – cryogenic process	Primary aggregates from crushed rock
Scenario 1 ferrous 1:0.51 non-ferrous 1:0.6	Number 1 Steel (2 foot) steel scrap process, substitution ratio 1:0.51	Aluminium foil production process, substitution ratio 1:0.6	UK based – cryogenic process	Primary aggregates from crushed rock
Scenario 2 ferrous 1:0.51 non-ferrous 1:1.01	Number 1 Steel (2 foot) steel scrap process, substitution ratio 1:0.51	Paper-backed aluminium foil production process, substitution ratio 1:0.1	UK based – cryogenic process	Primary aggregates from crushed rock
Scenario 3 UK based PSA process	Steel plate production process, substitution ratio 1:1	Aluminium clean scrap melting and casting process, substitution ratio 1:0.99	UK based-PSA process	Primary aggregates from crushed rock
Scenario 4 primary aggregates from marine sands	Steel plate production process, substitution ratio 1:1	Aluminium clean scrap melting and casting process, substitution ratio 1:0.99	UK based – cryogenic process	Primary aggregates from marine sand

Table 5.2.3. Description of the scenarios analysed as part of the scenario analysis (main changes in each scenario are highlighted in *italics*).

5.2.2.3 Scenarios analysis

A scenario analysis on some key operations was undertaken for the baseline waste stream (MSW1). Four different scenarios were identified (see Table 5.2.3):

- Scenario 1. This scenario explores the effect of employing different processes to recycle the recovered metals and assumes different substitution ratio. In this scenario, substitution ratio of 0.51 and 0.6 were chosen for ferrous and non-ferrous material respectively, based on the economic value of end-of-life scrap in the scrap commodity market in 2014, i.e. the exchange of end-of-life scrap for value, according to the Thinkstep database values [416]. For ferrous material, the process substituted is the same as used in the baseline scenario, whereas for non-ferrous material, production of aluminium foil is assumed to be substituted [416]. All other parameters are the same as the baseline scenario. This scenario is later referred to as ferrous 1:0.51-non-ferrous 1:0.6.

- Scenario 2. In this scenario all parameters are the same as the baseline scenario except for metal recycling; the rates chosen were 0.51 for ferrous and 0.1 for non-ferrous metal. The processes avoided in this case are the same as the baseline scenario for ferrous material, with paper-backed aluminium foil production for non-ferrous metal [416]. This scenario is later referred to as ferrous 1:0.51- non-ferrous 1:1.01.
- Scenario 3. Figures for Pressure Swing Adsorber (PSA) rather than cryogenic air separation for oxygen production were adopted in this scenario. The data for the electricity consumption were based on APP. All other parameters were assumed to be the same as the baseline scenario. This scenario is later referred to as UK based – PSA process.
- Scenario 4. The production of Plasmarok was assumed to replace extraction of marine sand and gravel for primary aggregates production [415]. All other parameters were assumed to be the same as in the baseline scenario. This scenario is later referred to as primary aggregates from marine sands.

The efficiency of the gasification reactor and plasma converter has not been taken into account in the scenario analysis, because this parameter is not considered to vary during the operation of the plant. In fact, as demonstrated in literature [177], the presence of the plasma and its variable power during plant operations allows the process to self-compensate any parameter variations (in terms of temperature or oxygen/steam supply) which can occur in the gasification reactor.

5.2.3 Results and discussion

Figure 5.2.3 shows a comparison of the environmental impacts associated with the two-stage gasification and plasma process treating the seven different feedstocks specified in Table 5.2.1.

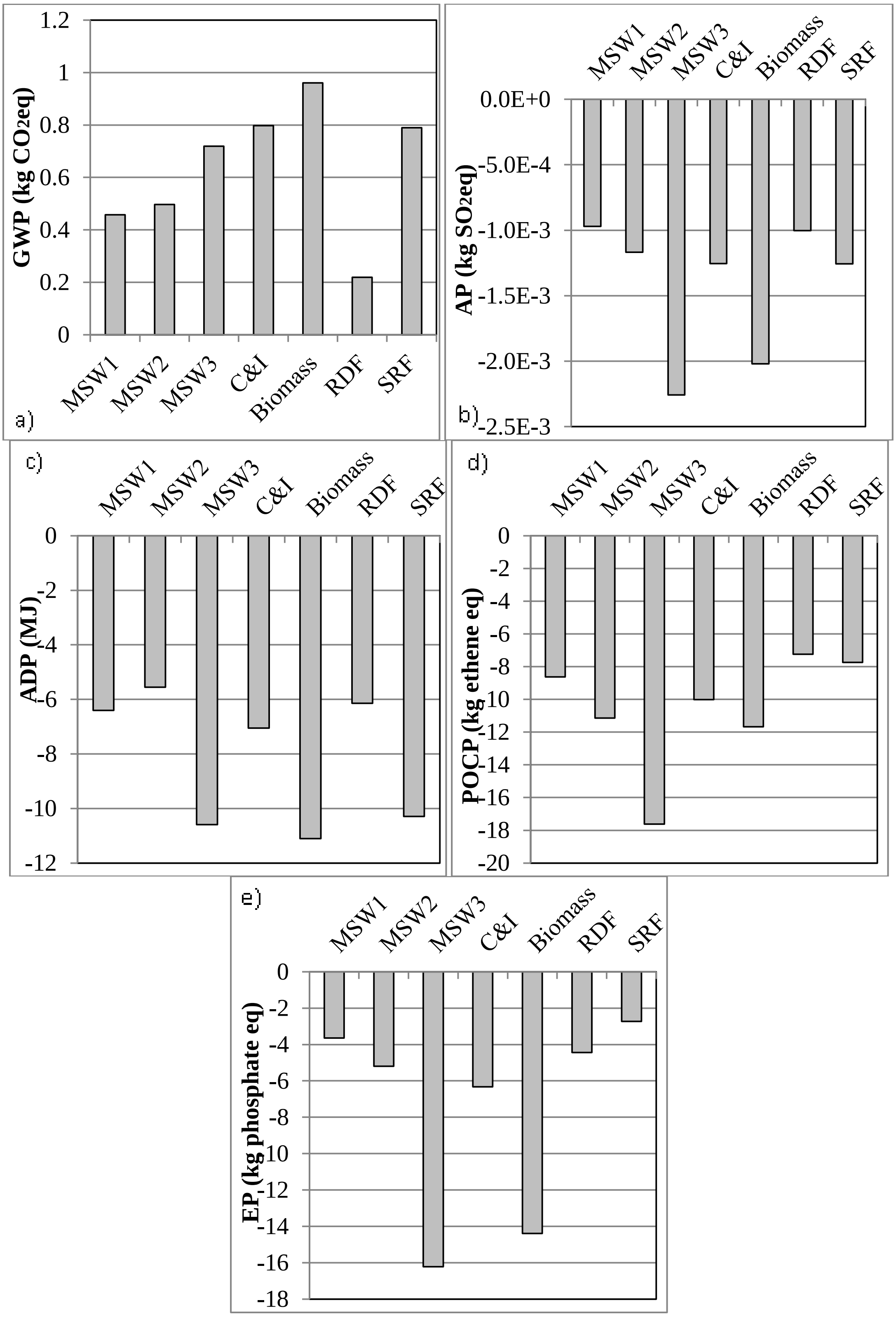

Figure 5.2.3. Environmental impacts of the seven feedstocks treated in the G-Pl process: a) GWP; b) AP; c) ADP; d) POCP and e) EP. The functional unit is 1 kg of waste.

The assumptions concerning recycling of recovered metals, oxygen production technology and the use of Plasmarok are based on the baseline scenario (Table 5.2.3).

Only significant results are shown here, although the analysis was performed for more indicators as shown in Table 5.2.4 where normalised results are presented according to the factors reported in the Appendix.

Normalised results (10^{-15})	*MSW1*	*MSW2*	*MSW3*	*C&I*	*Biomass*	*RDF*	*SFR*
Abiotic Depletion	-182	-158	-302	-201	-316	-175	-293
Acidification Potential	-57.7	-69.4	-134	-74.5	-120	-59.6	-74.7
Eutrophication Potential	-1.97	-2.81	-8.77	-3.42	-7.77	-2.4	-1.47
Freshwater Aquatic Ecotoxicity Pot.	19.6	11.0	-5.58	15.8	-1.71	12.6	66.4
Global Warming Potential	87.9	95.3	138.	153	184	42.1	152
Human Toxicity Potential	-66.9	-83.0	-168	-0.899	-1700	-743.0	-1160
Ozone Layer Depletion Potential	0.117	0.0685	0.0247	0.107	0.0648	0.0368	0.298
Photochem. Ozone Creation Potential	-49.8	-64.4	-102	-57.9	-67.4	-41.9	-44.8
Terrestric Ecotoxicity Potential	-458.	-490	-816	-563	-871	-479	-850

Table 5.2.4. Normalized results. The normalisation is done based on CML, IPCC, ReCiPe (region equivalents), EU25+3, year 2000 [310]. Functional unit 1 kg of waste.

The GWP impacts (Figure 5.2.3) reflect the carbon content of the waste offset by the avoided burdens. They range between 0.220 kg of CO_2 eq (RDF) and 0.960 kg of CO_2 eq (wood biomass) with the value for the baseline waste composition, MSW1, in the middle of this range. Although both the woody biomass and SRF have the highest calorific values compared with the other feedstocks, their GWP is the highest, while treating RDF in a two-stage gasification and plasma plant gives the lowest impact. It is worth noting that the results shown in Figure 5.2.3 include the biogenic and non-biogenic carbon contributions to the environmental impact, which increases the global warming of the biomass scenario. The GWP impact excluding biogenic carbon is later shown in Figure 5.2.11. Moreover the avoided burdens associated with the Solid Fuel Preparation unit are not significant for biomass and SRF: no metals are recovered from these two feedstocks as it is assumed that the biomass and the SRF arrive at the plant already sorted and are simply dried before being fed to the gasifier. Hence, they receive no credits for metals recovery.

For all feedstocks, the AP indicator shows negative values thanks to the electricity production (see Figure 5.2.3), with values ranging between -0.00226 kg of SO_2 eq (MSW3) and -0.00097 kg of SO_2 eq (MSW1). The result obtained for MSW3 is mainly due to the low sulphur content of this waste, as shown in the ultimate analysis in Table 5.2.1.

The ADP for all the seven feedstocks is also negative (see Figure 5.2.3); MSW3, Biomass and SRF show the best environmental performance thanks to a higher net electricity production (see Table 5.2.2).

As shown in Figure 5.2.3, the MSW3 performs better regarding the POCP and the EP thanks to the higher amount of metals recovered from the front end section and to the higher amount of electricity produced by the plant. As shown in Table 5.2.4 for the toxicity categories, the impacts are negative for the terrestric and human effects. Conversely, the freshwater toxicity potential shows a positive impact for all waste streams, except for MSW3 and biomass thanks to the lower amount of sodium hypochlorite requested by these two feedstocks in the alkali scrubber to clean the syngas.

Overall, the results show that the environmental impact of the feedstock evaluated depends on the category analysed, rather than identifying a single waste stream which is the best for all the impacts. In general, MSW1 shows an average impact amongst the waste streams analysed.

5.2.3.1 MSW1: hot spot analysis

A hot spot analysis has been carried out to show how the four main sections of the two-stage gasification and plasma process (G-Pl) contribute to the environmental impacts. The results for the baseline feedstock, i.e. MSW1, are shown in Figure 5.2.4.

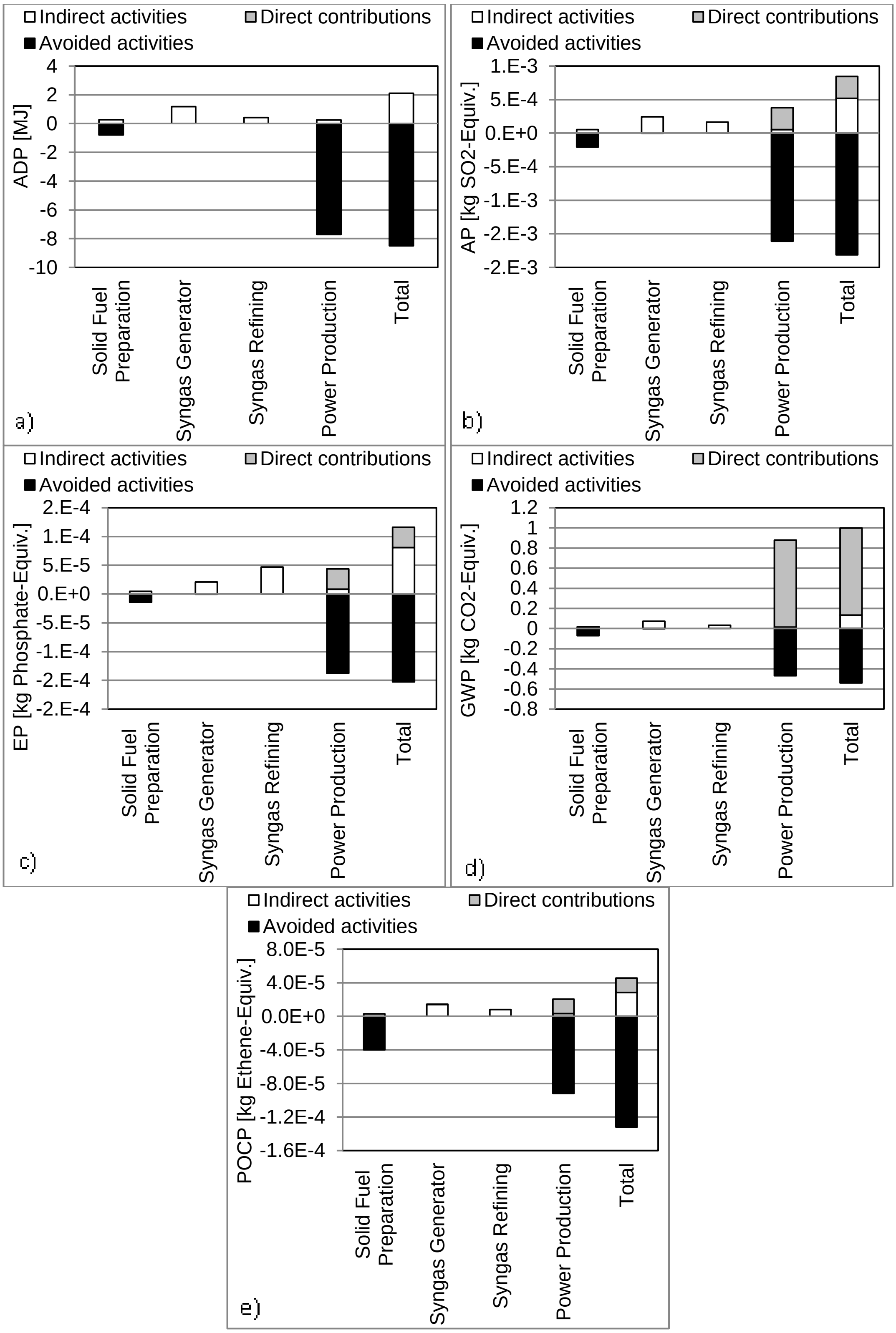

Figure 5.2.4. Hot spot analysis for the four main sections of the two-stage gasification and plasma process treating MSW1. Impacts are per kg of waste received at the plant. a) ADP; b) AP; c) EP; d) GWP; e) POCP.

Negative impacts refer to the avoided burdens, while positive impacts refer to direct and indirect burdens. The Syngas Generator and Refining sections determine a positive contribution to all the environmental indicators. The greatest avoided burdens are associated with the Power Production unit (accounting for 86% of the total avoided burdens for the GWP) and the Solid fuel Preparation unit where the metals are recovered from the MSW (accounting for 13% of the total GWP avoided burdens). Plasmarok production and substitution as secondary aggregates contribute less than 1% to the avoided burdens when crushed rocks are assumed to be substituted. Despite the avoided burdens associated with the electricity generated and exported to the grid, the total contribution of the Power Production unit to the total GWP is still positive – thus negative for the environment, because of the CO_2 emitted to the atmosphere in the flue gas.

Figure 5.2.5 shows the detailed hot spot analysis for the GWP impact indicator when only direct and indirect burdens are considered.

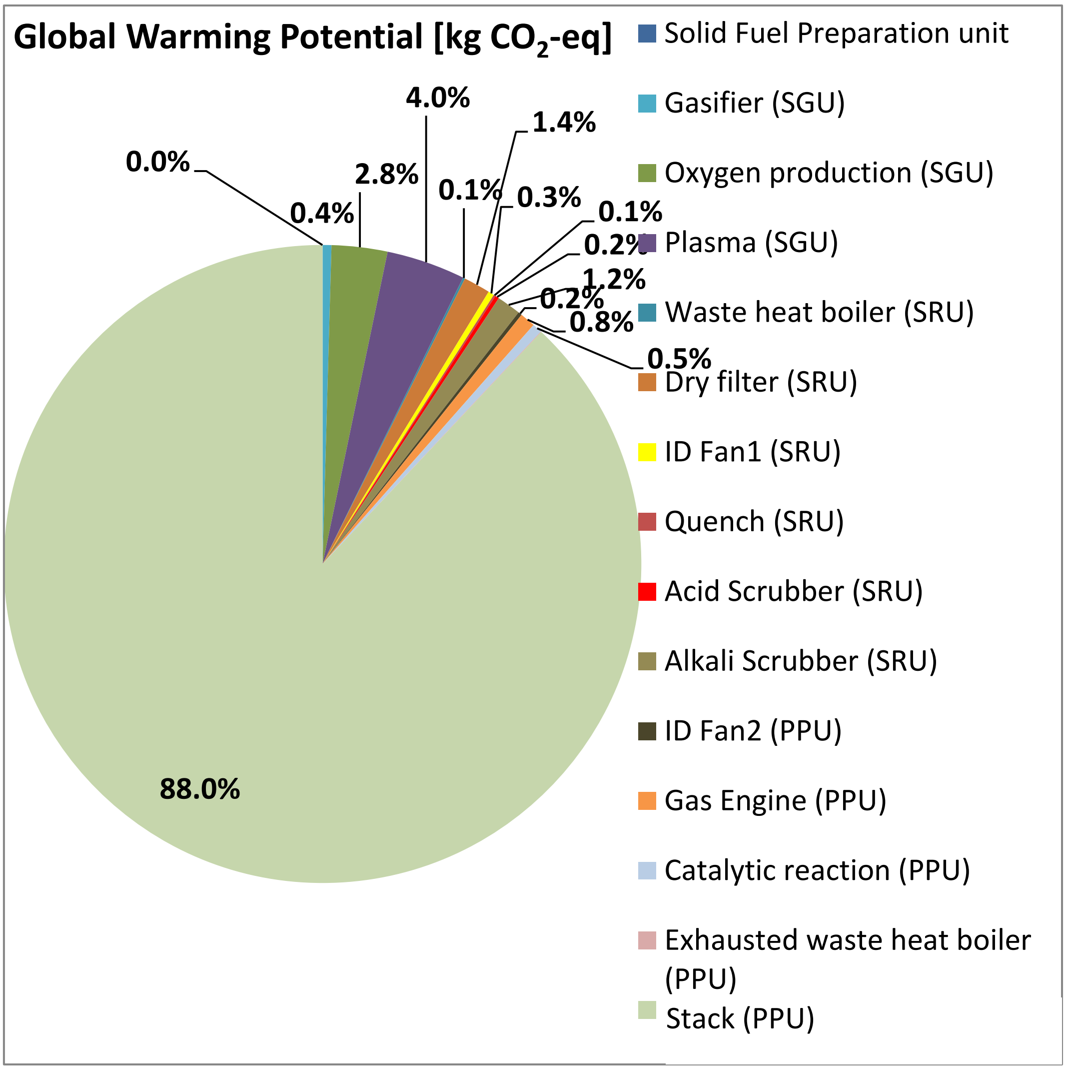

Figure 5.2.5. Detailed hot spot analysis for the GWP of the two-stage gasification and plasma process treating MSW1.

As already stated, the main impact is due to the flue gas released to the atmosphere at the stack (88%), while the oxygen supplied to the gasifier and the plasma accounts for less than 3%.

Although electricity is required by the plasma torch in the second stage of the process, this contributes only for 4% to the GWP, being a fundamental step in the removal of the tars from the syngas. In fact, energy requirement for the plasma torch accounts for 70 kWh/ton of RDF entering the gasifier. It is reported [24] that electricity requirements for plasma gasification of MSW are between 400 and 845 kWh/ton wastes. This was, however, referring to a single-stage plasma gasification process, where the gasification is sustained by applying thermal plasma directly onto the waste material, with all of the energy required for decomposition coming from the plasma [177]. In a two stage gasification and plasma process, such as the one presented here, the plasma arch is applied to fuel gas and carbonaceous particles, both produced in the gasifier and this reduces the energy required by the plasma itself.

Figure 5.2.6 shows a detailed hot spot analysis for the acidification category, as an example of a regional impact category, considering direct and indirect burdens.

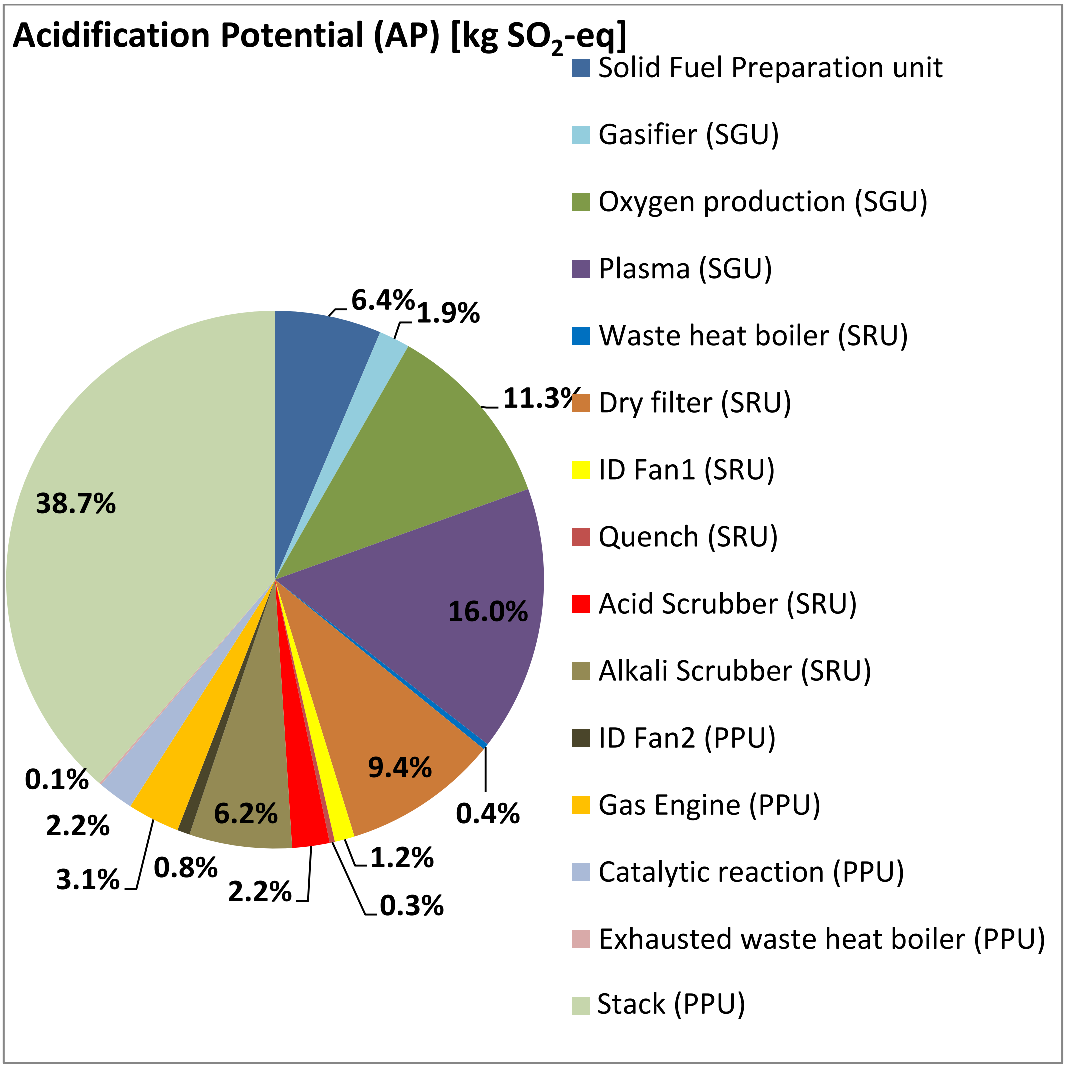

Figure 5.2.6. Detailed hot spot analysis for the AP of the two-stage gasification and plasma process treating MSW1.

The main contribution again arises from the flue gas but with a lower percentage (39%) compared to GWP, followed by the electricity supplied to the plasma (16%), oxygen production (11%) and finally production of sodium bicarbonate for gas cleaning (9%). Figure 5.2.7 shows the detailed hot spot analysis for the fresh water aquatic ecotoxicity potential.

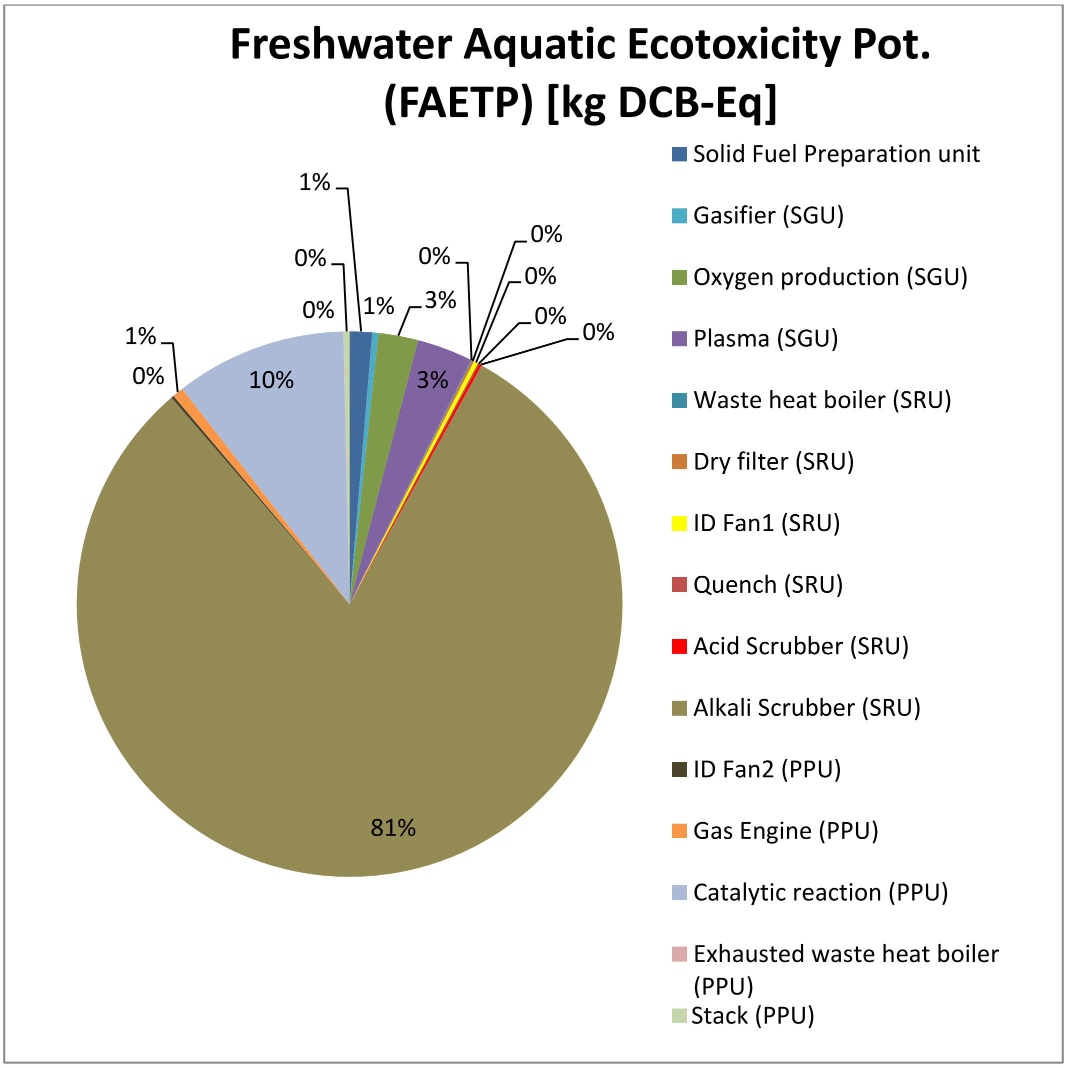

Figure 5.2.7. Detailed hot spot analysis for the FAETP of the two-stage gasification and plasma process treating MSW1.

One of the main contributions to water pollution is the sodium hypochlorite production supplied to the alkali scrubber which represents ~81% of the total FAETP impact. This is due to the chloride emissions to fresh water involved in the production process of this chemical which is needed to obtain a low sulphur content syngas as requested by the gas engine [345].

5.2.3.2 Scenario analysis

The results based on the scenarios presented in Table 5.2.3 are shown in this section.

5.2.3.2.1 Metal recovery process

Figure 5.2.8 shows the results of the scenario analysis for the metals recovery processes. The results refer to the total impacts per kg of MSW1 treated at the plant. The variation compared with the baseline scenario depends on the impact categories analysed. For the climate change indicator (i.e. GWP) only +/- 6% is observed when the type of process and the recycling rate are changed. A more significant variation is obtained for the AP – i.e. a 30% reduction of the impact, when it is assumed that the non-ferrous material is made by aluminium foil. The highest variation is shown for the human toxicity indicator where the ferrous 0.51 – non-ferrous 0.6 scenario shows an increase of 140% of the impact because of the assumption made on the kind of aluminium which is separated from the waste (i.e. aluminium foil).

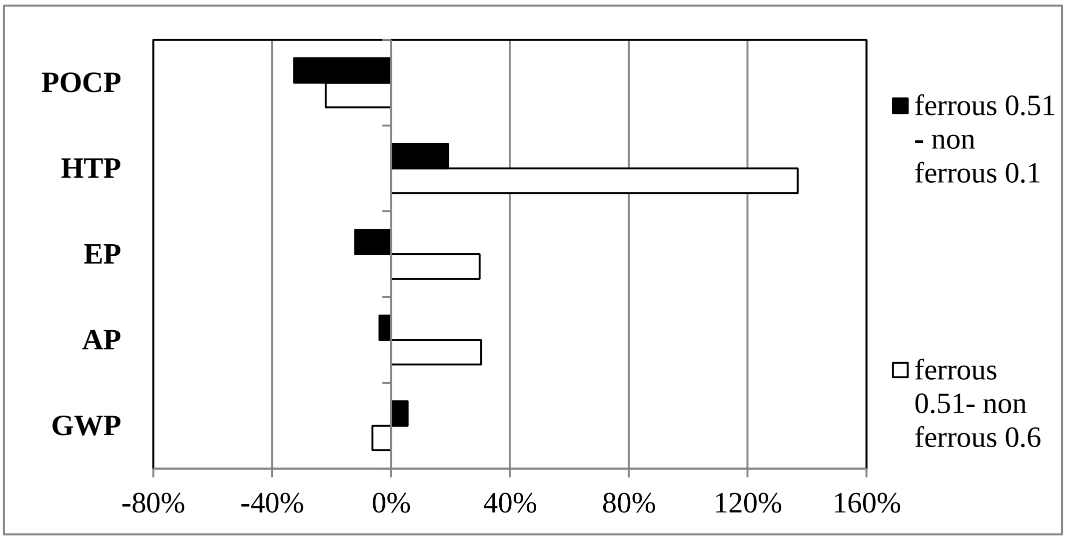

Figure 5.2.8. Scenario analysis for the metals recovery process. Results are shown as a variation compared to the baseline.

5.2.3.2.2 Oxygen production process

In the baseline scenario the oxygen is assumed to be produced by a cryogenic process based in the UK, which is the process likely to be used for an industrial scale plant (20MWe) although pressure swing adsorption (PSA) might be more appropriate at smaller scales. Figure 5.2.9 shows the results of the scenario analysis for the oxygen production process. The variation here is even lower compared with Figure 5.2.8, and only a +2.2% change is observed for the GWP. The highest variation is obtained for the eutrophication category, which shows an 8% increase of the impact. This means that the technology used to produce the oxygen supplied to the gasifier and plasma has a nugatory effect on the total environmental impacts of the process, although the electricity consumption associated with the oxygen production with PSA technology is higher compared to a cryogenic process.

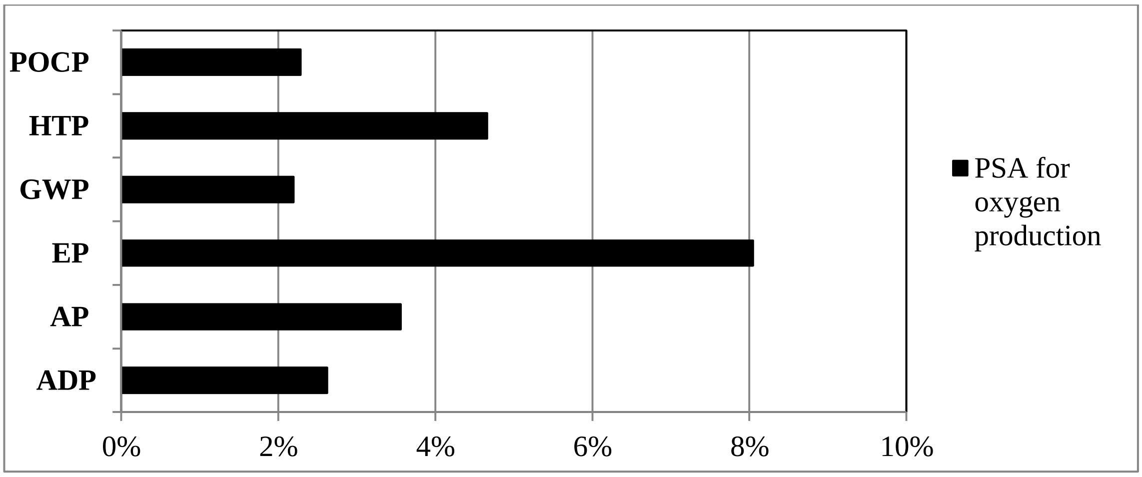

Figure 5.2.9. Scenario analysis for the oxygen production process. Results are shown as a variation compared to the baseline.

5.2.3.2.3 Vitrified slag substitution process

Finally, Figure 5.2.10 shows the results for the process substituted by Plasmarok production. The influence of this variation is negligible for the GWP, mainly because the amount of Plasmarok produced is small so that it does not contribute significantly to the total carbon footprint of the process. However, the scenario analysis shows that assuming a primary aggregates production from marine sand and gravel as substituted process for the plasmarok production decreases the eutrophication impact of 20% compared with the baseline (crushed rock as primary aggregates), and in general is associated with a reduction of the environmental impact of the process.

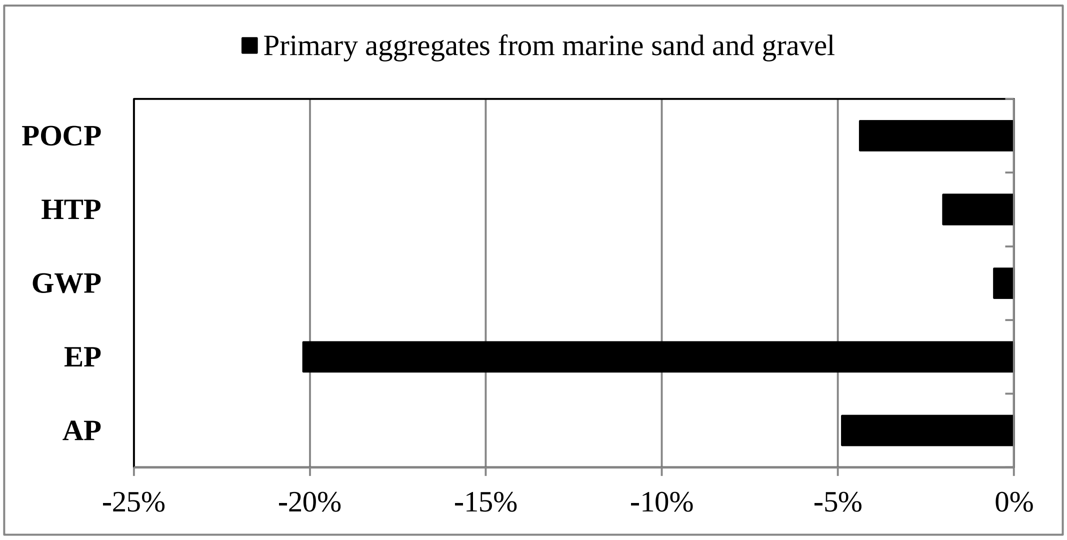

Figure 5.2.10. Scenario analysis for the substitution process allocated to the production of the vitrified slag from the plasma converter. Results are shown as a variation compared to the baseline.

5.2.3.3 Accounting for biogenic carbon

The GWP presented in Figure 5.2.5 includes the biogenic carbon content of the waste. However, an evaluation of the expected biogenic content for each feedstock is presented in Table 5.2.1, based on the specific composition of each feedstock and on data from literature [417]. Figure 5.2.11 shows the reduction obtained for the GWP when the biogenic carbon is excluded for all the feedstocks analysed. As observed, the impact becomes negative for the fuels with high biogenic carbon content: MSW1, biomass, RDF and SRF (i.e. showing a reduction of more than 100%). As expected, biomass and RDF represent the lowest environmental impact cases due to the composition of the feedstock and their calorific values (see Table 5.2.1). In general, if biogenic carbon is excluded, RDF is again the most environmentally friendly waste stream for the climate change category.

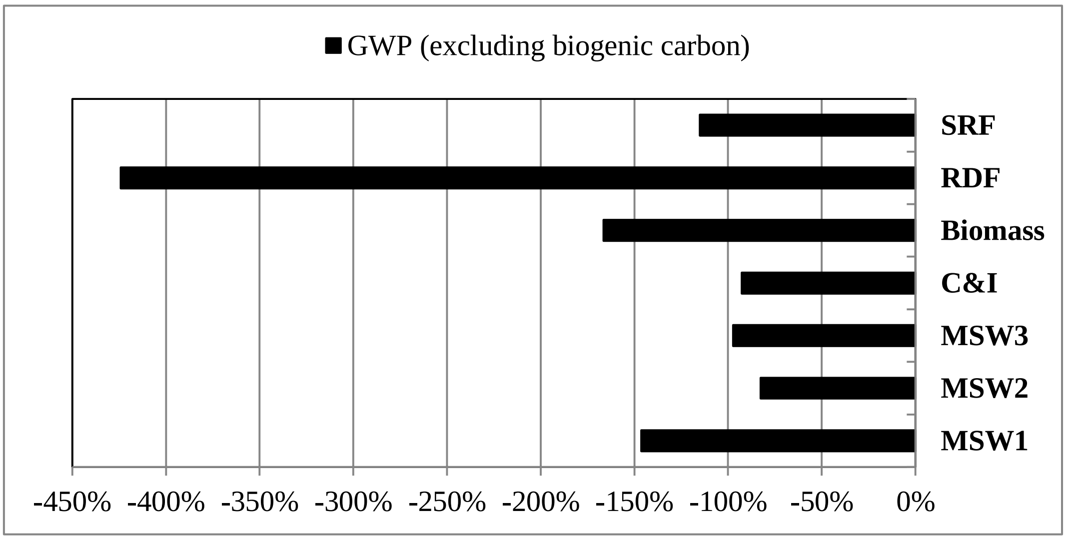

Figure 5.2.11. GWP excluding biogenic carbon. Results are shown as a variation compared to the baseline.

5.2.3.4 Functional Unit – GWP based on electricity production

In a LCA study, the choice of the functional unit is fundamental for the outcomes of the study and it has to reflect the goal and scope of the LCA. As a comparison, Figure 5.2.12 shows the GWP impacts when the seven feedstocks are considered as waste stream in a two-stage gasification and plasma process, and when the output of the plant is selected as functional unit, i.e. 1 kWh of electricity produced. This functional unit reflects better the secondary function of the gasification and plasma plant which is to produce electricity. RDF is again showing the lowest GWP impact, because of the high heating value of this waste stream. Overall the trend amongst the feedstocks does not change from the one shown in Figure 5.2.3. MSW1, MSW2, MSW3 and SRF show similar GWP impact, despite the heating value of the SRF being almost double that of the MSW1 heating value.

A comparison of the obtained results with the literature has also been performed. Muss *et al.* [418] performed a cradle-to-grave LCA of a plasma gasification process transforming construction and demolition derived biomass (CDDB) and forest residue into electricity. Their results showed a slightly higher GWP impact compared with the findings of our study, i.e., 1.8 kg of CO_2 eq per kWhel produced for biomass in Nuss *et al.* [418] compared with 0.6 kg of CO_2 eq per kWhel produced obtained in this study. This is mainly due to the different heating values of the feedstock, process parameters (electrical consumption) and system boundaries assumed in the two studies. Moreover, the two processes show different characteristics in terms of power generation unit: Nuss *et al.* [418] assumed the combustion of the syngas with a 10% of oil in a boiler and then the electricity production in a steam turbine, while here it is assumed that a gas engine directly uses the syngas for power generation.

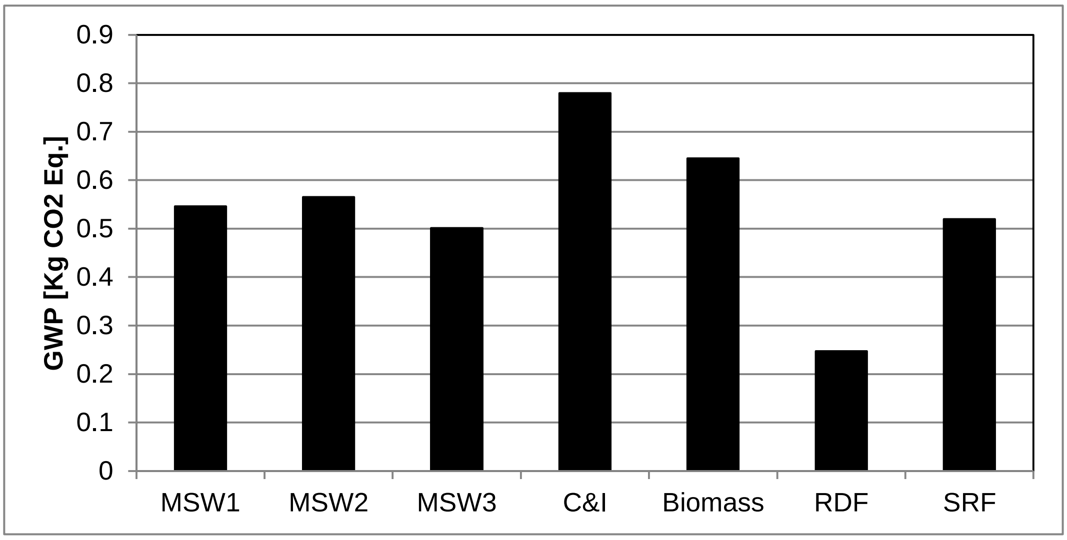

Figure 5.2.12. GWP impacts when a functional unit of 1 kWh of electricity produced at the plant is selected.

5.2.4 Conclusions

The environmental impact of a two-stage gasification and plasma plant under development has been evaluated when seven different waste streams are considered as feedstock. Overall, the environmental impact changes on the base of the characteristics of the feedstock treated and it is not possible to identify a single waste stream which is better than another for all the impact categories considered. The GWP varies from 0.220 (with RDF) to 0.960 (with biomass) kg of CO_2 eq per kg of feedstock treated. The treatment of RDF shows the lowest impact in terms of GWP, mainly due to the lower amount of feedstock needed to produce the net power output compared with the other feedstocks. For all the other impact categories, the two stage gasification and plasma process shows a negative environmental impact, mainly because of the avoided burdens associated with the production of electricity from the plant.

Based on the biogenic carbon content of each feedstock, the global warming potential impact excluding biogenic carbon has been estimated. On this basis, the GWP becomes negative for MSW1, biomass, RDF and SRF. As expected, biomass and RDF represent the most environmentally friendly cases when biogenic carbon is excluded, due to their composition and calorific values.

Detailed hot spot and scenario analyses have been carried out for processing a specific feedstock (MSW1). In terms of GWP, carbon dioxide in the flue gas contributes more than 88% of the total CO_2 eq. The production of the electricity supplied to the plasma torch and the production of the oxygen for the gasifier and plasma does not contribute significantly to the GWP (4% and 2.8%, respectively), despite their main role in the tars removal process. The main component of the flue gas is CO_2, and the amount emitted depends on the carbon content of the waste rather than the process used to convert the waste into energy. The performance of the overall system therefore depends primarily on the avoided burdens, i.e. on net electrical efficiency and recovery of usable ferrous and non-ferrous metals.

In terms of AP, the flue gas contributes 38% of the total; other relevant processes are production of electricity for the plasma torch and oxygen production. The acidification potential of the process could be improved by reducing the amount of chemicals used. In fact, production of urea used in the cleaning of flue gas contributes 9% of the total AP, while production of sodium hypochlorite for the alkali scrubber contributes 6%.

One of the main contributions to water pollution is the sodium hypochlorite in the alkali scrubber which contributes 81% of the Freshwater Aquatic Eutrophication Potential. Other chemicals should be investigated to obtain the same cleaning effect on the syngas with reduced impacts of production.

The robustness of these conclusions is limited because the gasification and plasma process is not yet fully commercialised so that the analysis had to be based on pilot plant results and simulations. The results of this study should therefore only be seen as a proxy of the actual emissions and the analysis should be revisited once data are available from full-scale operating plants.

5.3 Conventional and two-stage advanced electricity production technologies for municipal solid waste treatment

This part of the chapter aims to compare the dual stage advanced thermal technology for electricity production analysed in section 5.2 with other advanced dual stage technologies and with conventional technologies for waste treatment.

5.3.1 Goal and scope definition: functional unit and system boundary

The functional unit used for this comparative LCA is 1 kg of municipal solid waste (MSW1) as received at the plant. Figure 5.3.1 shows the boundary of the system considered. The black boxes represent the Foreground system while the grey boxes form the background. Four two-stage thermochemical processes for the treatment of MSW1 are analysed: a gasification-plasma process (G-Pl), a fast pyrolysis with combustion (FP-C), a high temperature gasification with direct melting and combustion (GM-C) and a gasification with secondary high temperature oxidation of the syngas (G-SC). The Foreground also includes conventional waste treatment technologies of the same scale, incineration and landfill.

The Foreground data used are site-specific wherever possible and based on experimental and modelling results. Otherwise, average data from the literature and specific datasets are used. The background data are regionalised in the sense that they refer to the UK rather than the European average [310]. Neither transport of waste from the generation point to the plant nor the generation of the waste are considered in the system as they are assumed to be the same for all the systems studied. The valuable outputs considered by system expansion are: the electricity generated from the waste; vitrified products generated by the gasification-plasma process and by the high temperature gasification with direct melting and combustion process; the incineration bottom ash (IBA) which can eventually be reprocessed as secondary aggregate; and the metals recovered from the waste in the four advanced thermo-chemical processes and from the incineration bottom ash (IBA).

The electricity produced is assumed to substitute electricity from the UK grid, which is based on an average mix of generating technologies [310]. Vitrified slag products as well as incineration bottom ash from the incineration plant are assumed to avoid production of crushed rock for the primary aggregate industry.

The avoided process allocated to ferrous and non-ferrous metals recovered from waste pre-treatment and incineration bottom ash have been modelled as already reported for the base case scenario in section 5.2.

Carbon dioxide from biogenic carbon is sometimes excluded from the comparison [296] because it forms part of the renewable carbon cycle, theoretically removed from the atmosphere in succeeding products. However, in this study carbon dioxide emissions from biogenic carbon are included in the estimates for the Global Warming Potential (GWP) because the assessment is based on existing waste streams with defined carbon content so that the production of the materials in the waste does not enter the analysis. Therefore the total carbon content of the waste is considered, with no distinction between biogenic and non-biogenic carbon.

The scale of the plants analysed is 20 MWe net, except for the GM-C for which the inventory is reported according to 12 MWe net output [401,402].

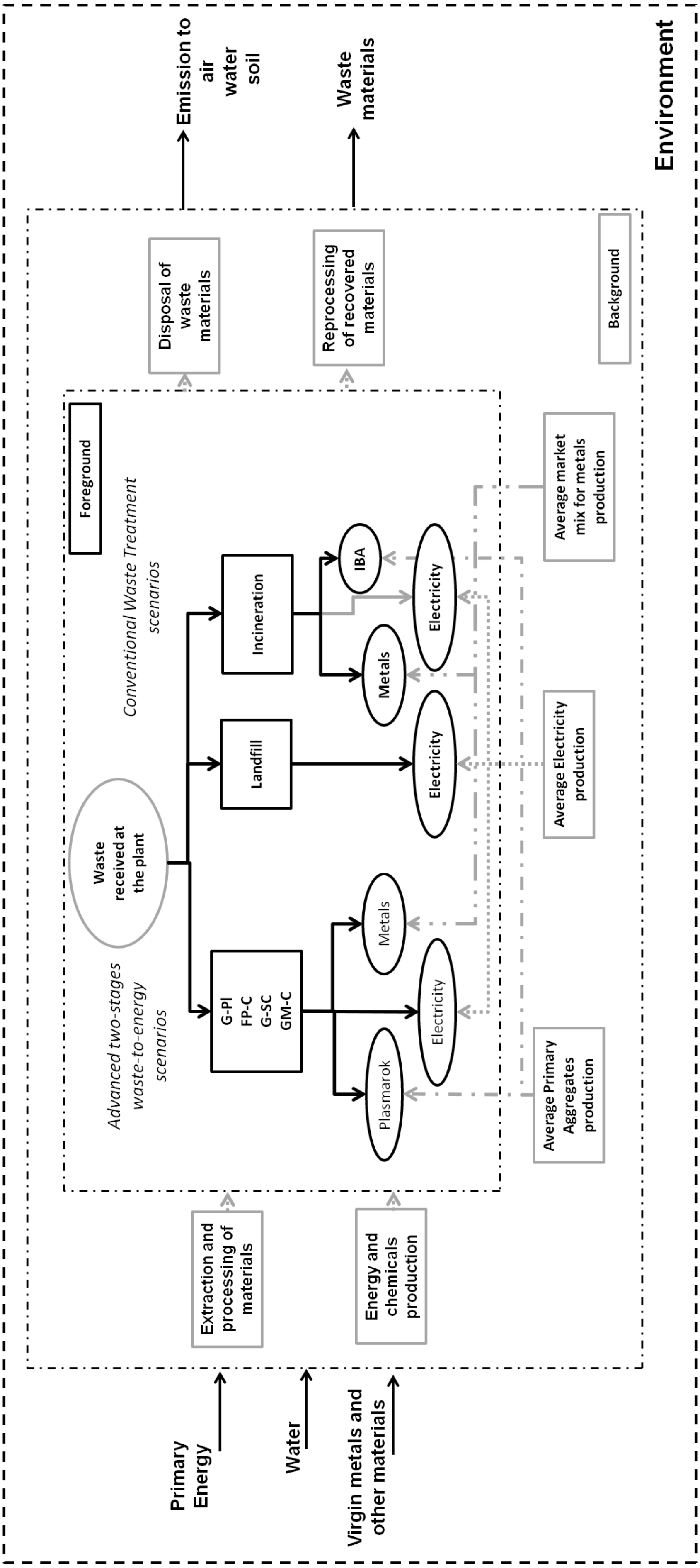

Figure 5.3.1. System boundary.

In this study the focus is on six impact categories (GWP, AP, ADP, EP, POCP, HTP) which are found to be most significant for the comparison between the different processes, as shown in the normalised results (Table 5.3.4).

5.3.2 Life cycle inventory

The mass and energy balances of the different technologies analysed are based on a set of input data generated from a variety of sources including research reports, personal communication with experts, literature, and recent environmental reports of leading companies in the related fields. Data related to indirect and avoided activities of the LCA models are mainly based on GaBi database [310]. Conversely, data for the direct activities included in the LCA models of the two-stage thermochemical processes are mainly based on process plant design using Aspen Plus software and validated, where possible, through several experimental results [313]. An inventory table of the elementary input and output flows of the processes analysed (G-Pl, FP-C, G-SC, landfill, incineration) is presented in the Appendix. Inventory data for the GM-C reported in literature [401,402] have been modified according to the purpose of this study (the composition of waste is fixed for all the processes analysed).

5.3.2.1 Waste composition

Municipal Solid Waste (MW) is a very heterogeneous feedstock, in particular in ash and moisture contents, and it has a high variability depending on its source and location, the collection methods involved and seasonal effects [417]. As the basis for the process comparison, a common MSW composition received at the plant as that of MSW1 is assumed in this work (average UK waste composition) [313]. The composition of the waste is not particularly tailored for any of the processes analysed.

Table 5.3.1 shows the ultimate analysis, moisture content and net calorific value of the MSW1 [313]. While the landfill, incineration and the GM-C plants can process MSW directly, the remaining three two-stage advanced processes need to convert the MSW into an RDF fluff, although no pelletisation is required.

Ultimate analysis	*MSW (as received at the plant)-dry basis*	*RDF (fuel entering the thermochemical process)-dry basis*
%Ash content	15.8	16
% C	41.8	42
%H	5.1	5
%O	35.3	35
%N	1	1
%S	0.25	0.3
%Cl	0.59	0.6
% Moisture content	40	12
Net Calorific Value (MJ/kg)	9	15

Table 5.3.1. Ultimate analysis, moisture content and net calorific value of MSW received at the plant and final RDF entering three thermochemical processes (G-Pl, FP-C and G-SC) on a dry-basis. The data for MSW are the same as Table 5.2.1 for MSW1 but on a different basis.

kg of metals per kg of MSW	*Metals for reprocessing in the Incineration (Sheffield plant)*	*Metals for reprocessing in the Incineration (North Hykeham plant)*	*Metals for reprocessing in the G-Pl; FP-C; G-CS*	*Molten metals for direct use in GM-C*
Ferrous metals	0.023 (collected from incineration furnace)	0.019 (collected from incineration furnace)	0.0289 (collected from waste pre-treatment)	0.036 (collected from gasifier direct melting)
Non-ferrous metals	0	0.004 (collected from incineration furnace)	0.0098 (collected from waste pre-treatment)	0

Table 5.3.2. Metals recovered in the incineration processes and in advanced thermochemical processes.

5.3.2.2 Conventional waste management system

The two conventional waste management treatments compared with the advanced technologies are landfill with electricity production and two incineration processes.

5.3.2.2.1 Landfill with electricity recovery

The inventory data for landfilling with electricity recovery are taken from the GaBi database [310] representing a typical MSW landfill with surface and basic sealing, meeting European limits for emissions. The site operations include landfill gas treatment, leachate treatment, sludge treatment and deposition. Part of the landfill gas is assumed to be flared (22%), part of it to be used for electricity production (28%) in a combined heat and power system and the rest emitted to the environment (50%). All manufacturing processes of the sealing materials, as well as energy requirements for the site, are included within the system.

5.3.2.2.2 Incineration

As reported by the England's Waste Infrastructure Report [419], in 2010 there were 73 permitted Incinerators and co-Incinerators in England, of which 18 processed MSW. The total MSW treated was 4,521,600 tpa, with a range in annual throughput of waste at these facilities from 3500 tpa up to 675,000 tpa.

The comparison is based on two incineration plants: one currently operating in Sheffield, South Yorkshire since 2006 [420]; the other a new plant still under commissioning in North Hykeham, Lincolnshire [421,422]. The high level diagram for the incineration processes is shown in Figure 5.3.2.

The Sheffield plant is considered as the best established incineration technology in England, with moving grate combustor, emissions control using urea (NO_x emissions are reduced because NO_x react with urea to form N_2 and water), hydrated lime and activated carbon injected into the flue gas, and particulate removal with filter bags [423]. Ferrous metals are assumed to be recovered from the bottom ash as shown in Table 5.3.2. The gross electrical efficiency of the process (allowing for the electrical efficiency of the steam turbine) is assumed to be 17% [423].

The North Hykeham plant is new; it has a reverse acting grate with ammonia injection to reduce the NO_x and meet the required limits. Bottom ash is recovered on site, and ferrous and non-ferrous metals are recovered in the IBA facility as shown in Table 5.3.2. Air pollution control (APC) residues are produced in the gas cleaning section, for treatment as landfilled hazardous waste. The gross electrical efficiency of the process is assumed equal to 26%, while the net electrical efficiency is 23% [421].

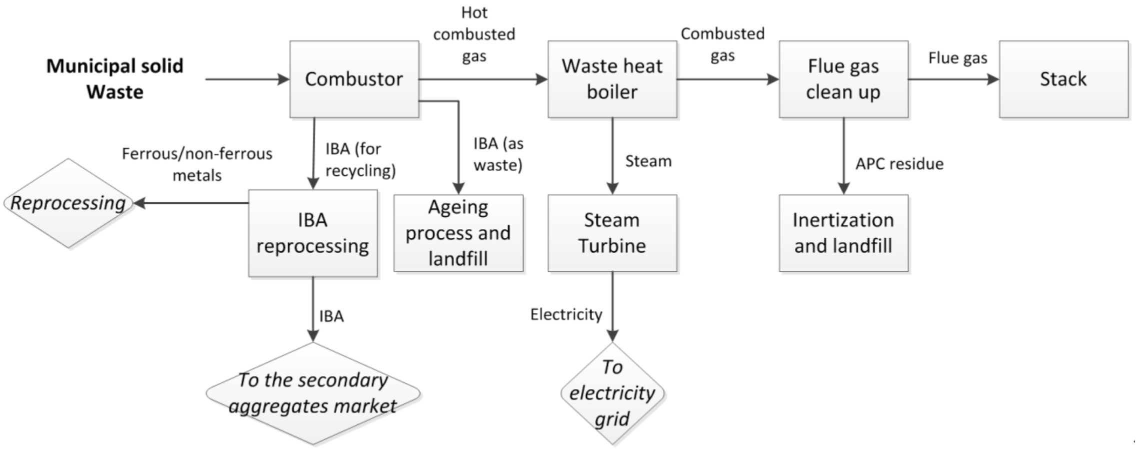

Figure 5.3.2. High level diagram of the incineration processes. The schematic is valid for both the Sheffield and the North Hykeham plants. Non-ferrous materials and bottom ash as secondary aggregate are recovered only in the North Hykeham process.

5.3.2.3 Advanced thermal technologies

The characteristics of the four two-stage thermo-chemical processes are summarised in Table 5.3.3. For consistency, the LCA study conducted on all processes is performed with the same boundaries (see Figure 5.3.1). The same amount of metals recovery has also been assumed for the FP-C, G-SC, and G-Pl as they are all advanced technologies that can treat RDF of the same composition as that treated in the G-Pl (Table 5.3.2). The different electrical efficiencies of the processes analysed are due to the nature of the technologies involved and their achievable efficiencies. In the FP-C and G-Pl, electricity is produced from the combustion of the syngas-a high calorific value fuel, mainly composed by CO and H_2,- in a gas engine. Conversely, in the G-SC and GM-C, electricity is produced in a steam turbine fed with steam raised from the heat recovered from a completely oxidised flue gas.

	FP-C	**G-SC**	**G-Pl**	**GM-C**
Feedstock	MSW	MSW	MSW	MSW
Main technology	Pyrolysis and combustion	Gasification and combustion	Gasification and plasma	High temperature gasification with direct melting and combustion
Type of reactor	Internal Circulation Fluidised bed	Moving grate	Fluidised bed	Vertical shaft
Oxidising agent	Air	Air	Steam/Oxygen	Enriched air for gasifier direct melting and air for combustor
Gas cleaning technology	Oil scrubber	Dry cleaning	Wet and Dry cleaning	Dry cleaning
Cleaning stage	Pre- combustion	Post-combustion	Pre-combustion	Post-combustion
Energy recovery system	Gas engine	Steam turbine	Gas engine	Steam turbine
Bottom ash post-treatment	Aging process and landfill	Aging process and landfill	-	-
APC residues treatment	Inertisation and then landfill	Inertisation and then landfill	Inertisation and then landfill	Inertisation and then landfill
Net Electrical efficiency (based on MSW)	26%[5]	8%	28%	18%

Table 5.3.3. Summary of the alternative thermochemical processes assessed in this study.

5.3.2.3.1 Dual stage, gasification and plasma process (G-Pl)

This process is analysed in detail in chapter 2 and section 5.2.

5.3.2.3.2 Two stage fast pyrolysis and combustion process (FP-C)

Fast pyrolysis-combustion process (FP-C) is a two stage process, deploying a pyrolyser and a combustor as shown in Figure 5.3.3. As reported in the literature [424], pyrolysis application to waste, for energy recovery is limited to few specific waste flows. In particular pure and homogeneous waste streams are required to produce good quality oil, which can be used in highly efficient energy conversion devices. The pyrolyser considered in this study converts the waste in the syngas generator unit using an internally circulating fluidised bed; the char produced by pyrolysis is converted in the

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[5] This figure does not take into account the energy content of the oil used in the cleaning

bubbling fluidised bed combustor. After cooling and partial dust removal by a cyclone, the tars and remaining dust are removed from the syngas by multistage scrubbing with a hydrocarbons scrubbing oil. Very limited data are available for this type of cleaning technology because of the novelty of the system and its limited application to waste treatment. The syngas cleaning process considered in this study consists of a collector, an absorber, and a stripper. The collector quenches the syngas with oil, cooling the gas to a temperature above the water dew point. This allows part of the tar to condense and mix with the scrubbing oil. The absorber further removes the tar vapour by absorption in oil at a constant temperature. Finally, the stripper operates at higher temperature to drive the absorbed hydrocarbons from the absorber oil [193]. The recovered mix of hydrocarbons is then recycled to the combustor. A further scrubbing system is used to remove most of the water, chloride and ammonia from the gas. The oil cleaning system and the scrubbers form the syngas refining unit. The cleaned syngas is finally used in the gas engine to produce electricity. More electricity is recovered in a steam turbine from the sensible heat of the hot flue gases.

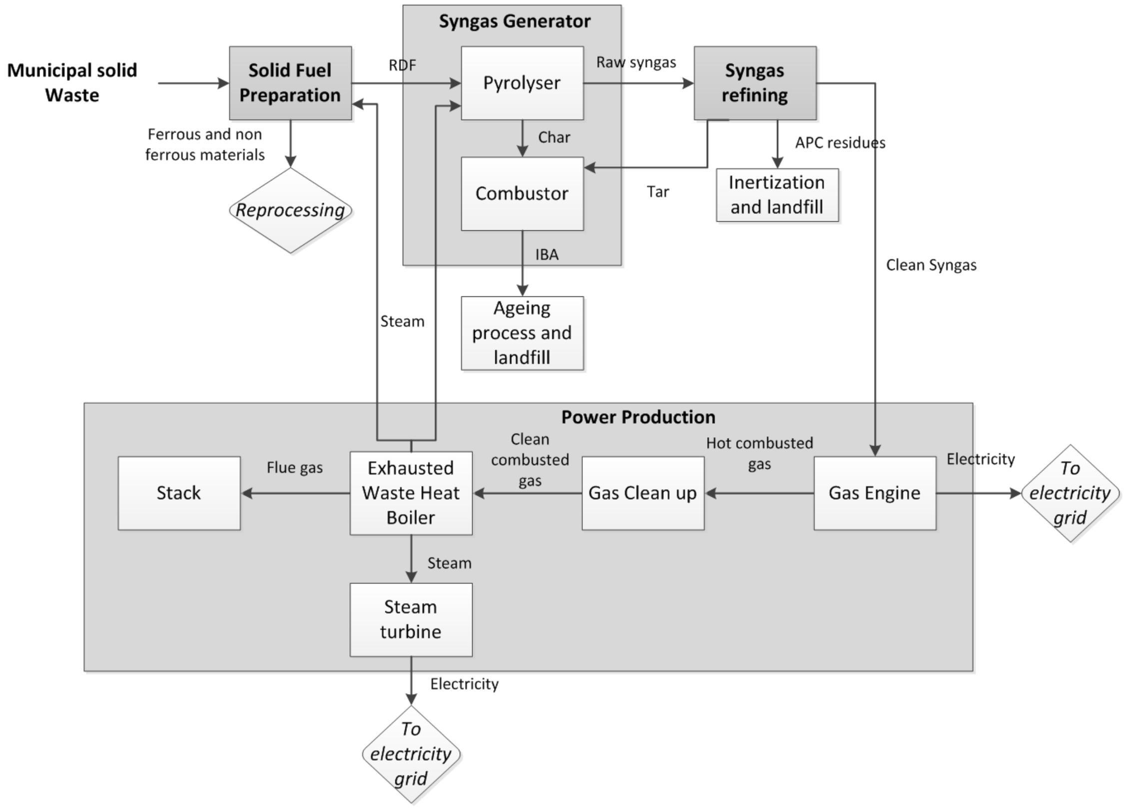


Figure 5.3.3. High level diagram of the fast pyrolyser-combustor process (FP-C).

5.3.2.3.3 Two stage gasification and syngas combustion process (G-SC)

The gasification-syngas combustor process (G-SC), shown in Figure 5.3.4, uses a moving grate gasifier to produce a synthesis gas which is then oxidised at high temperature in a secondary chamber. The system incorporates a dry flue-gas cleaning, which involves the injection of lime and activated carbon. Hot gas from the secondary chamber is recovered for steam production and a steam turbine is then used to produce electricity. Bottom ash produced in the gasifier as well as APC residues from the flue-gas cleaning system are treated and sent to landfill [425]. The net efficiency of the G-SC process considered here is 8% allowing for power use in pre-treatment to transform the residual MSW to RDF and recover the ferrous and non-ferrous materials; excluding the pre-treatment section, the net efficiency of the G-SC process itself is higher at 15%.

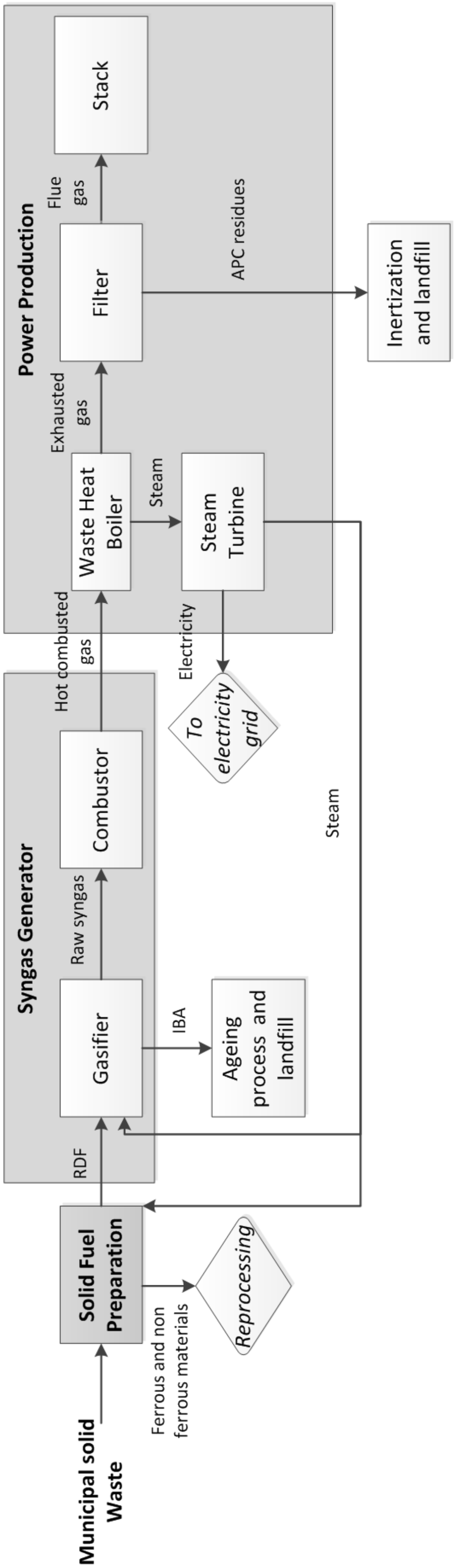


Figure 5.3.4. High level diagram of the gasifier-syngas combustor process (G-SC).

5.3.2.3.4 High temperature gasification with direct melting and combustion (GM-C)

High temperature gasification of unsorted waste with direct melting is nowadays the most widely spread gasification technology for commercial applications, mainly in Japan [426]. This technology includes a vertical shaft gasifier with direct melting and combustion (see Figure 5.3.5 for the high level flow-sheet of the process). In this type of plant, MSW is directly charged into a gasification and melting furnace from the top with coke (about 50 kg of coke is added for each ton of MSW to melt ash stably, prevent cool-down of slag and accelerate thermal devolatisation and gasification [401,402]) and 5% by weight limestone (to provide some pH buffering of the melt and to form fluid slag that can be easily discharged from the furnace bottom gasification [401,402]). From the top to the bottom of the gasifier it is possible to identify a pre-heating and drying region, a gasification region, a combustion region and a fusion region. Air or O_2 enriched air is injected into the decomposition regions (that operates at 600-800 ˚C) and in the combustion and melting region (that operates at 1000-1800 ˚C) [426,427]. The produced syngas is injected into a swirling combustor; heat is recovered from the hot flue gas and electricity is produced in a steam turbine. Then, the gas is cleaned and APC residues are filtered and the clean gas is emitted to environment. A vitrified slag is obtained as solid residues from the process, which can directly be used as secondary aggregate for road construction [428].

The inventory data to model this process are reported in literature [401,402] for a processing plant treating 200 ktpa of waste. The mass and energy balance and the emissions of this plant have been modified according to the specific waste composition adopted in this study. Parasitic loads of the process have been calculated according to literature data [426]; a gross power generation efficiency of 23% for gasification with direct melting technology, referred to a low calorific value of the MSW of 9 MJ/kg is reported. It is assumed that only electricity is produced and no heat is recovered as the heat network is not developed in the UK.

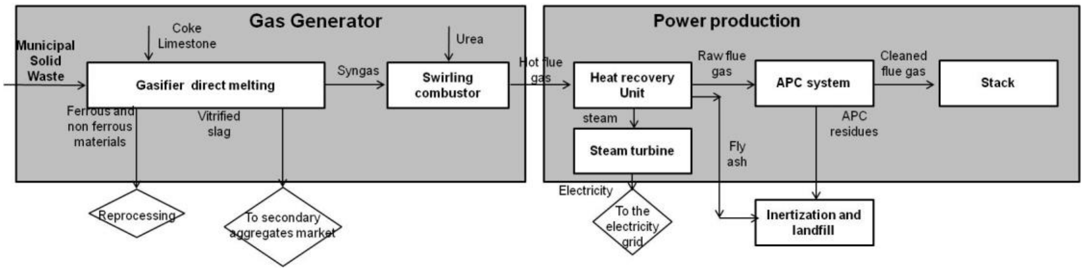


Figure 5.3.5. High level diagram of the high temperature melting gasification process (GM-C).

5.3.3 Results

The impact categories presented in this paragraph have been selected based on the normalised results, as shown in Table 5.3.4. The selected categories are highlighted in bold. The normalisation is done based on CML, IPCC, ReCiPe (region equivalents), EU25+3, year 2000 [310]. The impacts used to normalise the results are shown in the Appendix.

Normalised results (10^{-15})	**Landfill**	**Incineration (Sheffield)**	**Incineration (North Hykeham)**	**FP-G**	**G-SC**	**G-PI**	**GM-C**
ADP	1.4	3.2	-1.5	-2.6	0.39	40	-19
AP	-5.5	-89	-96	-120	-68	-180	-33
EP	3.7	-1.6	-57	-42	-22	-58	44
FAETP	53	120	27	9.6	51	-2	615
GWP	2.1	2100	510	220	710	20	206
HTP	210	160	160	130	-25	88	925
MAETP.)	-14	3200	720	290	1100	-67	2800
ODP	130	8800	17000	5700	2700	4500	0.149
POCP	0.0021	0.47	-2500	0.16	0.13	0.12	-32
TEPT	140	-28	-87	-39	-34	-50	17.9

Table 5.3.4. Normalised results.

Figure 5.3.6 shows the environmental impact of the four advanced thermochemical processes compared with conventional treatments for MSW. The results are expressed per functional unit, i.e. 1 kg of MSW received at the plant. The burdens are presented as direct+indirect, avoided and total to show the contribution of the system expansion.

As shown in Figure 5.3.6, the two-stage gasification and plasma process shows lower impact for the *global warming potential* (GWP) compared to any of the alternative technologies. For the landfill plant, the main contributor to GWP comes from the landfill gas released to the atmosphere (50% of the total), which is primarily methane and carbon dioxide. Per kg, methane has a GWP 25 times that of carbon dioxide [281]. For the incineration processes and the FP-C, G-SC and G-Pl, the main contribution to GWP comes, instead, from the flue gases released from the stack. The incinerations show a higher GWP as a result of a lower net electrical efficiency; they both determine a very similar GWP, 0.85 and 0.81 kg of CO_2 eq. for Sheffield and North Hykenham, respectively. The G-Pl process shows better performance than the three other thermochemical processes considered. This is again mainly due to the higher net electrical efficiency of the G-Pl process leading to higher avoided burdens compared with the thermochemical processes analysed here. In fact, although the parasitic loads of the G-Pl process, defined here as the total power consumed by the equipment operating in the plant, are slightly higher compared with the FP-C, GM-C and G-SC processes, the gross electrical efficiency more than compensates, as shown in Figure 5.3.7. Finally, GM-C shows an overall GWP impact of ~1 kg CO_2 eq per kg of MSW, mainly due to the C composition of the waste (that is the same for all processes analysed) and the utilisation of metallurgical coke to obtain stable high temperatures in the molten section. This increases the CO_2 emissions at the stack when compared with the alternative technologies.

Figure 5.3.6 shows the *acidification potential* (AP) of the G-Pl plant, compared with the alternative waste management scenarios investigated. The gasification and plasma process shows lower AP impacts than any other technology. In fact, it shows a negative value for AP, meaning that it avoids the impacts of emissions from conventional production of electricity (90% of the avoided burdens), metals (10%) and aggregates (less than 1%). The impact of the Sheffield incineration plant is also negative (-2.8×10^{-5} kg of SO_2 eq), whereas the AP of the North Hykenham plant shows a positive AP (4.17×10^{-4} kg of SO_2 eq). This is due to a difference of SO_2 emissions at the stack as reported in Table 5.3.5. The emissions of the North Hykenham plant are almost one order of magnitude higher than the emissions of the Sheffield plant. For the Sheffield plant, the AP is mainly due to impacts from the production of chemicals supplied to the cleaning system in the incineration plant, and to the content of NO_x and HCl in the flue gas at the stack, as shown in Table 5.3.5. (see Jeswani *et al.* [423] for further information). The Sheffield incineration plant is assumed to produce electricity only. However, even if the combined production of heat and power is considered, which reflects the current best established technology in the UK where some existing incineration plants are operated for CHP, the resultant acidification potential impact of the incineration plant is ten times higher than the gasification and plasma process (-0.9×10^{-4} kg SO_2eq and -10×10^{-4} kg SO_2eq for the incineration with CHP and G-Pl process, respectively). The North Hykenham incineration plant is provided with an advanced IBA treatment facility which can recover up to 0.2 kg of IBA as secondary aggregates per kg of waste, which is more than one and a half times the amount of Plasmarok produced in the G-Pl process [422]. However, although the avoided burdens due to the large amount of IBA which can be recycled as secondary aggregates from the North Hykeham plant are much higher than those allocated to the other thermal processes (see Table 5.3.6), higher emissions of HCl, SO_2, NOx and HF (which are the main contributors to AP impact) than the Sheffield plant and lower avoided burdens for electricity production and metal recovery than the dual stage processes, determine the higher overall value of the acidification potential for the North Hykenham plant. Overall, the AP of the incineration plants show less benefit than the G-Pl process. The acidification potentials of the FP-C, G-SC and the GM-C processes are equal to -7×10^{-4} kg SO_2 eq, -3.8×10^{-4} kg SO_2 eq and -5.58×10^{-4} kg SO_2 eq, respectively. The worse impact of the G-SC process is mainly due to two reasons: first, the gasifier-syngas combustion process has a lower electrical efficiency compared to the G-Pl process (see Table 5.3.3) and second, the secondary combustion stage leads to higher SO_2 emissions (see Table 5.3.5).

The *abiotic depletion potential* (ADP) shown in Figure 5.3.6 is an indicator of fossil energy consumption. All the processes analysed give a negative AD impact (i.e. an improvement), due to the system expansion methodology applied. However, the two stage gasification and plasma process gives the greatest saving, corresponding to a larger saving of fossil fuel resources, because of its higher net electrical efficiency. The GM-C shows an overall ADP higher than that of the other two-stage advanced processes as a result of the use of metallurgical coke. The ADP highlights a high consumption of fossil fuel resources for the gasification with direct melting technology as already observed in literature [401,402].

Figure 5.3.6 reports the *photochemical ozone creation potential* impact category. The landfill scenario has a high impact due to the higher production of certain primary air pollutants, such as VOCs, CO and NO_x emitted in the landfill gas, as shown Table 5.3.5 [267]. The analysis on the avoided burdens allocated to the use of IBA as secondary aggregates reported for the acidification potential of the North Hykenham plant applies also for the photochemical ozone layer creation potential.

For the ***eutrophication potential*** (EP) impact category, shown in Figure 5.3.6, the G-Pl process shows the best performance, being the only process determining a negative EP, followed by the FP-C. This is mainly due to the potentiality of the plasma torch which allows the conversion of almost all the ash and tars created during the gasification stage to Plasmarok [177]. This results in the production of an amount of APC residues per kg of waste treated which is 40% less than the APC residue produced in the FP-C process, as shown in Table 5.3.6.

Finally, Figure 5.3.6 compares the ***Human Toxicity Potential*** (HTP) of the processes. Again, the two-stage gasification and plasma process shows the best result, being the only advanced technology with a negative HTP impact. This category is in fact mainly related to the capacity of the process to convert the ash and tars which are formed in the first stage and to the amount of toxic substances released to the ecosphere.

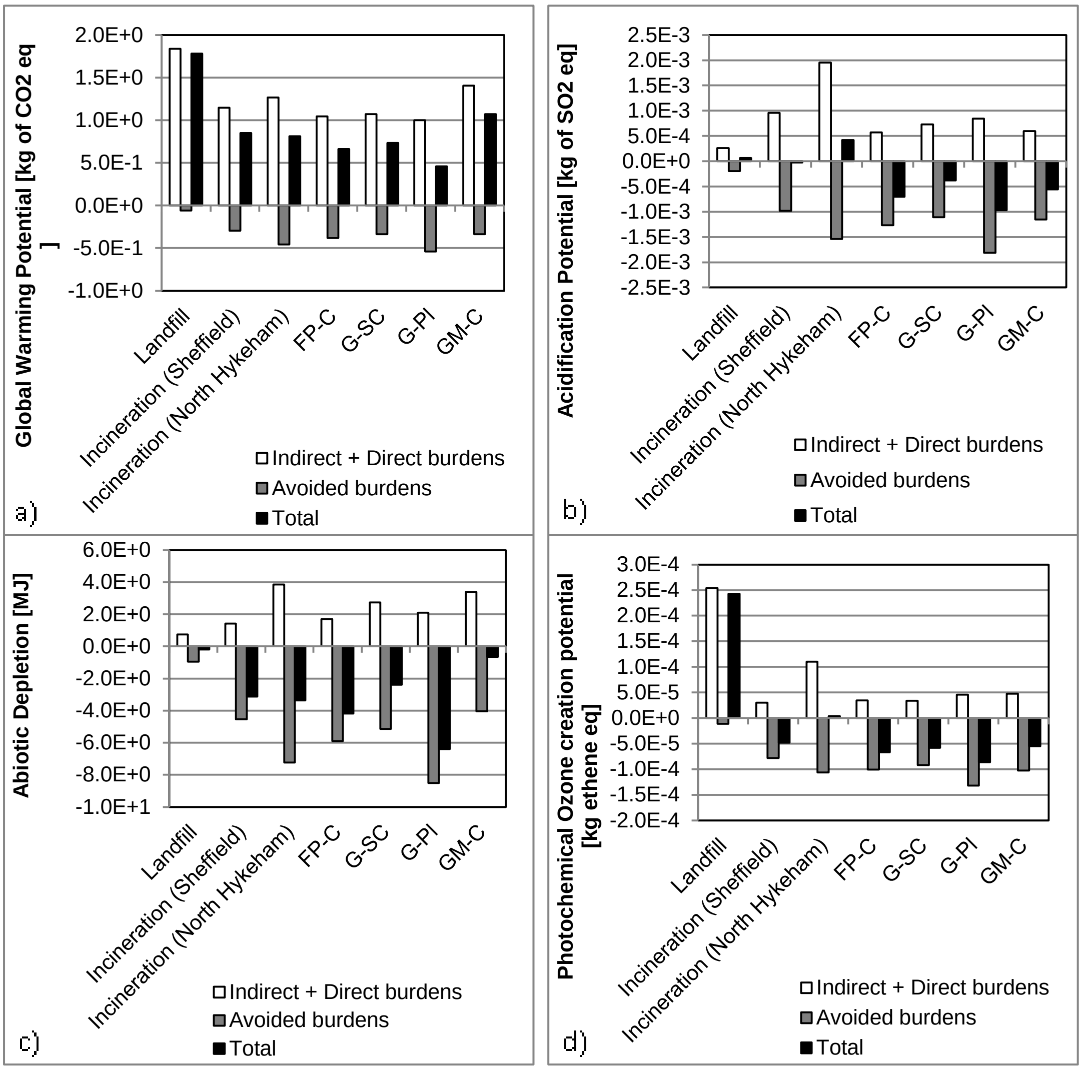
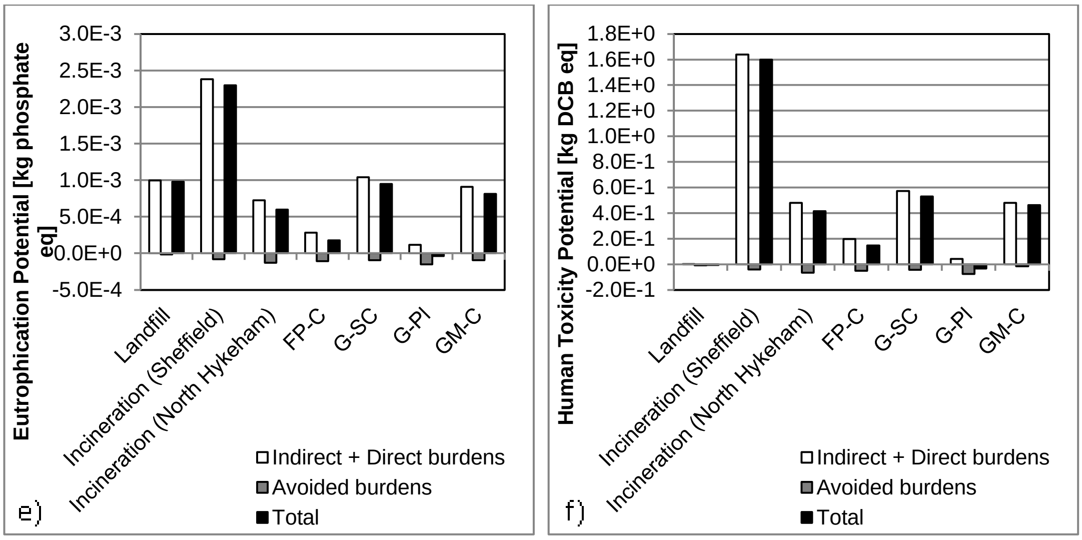


Figure 5.3.6. Environmental impacts of the four advance thermochemical processes against conventional. a) GWP; b) AP; c) ADP; d) POCP; e) EP; f) HTP.



(Continued) Figure 5.3.6. Environmental impacts of the four advance thermochemical processes against conventional. a) GWP; b) AP; c) ADP; d) POCP; e) EP; f) HTP.

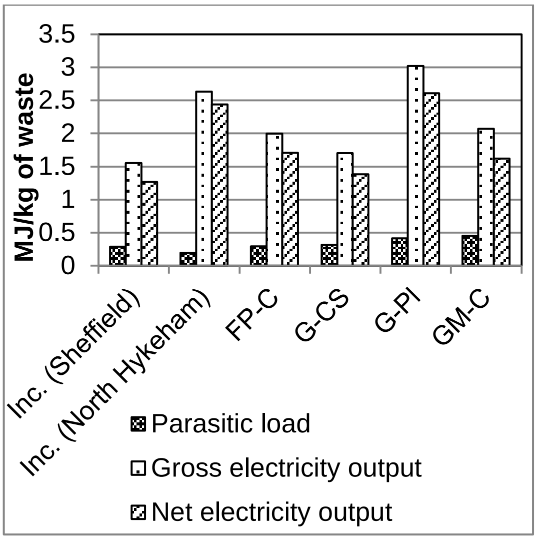


Figure 5.3.7. Parasitic loads (defined as the total power consumed by the equipment operating in the plant), gross and net electricity production of the processes analysed per kg of waste treated.

[10^{-6} kg per kg of waste]	**Landfill**	**Incineration (Sheffield)**	**Incineration (North Hykeham)**	**FP-G**	**G-SC**	**G-Pl**	**GM-C**
HCl	0.8	40	87	1.28	n.a.	0.57	13
Particulate	290	4	8400	n.a.	610	n.a.	3.5
CO	1500	2	430	300	n.a.	130	22
SO_2	97	80	430	300	920	150	12
NOx	280	700	870	500	5200	220	65
HF	0.0313	0.3	14000	11	310	10000	0.84

Table 5.3.5. Emissions to the atmosphere at the stack for the processes analysed. Only the main chemical compounds are listed.

[kg per kg of waste]	**Incineration (Sheffield)**	**Incineration (North Hykeham)**	**FP-G**	**G-SC**	**G-Pl**	**GM-C**
APC residue	0.028	0.032	0.02619	0.0228	0.0151	0.018
IBA as secondary aggregates	0	0.218	0	0	0.079	0.20
IBA residue sent to disposal[6]	0.197	0.011	0.00748	0.0532	0	0

Table 5.3.6. APC residue and IBA produced in the processes analysed.

5.3.4 Discussion

In the systems analysed in this study, the direct burdens that contribute to greenhouse warming correspond mainly to the CO_2 emissions from the stack and are directly proportional to the carbon content of the waste oxidised in the plant. The differences in contribution to global climate change between the technologies therefore result from the indirect and avoided burdens, of which the avoided burdens are expected to dominate. This means that, for climate forcing, the life cycle comparison between different technologies depends not only on the energy produced from the waste, but also on the valuable materials (i.e., metals) which can be recovered through the process. Different weights determine the total GWP of the GM-C process. Overall, the environmental impacts of the advanced two-stage thermochemical processes analysed here are lower compared with gasification with direct melting. This is mainly due to the additional requirements of metallurgical coke that significantly increases the results of the global warming potential and the abiotic depletion potential.

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[6] After metals and secondary aggregates are recovered.

Figure 5.3.8 shows the Global Warming Potential of the two-stage advanced technologies and the conventional MSW treatments expressed per kWh of net electrical output. The trend mirrors the results shown in Figure 5.3.6 and this means that the efficiency of the processes does not play the major role for the overall result score. The high temperature gasification with direct melting and combustion technology determines the highest GWP among all the dual stage technologies as a result of the additional emission of CO_2 due to the metallurgical coke, independently from the amount of MSW processed. When considering only the three advanced dual stage processes that do not require additional coke, the relatively poor performance of the G-SC process is due to the low net electricity generation, as shown in Figure 5.3.8. On this basis, the different impacts between the technologies are amplified. For the G-Pl process the GWP is 0.63 kg CO_2eq/kWh el, only 26% that of the Sheffield incineration plant; 50% of the North Hykeham incineration plant; 33% the impact of the air blown gasifier system (G-SC); 45% of the fast pyrolyser with gasifier (FP-C); and 26% of the high temperature gasification and combustion (GM-C).

The results of this study show that the two-stage gasification and plasma process is a better environmental solution for the treatment of municipal solid waste compared with incineration technologies, for all the impact categories analysed here. The comparison can alternatively be framed to show that an incineration plant, based on the North Hykeham technology, must achieve at least 30% net electrical efficiency to display the same GWP impact (i.e. kg CO_2eq per kg of MSW treated) as the two-stage G-Pl process. None of the incineration plants currently in operation in the UK reports a net electrical efficiency higher than 27% [429]. This is mainly due to the nature of the technologies involved and their achievable efficiencies: in the two-stage gasification and plasma process, electricity is produced from combustion of the syngas-a high calorific value fuel, mainly composed by CO and H_2,- in a gas engine, whereas in an incineration plant, electricity is produced in a steam turbine fed with steam raised from the heat recovered from a completely oxidised flue gas. Hence, the results show that the North Hykeham process has higher environmental impacts compared with the two stage G-Pl. However, neither of these plants is fully commercialised; the inventory data used in this study are based on pilot plants and feasibility studies, so that the comparison should be revisited once data are available from operating plants.

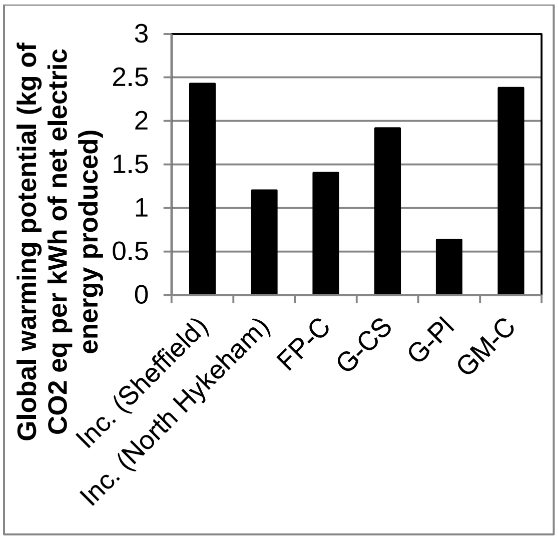


Figure 5.3.8. Global warming potential of the processes analysed for 1 kWh of net electrical output.

5.3.5 Conclusions

A life cycle assessment of four different two-stage advanced thermochemical processes has been conducted to evaluate their environmental performances in terms of carbon footprint and other environmental impacts against conventional waste treatment technologies. The two-stage advanced thermochemical processes considered in this study, specifically gasification and plasma torch (G-Pl), a fast pyrolyser with combustor (FP-C) and air blown gasifier system (G-SC), have a scale of 20 MWe net output whereas the widely spread high temperature gasification with direct melting and combustion (GM-C) has a scale of 12 MWe net output. They have been compared with two conventional waste treatment technologies, i.e. incineration with energy recovery based on two different plants and landfill with electricity recovery.

Overall, the results show that the environmental impact of the processes analysed is not only related to the electrical efficiency of the plant, but several other key factors determine the burden associated with the single waste treatment analysed. For example, treatment of MSW in the two-stage gasification and plasma process has substantially lower GWP and AP impact than landfilling (given the same amount of waste) and this is primarily due to avoiding emissions of methane from landfilled organic material. Hence, the nature of the treatment involved i.e. thermochemical versus biological, determines the environmental impact of the process itself. The two-stage G-Pl process shows a much lower acidification impact compared to incineration, mainly due to the higher SO_2 emissions when incineration technology is used. Moreover, the amount of metals recovered in the process can also play an important role in determining the overall environmental impact of a process plant. In this study, it is shown that the amount of metals recovered in the two-stage advanced processes is higher than in the incineration plants; this is because, while in the former, metals are recovered directly from the waste in the Solid Fuel Preparation unit, in an incineration plant, metals are recovered solely from the bottom ash, following combustion.

The net electrical outputs and stack emissions have also been analysed to support the broader environmental results reported.

Although the results presented in this chapter are mainly based on pilot scale experiments and process simulations for the larger scale of operation, they show that the G-Pl process may be regarded as a benchmark compared to other two-stage gasification technologies, when developing high efficiency advanced technologies for the treatment of municipal solid waste in future.

5.4 General conclusions

Advanced waste to energy technologies are currently supported by the UK government because they contribute to alleviate the waste disposal problem and at the same time they produce renewable energy. In this context, this chapter explored the environmental burdens of electricity production from waste, considering advanced and conventional technologies. The treatment of different feedstocks in the dual stage advanced process developed by APP was analysed. The analysis showed how feedstock associated with a higher calorific value and metal recovery determine lower GWP. Overall, the dual stage process determined lower environmental impacts than incineration, landfill and also other advanced gasification processes, especially when the alternative waste treatments do not consider separation and recycling of metals from waste. The energy efficiency was also shown to be a key factor for developing processes as it is strictly linked to lower environmental impacts.

Chapter 6. Methane from waste: thermal and biological technologies

This chapter aims to develop a LCA framework able to analyse the environmental impacts of renewable methane production from waste. In the first part of the chapter, the dual stage gasification and plasma thermal technology developed by APP is analysed for the treatment of MSW and production of Bio-Substitute Natural gas (Bio-SNG). Then, this technology is compared to alternative biological technologies for bio-methane production from waste.

Future energy mixes and different country-specific energy mixes are analysed to identify the effects of changing the reference system when comparing methane production to electricity production. For each section the modelling principles, the assumption and the inventory on which the LCA models are based are reported. Then, the environmental burdens are calculated in order to identify the most polluting sections of the processes analysed and the process that determines the lowest environmental impacts.

APP who is sponsoring this project alongside with its partners National Grid, Progressive Energy and Carbotech, won funding from both Ofgem's Network Innovation Competition and from the European BESTF-ERANET programme to build a pilot plant that demonstrates renewable gas production by thermal gasification of mixed waste [230]. Furthermore, after a national competition, APP together with its consortium formed by National Grid, Progressive Energy and CNG Service, was awarded by the Department for Transport governmental funding to develop a semi-commercial plant that will produce compressed Bio-SNG from waste using the gasification and plasma technology to power heavy goods vehicles [231]. The results reported in this chapter have been included in a wider analysis performed by APP for the European and national projects.

Part of the technical knowledge required to develop this LCA analysis on advanced waste treatment options has been developed whilst the author was working part-time as a process engineer at Advanced Plasma Power (APP). Mass and energy balance required for the development of this chapter have been generated using Aspen simulations complemented by experimental data.

The content of this chapter was partially published in:

Tagliaferri, C., Evangelisti, S., Clift, C., Chapman, C., Taylor, R., Lettieri, P. Life cycle assessment of conventional and advanced two-stage energy-from-waste technologies for methane production, Journal of Cleaner Production, in press.

National Grid, The future of gas. Supply of renewable gas. February 2016.

6.1 Introduction

When waste is treated in alternative technologies and a deviation from the waste hierarchy [148] is applied, Life Cycle Assessment (LCA) should be used to assess the environmental burdens of the developing alternatives [150]. Extensive LCA work is needed to assess the environmental performance of thermal and biological technologies producing methane from waste. In fact, the technological and environmental assessments of thermal technologies -mainly gasification- treating MSW for Bio-SNG production are rarely analysed in literature, whereas more studies focus on the analysis of thermal processes treating biomass.

Very few studies report on the technological performance and energy efficiency of methane production from MSW gasification. For example, Sues et al [430] modelled different routes for the production of bio-fuels, including, between others, SNG from MSW and other feedstocks to identify the mass conversion and exergy efficiency of each process. Juraščík *et al.* and Vitasari *et al.* [431,432] presented the analysis of the exergy efficiency of SNG production from wood gasification.

To the author’s knowledge, no studies report on environmental assessment of thermal technologies for methane production from the entire fraction of municipal waste. Conversely, wood and agricultural biomass [433–436] and also manure [437] treated in gasification technologies are usually considered. For wood waste, some studies [433,434] showed that the impact of the entire life cycle of the SNG process, from wood growth to heat and electricity production, was mainly due to the SNG production stage: the low overall chain efficiency of the SNG production process, resulting from additional processing, and the need for substantial energy for gas compression, limited the performance of the SNG system when compared with fossil alternatives.

Furthermore, many LCA studies on waste management assess the environmental impact of a single technology only, either biological (anaerobic digestion) [267,438–441] or thermal [176,374,378] and accordingly a single feedstock and product is analysed. Hospido *et al.* [442] are alone in analysing the environmental impacts associated with disposal of sewage sludge through anaerobic digestion or thermal processes but only pyrolysis and incineration were considered.

This chapter presents an attributional life cycle assessment of an advanced thermal technology treating the entire fraction of MSW for production of methane. Waste is first transformed into a clean syngas in an advanced dual stage gasification and plasma technology [374]; then, methane is produced using the technologies of water gas shift and methanation. The latter two technologies are already widely used in industry, for example, for production of hydrogen and ammonia from fossil resources [233,234] but they have never been previously proven for the production of methane from MSW.

In the second part of the chapter, the advanced gasification technology is compared to biological alternatives including i) mechanical pre-treatment of MSW associated with the anaerobic digestion of the organic fraction and landfill/incineration of residual waste; ii) anaerobic digestion of source separated waste and landfill/incineration of residual waste. Two different perspectives are analysed in this study: a waste management and an energy production perspective, where two different functional units are used, 1 kg of waste treated and 1 MJ of gas produced, respectively. For each perspective, the comparison is firstly performed considering the current UK energy mix and then extended to include future energy mix scenarios in the UK. On the one hand, the first approach represents the usual methodology applied in LCA studies [176,252,378,408]; on the other hand, the second approach is the most appropriate when the analysis focuses on developing technologies.

To the authors’ knowledge, this is the first study which attempts to analyse the impact of developing thermal and biological systems treating MSW for renewable methane production in the context of future energy scenarios. This work, focusing on Bio-SNG production from waste and future energy mixes, complements and expands chapter 5 [374] which focused solely on the production of electricity from waste in the current energy mix framework.

Furthermore, it is worth emphasizing that whilst many studies dealing with the environmental impact of waste to electricity and to bio-methane often analyse only the greenhouse gas emissions [47–50], this study presents a complete environmental assessment including a wide range of environmental impacts.

This chapter addresses the following questions:

- Which is the environmental impact of an advanced dual-stage gasification-plasma technology producing methane?
- How does this technology compare to the same technology when electricity only is produced in a gas engine?
- How does the dual stage advanced thermal technology for Bio-SNG production compare with alternative processes for bio-methane production?

6.2 Integrated gasification and plasma cleaning for Bio-SNG production

6.2.1 Goal and scope definition: functional unit and system boundary

In this section an attributional LCA is performed to analyse the environmental impacts of a dual stage advanced thermal technology including gasification and plasma for the production of Bio-SNG from MSW according to the technology developed by APP. To analyse the environmental burdens of this process, the following points are addressed:

1. Perform a hot spot analysis of a dual stage gasification and plasma waste treatment process for Bio-SNG production and identify the hot spots.
2. For different functional units, compare the dual stage advanced thermal technology producing Bio-SNG (this technology is identified as G-Pl-Bio-SNG) to the same technology producing only electricity from syngas in a gas engine (this technology is identified as G-Pl-El).
3. Perform a scenario analysis on the oxygen production technology and on the metal and vitrified slag substitution processes.
4. Analyse how the environmental burden of the G-Pl-Bio-SNG and G-Pl-El changes according to future energy mix in the UK until 2035.

The results are analysed per kg of MSW treated, unless otherwise specified.

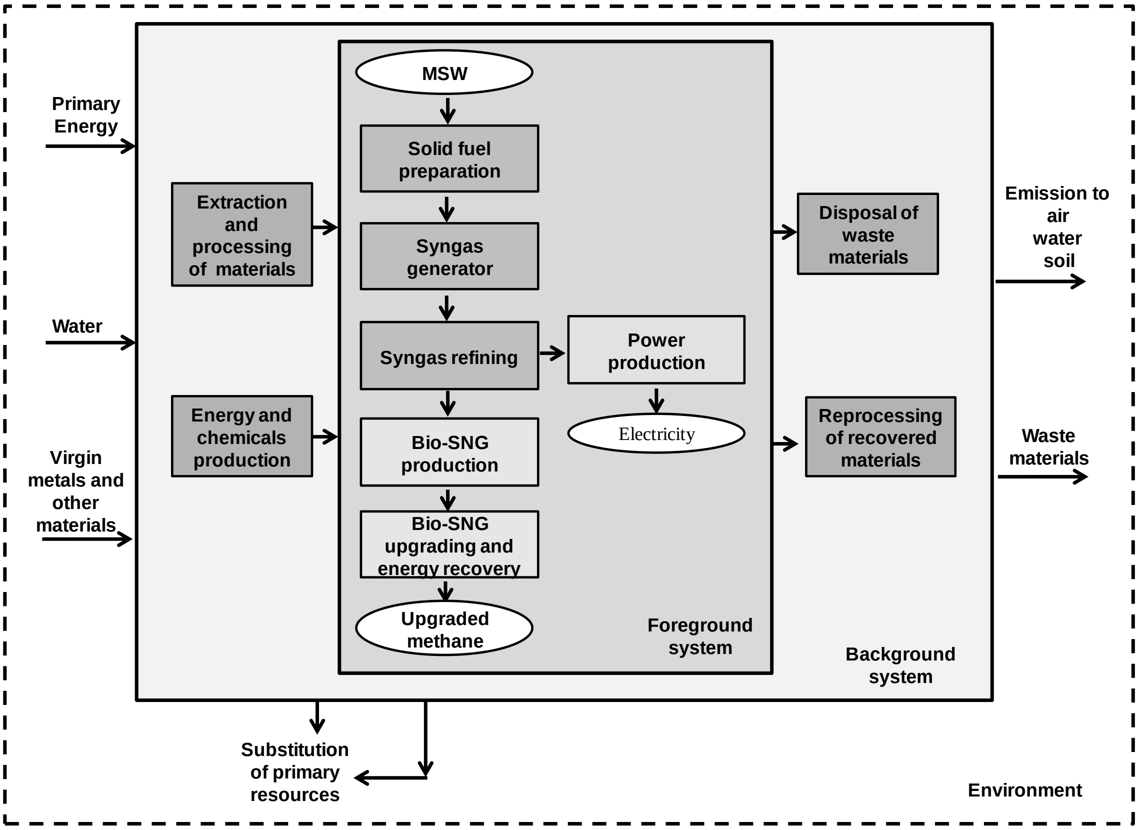


Figure 6.2.1. System boundary.

Figure 6.2.1 shows the system boundaries: direct indirect and avoided burdens are considered in the analysis and the foreground includes the G-Pl-Bio-SNG and the G-Pl-El. The G-Pl-Bio-SNG and the G-Pl-El are both treating feedstock having the same composition (see MSW1 composition in chapter 5). The 'zero waste approach' is assumed.

6.2.2 Life cycle inventory

The first three sections of the dual stage thermal treatment plant (that are solid fuel preparation, syngas generator and syngas refining, see Figure 6.2.1) are the same whether Bio-SNG or electricity are produced as the final product. The description of those sections together with the descriptions of the G-Pl-El and the modelling assumptions for the base scenario has already been reported in chapters 2 and 5. When electricity only is produced, the first three sections are followed by the power production section (its description and modelling assumptions are reported in chapters 2 and 5) whilst when Bio-SNG is produced the first three sections are followed by the Bio-SNG production and Bio-SNG upgrading and energy recovery sections. The two processes (for Bio-SNG and electricity) are designed to treat ~ 155ktpa of MSW.

The valuable outputs considered by the system expansion are the same as those reported in chapter 5 but upgraded Bio-SNG and steam are also considered Those are assumed to substitute natural gas from the UK grid, which is based on an average mix of generating technologies described by data from GaBi database [310] and steam production from natural gas, respectively. For ferrous metal the substitution ratio assumed in this chapter is 0.51 whereas for non-ferrous metal it is 0.6.

Figure 6.2.2 shows the high level diagram of the G-Pl-Bio-SNG process. For this process, the disposal of the condensate from the vapour-liquid separators is modelled according to what was already reported on the water effluents in chapter 5. In addition, the ZnS formed in the guard beds before the methanation stage is assumed to be regenerated with oxygen to produce a concentrated stream of SO_2 which in turn, can be converted into sulfuric acid, sulfur, or gypsum [443]. The environmental burden due to the process of regeneration and the consequential SO_2 conversion are not accounted in the LCA model and are considered negligible as the amount of the catalytic retention is small compared to the total amount of MSW processed (0.00132 kg of retention per kg of MSW). Key inventory data of the G-Pl-Bio-SNG are reported in Table 6.2.1. A more detailed description of the Bio-SNG production and Bio-SNG upgrading and energy recovery sections is reported in chapter 2.

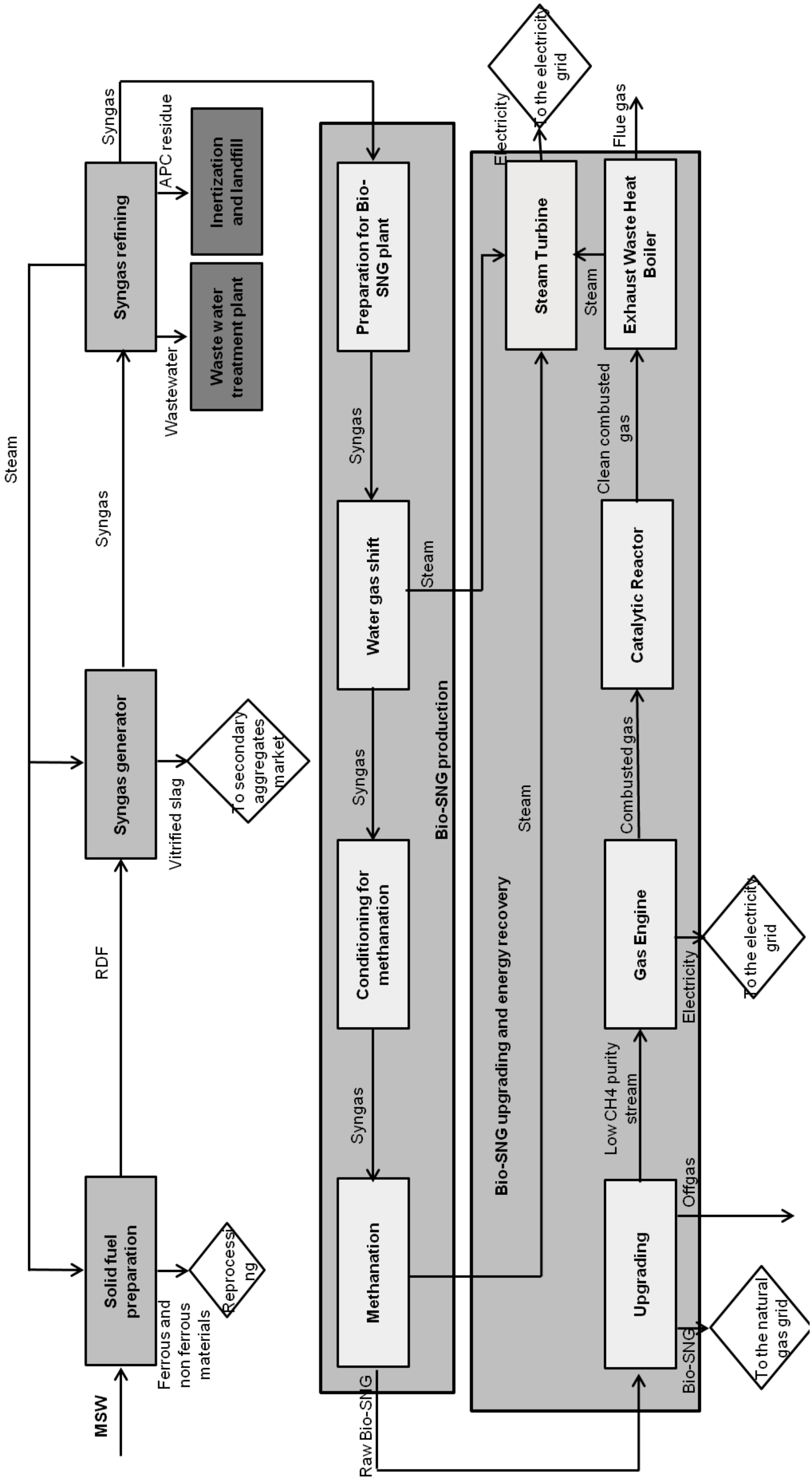


Figure 6.2.2. High level diagram of G-Pl-Bio-SNG.

Key inventory data of the G-Pl-Bio-SNG	**Value per 1 kg of MSW**
Steam used in water gas shift	0.157 kg
Power requirements for compression	0.938 MJ
Bio-SNG injected into the grid	0.107 kg
Steam to steam turbine from exhaust waste heat boiler, waste heat boiler, methanator	0.3913 kg
Electricity produced in the steam turbine	0.293 MJ
Electricity produced in the gas engine	0.287 MJ

Table 6.2.1. Key inventory data of the G-Pl-Bio-SNG per functional unit.

6.2.3 Results and discussion

6.2.3.1 Hot spot analysis of the G-Pl-Bio-SNG

GWP (Figure 6.2.3)

More than 50% of the direct+indirect GWP is due to the CO_2 emitted to the environment, mainly at the end of the PSA system and at the stack after the gas engine (contributing only for 5% to the total GWP). The direct activities determine 66% of the total positive burden. Indirect burdens are allocated mainly to the Bio-SNG production section (44% of the indirect GWP is due to the power requirements for gas compression before the methanation stage) and to the syngas generator section (power and oxygen requirements). 21 % of the total direct+indirect GWP is offset by the avoided burdens allocated to the fuel preparation unit. Except for the power required for gas compression and for the avoided burdens allocated to the Bio-SNG production, the results mirror what has already been reported for the G-Pl-El. If the ‘zero waste approach’ is neglected and the composition of the waste is accounted for in the analysis, it is possible to calculate the biogenic carbon content of the waste that is considered neutral for the GWP. Based on the specific composition of the waste treated in the G-Pl-Bio-SNG and on data from literature [417] the biogenic content of the feedstock determines 0.67 kg of biogenic CO_2 per kg of MSW treated.

ADP (Figure 6.2.3)

Indirect and avoided burdens only determine the ADP of the G-Pl-Bio-SNG. No direct burdens contribute to this indicator. The parasitic loads determine 80% of the direct+indirect burden. The total ADP is negative (-2.51MJ) thanks to the avoided burden allocated to the Bio-SNG upgrading and energy recovery section (to the Bio-SNG and electricity recovered) and to the solid fuel preparation unit. The avoided ADP allocated to the G-Pl-El (chapter 5) is almost three times higher than the avoided ADP allocated to the G-Pl-Bio-SNG because the environmental burden for the production of 1 MJ of electricity is higher than the environmental burden for the production of 1 MJ of methane according to the UK energy mixes in 2014. The analysis of paragraph 6.2.3.4.4 shows how future energy mixes can alter the results.

AP (Figure 6.2.3)

Direct emissions negligibly contribute to the total AP. This result is achieved thanks to the cleaning operations involved in the process, in particular thanks to the guard beds used to decrease the amount of S in the gas phase to avoid catalyst deactivation. The flue gas emitted to the environment after the gas engine (used to recover electricity from the low methane purity stream) has a SO_2 content of two orders of magnitude lower than the SO_2 content of the flue gas in the G-Pl-El. The indirect burdens are mainly due to the electricity requirements of the solid fuel preparation unit, plasma and compression. The avoided burdens allocated to the section of solid fuel preparation for metals recovery determine 75 % of the total avoided contributions; the rest is allocated to the Bio-SNG production and the electricity production. The total acidification potential of the Bio-SNG process is a positive number as the avoided contributions are only 71 % of the direct+indirect burdens and do not offset them. Conversely, the AP of the G-Pl-El is shown to be negative in chapter 5.

EP (Figure 6.2.3)

As for the AP, the direct contributions to the EP are negligible. 74% of the indirect contributions are due to the power requirements of the syngas generator and bio-SNG production unit. The EP of the syngas refining unit is due to the indirect activities of chemicals production as already found for the G-Pl-El in chapter 5. The production of sodium hypochlorite supplied to the alkali scrubber determines 19% of the total direct+indirect burdens. Conversely, the avoided burdens are mainly due to the metal recovery in the solid fuel preparation unit and to the production of Bio-SNG and electricity in the Bio-SNG upgrading and energy recovery section. The avoided contributions do not offset the positive burden.

FAETP (Figure 6.2.3)

This indicator is strongly increased by the indirect burden allocated to the production of chemicals in the refining section (68% of the total positive FAETP). Alternative chemicals or also alternative cleaning systems can help in the reduction of the FAETP. The electricity consumption for gas compression causes the second main contribution.

HTP (Figure 6.2.3)

The total HTP is slightly negative thanks to the avoided burden allocated to the recovery of metal in the solid fuel preparation unit that offset the direct+indirect contributions. The direct+indirect values of the HTP are completely driven by indirect burdens and in particular by power requirements. In fact, 43% of the indirect and direct burdens are due to the gas compression before methanation in the Bio-SNG production unit.

TEPT (Figure 6.2.3)

The terrestric ecotoxicity potential is driven by the electricity consumption and production. The total TEPT is a positive value because of the impact due to power requirements of the G-Pl-Bio-SNG.

ODP (Figure 6.2.3)

The production of sodium hypochlorite and urea are the main impacting processes of the syngas refining and Bio-SNG upgrading and energy recovery sections, respectively. Avoided burdens are negligible.

POCP (Figure 6.2.3)

Indirect activities only determine the total direct+indirect burdens. The avoided burdens, allocated to the metals recovery, Bio-SNG recovery and electricity recovery offset the positive impact.

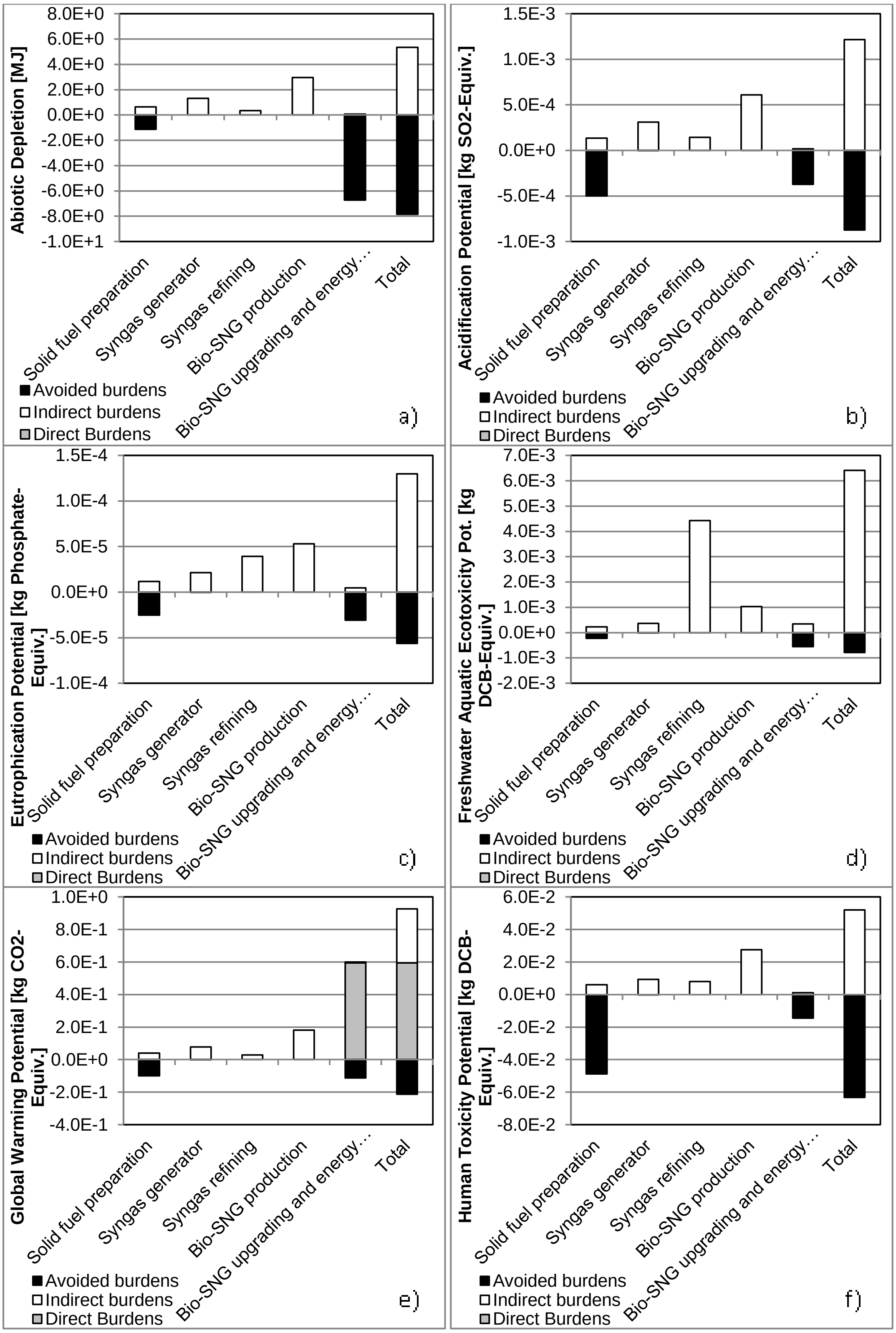
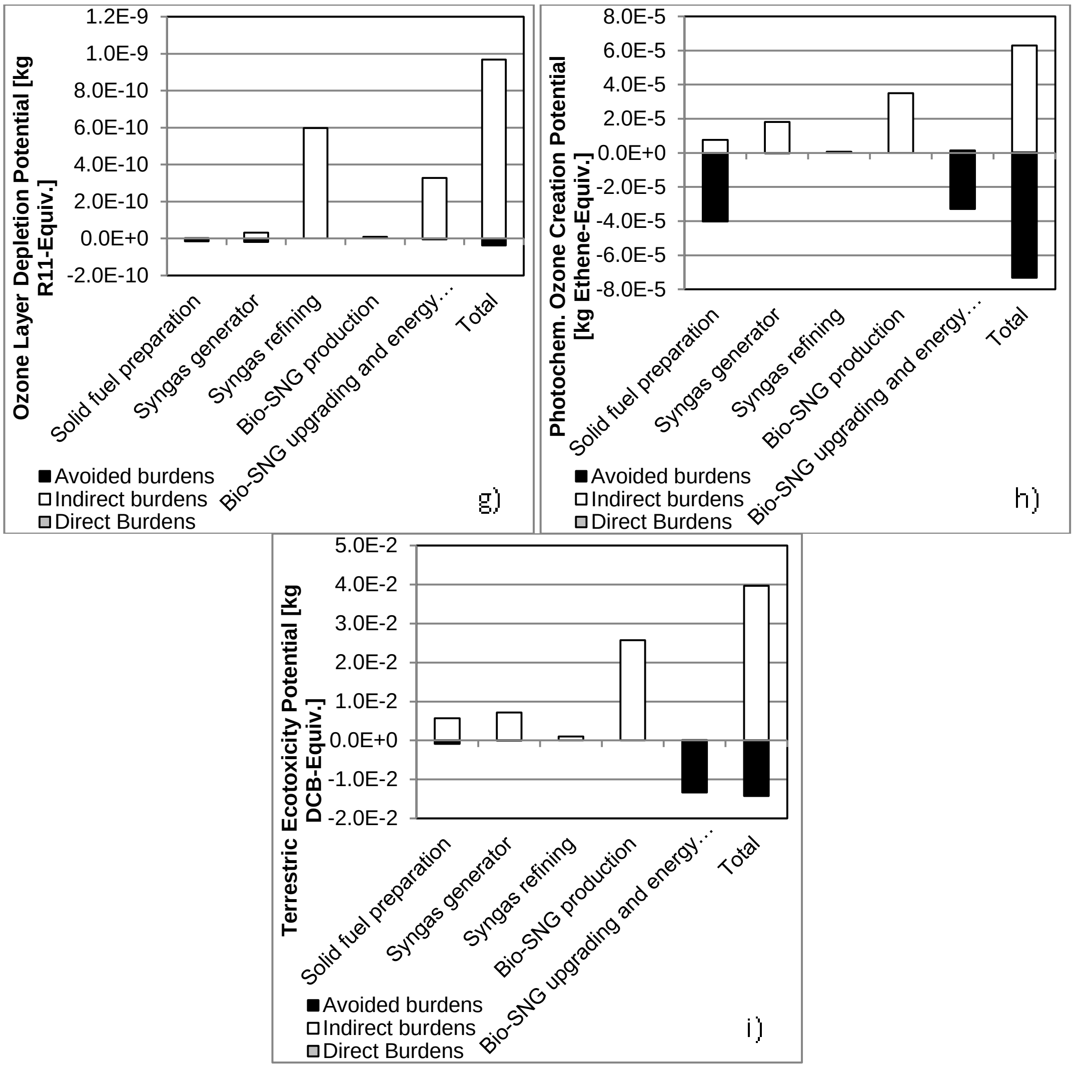


Figure 6.2.3. Hot spot analysis of the G-Pl-Bio-SNG process. a) ADP; b) AP; c) EP; d) FAETP; e) GWP; f) HTP; g) ODP; h) POCP; i) TETP.



(Continued) Figure 6.2.3. Hot spot analysis of the G-Pl-Bio-SNG process. a) ADP; b) AP; c) EP; d) FAETP; e) GWP; f) HTP; g) ODP; h) POCP; i) TETP.

The solid fuel preparation and the upgrading and energy recovery sections mainly show beneficial contributions to the environment thanks to the allocation of the avoided burdens for Bio-SNG, electricity and metal recovery. On the other hand, the syngas generator and the syngas refining sections always show disruptive contributions to the environment as no, or little avoided burdens are allocated to those sections. These results mirror what has already been found for the G-Pl-El in chapter 5. In fact, the hot spot analysis of the G-Pl-Bio-SNG and G-Pl-El shows how the sections that the two processes have in common have exactly the same role in the contribution to the burden (in both cases the solid fuel preparation determines avoided burdens and the syngas generator and syngas refining determine a disruptive contribution). The power production section of the G-Pl-El and the Bio-SNG upgrading and energy recovery section of the G-Pl-Bio-SNG determine the same type of contribution to the environment (both mainly contribute to the total burden with avoided impact) even if the sections of the two processes are not the same. For the G-Pl-Bio-SNG, the additional section of the Bio-SNG production determines a disruptive burden allocated to all the indicators.

Furthermore, the results presented so far show how the main difference between the G-Pl-El and the G-Pl-Bio-SNG is the consumption of electricity. The indicators affected by the burden of the electricity production process show a significant difference between the two processes whereas the indicators that are not influenced by the burden of the electricity production process show very similar results. The study also highlights how it is important to include in the assessment the burden of the indirect activities as for the majority of indicators the main burden is due to the indirect contributions.

6.2.3.2 Scenario analysis

A scenario analysis of the G-Pl-Bio-SNG has been performed on the oxygen production technology (the oxygen that is fed to the syngas generator section), the vitrified slag substitution process and on the metal recovery process according to scenario analysis described in chapter 5.

For the metal recovery process, some indicators (FAETP, GWP, and TEPT) show a negligible variation in the results (lower than 10%). Conversely, the indicators that are highly influenced by the metal recovery process, such as the HTP, show an increased variation in the results of the sensitivity analysis.

As for the G-Pl-El, negligible variation of the results is calculated for a change in the oxygen production technology and the vitrified slag substitution process.

The model regarding the dual stage advanced process for electricity production is more robust than the model regarding the process for Bio-SNG production as the metal recovery has a lower impact on the total burden in the former case. Further data on the scenario analysis are reported in the Appendix.

6.2.3.3 Carbon capture and storage

Recently, the concept of carbon capture and storage (CCS) as a mean for reducing CO_2 emissions has emerged in the climate mitigation policies. CCS is the process of capturing waste carbon dioxide from large point sources, compressing it and transporting it to a storage site usually using pipelines, and depositing it where it will not enter the atmosphere, normally an underground geological formation. The option of capturing CO_2 and storing it offers a mean of controlling GHG emissions. CCS is an energy-intensive process, which lowers the overall efficiency of the production plant.

While capturing CO_2 from the flue gas can reduce direct emissions from the plant itself, upstream emissions resulting from fuel and material procurement and downstream emissions resulting from waste disposal cannot be captured. These upstream and downstream emissions are usually small when compared with the direct emissions of the plant. However, when CCS is considered, the upstream and downstream emissions can become dominant and so they must be included in the environmental assessment of the process.

One of the possible technologies used for carbon capture is the adsorption of CO_2 in PSA. In the advanced thermal technology for Bio-SNG production, the first functionality of the PSA system is to produce a high pure methane stream to be injected into the grid but at the same time it produces a CO_2 rich stream. This stream can be used for storage in order to avoid CO_2 emission to the atmosphere. In light of the above, the environmental impact of the G-Pl-Bio-SNG process when the off stream of the PSA system rich in CO_2 is compressed for carbon storage is analysed.

In the analysis of the CCS system with PSA, the CO_2 emissions arising from the energy requirements for the CO_2 compression and transport need to be included. It is assumed [444] that the captured CO_2 is compressed to 13.5MPa and transported via a 300-km pipeline to the Southern North Sea where it is injected in gas fields [445]. The estimated power requirement for CO_2 initial compression is 0.04 kW per kg of MSW. In addition, electricity requirements for CO_2 re-compression along the pipeline are considered. The power requirement of 3 kW of electricity per km of pipeline is used based on a calculation from literature [446]. CO_2 leakage from the pipeline and emissions and energy requirements for the injection of CO_2, as well as the leakage from the reservoir over the lifetime of the Bio-SNG plant are considered negligible [444].

Figure 6.2.4 shows the comparison between the environmental impact of the G-Pl-Bio-SNG process as described in section 6.2.2 and the environmental impact of the G-Pl-Bio-SNG process when CCS is included. The aim of the CCS is to reduce the GHG emission and as shown in Figure 6.2.4, the GWP decreases by more than 1/3 if the storage of the CO_2 is accounted in the analysis. On the other hand, all other indicators become worse. This is due to the high electricity demand for gas initial compression and re-compression along the pipelines. Even if re-compression along the pipelines is not considered the CCS case shows the highest environmental impact for all indicators except the GWP (examples of this are the EP, ADP, AP).

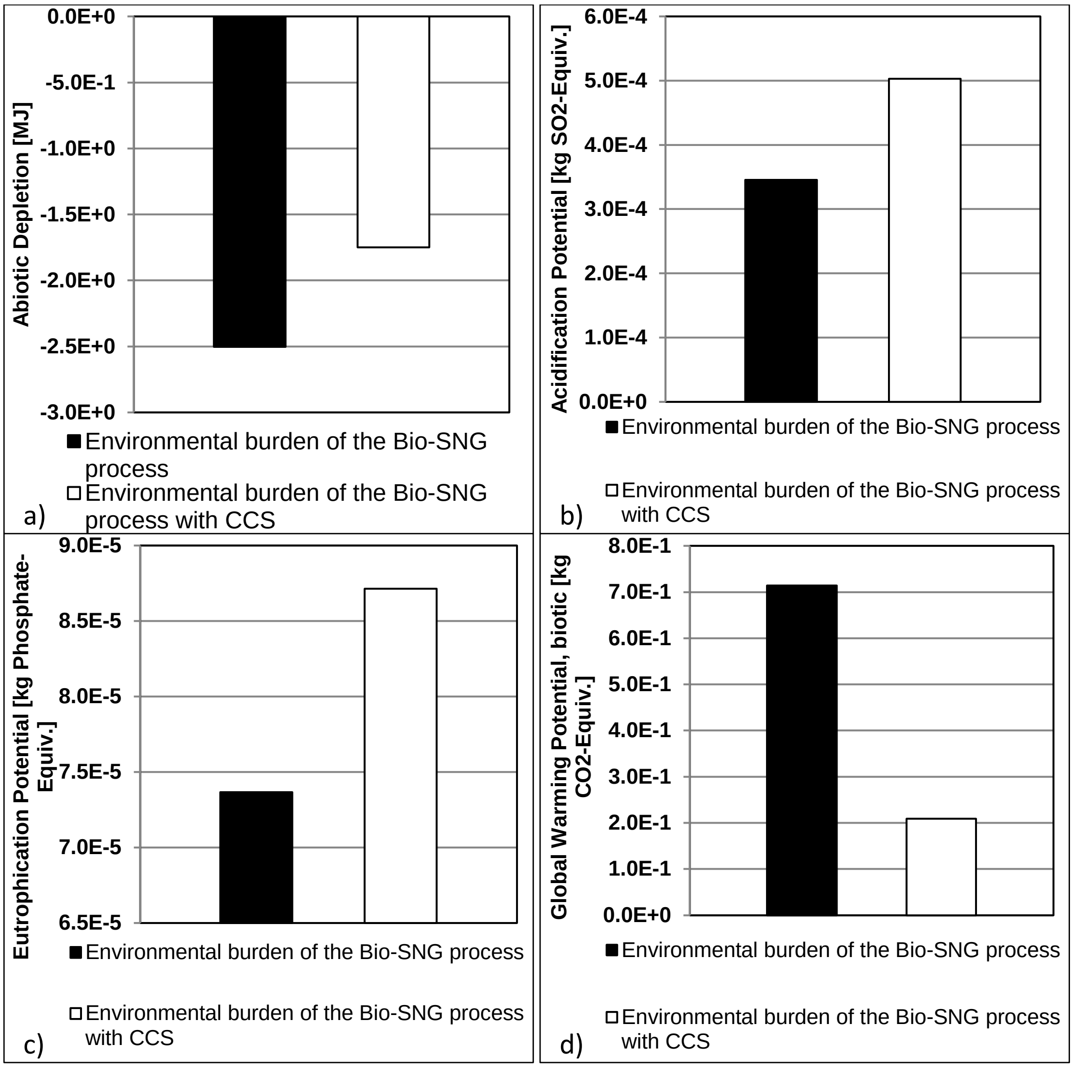


Figure 6.2.4. Comparison between the environmental burdens of the dual stage advanced thermal process for Bio-SNG production accounting and not accounting for the CCS. a) ADP; b) AP; c) EP; d) GWP.

6.2.3.4 Advanced thermal treatment: electricity or Bio-SNG

6.2.3.4.1 For 1 kg of MSW

Table 6.2.2 shows the burdens of the G-Pl-El (base case) compared to the G-Pl-Bio-SNG (base case). All the environmental impacts of the Bio-SNG process are remarkably higher; this is mainly due to the electricity requirement for gas compression before methanation. As already reported in literature [433,434], the entire life cycle of the Bio-SNG production process shows that a large fraction of the total impact is due to the SNG production stage. If new technologies achieve higher efficiency, the environmental performance of the plant could be improved. The low overall chain efficiency of SNG production process, resulting from additional processing, and the need for substantial energy for the compression of the gas, usually limits the performance of the SNG systems when compared with electricity only production processes.

Indicators	**G-Pl-El**	**G-Pl-Bio-SNG**
Abiotic Depletion [MJ]	-8.02	-2.50
Acidification Potential [10^{-3} kg SO2-Equiv.]	-2.62	0.345
Eutrophication Potential [10^{-3} kg Phosphate-Equiv.]	-0.12	0.0737
Freshwater Aquatic Ecotoxicity Pot. [10^{-3} kg DCB-Equiv.]	2.61	5.62
Global Warming Potential [kg CO2-Equiv.]	0.321	0.714
Human Toxicity Potential [10^{-3} kg DCB-Equiv.]	-90.6	-11.2
Ozone Layer Depletion Potential [10^{-9} kg R11-Equiv.]	0.955	0.931
Photochem. Ozone Creation Potential [10^{-6} kg Ethene-Equiv.]	-166	-10.1
Terrestric Ecotoxicity Potential [10^{-3} kg DCB-Equiv.]	-51.5	25.5

Table 6.2.2. Comparison between the environmental burdens of the G-Pl-Bio-SNG and the environmental burdens of the G-Pl-El. The functional unit is 1 kg of MSW treated in the two processes.

6.2.3.4.2 For 1 MJ of exergy

The burdens of the two processes are compared when the functional unit is the exergy content of the gases produced in the two processes. In detail, the functional unit of the G-Pl-El is 1 MJ of exergy of the syngas produced before the gas engine. The functional unit of the G-Pl-Bio-SNG is assumed to be 1 MJ of exergy of clean Bio-SNG produced and injected into the grid. The exergy content of the gases produced has been calculated as the sum of the exergy content of each molecular species forming the gases [447]. The exergy of the electricity produced and the exergy of the heat produced using the Bio-SNG were not chosen as functional units because the Bio-SNG is injected into the grid and can be used for different purposes, not only for heat production.

Table 6.2.3 shows the comparison between the total burden of the electricity process and the Bio-SNG process. The trend of the results is exactly the same as the trend reported in Table 6.2.3 even if the functional unit is different.

Indicators	**Total for electricity production**	**Total for bio-SNG production**
Abiotic Depletion [MJ]	-1.07	-0.49
Acidification Potential [10^{-3} kg SO2-Equiv.]	-0.35	0.0681
Eutrophication Potential [10^{-6} kg Phosphate-Equiv.]	-16.1	14.5
Freshwater Aquatic Ecotoxicity Pot. [10^{-3} kg DCB-Equiv.]	0.348	1.1
Global Warming Potential [kg CO2-Equiv.]	0.043	0.141
Human Toxicity Potential [kg DCB-Equiv.]	-0.0121	-0.0022
Ozone Layer Depletion Potential [10^{-9} kg R11-Equiv.]	0.13	0.186
Photochem. Ozone Creation Potential [10-6 kg Ethene-Equiv.]	-22.2	9.817
Terrestric Ecotoxicity Potential [10-3 kg DCB-Equiv.]	-6.89	5.0

Table 6.2.3. Comparison between the environmental burdens of the G-Pl-Bio-SNG and the environmental burdens of the G-Pl-El. The functional unit is 1 MJ of exergy of the gases produced in the two processes.

6.2.3.4.3 For different energy mixes and countries

The type of marginal electricity production technology considered strongly influences the LCA results [269,448]. In this paragraph, different energy mixes are explored to account for indirect and avoided burdens; only the GWP has been considered for this analysis.

Either of the following assumptions can be valid: i) the G-Pl-Bio-SNG and the G-Pl-El are two processes that stably contribute to the production of the UK bulk energy requirements (attributional approach); ii) the G-Pl-Bio-SNG and the G-Pl-El are marginal technologies only used in peak periods for energy production (consequential approach).

In the first case, the average energy mix in the UK should be used to allocate the avoided burden to the processes. Nowadays in the UK, the bulk requirements of electricity are supplied mainly using coal, natural gas and nuclear. This situation is shown in the first two bars of Figure 6.2.5a.

If the advanced thermal processes only supply peak energy and the consequences of changes in the level of output are considered in the LCA study (and consumption and disposal), the avoided burdens allocated to the energy production should reflect the UK marginal mix. In 2013, 2014 and 2015 wind power provided the majority of the additional energy required during peak hours over the base load (wind power was the main marginal electricity technology in the UK [62]). The environmental burdens of the G-Pl-El and of the G-Pl-Bio-SNG accounting for renewable marginal electricity from wind in the system expansion are shown in the second two bars of Figure 6.2.5a. The use of wind power as marginal electricity technology determines a decrease in the avoided burdens allocated to the electricity production as this production technology is cleaner. Therefore, the total GWP of the G-Pl-El increases, see Figure 6.2.5a. The GWP of the Bio-SNG production process increases as well (less than the GWP of the G-Pl-El) because a small amount of electricity is produced also in this case. The GWP of the G-Pl-Bio-SNG is shown to be only slightly lower than that of the G-Pl-El.

Figure 6.2.5b shows how, nowadays, the production of Bio-SNG through the advanced thermal process can be environmentally favourable over the production of electricity in countries where the electricity grid mix is cleaner than that of the UK. In 2014, in Norway, 92 % of the electricity was produced using water resources [310]. Norway in 2014 had the highest percent of electricity produced from hydropower in Europe and has therefore been chosen as the reference country in the following analysis. The burdens of the G-Pl-Bio-SNG and of the G-Pl-El are calculated assuming that those two processes are built in Norway; hence, indirect and avoided burdens are Norwegian specific; the Norwegian electricity grid mix is considered. The GWP of the Bio-SNG production process is lower than the GWP of the G-Pl-El as shown in Figure 6.2.5b.

The comparison is reported also assuming that the G-Pl-Bio-SNG and the G-Pl-El are built in Sweden. In Sweden 45 % of the electricity is produced using water resources and 34% is produced in nuclear plants [310]. The G-Pl-Bio-SNG process performs better again but the variance between the two GWPs is lower, see Figure 6.2.5b.

This analysis shows how the production of Bio-SNG is the environmentally preferable option in countries where electricity is mainly produced using renewable sources. Cleaner electricity sources decrease the avoided burdens allocated to the G-Pl-El but do not determine a significant variation of the GWP of the G-Pl-Bio-SNG (as the avoided burdens are mainly due to the Bio-SNG injected into the grid). Therefore, in countries where the electricity mix is cleaner the G-Pl-Bio-SNG is the best environmental option.

The Norwegian electricity system is not isolated but it exchanges electricity on the North Pool market. Lund *et al.* [449] report that as the Norwegian electricity production in Norway is fully used, in this country, the marginal electricity is supplied by Denmark. Therefore, if the advanced thermal processes are substituting only marginal peak technologies, when calculating the burden of the processes located in Norway, the Danish marginal electricity supply should be used. Figure 6.2.5b reports the GWP of the two advanced thermal processes when those are assumed to be built in Norway, and accounting for the Danish marginal electricity. Electricity in Denmark is mainly produced from hard coal and natural gas. The avoided burdens allocated to the production of electricity increases and the inversion of the results is shown (the G-Pl-El becomes again the preferred environmental option).

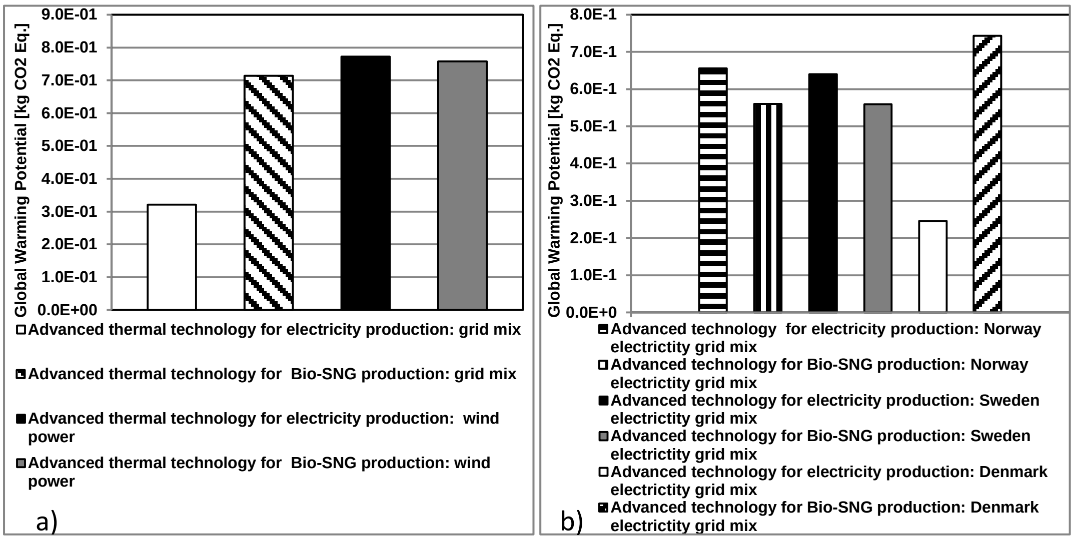


Figure 6.2.5. a) Comparison between the environmental burden of the G-Pl-Bio-SNG and the G-Pl-El in the UK according to different types of electricity production technologies. b) Environmental burdens of the G-Pl-El and G-Pl-Bio-SNG assuming that the plants are built in i) Norway, ii) Sweden iii) Sweden accounting for Danish marginal electricity technology.

6.2.3.4.4 For future energy scenarios in the UK

According to the four scenarios described by National Grid [62], (Gone Green, Low Progression, No Progression, Low Carbon Life, see chapter 2), the GWPs of the G-Pl-El and of the G-Pl-Bio-SNG have been calculated until 2035. The evolving energy mixes are taken into account for indirect and avoided burdens.

All the sources and technologies constituting the electricity and natural gas mixes (that includes between others, wind power, biomass, nuclear, offshore natural gas, etc., see chapter 2) are modelled using the Gabi database [310] except for shale gas and LNG. The environmental burdens of those two technologies have been calculated by the author (see chapter 4).

The aim of this analysis is to compare the advanced dual stage thermal processes for electricity and Bio-SNG production from 2014 to 2035 in the UK. First, only future electricity mix scenarios are considered in the indirect and avoided burdens of the two dual stage processes and then, in a second analysis both electricity and natural gas future mixes are considered.

Figure 6.2.6 reports the GWPs of the G-Pl-Bio-SNG and G-Pl-El for the four scenarios analysed by National Grid [62], accounting only for a change in the electricity grid mix (indirect and avoided burdens). In the UK grid mix, cleaner renewable energy, such as wind, nuclear etc, increases its importance within time and therefore the GWP of the G-Pl-El increases (as already seen for the 2014 mixes of NO and SE). The same trend can be seen for all the four scenarios analysed but it is stronger for the Gone Green scenario where sustainability is given high importance. One peculiar aspect of Figure 6.2.6 is that the GWPs of the two processes never cross at any point; this means that the Bio-SNG production always determines a higher environmental burden than electricity production. The main variation in the results is shown before 2020 whereas after this year the two GWPs keep almost constant. The Gone Green scenario shows very similar GWPs starting from 2020 ca.

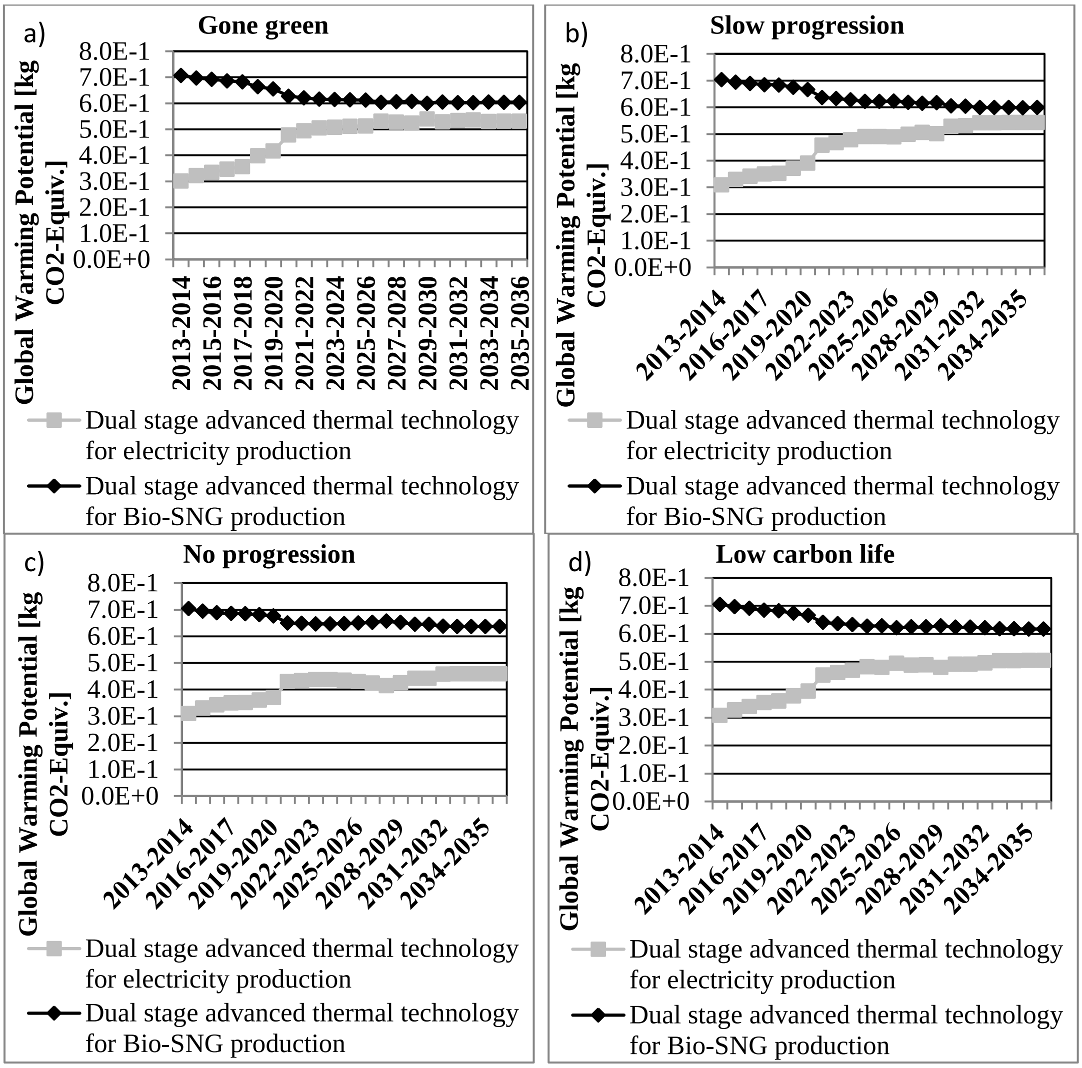


Figure 6.2.6. Environmental burdens of the G-Pl-El and the G-Pl-Bio-SNG in the UK accounting for future electricity mixes. a) Gone Green scenario. b) Slow Progression scenario. c) No Progression scenario. d) Low Carbon Life scenario.1 kg of MSW as functional unit.

Figure 6.2.7 shows the GWPs of the G-Pl-El and G-Pl-Bio-SNG within time accounting for future electricity mixes and natural gas mixes in the UK. The trend is exactly the same as that shown in Figure 6.2.6. However, in this case the increase in the GWP of the G-Pl-El and the decrease in the GWP of G-Pl-Bio-SNG are accentuated. This is due to a ‘dirtier’ natural gas mix that includes the imports of LNG and the use of local shale resources (that increase the environmental impacts of the natural gas mix). Again the inversion of the results is not shown before 2035. In the Slow Progression scenarios the two GWPs become equal whereas in the No Progression scenario the two lines remain almost parallel.

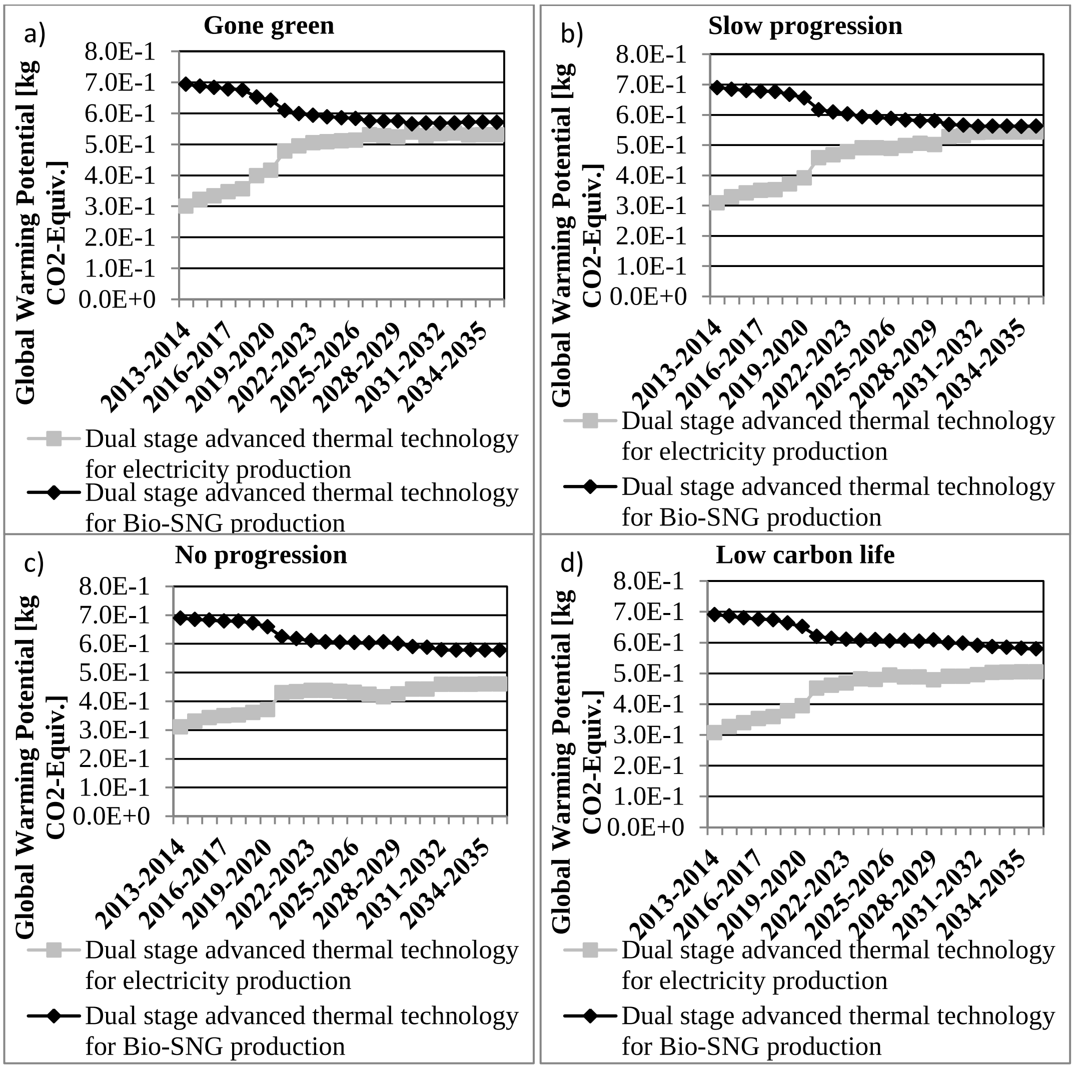


Figure 6.2.7. Environmental burdens of the G-Pl-El and the G-Pl-Bio-SNG in the UK accounting for future electricity and natural gas mixes. a) Gone Green scenario. b) Slow Progression scenario. c) No Progression scenario. d) Low Carbon Life scenario.

If the government policies prioritise sustainability within an increased economic growth the better environmental impact of Bio-SNG is assured thanks to an increased use of nuclear power after the 2035. The UK electricity grid mix needs to have a GWP lower than 0.7 kg CO_2/kWh of electricity and the UK natural gas grid mix needs to have a GWP higher than 0.5 kg of CO_2/kg of gas, for the Bio-SNG to become the best environmental option. This can be easily achieved, for example, by an increased production of power from nuclear (that needs to be at least 20% from the 16% of the 2035) and a reduction in the electricity production from natural gas and coal.

6.2.4 Conclusions

This analysis explored the environmental impact of a dual stage advanced thermal technology for Bio-SNG production at commercial scale, in terms of carbon footprint and other environmental impacts. The hot spot analysis of the process has been performed to identify the most polluting section of the process. The robustness of the LCA model has been assessed by performing a sensitivity analysis on key parameters, such as the avoided process and substitution ratio of metal recovery, the oxygen production technology, and the avoided process allocated to the production of the vitrified slag. The comparison between the burden of the G-Pl-Bio-SNG and the G-Pl Bio-SNG has been performed for a number of different cases and functional units. The GWPs of the G-Pl-Bio-SNG and the G-Pl-El have been analysed accounting for marginal renewable energy technologies in the UK and for Norwegian, Swedish and Danish electricity mixes. Finally, the environmental burdens of the advanced technologies were calculated accounting for the evolving energy mixes in the UK till 2035.

The production of Bio-SNG requires an additional section for the catalytic conversion of syngas in CH_4 when compared to the electricity production only. The Bio-SNG production section adds its burden to the total impact of each indicator. In particular, energy requirements for gas heating and compression need to be carefully considered when looking at the overall environmental performance of the process as these operations strongly influence all environmental categorises. The gases emitted to the environment at the stack and at the PSA systems are the main contributors to the GWP together with the indirect contributions of the syngas generator and Bio-SNG production sections. The carbon footprint of the process could be decreased considering the option of CCS, but at the same time all other indicators would become worse. Indirect burdens strongly impact the AP and the sodium hypochlorite production process dominates the FAETP and the ODP. The indicators whose impact is not strongly related to the production of the electricity requirements show the same results for the G-Pl-Bio-SNG and G-Pl-El.

The analysis has then shown how the comparison between the G-Pl-Bio-SNG and G-Pl-El is not unique but depends on the functional unit, energy technologies for the production of indirect and avoided energy and the location of the study. Cleaner energy mixes improve the environmental burden of advanced thermal technologies treating waste for Bio-SNG production over advanced technologies treating waste for electricity production. This has been shown when the system expansion accounts for future UK energy mixes evolving towards cleaner mixes and also for alternative locations than the UK, where the actual energy mix is cleaner.

The environmental burden of a process is not unique but it is strongly related to the inventory data used in the LCA model. The market situation, the energy mixes, the country, etc. all play a substantial role in the determination of its environmental burdens. The boundary, the limitations and the time of a LCA analysis should be carefully considered by policy-makers in the development of future legislation. Care must be taken to define the scope of the study in order to provide impartial results which would be then used to draw guidelines tackling the environment challenges that we face.

6.3 Biological and two-stage advanced methane production technologies for municipal solid waste treatment

In this section a comparative LCA is performed to analyse the environmental impacts of an advanced dual stage thermal technology and a biological process for waste treatment and production of renewable methane.

6.3.1 Goal and scope definition: functional unit and system boundary

The analysis starts from the waste stream (referred to as MSW in this study) exiting a material recovery facility (MRF), through to the production of methane, which is suitable for grid injection according to the Gas Safety Management Regulation [172]. The life cycle of the waste streams separated from the waste is omitted in this assessment as assumed to be identical in all scenarios investigated.

As shown in Figure 6.3.1, 5 different scenarios are analysed:

1. In scenario 1 (S.1), the residual waste is assumed to be mechanically sorted and then the centrally separated organic fraction is biologically treated in an anaerobic digestion plant at the same site. The separated non-biodegradable waste is partially recycled and partially sent to incineration as later specified.
2. Scenario 2 (S.2) is the same as scenario 1 but the separated waste is assumed to be partially recycled and partially sent to landfill as later specified.
3. In scenario 3 (S.3) we account for a higher source separation of bio-degradable waste and therefore the organic fine fraction of the residual waste is assumed to be source separated and treated in an AD plant whereas the rest is sent directly to incineration without further treatment.
4. Scenario 4 (S.4) is the same as scenario 3 but residual waste is assumed to be sent to landfill.

5. In scenario 5 (S.5) the waste is treated in an advanced thermal treatment technology, such as a two stage gasification and plasma process, based on the technology developed by APP [220].

Figure 6.3.1 shows the system boundary of this analysis and identifies the different scenarios, where circles identify flows whereas squares identify processes. Indirect activities of the supply chains and waste disposal processes constitute the background, whereas the scenarios investigated are the foreground. Avoided burdens are allocated to valuable substances production/recovery and emissions and residual waste material disposal are included in the assessment.

The main goals of this study are:

- To compare the environmental burdens of the different scenarios analysed and identify the hot spots.
- To compare the environmental burdens of the scenarios analysed according to the UK future energy mixes, till 2035 [62].
- To assess the impact of the functional unit on the results.
- To compare the environmental impacts of the anaerobic digestion process treating source-separated waste against centrally separated waste.

Two different perspectives are analysed in this work. Hence, the results are reported according to the functional units of 1 kg of MSW and 1 MJ of methane produced. When 1 kg of MSW is chosen as functional unit, the targeted question that the analysis is trying to answer is ‘what is the best waste management option given a certain amount of MSW?’ On the other hand, when 1 MJ of clean gas produced is chosen as functional unit, the study is trying to answer the following question ‘what is the best technology for the production of a given amount of methane?’ The results are always reported per 1 kg of waste unless specified otherwise. A key factor that differentiates the technologies analysed is the efficiency in methane production. Table 6.3.1 reports the yield in methane production for the scenarios analysed.

Scenario	Kg of MSW treated/MJ of methane produced
Scenario1-2	1.69
Scenario 3-4	0.92
Scenario 5	0.204

Table 6.3.1. Yield in biomethane production of the scenarios investigated.

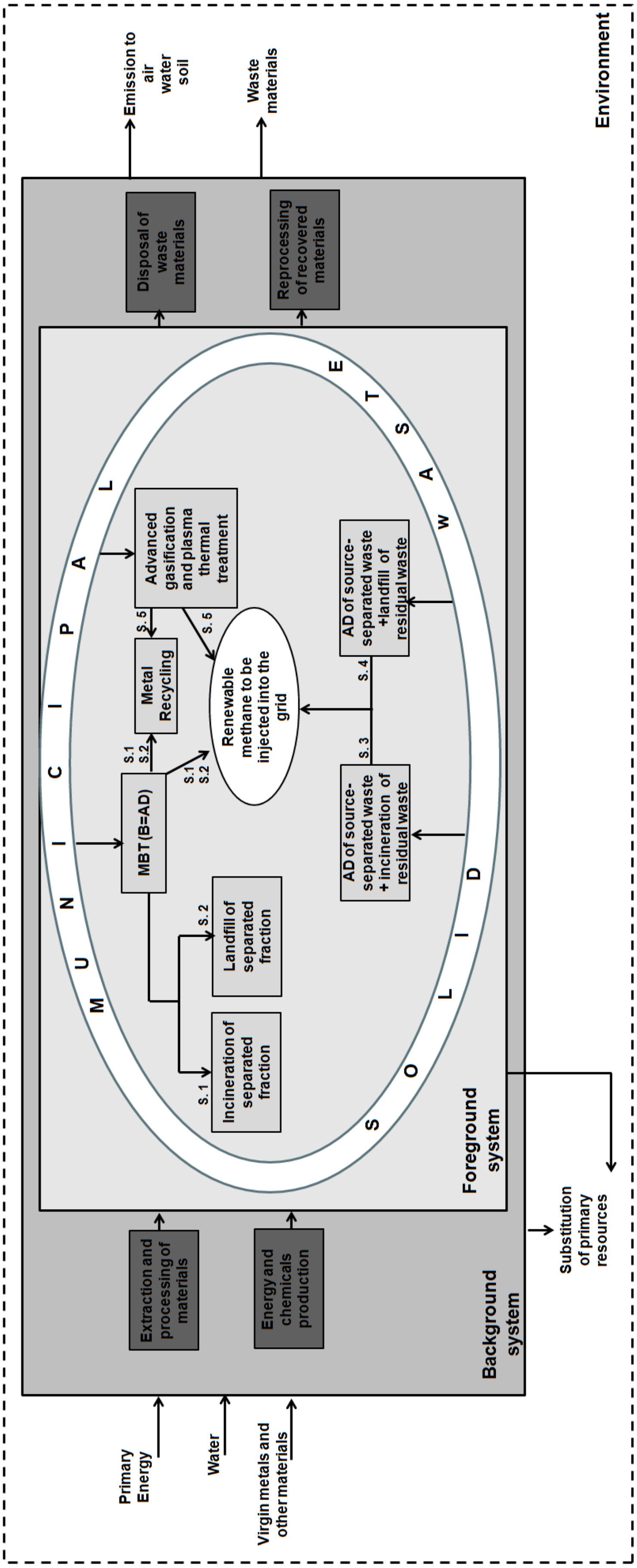


Figure 6.3.1. System boundary.

6.3.2 Life cycle inventory

The inventories of the processes analysed have been collected for commercial scale plants. Both the primary and secondary data used are regionalized and refer specifically to the UK. Key inventory data are reported in Table 6.3.2 and are further analysed in the following paragraphs and in the Annex to chapter 6. The models for incineration and landfill have been built according to GaBi database [310] and more information on those two processes and transport of waste is reported in the Appendix.

The residual waste composition and its heating value are the same as those reported for MSW1 in section 4.2 of chapter 5; they are based on typical waste collected in south-west England. The same waste composition is assumed for all the scenarios analysed.

		Modelled parameter	Value	Reference
AD of S.1, S.2	**Pre-treatment and digester**	Continuous, single-stage, mixed tank mesophilic reactor operating at a temperature of 35 ˚C	-	164,166,267,450
		Biogas yield	0.079 Nm^3/kg of centrally separated organic fraction	175
		Digester methane losses	3%	439,450–452
	Water and acids removal	Reaction of H2S with a catalytic bed of ZNO	-	170,171
		Water adsorbed on silica gel	-	170,173
	Biogas up-grading by PSA	Electricity consumption	0.8-0.88 kWh/Nm^3	171,173
		Methane losses	3%	167,173,453
	Digestate disposal	To incineration	-	345
AD of S.3, S.4	**Pre-treatment and digester**	Biogas yield	0.14 Nm3/kg of source separated organic fraction	174,267,454,455
	Digestate disposal	Fibres in the digestate	20%	456
		Liquor in the digestate	80%	456
		N of the liquor readily available to crops	80%	457
		P2O5 of the liquor readily available to crops	100%	457
		K2O of the liquor readily available to crops	100%	457
		Chemical fertilizer substituted by N	ammonium sulphate	458
		Chemical fertilizer substituted by P2O5	superphosphate	458
		Chemical fertilizer substituted by K2O	potassium chloride	458
		Nutrients dispersed to environment	-	174,439,441,459
S.5		Oxygen requirements	Average EU cryogenic oxygen production	310
		Vitrified slag: system expansion	Primary aggregates crushed rock	415,460
		APC residue treatment	-	310,345
		Water disposal	-	310
		Chemical requirements	-	310,345
		Direct and avoided burdens	-	Supplied by industrial developers

Table 6.3.2. Key inventory data.

6.3.2.1 Advanced thermal treatment: dual stage gasification and plasma process (S.5)

The description of this process and the high level diagram are reported in chapters 2 and 5.

6.3.2.2 Anaerobic Digestion of centrally separated waste (S.1 and S.2)

Archer *et al.* and Guinan *et al.* [461,462] refer to one particular layout of the MBT where no aerobic composting is used but the process is designed to deliver biogas using Anaerobic Digestion. AD cannot be directly applied to the entire fraction of MSW, therefore a mechanical treatment is needed to apply AD only to the organic fraction of the centrally separated MSW. In this case, extensive physical/mechanical separation and pre-treatment is necessary prior to digestion [175].

Many LCA studies analyse the impact of mechanical biological waste treatment (MBT) where the biological process is aerobic composting [176,306,378,408,463,464]. Conversely, very limited work has been done on the environmental impact of MBT processes where the biological treatment is AD. Some report on the software tools that can be used to calculate the burden of this process [465]; few others report on the results of the greenhouse gas impact [466] but none performs a comprehensive LCA study from cradle to grave, looking at all different environmental impacts.

Literature data have been used to build the models for scenarios 1 and 2 as referred to in Table 6.3.2; the high level diagrams of those scenarios are reported in Figure 6.3.2.

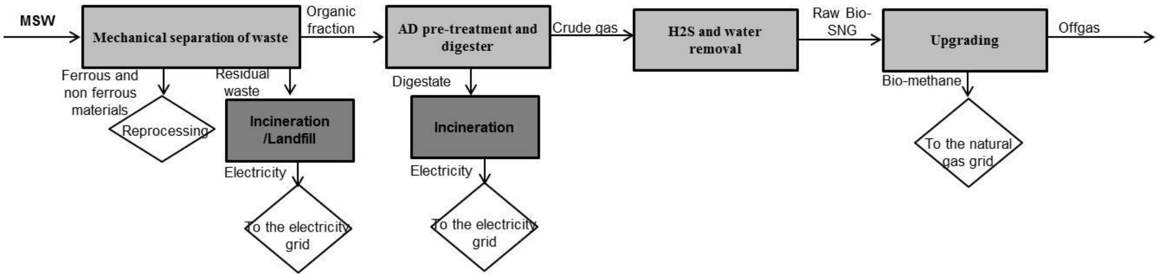


Figure 6.3.2. High level diagram of the anaerobic digestion process of centrally separated organic waste (S.1, S.2).

The outputs of the mechanical separation are i) organic fraction suitable for biological treatment in an AD plant; ii) recovered metals suitable for reprocessing and sales in the market; iii) inert material used as landfill cover; and iv) residual waste containing the remaining not separated MSW fractions sent either to incineration (scenario 1) or landfill (scenario 2). The unsorted remaining fractions are not transformed into RDF but are directly sent to the disposal facilities; no pelletizing is assumed as also reported in literature [176]. Defra [467] reports that recyclables (such as plastic and card) derived from the various MBT processes are typically of a lower quality than those derived from a separate household recyclate collection system and have a lower potential for high value markets. Therefore, for many mechanical separation systems, metals (ferrous and non-ferrous) are the only recyclates always extracted (as assumed in this study). The energy consumption for the mechanical separation of waste is based on literature [176,467,468].

Six operations are identified in the AD process (Figure 6.3.2): i) pre-treatment; ii) anaerobic digestion; iii) water and acids removal; iv) upgrading of the biogas in a PSA system; v) disposal of digestate to incineration. The characteristics of each section and the assumptions used in the LCA models are specified in Table 6.3.2 and in the Appendix.

6.3.2.3 Anaerobic Digestion of source separated waste (S.3 and S.4)

When planning for a sustainable new settlement, there is potential for increasing the sorting efficiencies [469]. In scenarios 3 and 4 the source separation of bio-degradable waste is higher than that of scenario 1 and 2 and this amount of waste is treated in an AD plant. The residual waste is assumed to be sent to incineration (scenario 3) or landfill (scenario 4). The high level diagrams of S.3 and S.4 are reported in Figure 6.3.3.

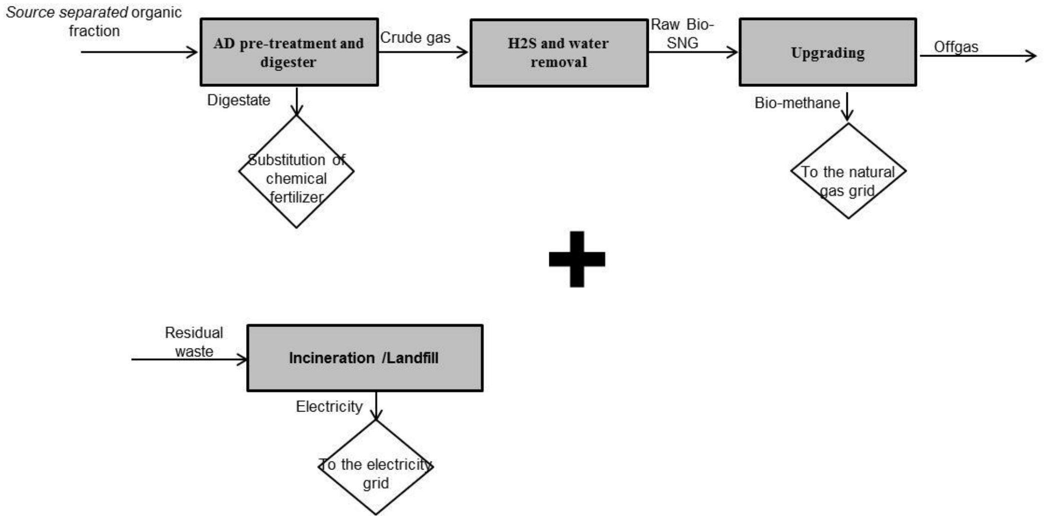


Figure 6.3.3. High level diagram of the anaerobic digestion process of source separated organic waste (S.3, S.4).

The substrate of the anaerobic digestion is kitchen source separated waste (the composition is reported in literature [454]); this is the substrate that determines the highest yield in biogas production. No card and paper are assumed to be anaerobically digested.

As the waste is assumed to be separated at source, the amount of mechanical separation and pre-treatment required (and thus the complexity and cost of the system) is reduced, although some mechanical separation is always necessary.

The model of AD for scenarios 3 and 4 is the same as the model used for scenario 1 and 2 except for the assumptions regarding the biogas yield and the digestate use. The raw biogas production has been assumed to be 0.14 Nm^3 per kg of bio-degradable fraction of MSW (wt%) (based on literature [267,454,455,470]). The whole digestate is separated in liquor and fibre as standard practice reported in Wrap [456] and the analysed separation method is physical [169]. The liquor separated from the whole digestate in the dewatering process is used as fertilizer, whereas the fibres are sent to incineration as inert material [456]. The system boundaries are expanded to include the avoided burdens allocated to the substitution of chemical fertilisers, and to the amount of carbon sequestered in the soil when the digestate is used as chemical fertilizer [470]. The emissions due to the organic fertilizers when those are on the soil are also included in the inventory. Further assumptions regarding the model are specified in Table 6.3.2 and in the Appendix.

6.3.2.4 System expansion

In scenarios 1, 2 and 5 the metals (ferrous and non-ferrous) are mechanically separated from MSW and recovered for future reprocessing and final sale as recycled metals. Therefore, avoided burdens are allocated to those processes according to the models already reported in literature [374] and in chapters 5 and 6.2.

In scenario 1 and 3 electricity is recovered from the incineration of waste; in scenario 5 the electricity is produced from the off gas of the Bio-SNG upgrading; in scenarios 2 and 4 electricity is recovered from captured landfill gas. Avoided burdens are allocated to the production of electricity based on an average mix of technology in the UK [310]. Avoided burdens have also been allocated to the production of upgraded methane because this is assumed to be injected into the grid and to substitute the UK natural gas mix [310].

In the analysis of the results, the current energy mix are substituted with future energy shares [62].

6.3.3 Results and discussion

Figure 6.3.4 shows a comparison of the environmental impacts associated with the five scenarios analysed. These results have been obtained using the current energy mix of the UK in the LCA models of indirect and avoided burdens. Only significant results are shown here, although the analysis was performed for more indicators as shown in Table 6.3.3 where normalised results are presented. It is not possible to identify a unique best scenario as the aspects influencing each indicator are different as explained in the following paragraphs. However, the scenarios where the metal recovery is considered show a better environmental performance for all the indicators analysed, except FAETP and ODP as shown in Figure 6.3.4. Those two latter indicators are driven by other factors as reported in the discussion of the results.

Normalised results (10^{-15})	**S.1**	**S.2**	**S.3**	**S.4**	**S.5**
Abiotic Depletion (ADP elements)	-70.04	4.92	-440.64	-12.56	34.50
Abiotic Depletion	-66.65	-27.14	-65.11	-13.46	-91.00
Acidification Potential	-33.40	-16.07	-9.81	13.15	19.50
Eutrophication Potential	19.82	34.77	24.90	40.23	4.21
Freshwater Aquatic Ecotoxicity Pot.	109.48	110.58	-10.81	-9.50	22.70
Global Warming Potential	93.25	171.51	104.60	187.68	138.00
Global Warming Potential, excl. biogenic carbon	-35.34	62.77	-24.00	88.12	9.23
Human Toxicity Potential	-81.27	-76.68	-8.41	-4.02	-63.00
Marine Aquatic Ecotoxicity Pot.	-2681	-2905	517.91	216.17	-2700
Ozone Layer Depletion Potential	0.28	0.18	0.30	0.04	0.09
Photochem. Ozone Creation Potential	-24.39	52.87	-16.50	70.77	-10.00
Terrestric Ecotoxicity Potential	-0.77	3.25	-0.21	3.73	1.32

Table 6.3.3. Normalised results. Results are reported for 1 kg of waste as functional unit.

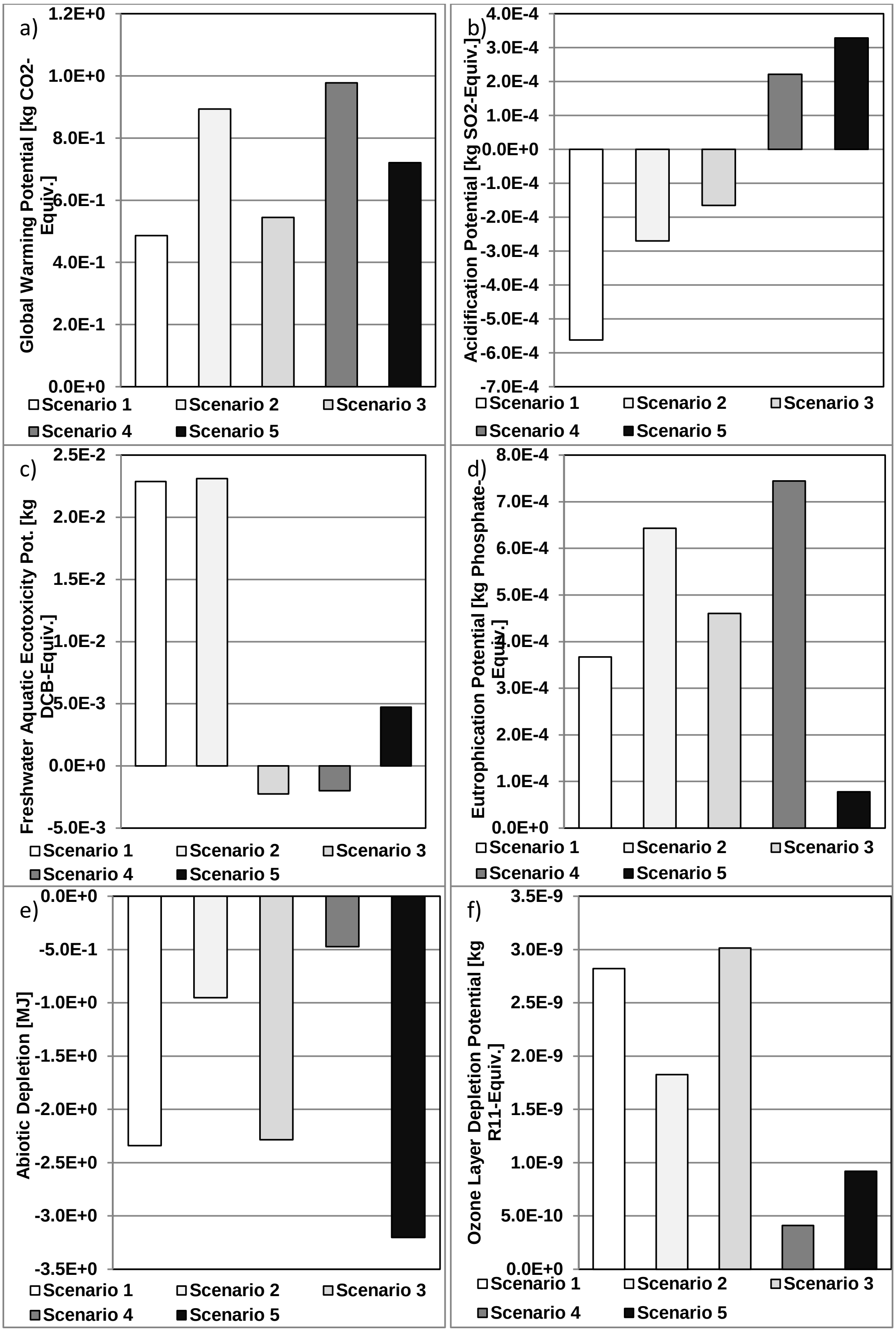


Figure 6.3.4. Environmental impacts of the scenarios analysed. Results are reported per 1 kg of waste as functional unit. a) GWP; b) AP; c) FAETP; d) EP; e) ADP; f) ODP.

6.3.3.1 Comparison scenarios 1-3-5

Figure 6.3.4 shows, among others, the environmental impacts of scenarios 1, 3 and 5. The results do not show a unique trend for all the indicators analysed.

GWP

Figure 6.3.4 shows that the dual stage process is the less favourable option. The value of the GWP for each scenario primarily depends on the CO_2 emissions at the stack and the avoided burdens allocated to the substitution of valuable products- that also means the efficiency in electricity and renewable methane production. As the waste treated in all scenarios has the same carbon composition, the avoided burdens mainly determine the relative balance of the results. The avoided burdens allocated to the production of electricity is contributing the most to the total GWP when they are compared to the avoided burdens allocated to methane production and metal recycling. This is due to the current highly carbonised electricity mix in the UK: the production of 1 kWh of the UK electricity mix determines 0.556 kg of CO_2 eq. whereas the production of 1 kWh of fossil methane determines 0.0014 kg of CO_2 eq. However, the production of Bio-SNG through thermal waste processes is not currently a fully developed technology but it will significantly contribute to the UK energy mix in future energy scenarios [62]. The latter will see an increased decarbonisation of the grid thanks to the introduction of renewable technologies and an increased footprint of the natural gas mix due to the introduction of LNG and possibly shale gas. Hence, the thermal production of Bio-SNG from waste might represent a valid alternative to decrease the burden of the UK gas mix when the analysis is performed according to the future energy mix.

AP

The AP (Figure 6.3.4) of scenarios 1 and 3 are both negative due to the allocation of avoided burdens to the recovery of metals and electricity production in the incineration processes. The indirect burdens related to the electricity recovery predominantly influence this indicator, whereas the avoided burdens allocated to methane production have a minor impact on the results (as also shown for the GWP). In scenario 5 the amount of electricity produced is smaller than the amount produced in scenario 1 and 3 and therefore the higher yield in methane production does not offset the positive burdens of the process. Scenario 1 shows an AP almost 3.5 times lower than the AP of scenario 3 even though its yield in methane is lower. This is due to the avoided burdens allocated to metal recovery in scenario 1 and not in scenario 3.

ADP

Figure 6.3.4 shows that the best option to avoid the depletion of fossil resources is the dual stage gasification and plasma process. The ADP of the advanced thermal process is

36% and 40% lower than the ADP of scenario 1 and 3, respectively. This is due to the higher yield in methane production per kg of MSW and consequently to the higher avoided burdens for methane production allocated to this process. For the ADP, hence, the aspect that determines the trend of the results is the avoided burdens allocated to the production of methane.

FAETP

FAETP (Figure 6.3.4) represents the most significant results within all the toxicity indicators and it has hence been chosen for discussion. Scenario 3 only shows a negative burden; this is due to the allocation of avoided burdens to the use of digestate as organic fertilizer substituting chemical fertilizer. In many LCA studies on AD [267,439,459,470] the allocation of avoided burdens for chemical fertilizer substitution is considered only for the GWP. Conversely, all the indicators analysed in this study account for these avoided burdens. Our results show how some indicators might be driven by the avoided burdens allocated to the chemical fertilizer substitution, hence for a complete LCA those impacts must be included in the boundaries. The FAETP value of 2.29E-2 kg of DCB Eq. allocated to scenario 1 is 100% due to the incineration of the digestate and its consequent emissions to air, water and soil through flue gas, bottom ash and APC residues disposal. Conversely, for scenario 5 the value of 4.73E-3 kg of DCB Eq. is due to upstream indirect emissions allocated to the production of chemicals used in the tertiary cleaning of the syngas.

EP

The significant difference in the EP (Figure 6.3.4) results -3.67E-4, 4.6E-4 and 7.79E-5 kg of phosphate Eq. for scenarios 1, 3 and 5, respectively- is mainly due to the difference in the emissions to the environment of the N compounds (see Table 6.3.4). Scenario 5 performs better than all other scenarios because the advanced thermal treatment causes lower emissions of NH_3. The disposal of digestate (either to incineration or as organic fertilizer for scenario 1 and 3, respectively) contributes almost wholly to this indicator. Further explanation is reported in the hot spot analysis of the anaerobic digestion.

Scenarios	**Emissions to air [10^{-6} kg]**		**Emissions to water [10^{-9} kg]**
	Ammonia	**Nitrogen Oxides**	**Total Nitrogen**
Scenario 1	13.4	-44.1	-1.83
Scenario 3	39.5	21.0	-1.24
Scenario 5	6.93	261.0	3.22

Table 6.3.4. Emissions of ammonia and Nitrogen Oxides to air and of Total Nitrogen to Fresh Water. Data are reported as per 1 kg of waste as functional unit.

ODP

Scenario 3 shows a higher ODP (see Figure 6.3.4) than scenario 1 because of the lack of avoided burden allocated to the metal recovery in scenario 3.

The previous results have been reported per kg of waste treated but the calculations for the functional unit of 1 MJ of methane produced have also been performed. The trend of the results is the same for all the indicators (therefore, those results are not reported) except for the ADP and GWP (see Figure 6.3.5).

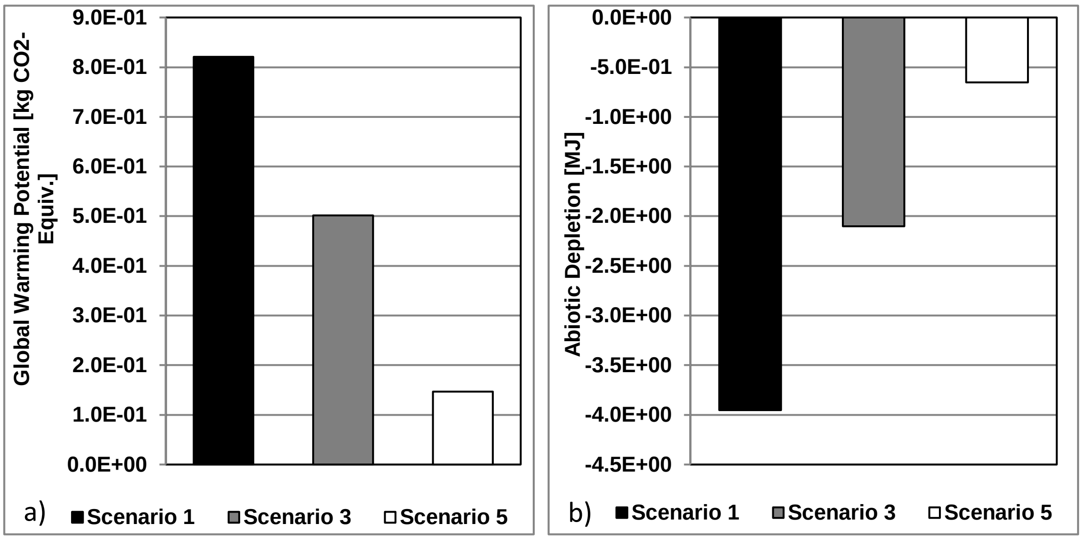


Figure 6.3.5. Environmental impacts of scenarios 1-3-5. Results are reported per 1 MJ of methane produced. a) GWP; b) ADP.

ADP (1 MJ of upgraded gas as Functional Unit)

Figure 6.3.4 shows that the best option is the dual stage process whereas Figure 6.3.5 shows that this process becomes the worst environmental scenario. Given 1 MJ as functional unit, the avoid burdens allocated to the production of the methane injected into the grid are the same for all the scenarios analysed and the aspects that prevail on the results are the avoided burdens allocated to the electricity production and metal recovery. Given a fixed amount of methane produced, different yields in methane production (as reported in Table 6.3.1) determine different amounts of MSW treated in the different processes. For 1 MJ of upgraded methane, the smallest amount is treated in the advanced thermal treatment process-0.2 kg- (as the yield in methane of this process is the highest); lower avoided burdens (compared to the avoided burden of scenarios 1-3) are, therefore, allocated to the metals recovered and to the production of electricity from the off gas in scenario 1. The amount of waste treated in scenario 1-3 is higher-2.8 kg and 1.6 kg, respectively. This results in higher avoided burdens allocated to the electricity recovery from the incineration of residual fractions in scenarios 1-3 and also in higher avoided burden allocated to the recovery of metal in scenario 1.

GWP (1 MJ of upgraded gas as Functional Unit)

A change in the functional unit determines an inversion of the results also for the GWP, but in this case Figure 6.3.4 shows that the dual stage process is the worst option whereas this process becomes the preferred choice in Figure 6.3.5 (for the ADP it was the opposite). When the functional unit is assumed to be 1 MJ of methane injected into the grid the avoided burdens allocated to the production of methane are the same for the three processes (Figure 6.3.5). The yield of methane production for the dual stage process is the highest and this corresponds to the lowest amount of MSW treated and therefore lowest direct burden of CO_2 from this process (emissions of CO_2 to the environment are based on the amount and composition of waste). For this case the avoided burdens allocated to the electricity and metal recovery do not have a significant influence on the results.

The other indicators do not show an inversion in the results because the avoided burdens allocated to the recovery of methane, electricity and metal are balanced and do not change the relative effect when the functional unit is changed.

Those results demonstrate how the choice of the functional unit is a key point of a LCA analysis as this may change the trend of the results.

6.3.3.2 Comparison scenarios 2-4-5

Figure 6.3.4 also reports the environmental results for scenarios 2 and 4. Even if the numerical results are not the same as scenarios 1 and 3, the relative trend of S.2, S.4 and S.5 is the same as S.1, S.3 and S.5 for the ADP, AP, EP and FAETP. For these indicators, the different environmental burdens allocated to scenarios 2 and 4 due to the landfill instead of incineration do not alter the preferred environmental choice. On the other hand, GWP and ODP do not show the same trend of the results.

GWP

When considering scenarios 1, 3 and 5 (Figure 6.3.4) the best choice to treat 1 kg of waste is scenario 1 (even if this scenario is not optimized for methane production, it is the one that determines the lowest environmental impact due to the avoided burdens allocated to electricity and metal production). Conversely, when considering scenarios 2, 4 and 5 (Figure 6.3.4), the best option is shown to be scenario 5. The methane that comes from the landfill gas released to the atmosphere (which is primarily methane and carbon dioxide) is the main contributor to GWP for scenarios 2 and 4 and this gives the poorest environmental performance. For scenario 5 the main contribution to GWP is instead coming from the off gases released from the upgrading system (which is primarily carbon dioxide).

ODP

This is the only indicator where S.2 and S.4 perform both better than S.1 and S.3. This is due to the lower contribution of indirect chemical productions for S.2 and S.4.

ADP, AP and GWP of scenario 1 and 3 are better than the same indicators for scenario 2 and 4 as expected (landfill is reported to have a higher environmental impact than incineration mainly because of the lower amount of energy recovered and higher emissions). However, EP and FEATP are shown to be the same for scenarios 1, 2, 3 and 4. The reason for this has to be found in the hot spot analysis of those processes. The main contributor to the EP and FAETP is due to the digestate disposal. Therefore, the other impacts of the processes, such as landfill, incineration or recovery of valuable substances become negligible and those do not affect the results.

6.3.3.3 Hot spot analysis of the anaerobic digestion processes

A hot spot analysis of the anaerobic digestion processes of centrally separated waste and AD of source separated waste is also performed. In this assessment, all the processes upstream of the biodegradable waste pre-treatment are not included, as the focus is only on the differences between the two AD processes.

Results in Figure 6.3.6 and Figure 6.3.7 are reported per 1 kg of organic waste.

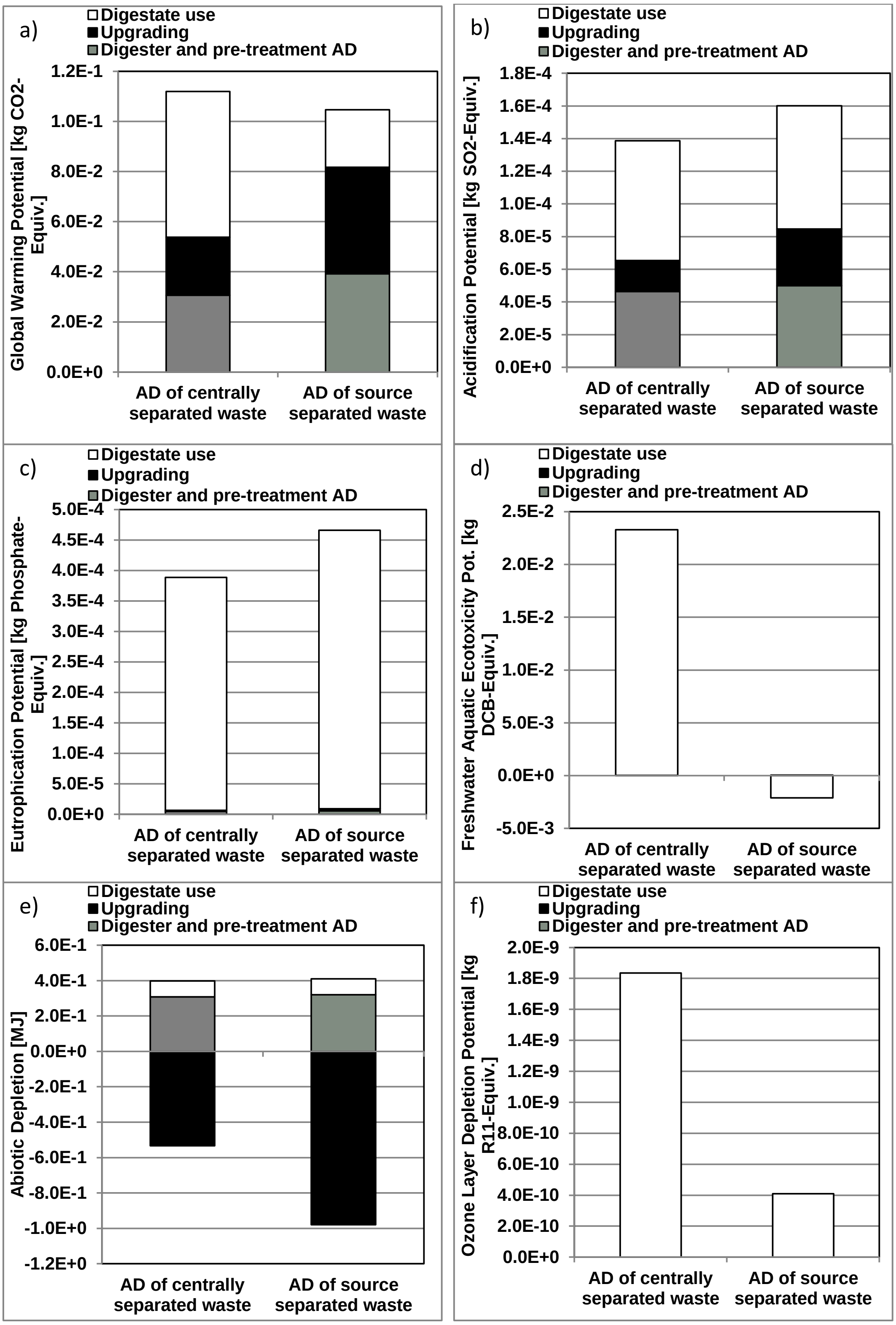


Figure 6.3.6. Hot spot analysis of the AD processes from centrally separated waste and source separated waste. Results are reported per 1 kg of waste as functional unit. a) ADP; b) AP; c) EP; d) FAETP; e) GWP; f) ODP.

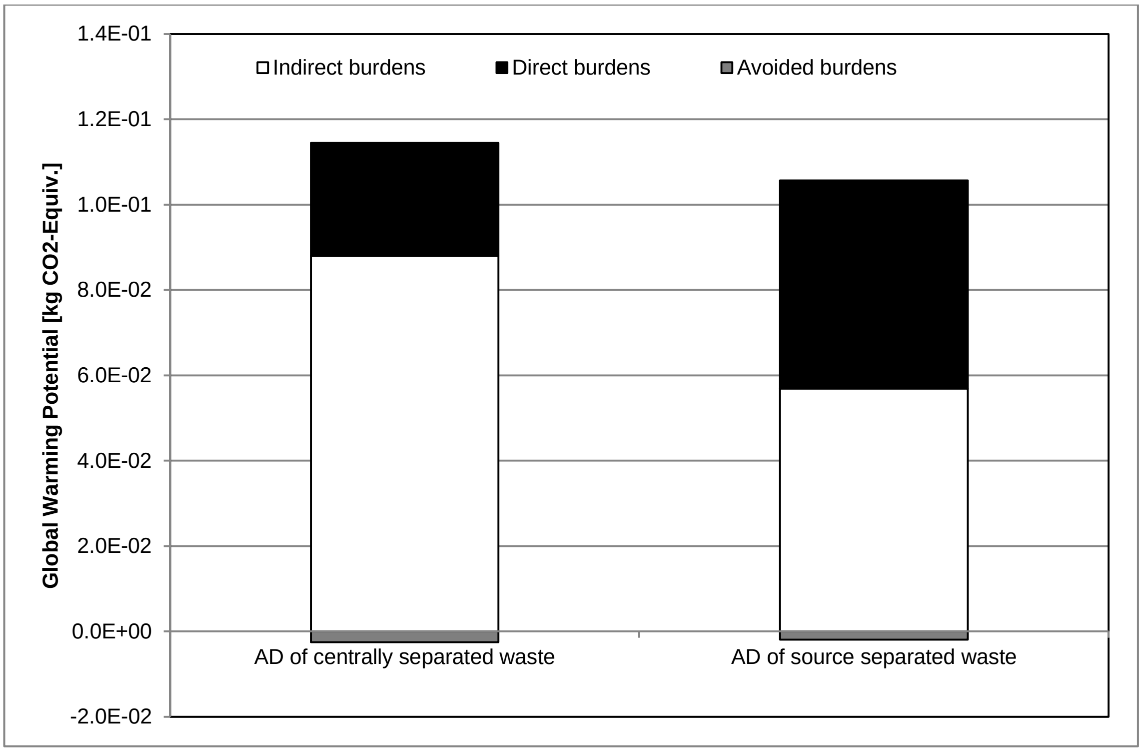


Figure 6.3.7. GWP of the AD processes from centrally separated waste and source separated waste. Indirect, direct and avoided burdens are identified. Results are reported per 1 kg of waste as functional unit.

- *Pre-treatment and digestion*. The pre-treatments and digestion sections of both types of AD determine a positive contribution to all the indicators. In both cases, this section mainly influences the indicators that strongly depend on the electricity consumption (ADP, AP, GWP) because the main environmental burdens are determined by indirect activities. For example, the AP of both processes is mainly due to the electricity consumptions. Conversely, the GWP is also due to the direct methane slips from the digesters accounted in the model. Pre-treatment and digester of the two types of AD are shown to have the same environmental impacts because the correlations to calculate the electricity requirements in the model are based on the amount of biodegradable waste in input (assumed to be the same in the two cases).
- *Upgrading*. Both upgrading processes show a highly avoided ADP (Figure 6.3.6) (in both cases the avoided value offsets the positive contributions) thanks to the avoided burdens allocated to the methane injected into the grid. However, the avoided ADP allocated to the AD of source separated waste is 83% lower than the ADP allocated to the AD of centrally separated waste (this is due to the difference in methane yield, see Table 6.3.1). The other indicators do not show any negative impact allocated to the upgrading processes because

the positive burdens due to the energy consumptions offset the negative values. The upgrading of the AD of source separated waste shows an AP 85% higher than that of an equivalent process operating on centrally separated waste: this is due to the higher yield in methane that determines also the higher energy consumption.

The burdens allocated to the digestate use are always positive (except for the FAETP of the source separated process).

- *Digestate use – source separated waste*. In the AD model of source separated waste, part of the nutrient content of the digestate is assumed to be lost after the spreading of the organic fertilizer on the ground. The avoided burdens of the digestate use are calculated as the difference of the positive burdens due to the application of the organic fertilizer to the soils (emissions due to the leaching, evaporation, run off, etc.) and the avoided burdens allocated to the substitution of the chemical fertilizers. Leaching of N into the soils, evaporation and run off constitute heavily polluting emission of nutrients to the environment and this is the main driver for the EP. For this indicator, the emissions of the organic fertilizer after spreading, are higher than the avoided burden allocated to the substitution of chemical fertilizers. The emissions occur also in the case where chemical fertilizers are used but in the LCA model the difference between the emission due to the organic fertilizer and the chemical fertilizer are included. The opposite result is shown for the FAETP; the avoided burdens allocated to chemical fertilizers offset the impact due to the emissions to the environment. Hence, for this indicator the weight of the substitution of chemical fertiliser is higher.
- *Digestate use - centrally separated waste.* In the case of AD applied to centrally separated waste the digestate is assumed to be co-incinerated with other waste. A mass balance indicates that the mass of nutrients in input to the incineration process needs to be found in the outputs as either emission to air or as ash. Therefore, those nutrients reach the environment and equally contribute to the EP. The same explanation can be applied to the ODP whereas the GWP is mainly due to the incineration of the fibres.
- *GWP- direct, indirect and avoided contributions.* Figure 6.3.7 shows the GWP of the two AD processes (from source separated and centrally separated waste, not including the processes that are upstream the biodegradable waste pre-treatment) and specify the contributions coming from direct, indirect and

avoided activities. The process of AD from source separated waste determines a lower impact than the process of AD from centrally separated waste because of the higher yield in methane- 1.04 10^{-1} and 1.12 10^{-1} kg of CO2 Eq., respectively. However, the direct burden contributes around 47% to the total GWP, whereas for the process of AD from centrally separated, this percentage decreases to the 24%. This disparity in the results is due to higher biogas yield and therefore higher direct leakage of CH_4 from the upgrading and digestion. The total avoided burdens allocated to the AD of source separated waste are smaller than the avoided burdens allocated to the other process even if the yield in methane of the latter is lower- -1.94 10^{-3} and -2.56 10^{-3} kg of CO2 Eq., respectively. The reason for this is that the avoided burdens of the AD from source separated waste does not only include the production of methane but also the substitution of chemical fertilizer and the emissions due to the evaporation, leaching and run off of part of the digestate nutrients. The higher indirect burdens of the AD of centrally separated waste are due to the higher parasitic loads allocated to the pre-treatment and digestion.

The electricity consumption for digestate dewatering in the AD process from source separated waste determines a negligible environmental burden to all indicators.

6.3.3.4 Accounting for the biogenic carbon content of waste

The most important results are analysed also considering the biogenic CO_2 emissions neutral for GWP according to the biogenic carbon content of the waste. The combustion of the upgraded methane injected into the grid is taken in consideration to calculate the biotic GWPs.

Figure 6.3.8a reports the GWP biotic for scenario 1-3-5 in case the functional unit is 1 kg of MSW. The trend of the results does not change whether the biogenic carbon is included or excluded because given 1 kg of the same MSW for all processes, the environmental burden of all processes is decreased by the same amount.

The same is not valid for Figure 6.3.8b where the results are reported according to the production of 1 MJ of upgraded methane. The feedstock of all three processes is the same but different amounts of MSW are treated (according to different yields in methane production). In this case the process with the lower efficiency in biogas production (that is the process that treats the higher amount of MSW) shows the lowest GWP because the amount of biogenic carbon in the feedstock is the highest.

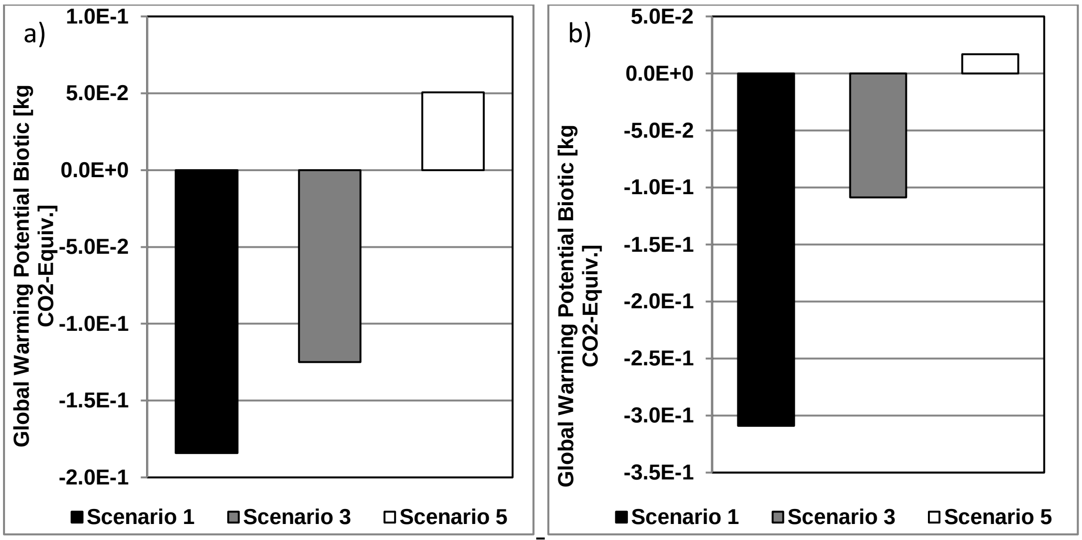


Figure 6.3.8. Biotic GWP. a) The functional unit is 1 kg of MSW. b) The functional unit is 1 MJ of upgraded methane.

6.3.3.5 UK future energy mixes

The energy supply (in particular electricity supply), is reported to strongly affect the results of a LCA analysis [269,448] and hence, a study of the environmental burdens of the processes analysed have been performed according to different energy technologies for indirect and avoided activities. Energy mixes have been considered as this is an attributional analysis; in consequential analysis, the use of marginal supplies of renewable energy would show the same trend of the results as that specified in Figure 6.3.9.

The aim of this analysis is to compare scenarios 1, 3 and 5 between 2014 and 2035 in the UK, according to the energy (both electricity and gas) mixes predicted by National Grid (four scenarios are studied: : i) Gone Green; ii) Slow Progression; iii) No Progression; iv) Low Carbon Life, see chapter 2). The modelling has been performed for the two different functional units, 1 kg of MSW treated and 1 MJ of methane produced.

In the first analysis, only future electricity mix scenarios have been considered while both electricity and natural gas future mixes have been included in a second analysis. The two cases do not show significantly different results, highlighting how a change in the electricity technology mix determines a higher variation of the results than a change of the natural gas mix. Only the coupled results regarding a change in natural gas mix and electricity mix are reported.

Figure 6.3.9 shows the GWPs of scenarios 1, 3, and 5 till 2035 for the four possibilities analysed by the National Grid, per kg of waste treated. The increase of the share of cleaner energy sources in the electricity mixes, for all four scenarios, determines an increase in the GWP for scenario 1 and 3. This is due to lower avoided burdens allocated to the production of electricity and hence higher total environmental burdens. On the another hand, scenario 5 decreases its environmental burden because of a lower influence of the electricity mix and higher environmental burdens allocated to the production of methane (the natural gas mix increases its environmental burden because of a higher use of LNG and shale gas). The same trend is depicted for all four scenarios but the GWPs of scenarios 1, 3 and 5 converge most closely in the Gone Green and in the Slow Progression cases. High economic growth and support to sustainability assumed in these two scenarios determines these results. For all four scenarios, from the year 2020-2021 the GWPs of all three processes become almost parallel, slowly converging toward the centre. The inversion of the results (between scenarios 5, 1 and 3) is not seen before 2035. The calculated GWP of the electricity grid which would determine an inversion of the results is determined to be 0.1 kg of CO_2 Eq. per kWh of electricity. This can be attained, for example, with a strong increase of the nuclear power in the grid mix, to greater than a 40% share. When the inversion of the results is attained, the GWP impact of producing methane from MSW would be less than the GWP of producing electricity.

Given 1 kg of MSW as functional unit, if the government policies prioritise sustainability within an increased economic growth, the evolving energy mixes determine a change in the environmental burden of the processes analysed.

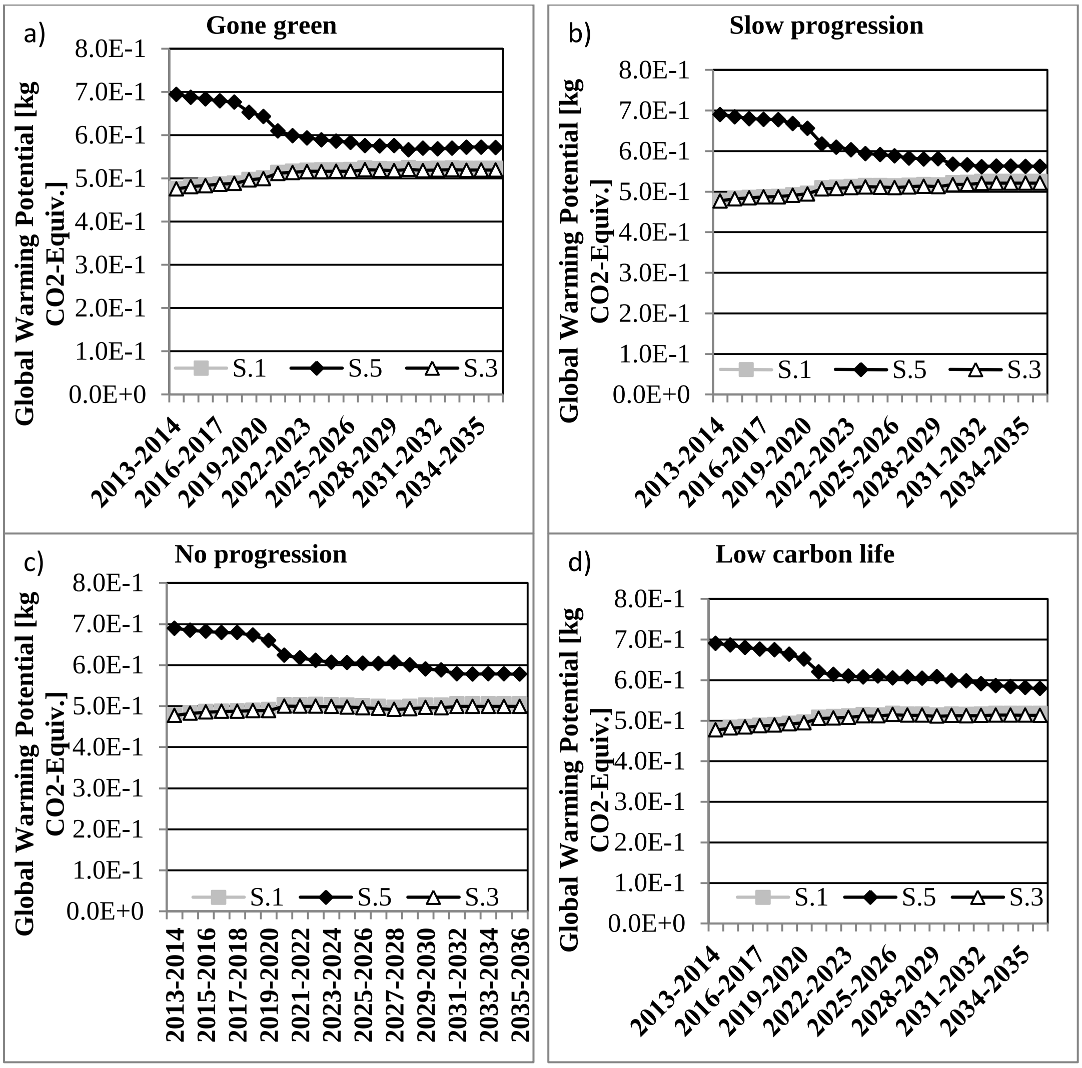


Figure 6.3.9. GWPs of S.1, S.3 and S.5 for future foreseen electricity and natural gas UK mix according to the a) Gone Green scenario; b) Slow Progression scenario; c) No Progression scenario; d) Low Carbon Life scenario. Results are reported per 1 kg of waste as functional unit.

The results of this analysis are also reported according to 1 MJ of methane injected into the grid. Future mixes of natural gas and electricity grid are included in the results of Figure 6.3.10 and Figure 6.3.11; these show the fossil and biotic GWPs, respectively. In this case both types of GWPs are reported because, given different amounts of MSW treated in S.1, S.3, S.5 the two show different trends.

The fossil GWPs of Figure 6.3.10 show the same trend for all four different scenarios analysed. The three processes analysed have almost parallel burdens showing how, in this case, the main contribution is due to the avoided burdens allocated to the methane produced. Fixing 1MJ of methane produced, it means the same avoided burdens for methane production are allocated to all three technologies; the avoided burdens allocated to the electricity have a minor environmental impact on the total score of the results and therefore no significant variation of the results is shown.

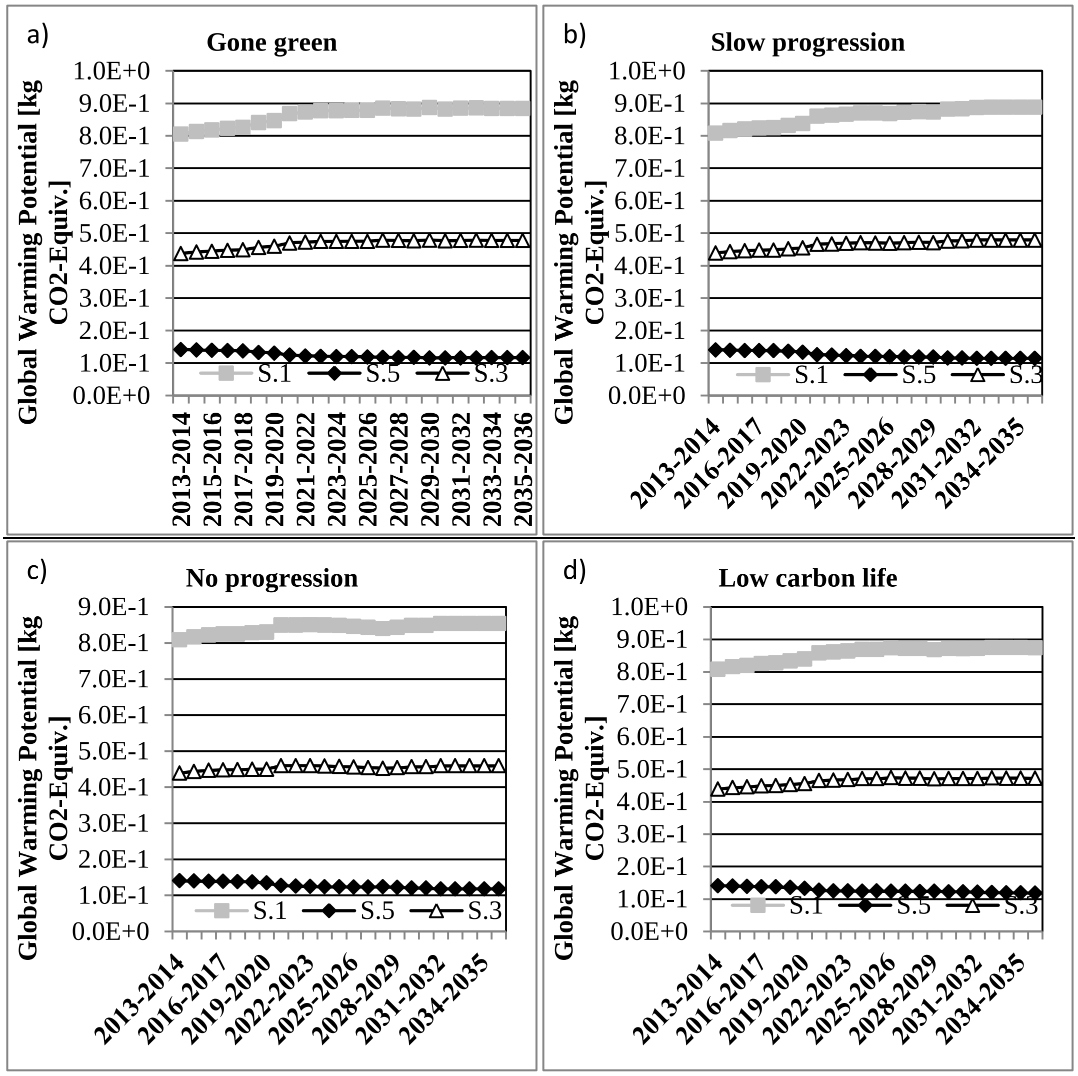


Figure 6.3.10. Fossil GWPs of S.1, S.3 and S.5 for future foreseen electricity and natural gas UK mix according to the a) Gone Green scenario; b) Slow Progression scenario; c) No Progression scenario; d) Low Carbon Life scenario. Results are reported per 1MJ of upgraded methane.

A slightly different trend is shown in Figure 6.3.11 where the biotic GWPs are reported. No significant variation is again shown for the four different scenarios but the results converge slightly. A different preferred technology is shown for this case: the best option for the fossil GWP is S.5 whereas for the biotic GWPs is S.1 as already previously reported for the current energy mix.

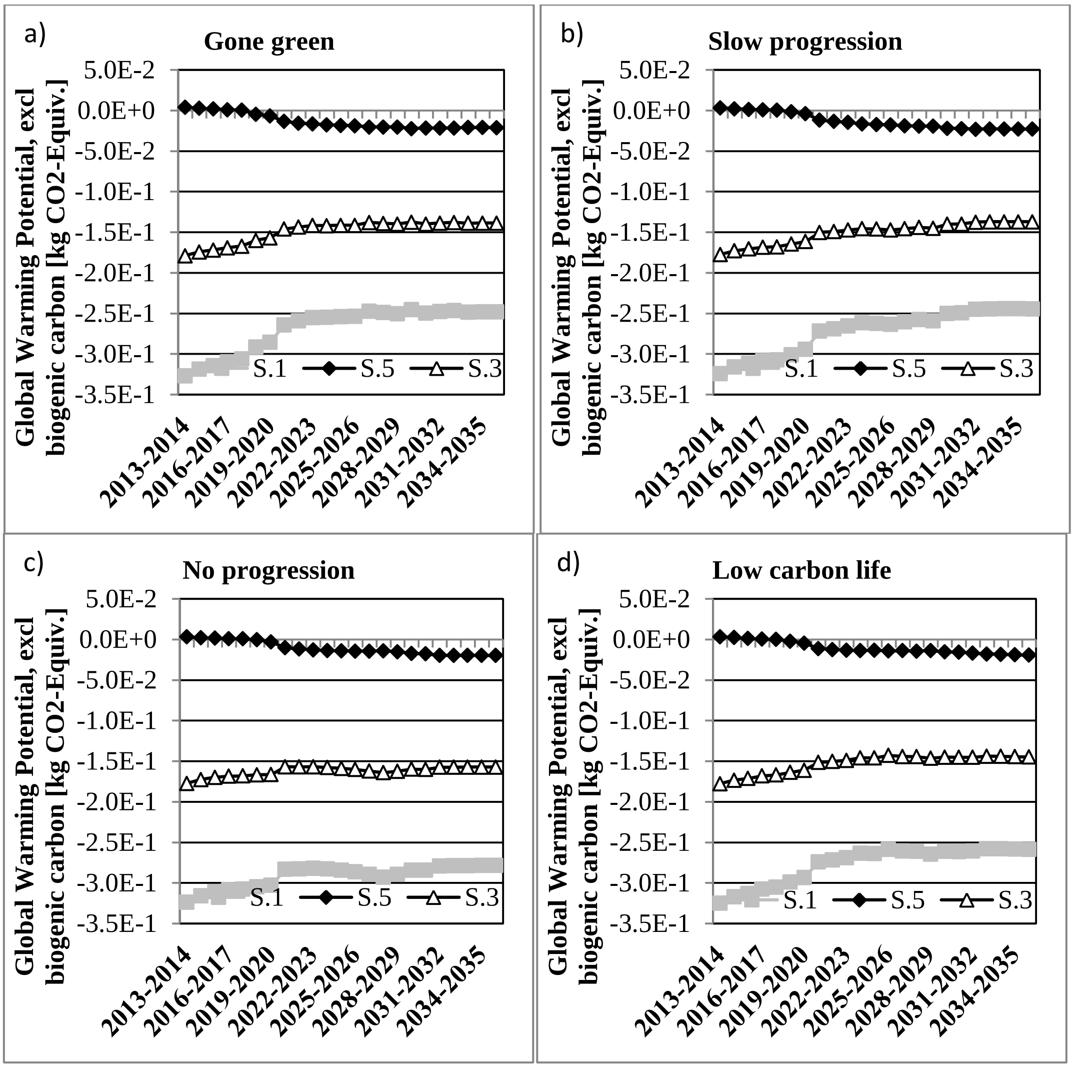


Figure 6.3.11. Biotic GWPs of S.1, S.3 and S.5 for future foreseen electricity and natural gas UK mix according to the a) Gone Green scenario; b) Slow Progression scenario; c) No Progression scenario; d) Low Carbon Life scenario. Results are reported per 1MJ of upgraded methane.

6.3.4 Conclusions

In this section the environmental performances of conventional and advanced treatment technologies of MSW focusing on the Bio-SNG production were analysed. Five scenarios have been identified, the main processes being: Mechanical Treatment associated with Anaerobic Digestion of centrally separated organic waste and landfill/incineration of the residual waste; source separation of food waste with landfill/incineration of residual waste; and a dual stage advanced thermal treatment process. The model for the inventory has been built based on literature and industry data and a complete environmental analysis has been performed. Furthermore, for the 5 scenarios analysed, two different approaches were considered. One was looking at the best environmental technology for treatment of waste, the other instead was focused on the renewable methane production. This was reflected on the choice of the functional unit, 1 kg of MSW and 1 MJ of methane produced, respectively for the two approaches. A unique trend in all the results cannot be identified but each process performs differently depending on the indicators analysed. Avoided burdens for energy production and direct emissions play the major role on the environmental burdens.

When the problem of waste management is approached, for the GWP, it is currently better to produce electricity from waste over bio-methane/Bio-SNG (as a result of the current UK energy mix) but this is bound to change for future energy scenarios. In fact, this work has also analysed the projection of GWP for the processes studied till 2035 accounting for future energy scenarios. Over this period of time it is predicted that there will be a strong decrease in carbon emissions for the electricity mix compared to the natural gas mix. In the context of waste to energy, this will enhance those technologies that produce renewable methane at high efficiency compared technologies that convert waste for electricity.

However, the functional unit was shown to be a key parameter for the overall trend of the results. In fact, when the problem of renewable energy production was tackled (functional unit 1 MJ of methane), the current GWP showed that the best option is the treatment of MSW in a dual stage advanced thermal treatment as a result of a higher efficiency in methane production. This trend is not due to change in the near future.

A hot spot analysis was performed for the AD processes from source separated and centrally separated waste. The pre-treatment and digestion processes determine a positive contribution to all the indicators, showing that no avoided burdens are allocated to them; the main environmental burdens of the pre-treatment and digestion are determined by their energy consumptions. However, the GWP is mainly due to the methane slips from the digester. ADP is the only indicator showing avoided burdens allocated to the two upgrading processes. For the digestate use of AD of source separated waste, the majority of the indicators are shown to be positive (mainly the EP, ODP and AP). This is because once on the soil, the burden due to the run-off, evaporation and leaching of N compounds from the organic fertilizer are higher than the avoided burden allocated to the substitution of chemical fertilizers. Those emissions strongly limit the environmental performance of this process when compared to the advanced thermal treatment of waste.

6.4 General conclusions

This chapter explored the environmental burdens of methane production from waste and analysed advanced thermal and biological technologies. The anaerobic digestion inevitably requires the disposal of the residual waste in thermal technologies as it cannot transform the entire fraction of MSW. This is instead possible for advanced waste to energy technologies and also for conventional mass burn processes that can treat a broader range of feedstock material. Except for the water contamination, the disposal of the residual waste and production of electricity hugely influences the comparison between biological and thermal processes producing methane and hence trade-offs between electricity and methane production have to be identified. Looking at the global warming, in the UK it is currently better to produce electricity from waste over methane from waste. This is strictly related to the current UK electricity mix that is still strongly based on fossil fuels. However, cleaner energy mixes improve the environmental burden of advanced thermal technologies treating waste and producing Bio-SNG over technologies treating waste and producing electricity.

Chapter 7. **Conclusions and future work**

7.1 Conclusions

The energy mix of a specific country evolves according to new research and technology developments, new sources becoming available, economic benefits and legislations. In this thesis the case of the UK has been considered. The evolution of the UK energy mix has been reviewed, historically since the medieval age, to modern times, including future projections.

The energy sector significantly contributes to the total environmental impact and is deemed responsible for the global temperature increase. To ensure the development of an affordable, secure and environmentally friendly energy supply, the UK Government has particularly focused its attention on possible domestic shale gas developments, imports of LNG and development of waste to energy technologies valuing waste as a resource that can significantly contribute to the total energy supply. However, as some of these energy sources and technologies are currently being developed, their contribution to the energy trilemma is currently under question, mainly when compared to conventional energy production. The aim of this thesis was to propose a comprehensive environmental framework using the Life Cycle Thinking approach to analyse the environmental impacts of the energy sources and technologies mentioned above. The approach developed in this thesis is for the UK, but it can be applied to any another country.

The key questions that this thesis has sought to address are:

1. How has the energy mix evolved through history in the UK? What are the characteristics of the current energy mix and the projections for the next 20 years for the electricity and gas supplies? What affects the energy shifts?

2. What are the future challenges for fossil energy and what renewable sources should the UK explore and develop?
   a. Is shale gas a valid environmental alternative in the UK when compared with conventional natural gas supply?
   b. Is LNG a clean fuel for energy supply?
   c. How does LNG compare with shale gas under an environmental perspective?
   d. Electricity from waste: how do advanced thermal treatments of waste compare against conventional waste treatment processes (incineration and landfill)? What type of waste feedstock determines the lowest environmental impacts when treated in advanced thermal processes?
   e. Methane from waste: how do biological and thermal processes compare?
   f. Electricity from waste vs. methane from waste: which alternative prevails as the energy mix evolves?

The main findings of this study are:

1. Coal has been the main component of the UK energy mix until the 90s, when a broader range of energy sources, including nuclear, gas, oil and renewables started developing. The UK is still strongly relying on fossil fuels but International and Local legislations are pushing towards the uptake of cleaner energy sources while ensuring energy security.
2. To meet stable and cleaner energy supply, shale gas, Liquefied Natural Gas and waste have been identified as key energy sources for the future of the UK energy mix.
   a. Shale gas production and distribution causes a greater environmental impact than the current supply of natural gas and this is mainly due to a lower overall gas recovery. However, shale gas determines a lower water use compared to the current gas supply.
   b. The complex and deep processing required to produce LNG and also the ship tanker transport limit the environmental performances of LNG supply to the UK.
   c. LNG has shown higher environmental impacts than shale gas.
   d. Advanced waste-to-electricity technologies treating high calorific value feedstock determine lower environmental burdens of conventional waste treatment processes thanks to higher energy efficiency and metal recovery rate.

e. The disposal of the residual waste not digested in an anaerobic digestion process hugely influences the comparison between biological and thermal waste treatments and it depends on the energy mix.
f. In the UK, it is currently more environmentally friendly to produce electricity over methane from waste, however this trend is projected to be inverted in the near future.

In the subsequent paragraphs, the key findings have been further detailed.

7.1.1 Fossil natural gas

The production of natural gas from fossil resources was discussed in chapter 4.

The first part of the chapter sought the answer to the question '*Is shale gas a valid environmental alternative in the UK when compared to the conventional natural gas supply*?' The environmental impacts of shale gas significantly depend on the procedures adopted during gas production.

Direct disposal of the waste water, produced during fracking operations, into fresh water is banned by law in the UK. However, this thesis analysed what the possible threats are of unwanted spills of flowback water when compared to the environmental impacts of the conventional gas supply to the UK. Improper waste water management can even double the toxicity impacts.

Water use, degradation and consumption are environmental indicators that should be carefully analysed when taking a political decision on shale gas. The water degradation of the conventional natural gas supply to the UK was shown to be even higher than that of shale gas. Conversely, the water withdrawn and used for the shale fracturing process significantly increases the water consumption of shale gas when compared to the water consumption of the gas grid mix. Hence, political investment should support the recycling of flowback water as this solution allows a reduction of the total water consumption associated with shale gas production. In particular, advanced researches to improve the efficiency of the recycling process should be strongly taken into consideration.

Furthermore, policy makers should carefully consider the increased environmental impact of shale gas regarding for example, Acidification Potential (AP), Global Warming Potential (GWP) and Photochemical Ozone Layer Creation Potential (POCP), due to the lower total amount of recoverable gas from shale gas wells when compared to conventional offshore wells. In fact, the EUR of shale wells, together with the flowback ratio and flowback disposal method discussed earlier, was shown to have the greatest impact on the total environmental burdens of shale gas exploitation: the low EUR forces the drilling and exploitation of a high number of wells.

The second part of chapter 4 answered the question '*Is LNG a clean fuel for energy supply?*' The environmental impacts of LNG delivery to the UK were explored. The analysis showed how the complex and deep processing required to produce the LNG and also the ship tanker transport can limit the environmental performances of this fossil energy supply, mainly when this is compared to shale gas production. This outcome clearly challenges the claim of LNG as a clean fuel.

When comparing shale gas production and LNG supply, it was shown that the environmental impacts of natural gas delivery to the UK are dictated by a well's total recovery, processing of the raw gas, and transport; they are significantly reduced when high well recovery are exploited, light processing is required and transport is reduced. Therefore, the supply of natural gas from close wells with high gas recovery is always to be preferred to the import of natural gas requiring deep processing.

7.1.2 Waste-to-electricity

Advanced waste to energy technologies are currently supported by the UK government because they contribute to alleviating the waste disposal problem and at the same time they produce renewable energy. In this context, Advanced Plasma Power (APP) has developed a two-stage thermal process (Gasplasma) in which the raw syngas and the ash generated in a conventional bubbling fluid bed gasifier are further treated in a plasma converter. This acts as a cleaning stage and cracks and reforms the tar and char species in order to provide a refined syngas suitable for energy production. Inorganic particulates are converted into a stable vitrified product that can be recycled as ceramic glass or road paving material. The syngas produced after refining is suitable for a wide variety of applications, such as production of electricity, methane, higher hydrocarbons and hydrogen. This thesis explored the generation of electricity and methane using the Gasplasma technology, on the basis of the current pilot and semi-Commercial plant developments.

Chapter 5 analysed the life cycle of the dual stage advanced thermal process for electricity production and answered the questions '*Electricity from waste: how do advanced thermal treatments of waste compare against conventional waste treatment processes (incineration and landfill)? What type of waste feedstock determines the lowest environmental impacts when treated in advanced thermal processes?*'

Firstly, the hotspot analysis of the gasification and plasma process was performed for different feedstocks, such as biomass, MSWs, Commercial and Industrial Waste, SRF and RDF. Overall, the environmental impact changed on the basis of the characteristics and composition of the feedstock treated and it was not possible to identify a single waste stream better than any other for all the impact categories considered. However, the treatment of RDF showed the lowest impact in terms of GWP, mainly thanks to its higher energy density.

The hot spot analysis of the dual stage gasification and plasma process showed that the carbon dioxide in the flue gas is the major contributor to the total GWP. The electricity supplied to the plasma torch and the production of oxygen for the gasifier and plasma did not contribute significantly to the GWP, despite their main role in the tar removal process. Conversely, these processes together with the emissions at the stack significantly increased the AP. It was shown how the chemicals required for the syngas refining are responsible for major impacts in water and toxicities categories.

To identify the role of gasification in the waste management sector, the dual stage process was compared to the more conventional waste disposal options of incineration and engineered landfill. Two processes of incineration were considered to represent the state-of-art and recent developments in the mass burn technologies.

The treatment of MSW in the two-stage gasification and plasma process had substantially lower GWP and AP impacts than landfilling. This was primarily due to avoiding emissions of methane from landfilled organic material. Compared to incineration, the two-stage process showed a much lower acidification impact, mainly due to the higher SO_2 emissions when incineration technology is used and a lower GWP due to higher energy efficiency. Moreover, the amount of metals recovered in the dual stage process can also play an important role in determining the overall environmental impacts mainly when looking at the toxicities and the remaining environmental impacts.

The potential of the dual stage gasification and plasma process was compared to that of other advanced waste to energy processes. In particular on the basis of latest developments in the field of advanced waste management, i) pyrolysis and combustion, ii) gasification and combustion, iii) one stage high temperature gasification with direct melting and combustion were also considered. The gasification and plasma process showed a better GWP than the three other thermochemical processes considered thanks to its higher net electrical efficiency. The same result was also shown for the other environmental impacts.

Overall, chapter 5 showed how the energy efficiency is key for developing processes and is strictly linked to lower environmental impacts.

7.1.3 Waste-to-methane

The consistent quality syngas produced by means of the Gasplasma technology can also be used for Bio-SNG production. The possibility was explored in chapter 6 that aimed to answer the question *'Methane from waste: how do biological and thermal processes compare?'*

Firstly, an attributional analysis of the dual stage process for Bio-SNG production was performed. It was shown how the production of Bio-SNG requires additional chemical conversion and upgrading compared to the electricity production only. These operations cause additional pressure on the total environmental impact. In particular, energy requirements for gas heating and compression need to be carefully considered when looking at the overall environmental performance and energy efficiency of the process. The indicators whose impact was not strongly dependent on the production process of the electricity requirements (Fresh Water Aquatic Ecotoxicity Potential, Ozone Depletion Potential, Terrestric Ecotoxicity Potential, etc.) showed the same result whether electricity only or Bio-SNG was produced from the advanced dual stage process.

The gases emitted to the environment during upgrading caused the main contribution to the GWP together with the indirect burdens due to production of the electricity required through the process. Therefore, the environmental impacts of Bio-SNG production were analysed also considering CCS. The results obtained for this particular technology can be extended to any other energy system that involves the production of pure CO_2. CCS helps in reducing the GWP of the process under study but at the same time all other indicators become worse because of the increased energy requirements. CCS is currently supported at political level, however, policy makers should carefully consider the decrease in the overall energy efficiency of the process.

Chapter 6 also analysed the environmental burdens of the anaerobic digestion process treating organic waste and producing bio-methane over advanced dual stage thermal technologies. To account for the entire fraction of municipal waste and to allow a direct and fair comparison between thermal and biological technologies, integrated systems were analysed. This included i) mechanical pre-treatment of Municipal Solid Waste (MSW) associated with the anaerobic digestion (AD) of the organic fraction and landfill/incineration of residual waste; ii) anaerobic digestion of source separated waste and landfill/incineration of residual waste; iii) advanced gasification-plasma technology for Bio-SNG production.

This study showed how the emissions of nitrogen, phosphorus and potassium components that contribute to the eutrophication potential (EP) and the fresh water aquatic eco-toxicity potential (FAETP) need to be considered when financial support is given to renewable methane production from waste. These emissions are significantly lower when waste is treated in advanced thermal technologies producing Bio-SNG.

Furthermore, it was shown how the anaerobic digestion inevitably requires the disposal of the residual waste in thermal technologies as it cannot transform the entire fraction of MSW. This is instead possible for advanced waste to energy technologies and also for conventional mass burn processes that can treat a broader range of feedstock material. Except for the water contamination, the disposal of the residual waste hugely influences the comparison between biological and thermal processes and hence trade-off between electricity and methane production has to be identified as discussed in the next paragraph.

7.1.4 Electricity vs. methane from waste treatment

Waste is a valuable resource for renewable energy production, both methane and electricity. Chapter 6 analysed both types of energy and answered the question *‘Electricity from waste vs. methane from waste: which alternative prevails as the energy mix evolves?’* The answer was shown to depend on the perspective approached, the time of the analysis, the country and the market in which the technology is developing. In LCA words, this means that the allocation of avoided burdens represents a significant part of the methodology that can totally drive the environmental results.

In the first part of the chapter, the dual stage gasification and plasma technology producing electricity was compared to the dual stage gasification and plasma technology producing Bio-SNG. Looking at global warming, in the UK it is currently better to produce electricity from waste over methane from waste. This is strictly related to the current UK electricity mix that is still strongly based on fossil fuels as analysed in chapter 2. However, cleaner energy mixes improve the environmental burden of advanced thermal technologies treating waste and producing Bio-SNG over advanced technologies treating waste and producing electricity. This was shown accounting for future UK energy mixes evolving towards cleaner technologies (the evolving mixes were reviewed in chapter 2) and also for alternative locations to the UK, where the actual energy mix is cleaner. Policy makers of the Northern European countries where the electricity mix is already strongly decarbonised should now financially support Bio-SNG production over electricity production from waste.

In the second part of chapter 6, the production of Bio-SNG from gasification-plasma was compared to the production of bio-methane from AD (the treatment of the biodegradable fraction in AD was considered in association with the treatment of the residual waste in thermal technologies producing electricity) and similar conclusions were drawn. Firstly, a waste management perspective was taken, in which a functional unit of 1 kg of waste to be disposed was used. Secondly, an energy production perspective, in which a functional unit of 1 MJ of renewable methane produced was considered. The results obtained from the waste management perspective demonstrated that when the current energy mix is used in the analysis (i.e. strongly based on fossil resources), the global warming potential (GWP) shows that processes with higher electric efficiency - thermal processes for electricity production - determine lower environmental burdens. However, as the electricity mix in the UK becomes less carbon intensive and the natural gas mix increases the carbon intensity, processes with higher methane yield - thermal technologies over biological technologies - were shown to achieve a lower global warming impact within the next 20 years, when the disposal of 1 kg of waste was used as functional unit. Conversely, if the attention of policy makers is pushed towards the challenge of renewable energy production (functional unit of 1 MJ of renewable methane produced), the GWP showed that more efficient technologies for renewable methane production - thermal technologies over biological technologies - cause a lower environmental burden for both the current and future energy mix.

Overall, the results of this study give credit to the current UK policy that is strongly supporting electricity production from waste. However, in about 8-10 years’ time, policy makers need to consider new legislations to push for methane production over electricity as this was shown to determine the lowest environmental impact for future energy mixes.

7.2 Future work

The technologies analysed in this thesis are all cutting edge technologies. The LCA models were all based on literature data, experimental and pilot plant runs data and industrial reports. This study has predicted the environmental impacts of these developing technologies before their full development. This study should be followed by a ‘post development’ analysis that considers the actual environmental burdens of fully built and operating plants in steady state conditions. In particular, for the analysis regarding the energy production systems from waste, new inventory data (mass and energy balance) should be based on industrial scale operating processes. The environmental burdens of shale gas production should be assessed based on data collected on site during actual gas production and the same is also valid for the analysis on LNG.

As previously reported, three comprehensive integrated aspects identify the sustainable approach: environmental, economic and social. This study has tackled only the environmental aspects of the developing energy sources and technologies in the UK but the other two aspects are equally important and need to be addressed according to the framework proposed in this study. In particular, the outcomes of the economic analysis will clearly have an impact as huge as that of the environmental analysis on the actual feasibility and development of the systems analysed in this thesis. The three analysis (LCA, LCC and SLCA) of the technologies studied in this thesis will be joined together to give a unique comprehensive assessment. The analysis of the three aspects of sustainability of key developing energy technologies have never been attempted before in the literature.

Appendices

Appendix to chapter 4.

A4.2 Shale gas: a life cycle perspective for UK production

A4.2-1. Modelling assumptions and system boundary

The following stages of the shale gas production process are analysed in order to build the hot spot analysis:

1. *Well site exploration and investigation.*
2. *Well pad and road preparation and construction.*
3. *Well vertical drilling.* Production of materials needed for drilling; transport of materials; energy required during drilling and emissions from machinery; emissions during drilling; casing and cementing; disposal of drilling wastes.
4. *Hydraulic fracturing of the well*. Horizontal drilling; production and transport of water, chemicals and sand needed for fracturing; energy used during the hydraulic fracturing and emissions from machinery; disposal of wastes.
5. *Well Completion.* Energy and materials required; disposal of flowback and produced water from the well; emissions of natural gas during well completion, workovers, unloadings; re-fracturing.
6. *Production.* Processing and cleaning.
7. *Pipe construction and transmission.*
8. *Post production phase.* Decommissioning, plugging and removing of equipment.

Avoided burdens have also been considered for the production of valuable hydrocarbon by-products other than natural gas. As widely reported in literature [130,333,343], it is assumed that extraction, processing and distribution of shale gas involve exactly the same processes as onshore extraction of conventional gas, but operations associated with hydraulic fracturing must be added. Therefore, two models have been built: the first (identified as the common operations model) accounts for the extraction of conventional gas and includes all the common processes between conventional and unconventional extraction: gas field exploration, natural gas production, purification, long distance transport and regional distribution. The second model (identified as the hydraulic fracturing model) includes all the processes specific to shale gas: horizontal drilling, fracking of the shale rocks, flowback disposal and handling of emissions associated with hydraulic fracturing. The emissions in the hydraulic fracturing model represent the difference in emissions between production of shale gas and conventional gas.

A4.2-1.1 Common operations model

In the common operations model it was not possible to differentiate between exploration, production (drilling and extraction) and purification. Hence, aggregate data for those processes are used [345] according to indirect, direct and avoided burdens UK site specific.

The gas distribution systems included in the analysis (long distance distribution at high pressure, regional distribution at high pressure and local distribution at low pressure) reflect the three gas pressure levels that can be found in the UK distribution system: i) national transmission system; ii) distribution network; iii) local distribution [370].

A4.2-1.2 Hydraulic fracturing model

A4.2-1.2.1 Horizontal drilling

Part of the drilling process (the vertical drilling) can generally be considered the same whether a conventional or an unconventional well are drilled. Therefore, the vertical drilling is included in the common operations model. However, the LCI model also accounts for the materials used during the directional drilling that is typical of shale gas recovery (conventional wells rarely require horizontal drilling).

The energy and materials usually required for both shale gas extraction and conventional gas extraction are reported in literature [121,325,349]. The difference between the two represents the material and energy required for horizontal drilling that is included in the LCI model built for this study. The emissions from drilling machineries and the emissions due to the fracturing machineries are modelled according to the values reported in Table A4.2-1.

The amount of materials (steel, water, betonite, cement etc.) included in this analysis for both vertical and horizontal drilling reflect the dimensions of a typical UK shale plays borehole as reported in Stamford *et al.* [343].

A4.2-1.2.2 Fracturing the shale formation

The amount of water needed for the rocks fracturing process (~15000 m^3) is calculated as the average of the values found in literature [129,130,325,333–335,341,343,346]. The same procedures was adopted for the amount of sand used for hydraulic fracturing [129,325,334,335,341,343]. The values reviewed for both water and sand were consistent and did not change significantly. Half of the fresh water used for the fracturing process is sourced from surface water and the other half from municipal water plants (as in [129]).

The amount of additives contained in the fracturing fluid is taken from Jiang *et al.* [129]. This represents a good assumption also for the UK situation where the chemicals are reported to be ~0.05% of the fracturing fluid; however, the production of the fracturing chemical is not expected to strongly influence the results.

The energy required to mix the water, sand and chemicals on the well site to produce the fracking fluid is considered negligible.

In this study, the amount of diesel consumed by the pumps during hydraulic fracturing is calculated based on the hydraulic horsepower delivered (12250), number of fracturing operations (15), hours of fracturing for each operation (2h) and the consumption of diesel per hydraulic horsepower delivered (250 g of diesel/kWh) according to the literature [130].

The emissions due to the use of diesel during hydraulic fracturing and horizontal drilling are included in the model and these are based on the amount of diesel consumed. The machines used for drilling and fracturing, such as diesel engines are probably the same, as well as the air pollutants emitted by these machines [347]. The emission of air pollutants from stationary diesel engines used for drilling and hydraulic fracturing are reported in Table A4.2-1according to literature [326,347]. For CO_2 emissions the calculations are based on the carbon content of diesel fuel. A constant relation of 3.175 kg CO_2 emitted/kg fuel consumed is assumed according to the diesel carbon content of 0.866 kg of C/ kg of fuel [310].

Typical emissions of air pollutants from stationary diesel engines used

for drilling and hydraulic fracturing		
	g/KWH mech.	**g/KWH diesel**
	Emissions per engine mechanical output	**Emissions per engine fuel input**
SO2	0.767	0.253
NOx	10.568	3.487
PM	0.881	0.291
CO	2.29	0.756
NMVOC	0.033	0.011

Table A4.2-1. Typical emissions of air pollutants from stationary diesel engines used for drilling and hydraulic fracturing [326,347].

A4.2-1.2.3 Flowback disposal

A sensitivity analysis on the fraction of flowback water produced (that is quite variable from well to well) and different disposal methods is performed. Four flowback disposal options have been identified: i) flow back recycling [128]; ii) flowback injection in deep well of Class II [118,129]; iii) direct disposal to fresh water [128]; iv) disposal to an industrial waste treatment plant [130].

When modelling the different scenarios later reported, the energy requirements of the different disposal options have been considered. For flowback recycling the amount of energy required to reprocess flowback water is assumed to be negligible compared to the total energy requirement of the extraction process [128]. The allocation of avoided burdens to the recycling of flowback water is taken into account in the model as a decrease of materials required for the fracking fluids of another well. The electricity requirement to run an electric pump for the injection of one kg of water in Class II wells has been calculated according to Clark *et al.* [121]. The electricity requirements for the treatment of flowback water in industrial plants are reported in Stephenson *et al.* [130]; the treatment involves reverse osmosis and evaporation or freeze-thaw evaporation, as described in Thomas [471]. The burden due to the production of the electricity required for the freeze-thaw evaporation process is added to the burden of an industrial waste water treatment plant as reported in GaBi database [310]. The direct disposal of flowback water to surface water is assumed not to require any further energy. In this case, the composition of flowback water [128] is considered for fresh water discharges.

A4.2-1.2.4 Completion and workover emissions

The emissions of natural gas associated with well completion, workover, well unloadings, 2 re-fracturing jobs, well equipment, transmission and storage and distribution are considered in the LCI model. The emissions due to unloading, well equipment, transmission, storage and distribution are considered to be the same for conventional and unconventional extraction [130,349] and hence those are included in the common operations model according to the Ecoinvent database [345]. In contrast, the potential emissions (the volume of natural gas that may leak, i.e. the potential emissions, not the actual emissions) due to well completion and workover are higher in the case of shale gas extraction because of the hydraulic fracturing phase and well maintenance.

The surplus of emissions due to the hydraulic fracturing process is included in the hydraulic fracturing model according to the values reported in literature [337,338,349,350]; re-fracturing of wells is also included in the assessment. The emissions reported in some literature work [349] includes the use of flaring and green technologies but in the first instance those are ignored in our model in order to be able to calculate the amount of potential emission associated with shale gas extraction. Then, in the sensitivity analysis different possibilities to reduce these potential emissions are explored.

As previously reported, completion of shale wells and workovers usually involves the hydraulic fracturing process and this can result in significant releases of natural gas and emission to the atmosphere. The handling method of the potential emission during completion and workovers is reported to have a dramatic impact on the carbon footprint of the shale gas production process [122]. Venting of natural gas emissions would significantly increase the carbon footprint of the process; alternative more challenging handling methods such as flaring or capture and injection in the grid decreases the potential warming of the process.

Reduced emissions completions (RECs) – also known as reduced flaring completions or green completions – is a term used to describe an alternate practice that captures gas produced during well completions and well workovers following hydraulic fracturing. Portable equipment is brought on site to separate the gas from the solids and liquids produced during the high-rate flowback, and produce gas that can be delivered into the sales pipeline [132]. Flaring is another option to handle the potential emissions but it only eliminates methane and other hydrocarbons contained in the natural gas; this is not always the preferred option as it does not avoid the emission of other polluting compounds (such as SOx, Nox, PM and CO).

A sensitivity analysis is performed on the fraction of potential emissions captured, flared and vented to assess the influence of the emission handling methods on the total environmental burden. Starting from the potential emissions released during well completion and workovers and the values of the parameters regarding the percent of potential emissions sent to flaring or to green technologies, the following equations are used to calculate the real emissions to the atmosphere:

Mass balance on methane

$$m_{CH4}^{OUT} = m_{CH4}^{IN} - m_{CH4}^{IN} x^F e^F - m_{CH4}^{IN} x^C e^C \qquad \text{eq.1}$$

Mass balance on ethane

$$m_{C2H6}^{OUT} = m_{C2H6}^{IN} - m_{C2H6}^{IN} x^F e^F - m_{C2H6}^{IN} x^C e^C \qquad \text{eq.2}$$

Mass balance on propane

$$m_{C3H8}^{OUT} = m_{C3H8}^{IN} - m_{C3H8}^{IN} x^F e^F - m_{C3H8}^{IN} x^C e^C \qquad \text{eq.3}$$

Mass balance on nitrogen

$$m_{N2}^{OUT} = m_{N2}^{IN} - m_{N2}^{IN} x^C e^C \qquad \text{eq.4}$$

Mass balance on CO_2

$$m_{CO2}^{OUT} = m_{CO2}^{IN} + \frac{m_{CH4}^{IN} x^F e^F}{MW_{CH4}} 44 + \frac{m_{C2H6}^{IN} x^F e^F}{MW_{C2H6}} 2 * MW_{CO2} + \frac{m_{C3H8}^{IN} x^F e^F}{MW_{C3H8}} 3 * MW_{CO2} \qquad \text{eq.5}$$

Where: m is the mass, x is the fraction of gas, F indicates flaring, C indicates captured, e is the efficiency of flaring or capture, IN indicates the potential emissions, OUT indicates the emissions released to the atmosphere and MW indicates the molecular weight.

The flaring efficiency is assumed to be 98% [122,129,130,330,335]. In the scenarios where part of the completion and workover emissions are assumed to be captured in green technologies, the capturing efficiency of those devices is assumed to be 90% according to literature [333,472] (90% of emissions from well completion are captured and utilised and the remaining 10% are vented).

A4.2-1.2.5 Transport

Road transport is widely reported to be one of the drawbacks associated with the production of natural gas from shale rocks. Air pollution, traffic and noise, are only a few of the main problems associated with massive road transport of fracturing water, sand and other materials.

The transport of materials needed for fracturing the rocks and for disposal of waste water are included in the LCI of the hydraulic fracturing model. A sensitivity analysis is then performed on the transport distances to explore the influence of these parameters on the total environmental impact of shale gas production. Return journeys are also considered.

The transport of material needed for horizontal drilling is considered negligible as this amount of material is small compared to the amount of materials required for fracking.

Municipal water is piped to site and surface fresh water is transported by truck prior to the fracturing operations. All the other materials needed on the well site are also trucked to the well site. The sand quarry is assumed to be close enough that no rail transport is needed [473]. In the case of flow back injection in class II wells, the waste water is transported by truck to the Southern North Sea where it is injected in gas fields. Recycled flowback water is transported only 10 km away as other shale wells are usually close to the well under study.

A4.2-1.2.6 Estimate ultimate recovery and processed natural gas

The EUR is a key parameter for the LCA model as a change of its value is reported to determine a significant variation in the results [122,335]. This is because the burdens of the production process are calculated according to a defined amount of gas (the functional unit) but primary inventory data of the gas production process are related to the gas extraction per well and do not depend on the productivity of the well. Hence, the inventory data calculated per functional unit are strictly related to the EUR and this makes the results very sensitive to this parameter. To contextualize the process of hydraulic fracturing in the UK, the US EUR is not considered suitable for this analysis, and the per well data are divided by the estimated UK amount of processed gas (strictly related to the EUR) that could be commercially produced in UK shale plays [341]. The raw and processed shale gas composition is reported in DECC [341]; usually, shale gas has the same composition as conventional gas [130,330]. The processed gas is assumed to be 100% methane pure with a calorific value of 52 MJ/kg (40MJ/m^3) and a density of 0.76 kg/m^3 [341].

The sensitivity of the results on the EUR is assessed by changing the value of this parameter and at the same time keeping fixed the density and composition of the gas produced.

Compound	Raw shale gas composition [% v/v]	Raw shale gas composition [% w/w]	Processed gas composition [%]
CH4	86	74.85136	100
CO2	3	7.180508	
C2H6	3	4.907225	
C3H8	1	2.398943	
N2	7	10.66197	

Table A4.2-1. Shale gas composition.

A4.2-1.3 UK conventional gas grid mix

The UK conventional supply of natural gas relies on the offshore extraction from the British North sea (53%) and on the imports from foreign countries, such as Norway (offshore extraction, 22%) and Qatar (Liquefied Natural Gas, 13%). The current UK gas grid mix is considered according to the GaBi database [310]. The whole supply chain of natural gas (production, processing, transport and distribution at low pressure to the consumer) is included in the LCI model.

A4.2-2 Results

A4.2-2.1 Degradative water use

Independently from the variation of the sensitivity analysis, the degradation of fresh water in shale gas extraction is almost completely due to the model of hydraulic fracturing (including horizontal drilling, fracturing of the shale rocks, flowback disposal and emissions) as shown in Figure A4.2-1a. The amount of degradate water released to rivers depends mainly on the flowback ratio and EUR, Figure A4.2-1b. According to the hot spot analysis (Figure A4.2-1b), degradative use of water is mainly due to the fracturing of shale rocks; the burden of this process is constant for all the scenarios except for S.15 and S.16 that explore different EUR. Flowback disposal to watershed is a source of water degradation in relation to the flowback ratio and disposal method.

The detailed hot spot of the fracturing rocks process is shown in Figure A4.2-1c. More than 50% of fresh water degradation is due to the excavation and processing of sand used in fracturing fluids. Frack sand must be of uniform size and shape and to achieve this, a deep processing is needed [359]. The processing plants wash, dry, sort, and store the sand and waste water is produced. This explains the indirect burden associated with the process of sand mining and processing. 25% of the water degradation in the fracturing process is associated with the production of fracturing chemicals.

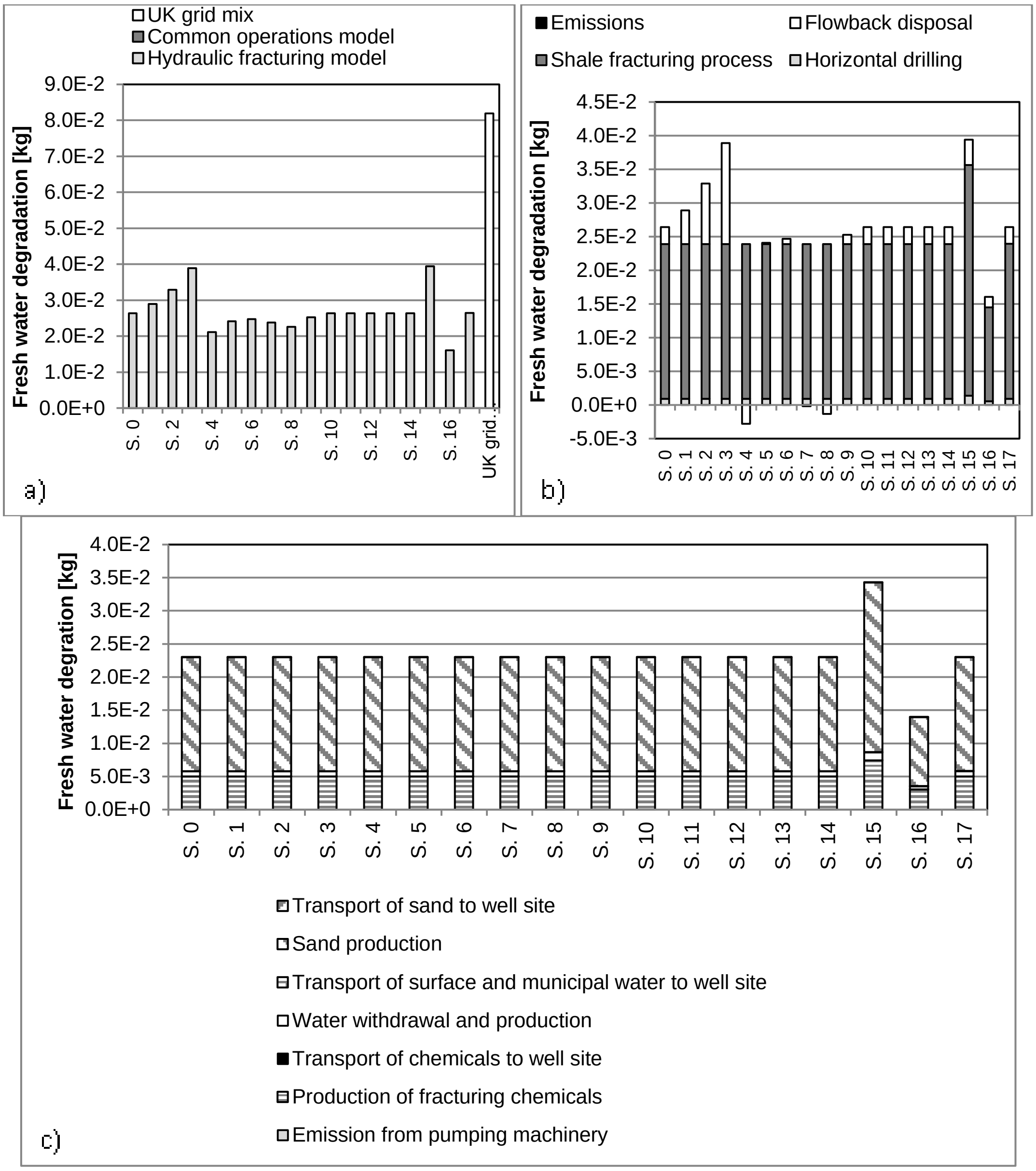


Figure A.4.2-1. a) Water degradation associated with the overall shale gas model. b) Hot spot analysis: water degradation associated with the hydraulic fracturing model. c) Hot spot analysis: water degradation associated with the hydraulic fracturing of shale formations.

A4.3 Liquefied natural gas for the UK: a life cycle assessment

A4.3-1 Modelling assumptions

A4.3-1.1 Sweetening and liquefaction process

The sweetening process has been modelled according to the data reported in Ecoinvent database [345]. The processing plant eliminates sulphur compounds to 99.99% from gas. Sour gas with less of 1% H_2S (lean gas) is treated with the Purisol process (physically), sour gas with more sulphur is treated chemically. The solutions used for the gas scrubbing are regenerated and warmed up to 130 °C. By this process a gas mix of H_2S and CO_2 is released. Subsequently, H_2S is transformed in elementary sulphur and water in a thermal process by 1000-2000°C and in a three-stage catalytic part.

The energy self-consumption rate for natural gas liquefaction is calculated follows [314]:

$$\begin{aligned}&Energy\ self\ consumption\ rate\ [\%]\\&\qquad=\frac{Fuel\ gas\ consumption\ in\ the\ liquefaction\ plant\ [MJ]}{Gas\ input\ in\ the\ liquefaction\ plant\ [MJ]}\cdot 100\end{aligned}$$

According to literature [139,314,316], this rate is 8.8%; conversely, according to the Ecoinvent dataset[345], 15% v/v of the natural gas that reaches the liquefaction plant is used within the plant itself for energy requirements. For this study, the inventory data reported in the Ecoinvent dataset [345] have been modified according to the more up-to-date data from literature. The refrigerants used in the cooling process are propane, a mixed refrigerant and nitrogen [474]. Plant and ship construction and maintenance, and also natural gas storage prior to ship transport, are included in the model according to the Ecoinvent database [345].

A4.3-1.2 LNG transport

A total of 45 new LNG tanker ships have been built for the new projects developed by the company Qatargas and the Q flex and Qmax are used to transport LNG from Qatar to the UK as a replacement for the steam turbine driven ships. The new tankers have 60% higher fuel efficiency and 30% lower emissions thanks to the new powered slow speed diesel engines that replace the steam turbines [135,138]. Therefore, the emissions due to LNG transport in steam turbine tankers reported in Ecoinvent database [345] were decreased by 30%. The tankers have an average cruise speed of 19.5 knots [139,316,324].

The distance between Ras Laffan and South Hook Terminal is 11821 km [367]; the voyage therefore lasts less than 13 days. However, 14 days of journey time was used in the model to account for eventual delays and for the waiting time at the entry of Suez Canal before the scheduled convoy time. There are multiple convoys passing the Suez Canal every day at fixed hours. Ships that booked the transit have priority. Those ships which booked the transit and arrive late at the canal (within a certain time range) can still transit but they have to pay an increased toll. Ships that did not book in advance the transit can still pass the canal but they do not have priority. Ships willing to pass through the canal have to arrive a few hours before the scheduled convoy time. The maximum allowed time to arrive at the canal before the convoys start depends on the boat type and on the direction of the journey but this time ranges between 3 and 5 hours. Therefore, accounting 24 hours delays is a conservative estimate.

The Qmax tanker is assumed to be used for transport; it carries LNG at 98.5% of its total volume capacity of 263000 m^3, corresponding to 110,000 tonnes at a density of 424 kg/m^3 at -163C and 1 atm.

To identify the power requirements for the reliquefaction systems, the amount of boil off gas is calculated according to the boil off rate of 0.14% (the amount of natural gas that needs to be reliquefied is 6.5 ton/h). The reliquefaction systems are working at the maximum of their capacity (7 ton/h for Qmax as reported in [138]) with an electric power requirement of 6 MW.

A4.3-1.3 On-board reliquefaction

The emissions to air associated with the electricity required by the reliquefaction system are included in the analysis. For this, the specific diesel propulsion is considered. The new ships of the Qatari fleet are equipped with five main diesel generator engines for all auxiliaries (7L32/40, MAN Diesel SE manufacture) and two main propulsion diesel engines (6S70ME-C, MAN B&W) [475,476]. Specification of the auxiliary generator engines and also the average emissions in the exhaust gas at rated power and without exhaust gas treatment are reported in the literature [477]. In this assessment, these emissions have been modified as described in the following paragraph to account for improved emission standards for marine transportation commonly referred to as Tier II and introduced in 2008 in the Annex VI of the IMO 1997 [368].

The maximum allowed NOX emissions for marine diesel engines according to IMO Tier II can be calculated as follows [478]:

$44 * n^{-0.23}$ g/kWh

Where n is rated engine speed in rpm and 130 ≤ n ≤ 2000. We assume that n equals 720 [477].

The limit for HC, PM and CO are reported in literature [478].

The 7L32/40 MAN Diesel engine is found to be fully compliant with the IMO Tier II exhaust emissions regulations [368], except for the sulphur emissions. Within the SOx emission control areas (that includes also the North Sea) the use of fuel oil with a sulphur content not exceeding 1% [479] or the application of an exhaust gas (SOx) cleaning system to reduce the total emission of SOx to 6.0g/kWh, (to be calculated as the total weight of sulphur dioxide emission [480]) is required.

Therefore, the emissions included in the inventory are those reported by Man [477] except for those associated with the sulphur content of the fuel. The fuel used to power the auxiliaries is assumed to be heavy fuel oil RM-B grade at 1% sulphur instead of 2.5% [477] (this value of 1 % complies also with the limit of 6 g/kWh). The fuel has an ash content of 0.1% and an ash content of the lube oil of 4.0% [477]. Lube ash oil and fuel ash oil are assumed to be PM 2.5.

The amount of fuel consumed by the reliquefaction systems is included in the inventory according to Man [477] (p. 97) (additions to fuel consumption for operation with marine gas oil at 100% load).

The reliquefaction electricity requirements of the return journey at ballast condition are considered negligible and not included in the inventory.

A4.3-1.4 Fuel consumption for propulsion

Burdens associated with fuel use for propulsion cover the outward journey with the payload of LNG and also return under ballast condition. The fuel oil consumption (assumed to be heavy fuel oil [323,481]) of the two main propeller engines is also considered. In the Ecoinvent database [345], the steam turbine of the tanker ship is assumed to be powered using only the boil off gas. As the boil off gas is assumed to be entirely reliquefied, we exclude from the Ecoinvent database [345] inventory the natural gas used to power the steam turbine but we included the fuel consumption to power the two main propellers. Q-Max carriers employ two MAN B&W 7S70ME-C engines, each rated at 21,770 kW at 91 rpm [482]. The specific fuel consumption for each engine is assumed to be 165 g/kWh [483]. Ships construction as well as maintenance is included in the model. Table A.4.3-1 reports some of the inventory data used in the LCA model.

Emission due to the reliquefaction systems	**Unit**	
Nitrogen N2	kg/tanker	10261440
Oxygen O2	kg/tanker	1945440
Carbon dioxide CO2	kg/tanker	1189440
Steam H2O	kg/tanker	635040
Inert gases Ar, He, Ne	kg/tanker	151200
Sulphur oxides SO2	kg/tanker	8064
Nitrogen oxides NO2	kg/tanker	18144
Carbon monoxide CO	kg/tanker	1209.6
Hydrocarbons HC	kg/tanker	1612.8
Soot (elemental carbon)	kg/tanker	604.8
Fuel ash	kg/tanker	60.48
Lube oil ash	kg/tanker	40.32
Additional amount of fuel needed to power the reliquefaction systems	ton/tanker	372.96
Amount of LNG transported by the tanker referred to the gas phase	Nm3	1.56E+08

Table A.4.3-1. Key inventory data per Nm3 of distributed gas.

A4.3-1.5 Evaporation process

According to the Ecoinvent dataset [345], 2% of the natural gas that reaches the evaporation plant is used within the plant itself. However, it is reported that the 3% of the gas is used to run the evaporation equipment [139,371]; the assessment has been based on the higher value. The burdens associated with natural gas distribution through pipes to the final consumer at low pressure, including compression energy and fugitive emission are also considered in the inventory according to Ecoinvent database [345].

Appendix to chapter 5.

A5.2 Integrated gasification and plasma cleaning for electricity production

		MSW1	MSW2	MSW3	C&I	Biomass	RDF	SRF
Input								
Waste		1 ton	1 ton	1 ton	1 ton	1 ton	1 ton	1 ton
Auxiliary materials								
Process water	kg	583.81	601.83	946.01	685.93	971.11	594.41	943.72
Oxygen	kg	236.70	266.41	363.28	314.06	538.29	269.72	403.72
Nitrogen	kg	33.22	31.74	39.00	45.01	51.81	32.40	46.87
Activated carbon	kg	0.10	0.11	0.14	0.12	0.19	0.11	0.17
NaHCO3	kg	2.62	2.01	1.64	2.63	2.29	2.07	7.76
Sulphuric acid	kg	3.53	2.55	7.40	2.97	1.15	3.41	4.04
Sodium hypochlorite	kg	11.68	8.48	4.96	11.25	7.63	8.81	35.78
Sodium hydroxide	kg	1.86	1.35	0.79	1.79	1.22	1.40	5.70
Urea	kg	1.34	0.96	5.03	1.12	0.47	1.27	1.52
Auxiliary energy								
Electric energy	MJ	442.95	239.61	651.03	501.12	459.21	453.24	708.05
Output								
Air emissions								
NH3	g	16.36	17.01	26.56	19.99	30.45	17.22	29.25
Argon	kg	10.29	11.76	15.79	13.66	21.21	11.73	17.55
CO	g	133.19	144.14	225.76	197.84	267.35	132.29	256.07
F2	g	0.29	0.31	0.00	0.34	0.00	0.00	0.00
HCl	g	0.57	0.93	1.59	1.03	2.44	0.60	1.66
HF	g	5.17	4.95	5.84	5.86	6.09	5.13	6.07
N2	kg	3701.81	3839.54	6099.28	4514.03	6920.48	3897.83	6594.17
NOx	g	225.90	163.63	97.21	190.95	73.69	218.67	258.83
O2	kg	672.71	698.15	1095.07	826.33	1254.70	706.54	1189.83
S	g	615.00	432.00	253.00	595.00	429.00	11.20	1950.00
SO2	g	149.84	108.88	63.74	144.42	98.05	113.26	459.16
H2O	kg	341.05	347.30	507.49	355.91	604.10	368.82	628.90
Residues								
APC	kg	15.10	6.60	7.45	11.43	4.55	20.82	21.23
wastewater	kg	11.64	12.01	15.30	14.24	20.90	11.84	19.22
Products								
Electric energy	MJ	3021.59	3162.47	5168.98	3685.22	5724.29	3187.68	5471.46
Metal scrap	kg	28.90	47.06	47.05	27.56	0.00	12.51	0.00
Non-metal scrap	kg	9.79	15.94	15.94	8.11	0.00	38.10	0.00
plasmarok	kg	79.42	30.45	50.29	72.37	3.20	88.98	65.23

Table A5.2-1. Inventory of the two-stage gasification and plasma process when different waste streams are considered. Only direct input and output are reported.

Impact category	**Impact per region-Europe (25+3) per year (reference year 2000)**
Abiotic Depletion (MJ)	3.51E+13
Acidification Potential (kg SO_2eq)	1.68E+10
Eutrophication Potential (kg Phosphate eq)	1.85E+10
Freshwater Aquatic Ecotoxicity Pot. (kg DCB eq)	2.09E+11
Global Warming Potential (kg CO_2eq.)	5.21E+12
Human Toxicity Potential (kg DCB eq)	5.00E+11
Marine Aquatic Ecotoxicity Pot. (kg DCB eq)	4.45E+13
Ozone Layer Depletion Potential (kg R11 eq)	1.02E+07
Photochem. Ozone Creation Potential (kg Ethene eq)	1.73E+09
Terrestric Ecotoxicity Potential (kg DCB eq)	1.16E+11

Table A5.2-2. Impact values used for normalisation. The normalisation is done based on CML, IPCC, ReCiPe (region average), EU25+3, year 2000 [310].

A5.3 Conventional and two-stage advanced electricity production technologies for municipal solid waste treatment

		G-Pl	**FP-C**	**G-SC**	**Landfill**	**Incineration (Sheffield)**	**Incineration (North Hykenham)**
Primary Energy resources	MJ	2.25E+00	1.04E+00	1.35	0.781	1.47	1.57
Non-renewable elements							
Aluminium	mg	1.40E+01	1.62E+00	4.97E+00	-	1.78E+01	3.81E+00
Chromium	mg	1.00E+01	-	1.50E+00	-	5.73E+00	1.20E+00
Copper	mg	1.60E+01	1.52E+00	3.09E+00	-	9.19E+00	2.48E+00
Iron	mg	1.78E+02	9.11E+00	4.39E+01	1.09E+03	2.40E+02	3.82E+01
Manganese	mg	2.46E+00	-	-	1.63E+00	1.51E+00	6.30E-01
Nickel	mg	2.40E+01	-	2.67E+00	-	1.39E+01	2.14E+00
Zinc	mg	3.86E+00	-	-	-	2.06E+00	2.00E-03
Non-renewable resources							
Clay	g	5.30E-01	3.95E+00	1.23E+01	5.97E+01	3.74E+01	9.10E+00
Inert rock	g	2.25E+02	6.57E+01	1.05E+02	3.79E+01	5.88E+01	7.48E+01
Limestone (calcium carbonate)	g	6.61E+00	1.09E+01	5.30E+01	2.05E+00	1.17E+02	3.94E+01
Natural Aggregate	g	3.36E+00	1.16E+01	3.58E+01	4.22E+01	1.08E+02	2.65E+01
Pyrite	g	1.90E-01	-	-	-	-	-
Quartz sand (silica sand; silicon dioxide)	g	4.90E-02	9.88E-03	1.49E+01	3.33E+01	9.30E-03	1.28E-03
Sodium chloride (rock salt)	g	1.10E+01	1.03E-02	7.49E+01	-	3.90E-02	3.95E-01
Soil	g	1.18E+00	2.37E-01	1.85E+00	2.06E+01	1.49E+00	2.74E-01
Renewable resources							
Water	kg	1.31E+02	-	8.78E+00	2.46E+01	6.28E+00	8.19E+00

Table A5.3-1. Input elementary flows for the processes analysed for 1 kg of waste treated. Elementary flows are flows which enter the techno-sphere from nature and the flows that exit the techno-sphere to the nature.

		G-Pl	FP-C	G-SC	Landfill	Incineration (Sheffield)	Incineration (North Hykenham)
Emissions to air							
Heavy metals to air							
Iron	mg	1.34E-01	1.58E-02	4.29E-02	9.40E-03	1.40E-01	3.91E-02
Lead	mg	2.64E-02	5.02E-03	9.06E-03	7.69E-03	2.00E-03	7.93E-03
Manganese	mg	2.61E-02	6.09E-03	8.53E-03	1.39E-02	7.91E-03	7.53E-03
Mercury	mg	6.09E-03	1.33E-03	3.39E-03	1.92E-02	4.71E-03	1.88E-03
Nickel	mg	2.46E-02	2.78E-03	4.18E-03	1.26E-03	2.10E-02	4.22E-03
Selenium	mg	1.18E-02	4.69E-03	6.21E-03	9.66E-04	3.80E-03	5.34E-03
Tin	mg	9.13E-03	4.07E-03	5.47E-03	5.10E-04	3.66E-03	0..463
Titanium	mg	8.14E-03	9.10E-03	2.54E-03	1.77E-04	8.17E-03	2.25E-03
Vanadium	mg	6.18E-02	6.37E-03	8.80E-03	2.55E-03	5.43E-03	9.49E-03
Zinc	mg	4.93E-02	9.10E-03	1.60E-02	-	3.18E-02	1.35E-02
Inorganic emissions to air							
Ammonia	g	-	-	-	-	1.42E-02	-
Argon	g	1.03E+01	1.17E+01	-	-	-	-
carbon dioxide	g	9.90E+02	1.09E+03	7.18E+00	1.00E+02	5.66E+00	7.11E+00
Carbon monoxide	g	5.30E-01	1.83E-01	1.46E+00	1.09E+00	1.31E+00	-
Hydrogen	g	-	-	2.25E-01	1.10E-01	1.93E-01	1.69E-01
Hydrogen sulphide	g	-	1.22E-02	1.57E-02	-	-	-
Nitrogen (atmospheric nitrogen)	g	3.70E+03	4.35E+03	-	-	-	-
Nitrogen monoxide	g	-	-	-	-	1.50E-01	1.32E-01
Nitrogen oxides	g	4.65E-01	3.89E-01	1.75E-01	2.32E-01	9.43E-01	1.04E-01
Nitrogentrifluoride	g	-	-	1.98E-01	-	-	-
Nitrous oxide (laughing gas)	g	-	-	-	-	3.11E-01	-
Oxygen	g	6.74E+02	7.79E+03	5.61E-01	9.95E-01	2.75E-01	4.77E-01
Sulphur	g	6.16E-01	-	-	-	-	-
Sulphur dioxide	g	4.31E-01	4.41E-01	2.06E-01	-	2.56E-01	6.13E-01
Organic emissions to air (group VOC)							
Group NMVOC to air	mg	2.04E+01	1.64E+01	1.89E+01	5.62E+01	4.06E+01	2.47E+01
Hydrocarbons (unspecified)	mg	8.34E-02	5.45E-02	7.21E-02			5.00E-01
Methane	mg	5.12E+02	1.38E+02	1.38E+03	2.60E+02	1.21E+03	1.07E+03
Methane (biotic)	mg	4.01E-01	2.99E-02	7.80E-02	2.13E+04	2.73E-01	6.89E-02
VOC (unspecified)	mg	-	-	-	-	-	8.70E+01
Particles to air	mg	3.80E+01	1.34E+01	2.43E+01	2.88E+00	5.15E+01	1.10E+02
Emissions to fresh water							
Inorganic emissions to fresh water							
Antimony	mg	-	1.72E+01	5.39E+01	-	1.63E+02	3.98E+01
BOD	mg	2.01E+00	1.17E+03	3.66E+03	-	1.11E+04	2.70E+03

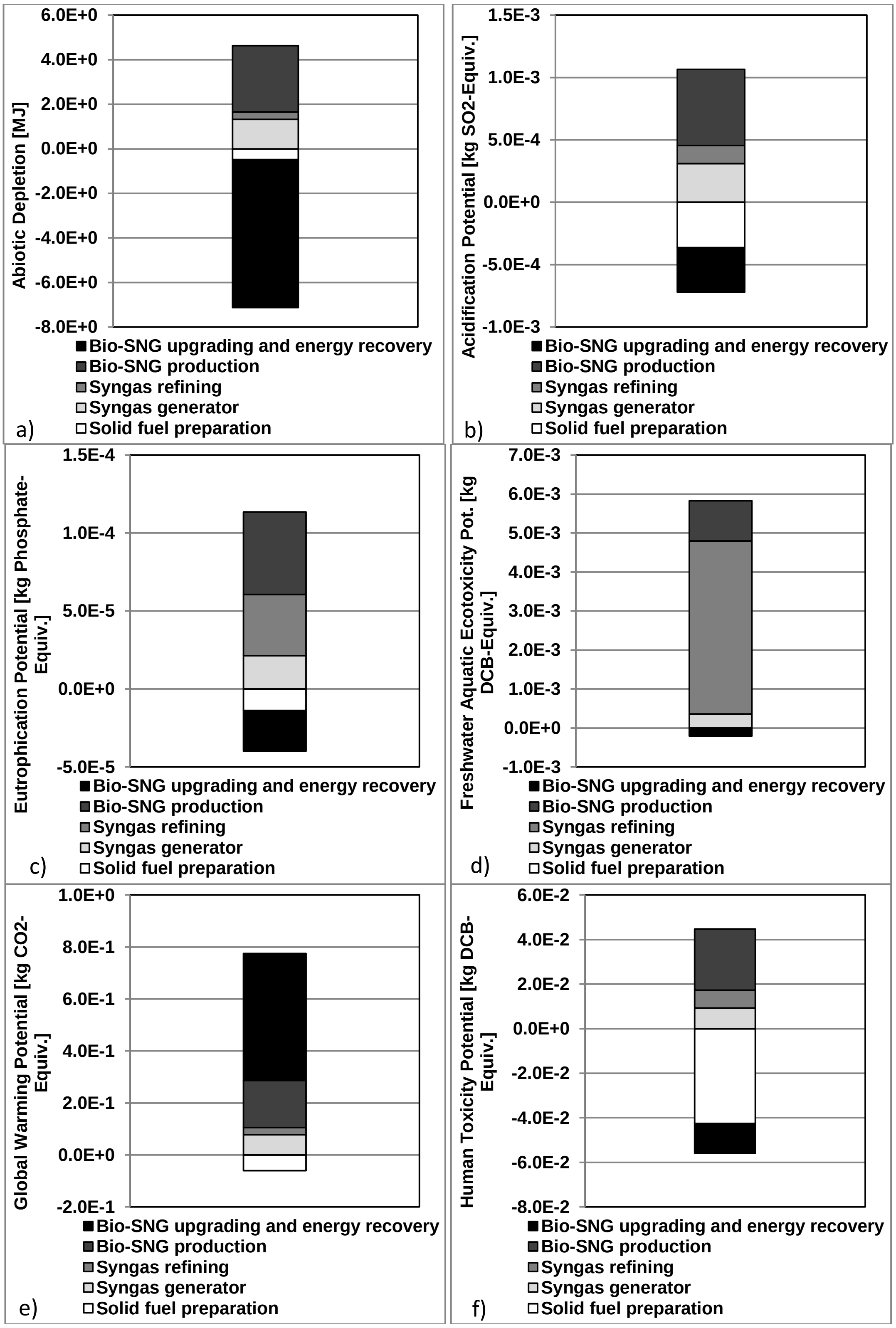
Boron	mg	2.27E+05	1.90E+05	2.16E+05	0.00E+00	4.43E+05	1.69E+05
Bromine	mg	0.00E+00	1.87E+06	5.82E+06	0.00E+00	1.76E+07	4.30E+06
Chloride	mg	7.98E+09	1.60E+02	4.98E+08	5.85E+07	1.05E+09	7.08E+08
Chromium	mg	-	1.11E+01	3.46E+01	-	1.05E+02	2.56E+01
Copper	mg	-	9.77E+00	3.05E+01	-	9.27E+01	2.26E+01
DOC	mg	2.51E+00	1.41E+03	4.20E+03	-	1.34E+04	3.27E+03
Fluoride	mg	6.69E+07	3.26E+07	4.64E+07	7.28E+06	3.79E+07	3.86E+07
Iron	mg	-	1.08E+01	3.30E+01	-	1.03E+02	2.47E+01
Lead	mg	-	2.83E+00	8.38E+03	-	2.68E+01	6.53E+00
Manganese	mg	7.76E+00	1.01E+00	2.93E+00	-	9.75E+00	2.24E+00
Nitrate	mg	1.15E+07	1.51E+07	2.23E+06	6.31E+06	1.47E+06	1.96E+06
Phosphate	mg	4.31E+06	4.81E+05	1.00E+06	1.30E+06	4.46E+06	1.04E+06
Potassium	mg	8.17E+06	3.66E+08	1.14E+09	3.69E+06	3.47E+09	8.44E+08
Sodium	mg	4.83E+08	3.38E+08	9.84E+08	2.25E+07	3.00E+09	8.23E+08
Solids (suspended)	mg	1.85E+02	3.37E+03	1.05E+04	-	3.20E+04	7.78E+03
Strontium	mg	2.77E+00	-	-	-	-	-
Sulphate	mg	1.80E+04	3.16E+08	9.37E+08	2.91E+07	2.73E+09	6.96E+08
Tin	mg	-	1.03E+00	3.22E+00	-	-	2.38E+00
TOC	mg	2.51E+00	1.41E+03	4.42E+03	-	1.34E+04	-
Vanadium	mg	-	1.97E+00	6.15E+00	-	-	4.54E+00
Zinc	mg	1.73E-06	2.32E+01	7.24E+01	-	-	5.35E+01
Aluminium	mg	1.75E+01	7.37E+02	2.30E+03		6.99E+03	1.70E+03
Ammonia	mg	2.64E-01	1.19E-01	-	-	-	1.33E-01
Ammonium / ammonia	mg	3.32E+00	-	1.05E-01	1.55E+02	6.73E-01	2.15E+00
Barium	mg	-	1.05E-01	-	1.02E-01	2.58E-01	1.29E-01
Bromate	mg	8.43E-01	-	-	-	-	-
Calcium	mg	8.65E+02	6.66E+02	2.05E+03	2.48E+01	6.20E+03	1.52E+03
Carbon disulphide	mg	-	-	-	5.10E+00	-	-
Carbonate	mg	1.28E+00	4.94E-01	7.20E-01	-	4.05E-01	2.56E+00
Chlorate	mg	6.44E+00	-	-	-	-	-
Chlorine (dissolved)	mg	7.74E-01	3.70E-01	4.51E-01	-	2.17E-01	3.86E-01
Magnesium	mg	9.82E+01	5.18E+01	1.54E+02	2.77E+00	4.69E+02	1.15E+02
Nitrogen	mg	2.47E-01	-	-	-	-	-
Nitrogen organic bounded	mg	4.63E-01	1.51E-01	2.27E-01	8.74E-01	-	3.32E-01
Phosphorus	mg	3.04E-01	-	-	1.56E+01	-	
Sodium hypochlorite	mg	9.63E-01	-	-	-	-	3.58E-01
Sodium sulphate	mg	7.22E+00	-	-		2.16E-01	2.55E-01
Sulphide	mg	3.87E-01	-	-	3.22E+00	-	4.68E-01
Heavy metals to fresh water	mg	2.20E+01	1.90E+01	6.11E+01	4.73E+00	1.79E+02	4.37E+01
Particles to fresh water	mg	2.15E+02	3.35E+01	6.27E+01	1.22E+02	9.64E+01	5.60E+01
Emissions to sea water							
Inorganic emissions to sea water	mg	9.89E+01	2.99E+01	4.62E+01	1.35E+02	6.60E+01	5.07E+02
Organic emissions to sea	mg	9.00E-01	1.11E+00	1.07E+00	8.00E-03	3.62E+00	1.40E+00

water							
Other emissions to sea water	mg	2.55E+06	1.56E+06	2.01E+06	5.40E+04	9.40E+05	1.72E+06
Particles to sea water	mg	1.05E+01	3.69E+00	5.24E+00	5.79E+00	-	-

Table A5.3-2. Output elementary flows for the processes analysed for 1 kg of waste treated. Elementary flows are flows which enter the techno-sphere from nature and the flows that exit the techno-sphere to the nature.

Appendix to chapter 6

A6.2 Integrated gasification and plasma cleaning for Bio-SNG production



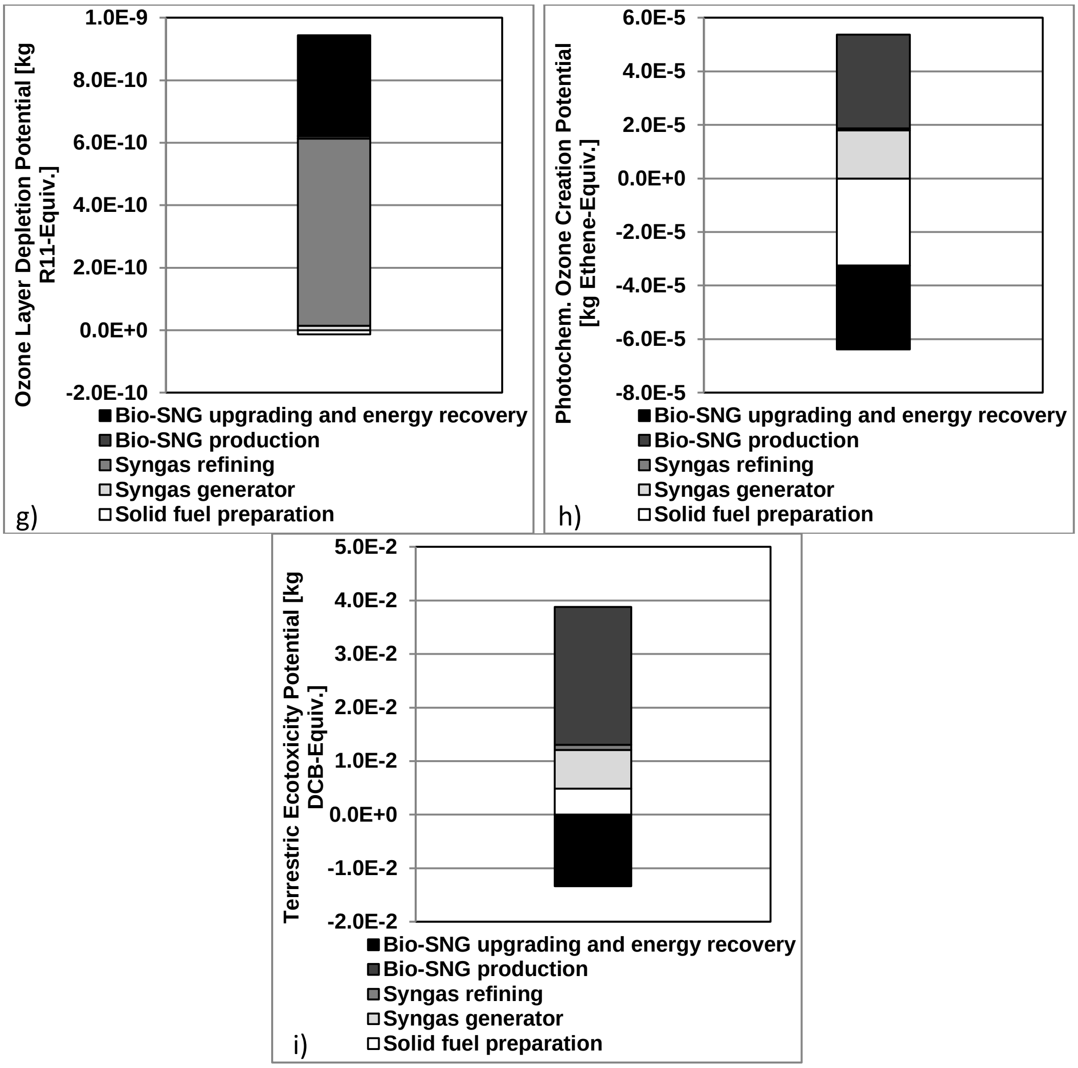


Figure A6.2-1. Hot spot analysis of the G-Pl-Bio-SNG process. a) ADP. b) AP. c) EP. d) FAETP). e) GEP. f) HTP. g) ODP. h) POCP. i) TEPT.

Results variation [%] from the base case described in the main text	**Dual stage gasification and plasma process for Bio-SNG production**	
	Ferrous 1:1-non-ferrous 1:0.99	**Ferrous 1:0.51-non-ferrous 1:0.1**
Abiotic Depletion [MJ]	-13.76	-25.19
Acidification Potential [kg SO2-Equiv.]	85.40	96.26
Eutrophication Potential [kg Phosphate-Equiv.]	14.77	20.74
Freshwater Aquatic Ecotoxicity Pot. [kg DCB-Equiv.]	3.10	3.03
Global Warming Potential [kg CO2-Equiv.]	4.03	7.60
Human Toxicity Potential [kg DCB-Equiv.]	-409.24	-351.99
Ozone Layer Depletion Potential [kg R11-Equiv.]	96.08	0.00
Photochem. Ozone Creation Potential [kg Ethene-Equiv.]	74.58	-36.48
Terrestric Ecotoxicity Potential [kg DCB-Equiv.]	3.12	2.61

Table A6.2-1. Dual stage advanced thermal process for Bio-SNG production from waste: results of the scenario analysis for the metal reprocessing process.

Results variation [%] from the base case described in the main text	**Dual stage gasification and plasma process for Bio-SNG production**	
	Cryogenic oxygen scaled on UK electricity figures	**PSA oxygen production**
Abiotic Depletion [MJ]	-1.97	-5.29
Acidification Potential [kg SO2-Equiv.]	-7.63	-2.67
Eutrophication Potential [kg Phosphate-Equiv.]	3.83	5.79
Freshwater Aquatic Ecotoxicity Pot. [kg DCB-Equiv.]	2.13	2.64
Global Warming Potential [kg CO2-Equiv.]	0.67	1.36
Human Toxicity Potential [kg DCB-Equiv.]	-31.29	-38.20
Ozone Layer Depletion Potential [kg R11-Equiv.]	-2.75	-2.69
Photochem. Ozone Creation Potential [kg Ethene-Equiv.]	-6.32	-2.89
Terrestric Ecotoxicity Potential [kg DCB-Equiv.]	18.50	21.35

Table A6.2-2. Dual stage advanced thermal process for Bio-SNG production from waste: results of the scenario analysis for the oxygen production process.

Results variation [%] from the base case described in the main text	**Dual stage gasification and plasma process for Bio-SNG production**			
	Land won sand and gravel aggregates	**Marine sand and gravel aggregates**	**Recycled aggregates**	**No aggregates**
Abiotic Depletion [MJ]	0.00	0.00	0.00	0.00
Acidification Potential [kg SO2-Equiv.]	0.20	-16.31	-0.10	0.23
Eutrophication Potential [kg Phosphate-Equiv.]	0.05	-11.86	-0.02	0.07
Freshwater Aquatic Ecotoxicity Pot. [kg DCB-Equiv.]	0.01	-6.85	0.01	0.01
Global Warming Potential [kg CO2-Equiv.]	0.02	-0.43	-0.01	0.02
Human Toxicity Potential [kg DCB-Equiv.]	-0.18	7.15	-0.14	-0.28
Ozone Layer Depletion Potential [kg R11-Equiv.]	1.85	1.84	-0.99	1.87
Photochem. Ozone Creation Potential [kg Ethene-Equiv.]	0.49	-44.74	-0.11	0.64
Terrestric Ecotoxicity Potential [kg DCB-Equiv.]	0.00	-0.01	0.00	0.00

Table A6.2-3. Dual stage advanced thermal process for Bio-SNG production from waste: results of the scenario analysis for the substitution of the vitrified slag.

A6.3 Biological and two-stage advanced methane production technologies for municipal solid waste treatment

A6.3-1. Life Cycle Inventory

A6.3-1.1 System expansion

Ferrous material is assumed to be substituted at a 1 to 0.51 rate and the recovered material is assumed to be recycled by electric furnace processing, as reported by the Worldsteel LCA Methodology report [412]. Non-ferrous metal is assumed to be substituted at a 1 to 0.6 rate. The recovered aluminium is assumed to be recycled by clean scrap melting and casting, as reported in the Environmental profile report for the Aluminium Industry [413]. A sensitivity analysis on the substitution ratio of the metals recovered has been performed but the variation in the results was negligible and the results have not been reported.

A6.3-1.2 Transport

For scenarios 1-2 MSW is assumed to be transported from transfer station to the processing plants (50 km distance) but the transport of the mechanically separated fraction to landfill/incineration is not considered. The environmental burden due to the use of trucks has been allocated to the direct burden of the mechanical treatment plant and the diesel production has been allocated to the indirect burden of this plant.

For scenario 3-4, source separated biodegradable waste is assumed to be transported from kerbside to the AD plant (50 km distance) and the residual waste is assumed to be transported from transfer station to incineration/landfill (50 km distance). The environmental burdens due to the use of trucks and production of diesel have been allocated to the section of incineration/landfill for the 75 % (amount of residual waste) and for the 35% (amount of source separated waste) to the digester and pre-treatment section of the anaerobic digestion plant. For scenarios 5 MSW is assumed to be transported from transfer station to the advanced thermal treatment process (50 km distance). All waste is assumed to be transported in an EURO 4-22t payload truck. The burden due to transport (including the diesel production) is always less than 5% of the total environmental burden of the processes, therefore, no sensitivity analysis has been performed on this parameter.

A6.3-1.3 Incineration

Waste incineration is modelled according to the average data for UK waste-to-energy plants taken from the database of GaBi 5.0 software [310]. Two different incineration models are used, respectively with wet and dry flue gas treatment (FGT). Different NOx-removal technologies are used to represent the application of different FGT systems in Europe; the data from GaBi represent averages over a number of European incinerators. The system includes the generation of steam to produce electricity and heat.

A6.3-1.4 Landfill

The inventory data for landfilling with electricity recovery were based on the GaBi database [310]. The data set represent a typical municipal waste landfill with surface and basic sealing, meeting European limits for emissions. The site operations include landfill gas treatment, leachate treatment, sludge treatment and deposition. Part of the landfill gas is assumed to be flared (22%), part of it to be used for electricity production (28%) and the rest emitted to the environment (50%). All manufacturing processes of the sealing materials, as well as energy requirements for the site, were included within the system.

A6.3-1.5 Advanced thermal treatment: dual stage gasification and plasma process

Avoided burdens are allocated to the production of Plasmarok as it can be used as a substitute for aggregate materials [415]. We assume that Plasmarok substitutes the production of primary aggregates from crushed rock as this is the most important source of primary aggregates in England [460]. We assume that the oxygen supplied to the process is produced using the technology of cryogenic separation of air. Average UK data are applied [310]. The inventory for the activated carbon used to remove the APC residue is reported in Noijuntira *et al.* [484].

A wet scrubbing system is used in the dual stage advanced process to further cool and clean the syngas from acid and alkali compounds. As the physical and chemical composition of the liquid effluents of this process do not exceed the limits reported in the WID directive [152], we assume that water effluents are treated in standard waste water treatment plants [310].

The inventory and the environmental burden of all chemicals used in the process (such as nitrogen, sodium hypochlorite, urea etc.) are reported in GaBi 6 LCA software [310] and in Ecoinvent [345]. The production of Bio-SNG is considered according to Table 6.2.1.

A6.3-1.6 Anaerobic Digestion of centrally separated waste

Six operations are identified in the AD process: i) pre-treatment; ii) anaerobic digestion; iii) water and acid compounds removal; iv) upgrading of the biogas; v) disposal of digestate to incineration. The characteristics of each section and the assumptions constituting the LCA models and the inventory data are specified below.

- Pre-treatment and anaerobic digestion phase (i and ii). After the mechanical separation of the MSW, the biodegradable centrally separated waste (the composition of the biodegradable part is reported in literature [454]) enters the pre-treatment section where the waste undertakes maceration and hygienisation. Further details of this phase (i.e. pre-treatment steps, vessel dimensions, etc.) are reported in literature [267,450,454]. The AD phase is assumed to be using a continuous, single-stage, mixed tank mesophilic reactor operating at a temperature of 35 ˚C in wet conditions given its broad application [164,166,267,450]. The majority of AD plants from centrally separated biodegradable waste actually in use operate under this condition. Monson *et al.* [175] reported that 73% of the anaerobic digesters treating centrally separated MSW analysed in their report use a wet system as well as 90% operate in mesophilic temperature range. The yield of biogas produced during this phase decreases if the organic fraction of centrally separated municipal solid waste is used instead of the source separated fraction. This is due to the higher content of plastic and impurity of the organic matter supplied to the digester and therefore a lower composition of volatile solid on which the yield of methane depends [485]. The yield of biogas is assumed to be 0.079 Nm^3 per kg of the centrally separated organic fraction [175]. The model accounts also for methane losses occurring in the digester [439,450–452], which are assumed to be 3 %.
- Cleaning (water and H_2S removal), up-grading and off gas flaring phase (iii and iv). The gas has to be cleaned and upgraded before grid injection. The cleaning of the biogas includes removal of H_2S and water that can cause damages to the subsequent upgrading unit and to the grid pipes. The most common method for H_2S removal from the crude biogas is through the reaction of H_2S with metal oxides [170,171]. In this study H_2S is assumed to be removed in a desulfuriser unit with a catalytic bed of ZNO, which is placed at the digester plant, where the

biogas is produced. Water is assumed to be adsorbed on silica gel in the upgrading unit in the pre-treatment phase. To achieve the strict regulation limits set in the GMRS [172], the raw methane is assumed to be upgraded in a pressure swing adsorption system. Petersson *et al.* [167] reported that the PSA electricity consumption including gas compression to 7 barg is 0.25 kWh/Nm^3 of raw biogas. Persson *et al.* [173] assumed instead that the electricity requirement for PSA was 0.5-0.6 kWh/m^3 of upgraded gas, not accounting for high pressure compression (those are the figures reported by the owners), whereas Persson [171] reported the same values as Persson *et al.* [173] for electricity requirements not accounting for compression and specified that the electricity needs to be increased to the values of 0.8-0.88 kWh/m^3 of upgraded gas if compression is considered as well. The latter value has been used in the LCA model. This value is in line with the value reported in the Ecoinvent database [345] for raw gas upgrading (CH methane, 96% from biogas purification). The model accounts also for a 3% methane losses (the amount of methane not recovered from the raw biogas, hence the methane content of the off gas of the PSA system) occurring in the upgrading system [167,173,453]. The PSA model does not include the production of a combustible stream for electricity production, as in literature this layout is not reported to be used for AD systems. The off gas of the PSA system is assumed to be flared before emission to environment.

- Digestate use (v). When anaerobic digestion is used to treat centrally separated organic wastes the low quality of the digestate prevents the use as fertiliser and it has to be disposed of either by thermal treatment or landfill [175,486]. In this study, the digestate produced in scenarios 1-2 is assumed to be incinerated according to Ecoinvent database [345].

A sensitivity analysis on main parameters regarding the AD process has not been reported here as significant results have already been published in Evangelisti *et al.* [267].

A6.3-1.7 Anaerobic Digestion of source separated waste and additional disposal

The whole digestate is assumed to be separated in liquor and fibre as standard practice [456] and the separation method is assumed to be physical [169]. The liquor is normally separated using a separator or centrifuge to remove coarse fibres. The fibres represent 20 % of the total digestate whereas the liquor is usually 80% in weight of the total digestate [456]. To calculate the nutrient of the liquor after dewatering the values for nutrient partition between liquid and solid phases as reported in Lukehurst *et al.* [487] are used. The electricity requirements for dewatering are taken from Wiliams *et al.* [488].

We assume that the liquor separated from the whole digestate in the dewatering section is used as fertilizer whereas the fibres are sent to incineration as inert material (as reported in Wrap [456]). The LCA model accounts for the burden associated with the use of liquor as fertilizer. Organic fertilizers coming for example from the anaerobic digestion can be used to improve the nutrient content of soils and therefore avoiding the use of chemical fertilizers (the use in agriculture of fertilisers with high available nitrogen content, i.e. digestate, is anyway, affected by restrictions according to the Nitrogen Vulnerable Zones as reported by the European Parliament [489]).

We assume that the nutrients are not lost during the anaerobic digestion phase; thus all the nutrients of the bio-degradable waste (N, K and P) remain in the whole digestate [174,267]. The nutrient content of the fertilizers is calculated based on the amounts of N, P_2O_5 and K_2O for N, P and K, respectively [458], 2010). The distinction between readily availability and crop availability of nutrients [490] is used to calculate the avoided chemical fertilizers and the emissions due to fertilizer spreading.

The amount of readily available nutrients assumed in this study is reported in Wrap [457] (80% of the N content of the digestate is readily available and 100% of K_2O and P_2O_5). Defra [458] reported the chemical fertilizers usually employed for N, K and P. The N readily available content of the digestate is assumed to substitute the chemical fertilizer of ammonium sulphate, the K_2O readily available content of the digestate is assumed to substitute the chemical fertilizer of potassium chloride and the readily available content of P_2O5 of the digestate is assumed to substitute the chemical fertilizer of the superphosphate. The results have been calculated equalling the amount of nutrients readily available in the digestate to the amount of chemical fertilizer needed. (i.e. 1 kg of N readily available in the digestate equal 1 kg of the chemical fertilizer $NH_4(SO_2)$ substituted).

In the LCA model we have also accounted for the emission due to the organic fertilizers when those are on the soil.

During the application of both chemical and organic fertilizers part of the nutrients is dispersed into the environment and is not crop available. This means that some amount of the readily available nutrients might be lost as air emission (run off or evaporation), water leakage (leaching) or might not be readily absorbed by the plants because they are chemically bound in a form of not easy uptake from plants. The amount of the nutrients really uptaken by the crops is defined as the nutrient crop availability. Bruun *et al.* [459]

reported the emission coefficients due to the use of the digestate as fertilizer. In particular those emissions represented the difference between normal agricultural practice only using inorganic fertilizers and use of digestate supplemented with inorganic fertilizers, according to the Danish legislation. Those coefficients have already been used in some recent works [174,267,439]. Nitrogen emissions from organic fertilizers are higher than those of chemical fertilizers for two reasons: i) chemical fertilizers are given to the plants when and where they need them and this reduces N evaporation; ii) not all the readily available N is absorbed by the plants. The emissions associated with the spreading of fertilizers, either chemical or organic, are highly variable and depend strongly on the soil and weather conditions, spreading practice and crop practice. Therefore, it is possible that the emission coefficients reported in Bruun *et al.* [459] do not exactly mirror the UK situation but as far as the author's knowledge these data are the only available. Wrap [456] reported that they are undertaking studies on the emission coefficients for digestate spreading applied to UK situation but no data have yet been released. New data on evaporation, leaching and loss of fertilisers might influence the results.

In the LCA model we also account for the part of the carbon of the feed waste not released as biogas. Part of the carbon content of the liquor used as fertilizer is sequestered in the soil and not released to the atmosphere as CO_2 during the timeframe considered [174]. This has been accounted as an actual removal of CO_2 from the atmosphere and therefore as a negative contribution to the global warming.

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