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TESI DI DOTTORATO

LIFE CYCLE ASSESSMENT OF DISTRIBUTED ENERGY PRODUCTION USING BIOFUEL FROM WASTE

Candidata: **Sara Evangelisti**

Supervisore interno Andrea Micangeli

Supervisori esterni Domenico Borello

Paola Lettieri

Ai miei nonni,

quelli che non ci sono più e quelli che resistono.

Grazie per la vostra memoria.

Preface

The work reported in this PhD thesis entitled '*Life Cycle Assessment of Distributed Energy Production Using Biofuel From Waste*' was carried out for the first two years at the Interuniversity Research Centre on Sustainable development (CIRPS) at Sapienza University of Rome (IT) and for the last year at the department of Chemical Engineering at University College London (UCL) (UK), from November 2009 to June 2013. The thesis was supervised by Dr Domenico Borello (department of Mechanical and Aerospace Engineering at Sapienza University of Rome) and by Dr Paola Lettieri (Chemical Engineering department at UCL). The internal supervisor was Dr Andrea Micangeli. The content of the thesis is mainly based on five scientific journal papers prepared in collaboration with the supervisors and other external partners. In particular, during the third year of the PhD the work has been carried out in collaboration with Prof Ronald Clift at the Centre for Environmental Strategy, the University of Surrey (UK). The papers are:

1. Borello, D., Evangelisti, S., Tortora, E., 2013. *Modelling of a CHP SOFC System Fed with Biogas from Anaerobic Digestion of Municipal Waste Integrated with Solar Collectors and Storage Unit*. International Journal of Thermodynamics 16, 28-35.
2. Evangelisti, S., Lettieri, P., Borello, D., Clift, R., 2013. *Life Cycle Assessment of energy from waste via anaerobic digestion: a UK case study*. Waste Management. Manuscript submitted.
3. Evangelisti, S., Lettieri, P., Borello, D., Clift, R., 2013. *Distributed Generation by Energy from Waste Technology: A Life Cycle Perspective*. Journal of Process, Safety and Environmental Protection. Manuscript Submitted.
4. Evangelisti, S., Lettieri P., Borello, D., Clift, R., 2013. *Environmental Impact of various waste to energy closed loop scenarios in the distributed generation paradigm*, Applied Energy. Manuscript in preparation.
5. Evangelisti, S., Lettieri P., Borello, D., Clift, R., 2013. *Life Cycle Assessment of Waste to energy systems in the distributed generation paradigm: a comparison between Italy and UK,* Applied Energy. Manuscript in preparation.

During my first year of PhD I worked on a different project on the measurement of the efficiency of a prototype of solar thermal collector with internal storage, for the production of domestic hot water. This work has been published in a scientific journal paper, Borello, D., Corsini, A., Delibra, G., Evangelisti, S., Micangeli, A., 2012. *Experimental and computational investigation of a new solar integrated collector storage system*, Applied Energy, 97, 982 – 989.

Abstract

Municipal Solid Waste can be a potential renewable and non-seasonal resource for energy production. Alternative uses of waste, in fact, become increasingly interesting both from a waste management perspective – to reduce the amount of increasing waste deposited at landfill - and from an energy system perspective – to get the targets in terms of share of renewable and greenhouse gas reduction.

In Europe, in recent years a lot of attention has raised around the possibility to use the Organic Fraction of Municipal Solid Waste (OFMSW) for the production of energy through anaerobic digestion. In this process, the bio-waste is metabolized by bacteria under anaerobic conditions producing a gas – i.e. biogas. Digestate, a by-product of the anaerobic digestion process, can be used as a valid substitute to conventional mineral fertilisers. Production and collection of OFMSW usually takes place at district level, hence making it a great potential as non-seasonal energy feedstock. The biogas can then be used as a fuel for Combined Heat and Power (CHP) production, through systems directly installed at dwelling level. This perfectly matches with the Distributed Generation paradigm, where the energy is produced at, or near to its point of use.

The aim of this thesis was to evaluate the environmental impact of a *waste-to-energy* system in a distributed generation paradigm. OFMSW was considered as main feedstock to produce biogas, which was fed to different micro CHP units to generate energy for those dwellings that generated the waste. Three different technologies were investigated: Solid Oxide Fuel Cell (SOFC), Micro Gas Turbine (MGT), Stirling Engine. A secondary objective of the work was to demonstrate the feasibility of *the waste-to-energy closed loop* at residential level.

In order to achieve this, a Life Cycle Assessment with system expansion was performed considering two case studies: the Borough of Greenwich in the Greater London area (UK) and the municipality of Livorno (IT), in Toscany region. They were compared in terms of energy generation and waste disposal strategies. The analysis was based on a comparative assessment of two sub – systems: Waste Management scenarios – where alternative waste treatments for the OFMSW were investigated and a Distributed Generation scenarios – where three different micro-CHP technologies were analysed, together with two different

ways of using biogas and three different operating strategies. Moreover, a reference scenario for the production of energy at a residential level was assumed in the two case studies and possible reductions in terms of several environmental impacts categories were evaluated.

Results showed that, although anaerobic digestion is potentially the best option in term of GHG emissions for the treatment of the organic waste, the amount of biogas produced with this fraction was not enough to cover the energy demand of the dwellings that generated the waste. Furthermore, when normalised per tons of OFMSW produced in the two geographical contexts, 851 and 856 kg CO_2 eq were saved in UK and Italy respectively, when AD was considered as alternative waste treatment compared to the landfill plant with energy recovery. The potential emissions' savings were reduced to 30 and 35 kg CO_2 eq per kg of OFMSW treated in UK and Italy respectively, if an incineration plant with electricity and heat production was considered as displaced process. The robustness of the first sub-model was investigated through a sensitivity analysis. The most critical assumption concerned the quantity and quality of the energy substituted by the biogas production.

Fuel cell micro – CHP could clearly reduce the environmental impacts of UK and IT homes compared with their current average value. There was however great difficulty in estimating the magnitude of these reductions, because of the influence of key parameters assumed in the design phase of the micro CHP units, i.e. the H to P ratio of the dwellings and the operating strategy adopted. Another important issue was the type of technology that substituted the new system. Results showed that this can determine when the micro CHP system represents a saving or a burden in terms of emissions. Conclusions showed that the definition of the future energy scenario in which the process will be embedded is a key issue to determine the actual environmental benefits due to the introduction of waste to energy systems in the distributed generation paradigm. This depends on macro- national and European strategies and highlights the importance of a holistic approach to inform decision-makers on the best solution for waste to energy policies.

Sommario

I rifiuti solidi urbani possono essere una potenziale risorsa rinnovabile e non stagionale per la produzione di energia. Negli ultimi anni, l'uso alternativo dei rifiuti sta diventano sempre più interessante sia sul piano della gestione stessa dei rifiuti - per ridurne la quantità da mandare in discarica - sia sul piano energitico - per arrivare agli obiettivi imposti dall'Unione Europea in termini di energia da risorse rinnovabili e riduzione dei gas serra.

Negli ultimi anni in Europa è stata sollevata molta attenzione attorno alla possibilità di utilizzare la Frazione Organica dei Rifiuti Solidi Urbani (FORSU) per la produzione di energia, attraverso il processo di digestione anaerobica. In questo processo, il rifiuto organico viene metabolizzato da batteri in condizioni anaerobiche per produrre il biogas. Inoltre è possibile utilizzare il digestato, un sottoprodotto della digestione anaerobica, come sostituto dei fertilizzanti commerciali. La FORSU viene generata e raccolta generalmente a livello di distretto urbano, rendendola una potenziale risorsa energetica non stagionale, da utilizzare all'interno del distretto stesso. Il biogas prodotto può infatti essere utilizzato come combustibile per la produzione combinata di calore ed elettricità, attraverso sistemi installati direttamente nelle singole abitazioni o a livello condominiale. Questo modello energetico rientra nel paradigma della generazione distribuita, in cui l'energia è prodotta nel - o vicino, al punto di utilizzo.

Lo scopo di questa tesi è stato quello di valutare l'impatto ambientale di un sistema chiuso 'rifiuti-energia' per la generazione distribuita. Il sistema considerato è così costituito: partendo dalla FORSU generata dai residenti del distretto urbano considerato, attravero il processo di digestione anaerobica, si produce biogas, che alimenta diverse unità di micro cogenerazione per fornire energia alle stesse abitazione che hanno generato il rifiuto. Sono state analizzate tre diverse tecnologie per la micro cogenerazione: celle combustibili ad ossidi solidi (SOFC), micro turbine a gas e motori Stirling. Un secondo obiettivo del lavoro di tesi è stato quello di dimostrare la fattibilità del ciclo chiuso 'rifiuti – energia' a livello di distretto urbano residenziale.

Al fine di raggiungere questi obiettivi, è stata applicata la metodologia di valutazione del ciclo di vita (Life Cycle Assessment) con l'espansione dei confini del sistema, considerando due casi di studio: il municipio di Greenwich nella zona di Londra (UK) e il comune di

Livorno (IT), nella regione Toscana. Le due aree sono state confrontate in termini di produzione di energia e strategie di gestione e trattamento dei rifiuti. L'analisi si è basata sulla costruzione di un modello rappresentante il sistema descritto, e sulla valutazione comparativa di due sottoinsiemi di scenari: il primo riguardante diversi scenari per il trattamento dei rifiuti, in cui sono stati analizzati trattamenti alternativi per la FORSU, e un secondo insieme di scenari riguardanti la generazione distribuita, in cui sono state analizzate le tre diverse tecnologie per la micro-cogenerazione con tre alternative in termini di strategia operativa, considerando due modi diversi di utilizzo del biogas. Inoltre, è stato assunto uno scenario di riferimento per la produzione di energia a livello residenziale, per valutare le potenziali riduzioni in termini di impatto ambientale nei due casi di studio.

I risultati hanno mostrato che, sebbene la digestione anaerobica sia potenzialmente la migliore alternativa per il trattamento dei rifiuti organici in termini di emissioni di gas serra, la quantità di biogas prodotto con questa frazione non è sufficiente a coprire il fabbisogno energetico delle abitazioni che hanno generato i rifiuti stessi. Tuttavia, normalizzando i risultati ottenuti rispetto alla quantità totale di FORSU trattata nelle due aree considerate, trattando i rifiuti attraverso il processo di digestione anaerobica è possibile ottenere un risparmio in termini di emissioni di gas serra uguale a 851 e 856 kg di CO_2 eq nel Regno Unito e in Italia, rispetto all'invio della stessa quantità di FORSU in una discarica con recupero di energia elettrica. Il potenziale risparmio in termini di emissioni si riduce poi a 30 e 35 kg di CO_2 eq per kg di FORSU trattata nel Regno Unito e in Italia, se il paragone è fatto con un impianto di trattamento termico per la cogenerazione.

Un'analisi di sensitività è stata compiuta per valutare la robustezza del modello rispetto a determinati parametri chiave. L'ipotesi più critica è stata la quantità e la 'qualità' dell'energia che è sostituita con l'utilizzo del biogas prodotto.

L'utilizzo di cella a combustibile per la micro - cogenerazione distribuita ha dimostrato grosse potenzialità di riduzione degli impatti ambientali sia nel caso inglese sia in quello italiano, se paragonate agli scenari di riferimento attuali. E' tuttavia difficile prevedere l'ordine di grandezza di queste riduzioni, a causa della variabilità di parametri chiave assunti in fase di progettazione e modellazione delle unità di micro cogenerazione e della loro influenza sui risultati finali, come il rapporto tra produzione di calore ed elettricità sia dal lato della domanda che da quello della fornitura di energia e le strategie di funzionamento adottate. Infine, dai risultati del lavoro di tesi è emerso che l'aspetto più importante è l'individuazione del tipo di tecnologia per la produzione energetica che è sostituita con il nuovo sistema. Questa rappresenta la base di confronto e determina se l'unità di micro cogenerazione rappresenta un risparmio o un altro aumento delle emissioni. La definizione dello scenario energetico futuro in cui il processo sarà collocato è quindi una questione chiave per l'individuazione dei vantaggi reali per l'ambiente, che si possono ottenere con l'introduzione di sistemi rifiuti-energia. Questo, a sua volta, dipende da macro strategie nazionali ed europee e pone l'accento sull'importanza di un approccio olistico per informare i decisori sulla soluzione migliore per le politiche di produzione di energia da rifiuti.

Table of contents

List of figures

List of tables

Nomenclature

AD	Anaerobic Digestion
AP	Acidification Potential
BMW	Biodegradable Municipal Waste
BSI	British Standard Institution
CCGT	Combined Cycle Gas Turbine
CHP	Combined Heat and Power
DEFRA	Department for Environment, Food and Rural Affairs
DG	Distributed Generation
El	Electricity (operating strategy)
EPA	Environmental Protection Agency (US Department)
EU	European Union
FT	Full Thermal (operating strategy)
FU	Functional Unit
GHG	Greenhouse Gas
GWP	Global Warming Potential
HF	Half Thermal (operating strategy)
HHV	High Heating Value
H to P	Heat to Power
ICE	Internal Combustion Engine
IPCC	International Panel on Climate Change
ISO	International Standard Organization
ISPRA	Istituto Superiore per la Protezione e la Ricerca Ambientale

LCA	Life Cycle Assessment
LHV	Low Heating Value
MBT	Mechanical Biological Treatment
MGT	Micro Gas Turbine
MSW	Municipal Solid Waste
NE	Nutrient Enrichment
NMVOC	Non-Methane Volatile Organic Compound
OFMSW	Organic Fraction of Municipal Solid Waste
PEMFC	Polymeric Electrolyte Membrane Fuel Cell
POCP	Photochemical Ozone Creation Potential
PSA	Pressure Swing Adsorption
RHI	Renewable Heat Incentive
ROC	Renewable Obligation Certificate
SDS	Sustainable Development Strategy
SE	Sterling Engine
SETAC	Society of Environmental Toxicology and Chemistry
SGP	Specific Gas Production
SOFC	Solid Oxide Fuel Cell
TS	Total Solids
VOC	Volatile Organic Compound
VS	Volatile Solids
WM	Waste Management
WRAP	Waste and Resources Action Programme

1. Introduction

1.1. Problem statement

Sustainable development means that "*the need of the present generation should be met without compromising the ability of future generations to meet their own needs"* (Burtland, 1987) It incorporates different global challenges and integrates economic activity with environmental protection and social concerns (Azapagic, Emsley & Hamerton, 2003; Azapagic, 2004). Humanity faces a number of challenges such as rapid depletion of finite natural resources and diminishing capacity of the planet to absorb impacts resulting from various human activities. In many geographical zones the carrying capacity of the planet has already exceeded its limits (Clift 2006). The European Union set up a framework for a long-term vision of sustainability in which economic growth, social cohesion and environmental protection go hand in hand and are mutually supporting. The *Renewed European Sustainable Development Strategy* (European Union, 2006) defined a strategy around seven key challenges to reconcile human activities with the carrying capacity of the planet. Above all, it identified climate change and clean energy as the two key challenges, with the aim to minimise the costs and negative effects of climate change on environment and society. In the EU, the energy sector is the largest contributor of greenhouse gas (GHG) emissions, accounting for more than 70% of the total; therefore measures to reduce emissions arising from this sector are at the centre of many climate change mitigation strategies (EUROSTAT, 2011a).

Promoting sustainable consumption and production as well as sustainable waste management options are other important points of the EU *Sustainable Development Strategy* (SDS). The Statistic report for 2011 reveals an increase in generation of municipal solid waste (MSW) in the European Union[1] compare to 1995 level. Municipal waste is the most important non-mineral waste category in terms of political importance despite the fact that it accounts for only 10% of the total waste generated in Europe in 2009 (Blumenthal, 2011). Its relevance is so high because of its composition and its link to

[1] Considering 27 countries.

consumption patterns and waste generators. In spite of increase in waste generation in EU-27, the amount of municipal waste sent to landfill has been reduced since 1995. This is due to the implementation of several European legislation and directives such as the *Waste Directive* (European Parliament, 2008) and *Landfill directive* (European Parliament, 1999). The latter, in particular, requires Member States to reduce the amount of biodegradable municipal waste sent to landfill. Among the so-called old Member States, UK (50%) and Italy (45%) reported the fifth and sixth highest landfill rates in 2008. In order to minimize the environmental impacts of waste, the *2008/98/EC European Framework Directive* on waste introduced a waste management hierarchy. In this directive, incineration without energy recovery and landfill are considered the least environmentally friendly methods.

The efforts in transforming the energy sector and diversifying the sources have changed the perception of waste (Fruergaard 2010). Even if prevention is the first option in the Waste Management Hierarchy (European Parliament, 2008), waste can play a significant role in meeting climate change reduction targets in terms of renewable energy production and associated environmental benefits.

Waste is produced at different levels: households, restaurants, hotels, hospitals, commercial facilities, industries. It is responsibility of local authorities to develop sustainable waste management systems, particularly in regards to municipal solid waste .Optimising waste management system and promoting reuse and recycle of waste can bring several advantaged to local communities in terms of economic, environmental and social benefits. Designing a waste treatment facilities that meet the needs of a particular community can provide energy to the same community, creating a *waste - to - energy closed loop* (Porteous, Jones, Frith, *et al.*, 2003).

This perfectly matches the *distributed generation* paradigm. The term distributed generation (Ackermann et al, 2001) indicates that power and heat are produced at, or near to, the points of use using small power unit when compared to the usual plants used for large scale production. Application of distributed generation concept means that in the future a local community could be self-supported in terms of electricity and energy for heating and cooling. This is in contrast with the current facilities that concentrate the production of products, thermal and electrical energy in large plants operating in a central location and connected with the final users via transmission and distribution networks. The centralized generation paradigm shows strong limitations due to the vulnerability of complex systems and the scarcity of the fossil fuels commonly used in large scale plants. On the other hand, small scale plant can advantageously use fuels locally available, often produced by Renewable Energy Sources, e.g. biogas from waste and wood biomass, improving the sustainability of the power generation. In particular, amongst all energy sector users, residential sector accounts for nearly one quarter of the total GHG emissions in Europe in 2010 (EUROSTAT, 2012c). Therefore, carbon reduction measures in this sector are of particular importance (Hawkes 2010).

A range of micro-generation technologies exists: systems which directly harness renewable energy from the natural environment - such as small wind turbines, hydro plants, solar thermal collectors or PV systems, or systems which generate heat and power form a fuel – such as internal combustion engine, Stirling engines, micro turbines, fuel cells.

Fuel cells are electrochemical energy conversion devices that convert chemical energy in fuel directly into electricity and heat without combustion process (Hawkes et al. 2009a). They are very promising energy conversion devices. They show a very high electrical efficiency, contain no moving parts and produce zero emissions at the point of use. Low temperature Polymeric Electrolyte Membrane Fuel Cell (PEMFC) and high temperature Solid Oxide Fuel Cells (SOFC) are amongst commercially available fuel cell types that are considered reliable for the application in micro-CHP generation (Calise, Dentice d'Accadia, Palombo, *et al.*, 2006). Despite a very low capital cost, PEMFC requires pure hydrogen to be fed. Therefore, bio-fuel such biogas must be converted to H_2-rich gas before feeding the PEMFC using reformer and CO removal, which dramatically increases system's complexity and decreases the overall electrical efficiency. On the other hand, SOFC are very flexible in burning several type of fuels due to the high working temperatures (700-1000°C) that allow them to convert hydrocarbons into hydrogen internally with an overall electrical efficiency of approximately 55-60%, even when fed by methane (Shiratori, Ijichi, Oshima, *et al.*, 2010; Larminie & Dicks, 2003), the absence of moving component (reducing mechanical stresses, noises and vibrations) and small efficiency reduction when working in off-design conditions. Moreover, this efficiency can be increased to up to 70% (Calise, Dentice d'Accadia, Palombo, *et al.*, 2006) when SOFCs are coupled with gas turbine or organic Rankin cycle turbine in hybrid cycles. Due to their modularity, SOFCs can be easily integrated in combine heat and power plants (CHP), reaching very high global efficiency (around 80 %) (Liso et al.

2011). On the other hand, technology of fuel cells (and of SOFC in particular) is not yet mature and many developments have to be done to increase reliability of such equipment before it will be possible for SOFCs to become economically competitive with ICE alternatives.

Aiming to develop a waste-to-energy closed loop in the residential sector, European Member States, and in particular UK and Italy, are investing in developing strategies for CHP technologies to penetrate into local energy schemes. However, consumers want to make the right environmental choices when paying for services or buying products, and policy makers want to promote sustainable consumption and production to respond to national and international environmental challenges (European Union, 2010). In the *Communication on Integrated Product Policy* (European Commission, 2003), the European Commission concluded that Life Cycle Assessment (LCA) provides the best framework for assessing the potential environmental impacts of products or services currently available. Taking a life cycle perspective helps to avoid burdens shift when decision makers have to evaluate different key strategies.

1.2 Objective of research and research questions

Municipal Solid Waste can be an alternative source of energy to fossil fuels and can reduce CO_2 emission as well as assist in meeting European and National targets in terms of Renewable Energy requirements. In particular, this thesis investigates the production of biogas via anaerobic digestion of the Organic Fraction of Municipal Solid Waste (OFMSW). In this process, the bio-waste is metabolized by bacteria under anaerobic conditions producing a gas – i.e. biogas, composed by 60% of CH_4 and 40% of CO_2, with traces of H_2S, N_2, O_2, and H_2O. Digestate, a by-product of the AD process can be used as a valid substitute to commercial fertilisers. Production and collection of OFMSW usually takes place at district level, hence making it a great potential as non-seasonal energy feedstock. Moreover, using OFMSW for energy production reduces the amounts of waste sent to landfill sites near urban areas where land is at its premium. Biogas obtained from the anaerobic digestion of OFMSW can be used as a fuel for CHP units through several energy systems.

A number of studies on micro-CHP systems for residential applications has been published investigating carbon footprint (Cockroft & Kelly 2006; A. Hawkes & M. Leach 2005; a. D. Hawkes et al. 2009a; a. D. Hawkes et al. 2011) and Life Cycle Assessment of SOFC related applications including analysis of manufacturing and use phase of a fuel cell (Giannopoulos & Founti 2011a; Halliday et al. 2005; Hawkes et al. 2007; Lunghi et al. 2004; Pehnt et al. 2003; Pehnt, 2008; Pöschl et al. 2010; Staffell & Ingram 2010; Staffell et al. 2011). Many works have also been published on the life cycle assessment of waste management systems and waste to energy solutions (Fruergaard et al. 2009; Patterson et al. 2011; Eriksson et al. 2007; Pöschl et al. 2010). However, full investigation on the entire life cycle of a waste-to-energy system in a distributed energy generation paradigm is still missing.

The aim of this thesis is to evaluate the environmental impact of a *waste-to-energy* system in a distributed generation paradigm. It investigates a system in which the OFMSW is used to produce biogas, which is fed to different micro CHP units to generate energy for those dwellings that generated that waste. A secondary objective of this work is to demonstrate the feasibility of *the waste-to-energy closed loop* at residential level. In order to achieve this, a Life Cycle Assessment is performed considering two case studies: the Borough of Greenwich in the Greater London area (UK) and the municipality of Livorno (IT), in Toscany region. The system under analysis is a multifunctional process. It deals with a waste management process and energy generation from waste: the boundaries of the system are expanded to consider the avoided emissions due to the production of energy and other alternative waste treatment options.

The main research questions of this work are the followings:

1. *How much CO_2 it is possible to save when bio-waste is diverted from landfill and sent to an anaerobic digestion plant.*

   In order to answer this question, alternative waste treatments for the OFMSW are investigated from a life cycle perspective, such as incineration with energy recovery and landfill with electricity production.
2. *What it is a reliable alternative for the use of bio-waste for micro CHP applications.*

   In order To answer this question, three different micro-CHP systems that supply energy at residential level are analysed. Two different ways of using biogas are investigated and three different operating strategies for the micro-CHP units are taken into consideration.

3. *How relevant is, the geographical context when we talk about waste management and energy strategy in terms of environmental impact,.*
   Two different geographical contexts within the European Union are considered as case studies: Italy and United Kingdom. They are compared in terms of energy generation and waste disposal strategy and conclusions on the best framework for the penetration of waste-to-energy closed loop systems are given.
4. *How much CO_2 it is possible to save with fuel cells fed with biogas in UK and Italy.*
   In order to answer this question, a comparative approach is taken in this study. A reference scenario for the production of energy at a residential level is assumed in two case studies and possible reductions in terms of CO_2 equivalent are evaluated.

1.3 Outline of the research

The framework of this work is related to Renewable Energy and Waste Disposal targets that European Countries have to meet in the next few years. Chapter 2 provides an overview of the main concepts adopted as a background for this work. Section 2.1 analyses the policy framework and the state of the art in waste disposal and micro generation in United Kingdom and Italy. The concept of distributed generation and related existing technologies are briefly described in Sections 2.2 and 2.3 respectively. In particular, Section 2.3 focuses on the description of the main features of three technologies chosen in this work for the assessment: Solide Oxide Fuel Cells, micro Gas Turbine and Stirling Engine. Section 2.4 investigates the importance of the Heat to Power ratio of the micro-CHP systems alone and in relation to the H to P of the dwelling. Section 2.5 introduces a summary of the main concept for a potential economic evaluation of the three technologies, considering the framework of incentives and supporting policies in the two countries. Section 2.6 introduces the state of the art in waste disposal presently implemented in the two areas, with a focus on anaerobic digestion process and use of the digestate as a by-product.

Chapter 3 focuses on the approach followed in this thesis. First, it summarises the basic concepts of Life Cycle Thinking and Assessment (Section 3.1). Section 3.2 presents a critical review of the state of the art in LCA studies applied to waste management and micro CHP applications. In Section 3.3 the approach taken respect to the renewable carbon – a topic of much debate amongst the LCA practitioners' community – is critically discussed and the position taken in this study is presented. Overall, chapter 3 highlights the methodological approach of the author in conducting this LCA.

Chapter 4 presents the methodology of the study conducted and provides the answers to the research questions. A detailed description of assumptions used in this study is provided in accordance with LCA standards. The overall goal and scope definition of the LCA is stated in Sections 4.1-4.3, along with the presentation of the two case studies (Section 4.4), the time perspective (Section 4.5) and the technological scope (Section 4.6). The environmental impact categories chosen for the scope of this work are presented in Section 4.7, while Section 4.8 introduces the software tool used to carry out an LCA. In order to analyse the concept of *waste-to-energy closed loop*, the study is divided into two sub-systems. Section 4.8 presents the systems developed to answer to the research question number one: different alternatives for the waste treatment are investigated in a LCA perspective in the two different countries. Section 4.9 is focused on the system developed to answer the research question number two: three micro Combined Heat and Power technologies are investigated to supply energy at residential level. Moreover, three operating strategies for the designing of the micro-CHP technologies are analysed when fed by OFMSW-based fuel. Finally, Section 4.11 reports the Life Cycle Inventory: all the data used for the LCA are described along with the range of different parameters found in the literature.

Chapter 5 presents the results of the LCA. The data is presented for the two sub-systems introduced in chapter 4 and presented separately for a specific case study in order to answer the question number three. Section 5.2 presents the results of the Waste Management scenarios, while Section 5.3 is dedicated to the Distributed Generation scenarios. Section 5.4 provides the sensitivity analysis carried out to test the robustness of the developed model and to identify key parameters that have the highest influence on the overall results. Section 5.5 presents the results of a further investigation carried out on the uncertainties related to LCA of waste management systems by comparing different landfill process models presently discussed in the literature. In fact, the behaviour of material in landfill remains the most uncertain part of the LCA applied to waste management due to uncertainties associated with the emissions produced over time and the challenge of modelling the decomposition of different types of materials.

Chapter 6 critically discusses results obtained for the two sub-systems, providing the answer to the question four. In this chapter the results are compared with findings from the literature. Although LCA results are strongly related to the specific boundaries of the study (which usually change from one study to another), it is possible to express the environmental impact of the system in terms of CO_2 savings per dwelling.

Finally, Chapter 7 draws the conclusions of the work, emphasizing the research questions presented in Chapter 1 and provides recommendations for further work.

2. Background

2.1 Waste management and micro-CHP development: policy framework and state of the art in Europe

2.1.1 Introduction

In this Section the European policy framework in terms of management of bio-waste and micro-CHP development is presented, along with the state of the art in European member states.

Almost 3 billion tonnes of waste were generated in the EU‑27 in 2010, which is 5 tonnes per capita. The quantity of waste generated amongst the Member States reflects differences in the economic structure and consumption patterns as well the different degree of implementation of waste prevention policy. However, differences between countries should be regarded with caution since these could also be caused by the methodologies used for the collection of data so far (EUROSTAT, 2010).

Waste policies in the European Union have been put in place since the 1970s. At present, it comprises three main elements. The European Union's waste legislation comprises three main elements. A horizontal legislation establishes the overall framework for the management of waste, including definitions and principles. Legislation on treatment operations, such as landfill or incineration, sets technical standards for the operation of waste facilities. Legislation on specific waste streams, such as batteries, packaging waste, end-of-life vehicles and waste electrical and electronic equipment, includes measures directed towards increasing recycling or reducing hazardousness.

Waste includes many different types of items and substances. Each kind of waste stream has its own characteristics which have different pressures on the environment and on human health. 68% of the waste generated in the EU‑27 in 2006, or almost 2 billion tonnes, was mineral and solidified waste, which comes mainly from mining/quarrying activities and construction/demolition activities. This type of waste also includes combustion waste, mainly from the production of energy (158 million tonnes), which alone accounts for 5 % of waste generated in the EU‑27 in 2006 (EUROSTAT, 2010). Figure 2.1 shows the type of waste generated per percentage of total waste.

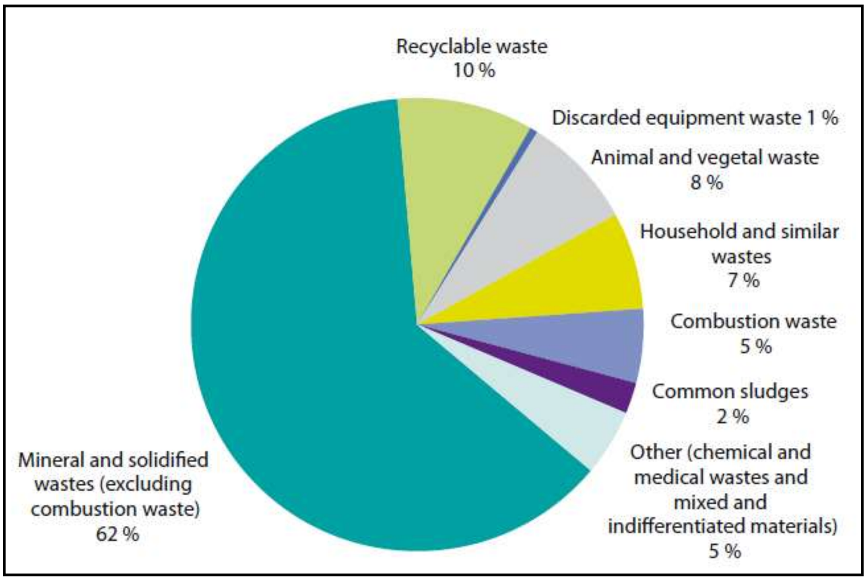


Figure 2.1. Waste generated by type in EU-27, 2006 (EUROSTAT, 2010).

Waste collected by municipal authorities includes all the waste collected and disposed of through the municipal waste management system. Municipal waste consists of waste generated by households and other wastes, which are similar in nature and composition, collected and managed by or on behalf of municipal authorities. The bulk of this waste stream is from households, though similar wastes from sources such as commerce, offices and public institutions are included. It includes many different types of materials including paper, plastics, food, glass and household appliances. This thesis is focused on the organic component of the MSW, directly collected at the source.

2.1.2 State of the art of Bio-Waste Management and biogas production in Europe

The most important European legislations and policies that affect the management of bio-waste, and that are important for the purpose of this study, are the following:

- the target of the *Landfill Directive* regarding the reduction of biodegradable municipal waste going to landfill and that obliges Member states to reach the target of 35% reduction compared to the 1995 levels by 2016 (European Parliament, 1999);
- the emission limit values of the *Waste Incineration Directive* (European Parliament, 2000);

- the provision of bio-waste and on end-of waste criteria in the *Waste Framework Directive* (European Parliament, 2008);
- the *Nitrate Directive*, which imposes limits on N loads on farmlands, that can affect the application of compost and digestate to land (European Parliament, 1991);
- the EU policy for Renewable Energy, which affects the incentives for the use of bio-waste as a renewable energy resource (European Parliament, 2009a).

Presently, the Waste Management in EU is characterized by 38% of Landfilling, 20% of Incineration, 22% of recycling and 20% of composting, as shown in Figure 2.2. UK and IT show a higher landfilling rate compared with the European average: 50% and 45% respectively.

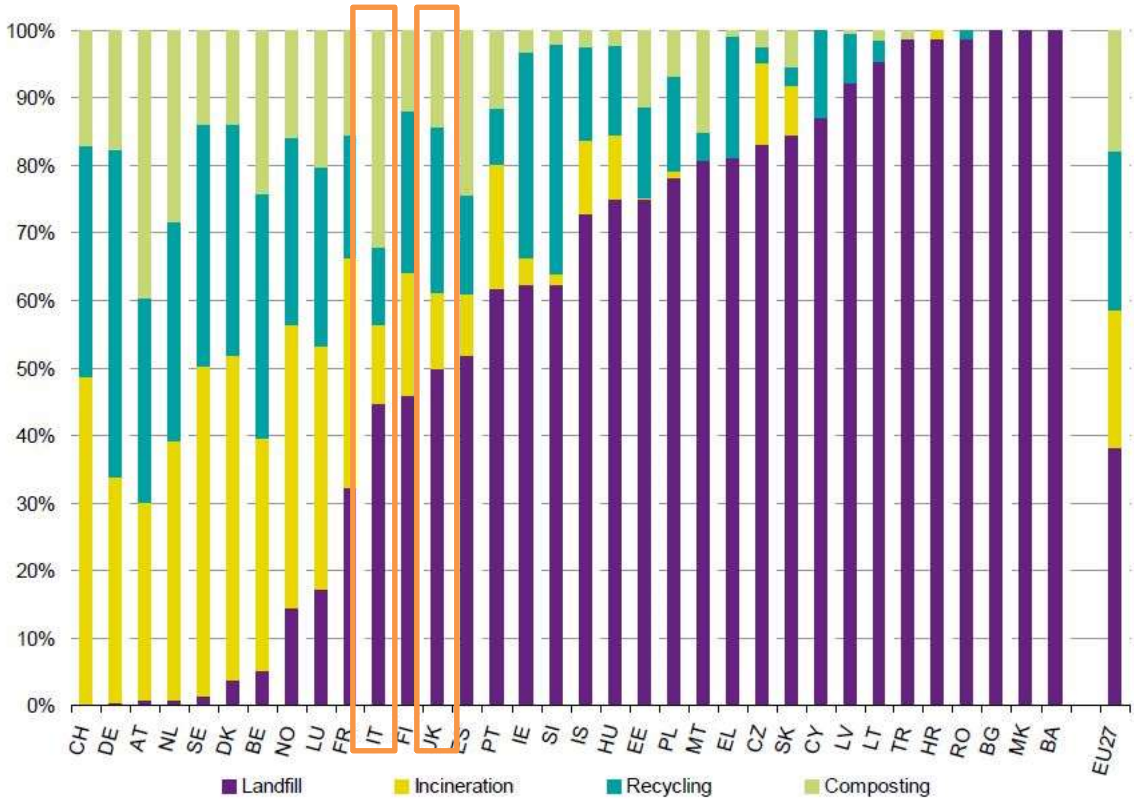


Figure 2.2. Waste Management in Europe in 2010 (EUROSTAT, 2011b)**.**

Organic Fraction of Municipal Solid Waste (OFMSW) is a type of *bio-waste*. The concept of bio-waste as used here reflects the definition of the Waste Framework Directive and it is different from the concept of *bio-degradable waste* as defined in the Landfill Directive. Indeed, the Waste Framework Directive defines bio-waste as '*biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and comparable waste from food processing plants*', whilst bio-degradable waste is defined in the Landfill Directive (European Parliament, 1999) as '*any waste that is capable of undergoing anaerobic or aerobic decomposition, such as food and green waste, and paper and paperboard*'.

In the European Union, between 118 and 138 million tons of bio-waste are produced every year, about 70% of which is in Municipal Solid Waste. It is estimated that this quantity is likely to increase by 10% by 2020 (EUROSTAT, 2011b). On average, approximately 40% of bio-waste in EU is still being sent to landfill. Therefore the treatment of bio-waste remains a big challenge, recognised in three different European policy papers: *Soil Thematic Strategy* (European Parliament, 2006), where the need to replenish carbon in degraded soil is underpinned; the already cited *Renewables Directive* (European Parliament, 2009a), which considers bio-waste as one of the energy sources to be exploited in moving away from fossil fuels and recommends Member States to consider AD as part of measures to meet their binding national renewable energy target for 2020; and the *Waste Framework directive* (European Parliament, 2008). A number of various waste management technologies are currently adopted by different EU Member States, as showed even in Figure 2.2. Sweden for example, relies heavily on incineration and material recovery to divert waste from landfills and employs other advanced technologies for the biological treatment of organic waste. Other countries, such as Austria, achieve high materials recovery and composting rates and therefore incineration rate is lower. Less developed countries, such as Romania, still heavily rely on landfill due to a lack of more complex waste management technologies. Bio-waste can be converted to electricity, heat or transport fuels at relatively low cost, thus limiting the use of fossil fuels and insuring energy security (European Commission, Joint Research Centre & Institute for Environment and Sustainability, 2010). Figure 2.3 summarises the different options for the treatment of OFMSW, highlighting weakness and strength points of each technology.

Figure 2.3. Options treatment for the OFMSW (WiP - Renewable energies 2013)**.**

In 2010, the EU released a report made by two consultant groups on the assessment of different policy scenarios for the treatment of bio-waste in the European Union (ARCADIS, 2010). In this report, a baseline scenario was developed based on the execution of the present policies on bio-waste management in the EU. Following this scenario, the authors of the report identified the generation of bio-waste by 2020 (Figure 2.4) and the trend of bio-waste management options (Figure 2.5) in the future.

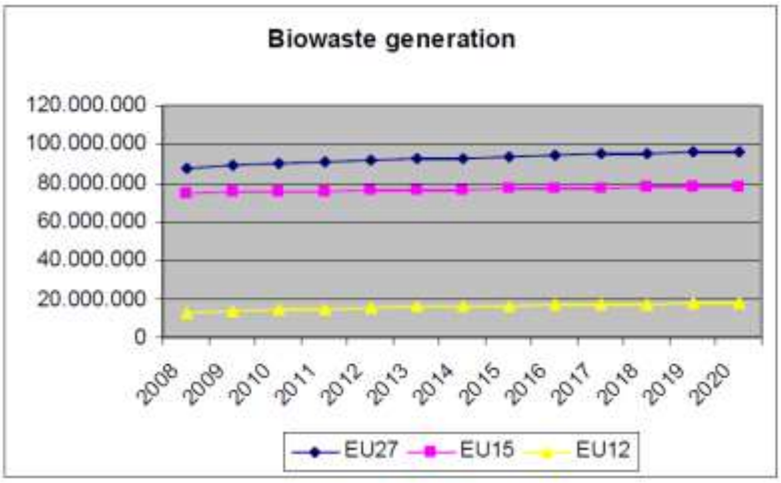


Figure 2.4. Generation of Bio-waste in EU for the next years (ARCADIS, 2010)

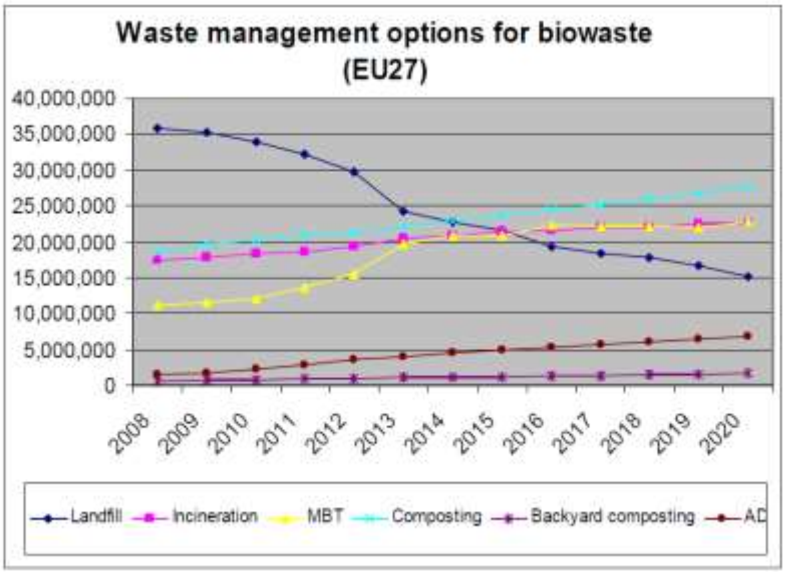


Figure 2.5. Waste management options for bio-waste in EU for the next year (ARCADIS, 2010)**.**

As showed above, the bio-waste is not expected to grow (or grow slightly) in the next years (see Figure 2.4). In terms of Waste Management options, landfilling of bio-waste is expected to decrease –mainly due to EU and nationals legislations, while composting, incineration and Mechanical Biological Treatment (MBT) are expected to be the first options (see Figure 2.5) in the future. Amongst other things, in fact, the Landfill Directive requires member states to develop a strategy to reduce the quantity of bio-degradable municipal waste sent to landfill to 75% (2006), 50% (2009) and 35% (2016), compared with the amount of bio-degradable municipal waste landfilled in the base year (1995) (Hill *et al.*, 2011). Regarding Anaerobic Digestion (AD), the biological treatment is supposed to increase rapidly in the next years, and it is expected to reach around 7,000,000 tons of bio-waste treated per year in 2020.

Organic fraction of municipal solid waste can be collected separately from other waste streams and transported to the centralised facilities. Organic waste from food industries and restaurants can also significantly contribute to the potential increase of biogas generation in EU.

As reported by Eurobser'ER (Eurobser'ER, 2012) in their *Biogas Barometer*, in 2011 the primary energy production from biogas was steady in the European Union Member states, excluding Germany[2]. In the previous year, thus between 2010 and 2011, the primary energy production from biogas grew by 19.3%, corresponding to 812 ktoe[3] of biogas added in EU. At the present, the biogas production comes mainly from purpose-designed energy recovery plant – equal to 56.7%, while 31.3% is producing from landfill biogas plant and the rest (12%) from wastewater treatment plant. Landfill biogas is mainly produced in UK, France, Italy and Spain, whereas purpose-designed energy recovery plant dominates in Germany, Dutch, Czech, Austrian, Belgian, Danish, Luxembourg and many of Eastern Europe's markets. Cogeneration, thus electricity and heat production, are the main forms of biogas recovery in EU. The electricity production increased by 18.2% between 2010 and 2011, while the heat production increased by 16% over the same period. Presently, most of the heat produced is still used directly on site to satisfy heat demand of the plant (drying sludge, heating buildings and maintaining the digester at the designed process temperature). In 2010, again as reported by the EUrobserv'ER report, biogas electricity production was over the target of 28.7 TWh set by the European Member states under their National Renewable energy action plans[4]. The same is expecting for 2011. Heat recovery as well appears to be in line with the 2015 and 2020 objectives.

In order to foster the development of anaerobic digestion, Eurobserv'ER experts suggest to European countries to put efforts through incentives and lifting regulatory barriers, as implemented by UK (see Section 2.1.4). Italy is identified as one of the most promise countries for the future AD development. One of the key factors identified for the developing of AD, it is the improvement of energy efficiency of the biogas plant. Until now, the sector has been driven mainly by incentives linked to electricity production, but there are evidences that show the importance of emphasising the heat recovery potential of biogas production, in line with the UK's current Renewable Heat Incentives (RHI) regulation. Another application that will develop in the next years is biogas injection into natural gas

[2] Germany, the main producer of biogas in EU in 2011, changed its primary energy calculation method for small cogeneration plants. In November 2013 a statistical review will be made of 2010 by Eruobserv'ER, to clarify the actual sector trend. Here, as suggested by the report, the biogas production from Germany is not taken into account (Eurobserv'ER, 2012).

[3] Kilo tons of oil equivalent.

[4] Article 4 of Directive 2009/28/EC on Renewable Energy requires Member States to submit national renewable energy Action Plans, that provide detailed roadmaps of how each Member State expects to reach its legally binding 2020 target for the share of renewable energy in their final energy consumption.

grid, allowing the gas to be stocked and used remotely from the production site. Some countries in EU already have set up a regulation enabling biogas to be injected into the grid.

Table 2.1. Gross electricity production from biogas in EU in 2010 and 2011 (Eurobser'ER, 2012)

	2010			2011*		
Pays/Country	Centrales électriques seules/ Electricity only plants	Centrales fonctionnant en cogénération/ CHP plants	Total/ Total	Centrales électriques seules/ Electricity only plants	Centrales fonctionnant en cogénération/ CHP plants	Total/ Total
Germany	14 847,0	1 358,0	16 205,0	10 935,0	8 491,0	19 426,0
United Kingdom	5 137,0	575,0	5 712,0	5 098,0	637,0	5 735,0
Italy	1 451,2	602,9	2 054,1	1 868,5	1 536,2	3 404,7
France	756,0	297,0	1 053,0	780,0	337,0	1 117,0
Netherlands	82,0	946,0	1 028,0	69,0	958,0	1 027,0
Spain	536,0	117,0	653,0	709,0	166,0	875,0
Czech Republic	361,0	275,0	636,0	535,0	394,0	929,0
Austria	603,0	45,0	648,0	555,0	70,0	625,0
Belgium	149,0	417,0	566,0	158,0	442,0	600,0
Poland	0,0	398,4	398,4	0,0	430,0	430,0
Denmark	1,0	352,0	353,0	1,0	342,0	343,0
Ireland	184,0	22,0	206,0	181,0	22,0	203,0
Greece	190,5	31,4	221,9	37,6	161,7	199,3
Hungary	75,0	21,0	96,0	128,0	55,0	183,0
Portugal	90,0	11,0	101,0	149,0	11,0	160,0
Slovenia	7,2	90,2	97,4	5,7	121,0	126,7
Slovakia	1,0	21,0	22,0	39,0	74,0	113,0
Latvia	5,9	50,8	56,7	0,0	105,3	105,3
Finland	51,5	37,8	89,2	53,6	39,4	93,0
Luxembourg	0,0	55,9	55,9	0,0	55,3	55,3
Lithuania	0,0	31,0	31,0	0,0	37,0	37,0
Sweden	0,0	36,4	36,4	0,0	33,0	33,0
Romania	0,0	1,0	1,0	0,0	19,1	19,1
Estonia	0,0	10,2	10,2	0,0	17,0	17,0
European Union	24 528,2	5 803,0	30 331,2	21 302,4	14 554,1	35 856,4

* Estimation. Estimate. Les décimales sont séparées par une virgule. Decimals are written with a comma. **Source: EurObserv'ER 2012.**

2.1.3 Micro cogeneration in Europe

The *Cogeneration Directive* 2004/08/EC (European Parliament, 2004) outlines a policy framework for the European Union to expand the deployment of cogeneration in Member States. The climate agenda, which has grown in importance since 2004, has added further impetus to the wider use of cogeneration. One of the main achievements of the Cogeneration Directive has been to codify for Europe what is meant by high efficiency cogeneration. Using the framework of the Cogeneration Directive, promoting cogeneration to meet additional electricity needs gives a Member State a quantifiable primary energy saving per unit of electricity generated.

The use of CHP system presents a potential for reducing environmental impact and increasing energy efficiency. As reported by the European Environment Agency in its assessment on Combined Heat and Power (EUROSTAT, 2012a), the preferred fuel in Europe for new CHP plants is natural gas due to its flexibility concerning the heat to power ratio as well as a better environmental performance compared for instance with liquid fossil fuels. In recent years, the development of CHP in EU Member states has suffered from increasing natural gas prices and falling electricity prices, which have diminished the cost competitiveness of these plants. This trend may change in the future given that both prices have now started to rise again (EUROSTAT, 2012b). Other potential barriers for the development of cogeneration technologies include high costs for grid connection to sell surplus electricity, relatively high start-up costs, and the design of the policy framework in some cases. Figure 2.6 shows the percentage of gross electricity generation in 2010 in EU with CHP systems. UK and Italy show the 9^{th} and 17^{th} countries with the lowest percentage of CHP production, respectively.

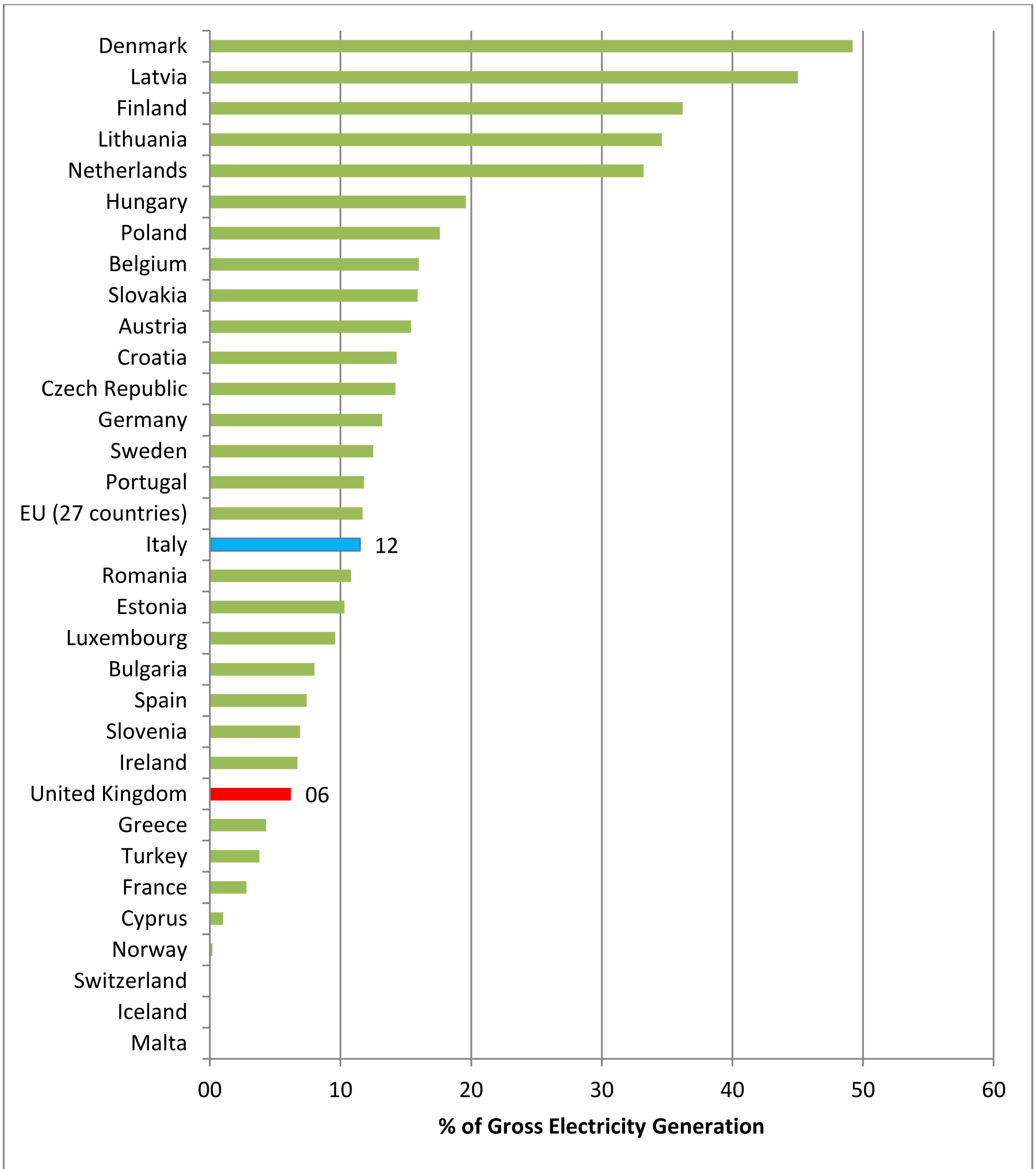


Figure 2.6. Gross electricity generation from CHP systems in Europe in 2010 (EUROSTAT, 2012b)

The trend is confirmed in Figure 2.7. As it is possible to observe here, Italy has increased the percentage of CHP production (electricity) over the last 2 years, while in UK it has decreased over the same period.

An overview on waste management and micro-CHP development policy framework is presented in the next two Sections, for the two case studies.

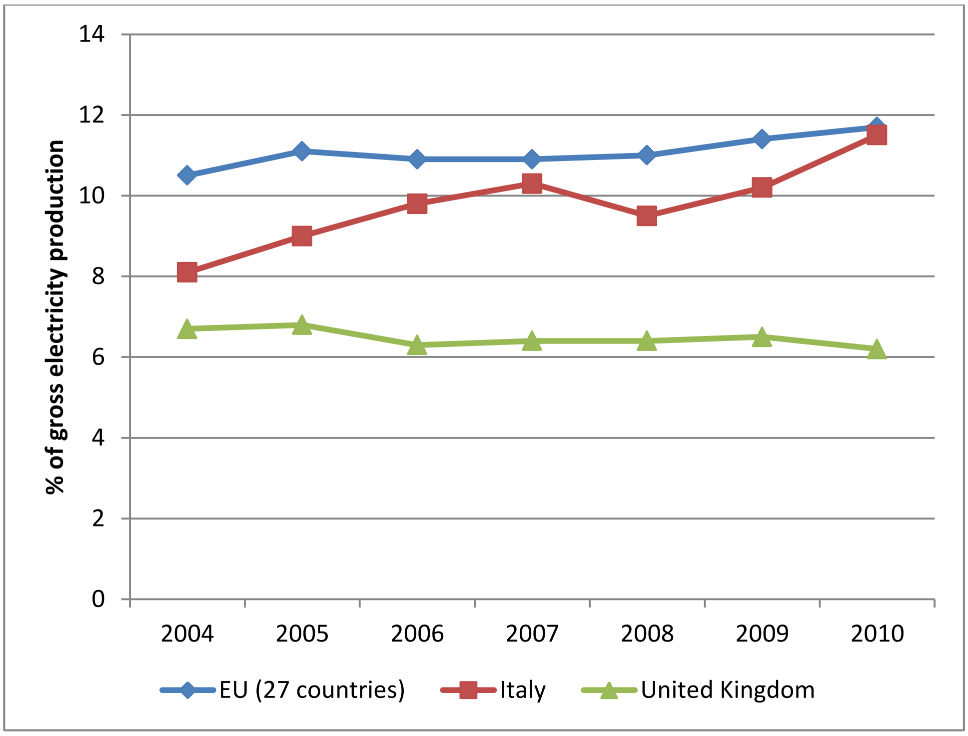


Figure 2.7. Trends of gross electricity production from CHP systems in Europe, Italy and UK, from 2004 to 2010 (EUROSTAT, 2012b)**.**

2.1.4 UK

Waste

UK has a large potential to increase biogas production through centralised and larger scale plants, with food waste being the main feed for the digesters. It is estimated that more than 8 million tonnes of food waste is generated by households in the UK per year (Quested & Johnson, 2009). The UK Government's *Structural Reform Plans* of July, 16, 2010 includes an action to '*set out steps to promote increased energy from waste through anaerobic digestion*' (DEFRA, 2010a). The Department for Environment, Food and Rural Affairs (DEFRA) incorporated this and on November, 30, 2010 a Framework Document was published which aims to set out the steps necessary to increase energy from waste through anaerobic digestion (DEFRA, 2011). In the *Anaerobic Digestion Strategy and Action Plan,* it was estimated that the potential for AD deployment for heat and electricity in the UK could be between 3 and 5 TWh by 2020. Moreover, to increase the development of AD technology in the UK, the Government, through the Waste and Resources Action Programme (WRAP), set up a new loan fund to help stimulate investment in AD infrastructure with a total of £10m over 4 years (DEFRA, 2011).

Moreover, in 2010 WRAP and Renewables East[5] sponsored the publication of a Publicly Available Specification (PAS), in collaboration with the British Standards Institution. The PAS 110 covers '*whole digestate from an anaerobic digestion system that accepts only source segregated bio-wastes and/or biodegradable non-waste materials. It also covers liquor and fibre fractions that may be produced by separating whole digestate, after the anaerobic digestion process'* (BSI, 2010). Digestate materials that conform to this PAS are suitable for use as soil improvers, and can be placed on the market for this purpose.

In the UK, landfill biogas is presently effectively supported, with 65% of the total landfill plants being engineered landfill with biogas recovery systems (Hill *et al.*, 2011). As reported by Eurobserv'ER, - in 2011, the number of AD plants in UK rose by 30% (not including AD plants used in the water treatment industry), generating a total electricity capacity of 75 MWe. Nowadays in UK there are 83 plants that use no water waste as feedstock. 61% of those use waste (industrial and not industrial, such as organic fraction of municipal solid waste) as feedstock. In 2005 there were only 2 plants outside the water industry. The last 4 years, in particular, have seen a big development of AD plants with no water feedstock This is a consequence of the new legislation on Renewable Heat Incentives (RHI) which is stimulating the market to invest in this technology (Eurobser'ER, 2012).

[5] Renewables East is the regional renewable energy agency for the East of England (www.renewableseast.org.uk).

Figure 2.8. Number and location of biogas plants (outside the water industry) in UK in 2012 (source *The Official Information Portal on Anaerobic Digestion (www.biogas-info.co.uk)*).

The Renewable Heat Incentive (DECC, 2011) is a financial helps from the UK Government to promote renewable heat. Renewable heat producers and bio-methane produced with size of installations below 200 kW are eligible to receive the incentive. New RHI options are under development and they will be applicable to installation bigger than 200 kWe in the near future.

The main drivers for AD development in UK are, beside the European policies described in Section 2.1.2 and adopted as national legislation:

- *Climate Change Act* (UK Parliament, 2008), where the targets of 34% reduction of GHG by 2020 and 80% by 2050 , based on the 1990 levels, are set;
- *Renewable Energy UK action plan* (DECC, 2012), where a 15% of renewable energy is set by 2020 and it is specified in 12% of Renewable Heat, 30% of Renewable Power and 10% of Renewable Transport Fuel.

At present, the financial instruments available for biogas producers in UK are:

- *Feed in tariff*: for plant smaller than 5 MWe and built after the 15th of July 2009, that assures a 14.7 p tariff for 1 kWhe generated for plant smaller or equal to 250 kWe; 13.6 p/kWhe generated for plants between 250 and 500 kWe and 9.9 p/kWhe for plants between 500 kWe and 5 MW. The export tariff is , presently, 4.5 p/kWhe (DECC & DEFRA, 2012);
- *Capital Grant Aid*: fund from the Rural development programme for England (DECC & DEFRA, 2012) for the construction of the plant;
- *Enhance Capital Allowance*: provides businesses with enhanced tax relief for investments in equipment that meets published energy-saving criteria. With CHP, case by case Certification is needed to ensure support is provided for 'good quality' CHP;
- *Gate Fees*: Charge levied upon a given quantity of waste received at a Waste processing facility (Longden, Brammer, Bastin, *et al.*, 2007);
- *RHI:* as explained above;
- *Renewable Obligations*: based on green certificates and applied to plants bigger than 5 MWe. The certificates are issued to accredited renewable electricity producers and the electricity suppliers, who are under obligation to submit an annual number of certificates deliver this energy to the consumer. Anaerobic digestion is one of the technologies that attract additional aid for production. Until April 2015, a methane digester operator will acquire 2 ROCs/MWh, and then in 2015/2016 the number will drop to 1.9 ROC/MWh and further to 1.8 ROC/MWh in 2016/2017.

The end of the RO system is in sight in UK. In 2014, a new electricity purchasing scheme will be introduced with a set price based on long-term contracts (FiT-CfD, Feed-in-Tariffs with Contract for Difference). Producers will then have the choice of the two systems until the RO scheme ends sometime after 2017 (Eurobser'ER, 2012).

Micro-cogeneration

The UK government has imposed some targets, as explained above, in terms of renewable energy consumption and GHG reduction. The use of micro- Combined Heat and Power (CHP) systems can help meeting these targets, especially at residential level.

The attention is particularly focused on the possibility to produce heat and electricity at the same time. Nowadays heating accounts for 47% of the total UK final energy consumption and for more than three-quarters (77%) of the energy use across all non-transport sectors. In terms of carbon emissions, heating accounts for 46%. The most recent data show that approximately 69% of heat is produced from natural gas. Oil and electricity account for 10% and 14% respectively, solid fuel 3% and renewables just 1.5%. Heat sold, i.e. heat that is produced and sold under a contract (including CHP plants and community heating schemes), accounted for 2% (see Figure 2.9). The possibility to reduce the amount of fossil fuel in this sector is extremely interested.

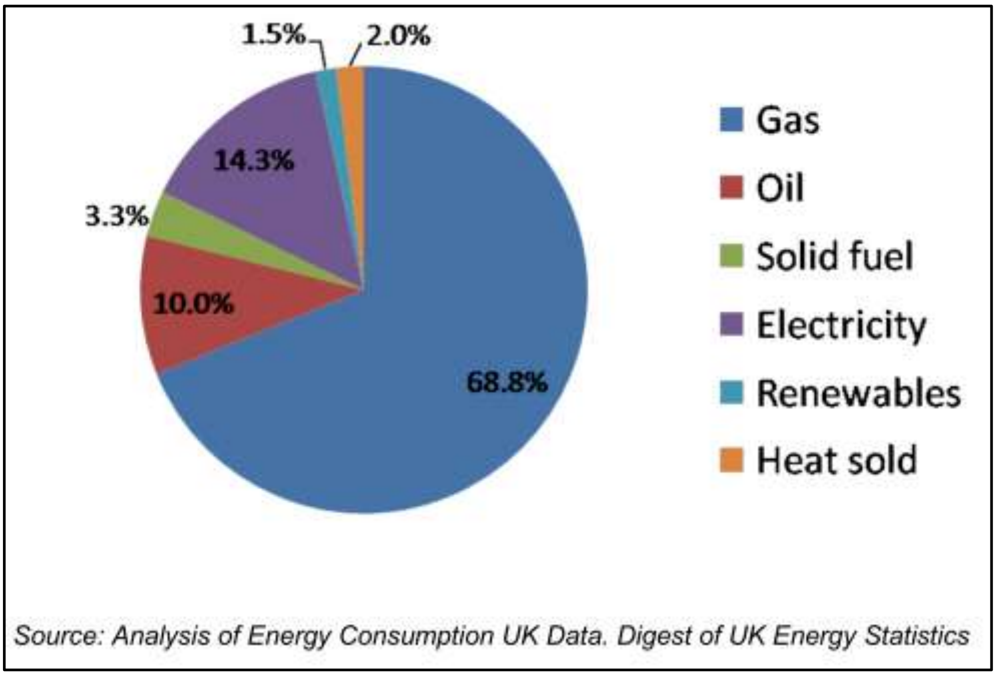


Figure 2.9. Heat use in UK by energy type in 2008 (DECC, 2011)

The UK Government has recognized micro-CHP systems as a strategic component towards the decarbonisation of the residential energy sector, as emphasized in the *Micro-generation Strategy* published by the Department of Energy and Climate Change (DECC). The strategy – one of the few adopted by EU Member States - highlights that *'the UK owns the largest European boiler's market, with 17 million systems currently installed and 1.6 million boilers sold annually*'. The DECC have estimated that the substitution of micro-CHP systems could reduce annual emissions of CO_2 by up to 2.1 tons compared to condensing boilers and electricity grid (DECC, 2011). A practical example is given by the Borough of Woking, which achieved a 49% reduction in energy consumption and 77% reduction in CO_2

emissions between 1991 and 2004. This was a consequence of the installation of photo-voltaic systems and the UK first fuel cell CHP system (200kWe), with a total network of sixty local generators for municipal buildings, social housing and town centre business (Allen, Hammond & McManus, 2008).

The *Carbon Trust*, a private consultancy company based in UK, launched a large field trial for micro-CHP systems in 2006. They tested 72 Stirling engines and 15 ICEs, along with 15 condensing boilers to determine a baseline for potential CO_2 reduction. The systems were installed in different type of houses in UK (Carbon Trust, 2011). The objects of the test were: to determine the real energy demand profile for a typical residential application, recording the energy demand every 5 minutes for 2 years and to evaluate the carbon savings when a micro-CHP unit is installed in replacing of a condensing boiler. Only the full thermal demand operating strategy was investigated. For the latter objective, results showed that an average reduction of 5% of CO_2 emissions was achievable, compared with a traditional system. However, a great variability was recorded, between -4 and 12%, based on single household. The results were better for heat demand bigger than 15,000 kWh per year, allowing a saving of 9% in carbon emissions.

2.1.5 Italy

Waste

In the Energy Position Paper of 2007, the Italian Government estimated a primary energy production from biomass equal to 16.5 Mtep by 2020: 3 Mtep are estimated for electricity final consumption, 9.3 Mtep for heat consumption and 4.2 Mtep for transport final consumption (biofuel) (Italian Government, 2007). Considering the last available data, the consumption of primary energy from biomass in Italy is presently equal to 17.9 Mtoe (EUROSTAT, 2012d). The main sector which is expected to expand is the energy production (electricity and heat) from biogas (Tuccinardi, 2010).

The Italian Government defined the implementation of the first seed of the European Renewable Energy Directive (European Parliament, 2001), in the D.Lgs n. 387/2003 (Italian Parliament, 2003b). In this law, biomass was considered as a renewable energy source and it was defined as '*the biodegradable part of waste or agricultural or agro-industrial residues'*. In this law the Green Certification scheme was set up for all the plants which are producing energy from renewable resources.

In Italy, the management of waste is strongly variable along the country. In the north regions of Italy, in fact, it is possible to identify high recycling rate (up to 70%), while in the south regions the landfilling of the waste is still the most favourable option (ISPRA, 2012). At the present, the 90% of the anaerobic digestion plants treating OFMSW are localized in the north and central regions of Italy. The total amount of OFMSW treated is 892,000 tons per year (Tuccinardi, 2010). The majority of these plants operate in wet regime and they use source-sorted OFMSW as feedstock.

Based on the Eurobserv'ER report, Italy was one of the most '*anaerobic digestion-friendly countries*', thanks to its attractive incentive legislation that allowed energy crop as an input of the plant (Eurobser'ER, 2012). As a consequence of the worldwide economic crises, in 2012 Italian government halved the feed-in tariff for plants smaller than 1 MWe using organic products to € 0.14/kWh by 2013. Plants using organic by-products will also see their payment levels fall by 36.4% to € 0.178/kWh. However, on the other hand, incentives increased for the production of biogas from waste, with an increase by 20% to € 0.216/kWh.

Current legislation has stimulated farming biogas production. According to Terna, Italy's energy transport operator, the number of biogas plants (all sources taken together) rose from 352 (342 MWe) in 2010 to 475 (418 MWe) in 2011 (TERNA, 2012). Most of the growth in biogas output comes from the farming sector. The number of farm digesters has almost doubled, from 114 (54.3 MWe) to 225 (127.6 MWe) and this figure is expected to increase in 2012 as investors hope to take advantage of the improved feed-in tariff. At the same time, Italy is making efforts to increase its biogas recovery from landfills. In 2011 primary energy production from landfills apparently doubled from 349.6 ktoe in 2010 to 755.6 ktoe (Euobser'ER, 2012).

Unfortunately the legislation for biogas plants fed by OFMSW in Italy is less organized and harmonized than the UK one and this causes a slow development of the market. First of all the territorial organization of the collection and management of waste is completely different in the two cases. In Italy the normative refers to Environmental Authorities[6]: they are independent authorities, dedicated to integrated public services, such as water system or waste management (Italian Parliament, 2011a). The areas of interests are identified by

[6] Ambito Territoriale Ottimale – ATO, from the Decreto Legislativo n.22, 5th of February 1997.

the regions with special regional laws, and they act on them with independent Authorities with specific legal state, which organize, control and entrust the management of the integrated services.

In terms of Waste Management, every Environmental Authority has to reach a target in terms of recycling rate; the targets for the energy recovery from waste are less defined by the normative. The D.Leg. n.133 of the 11^{th} of May 2005 (which comes from the European Directive 2000/76/CE) is the only law that sets the limits for the emissions from Waste-to-Energy plants, but no other regulations or strategic plans have been settled (Italian Parliament, 2011a). In fact, in Italy, given the backwardness of the sector, the regulation CIP 6 (Italian Government, 1992) accounted the waste used for the production of energy as a renewable source (at the same level as a biomass) (ENEA, 2009). This changed then during time, with the following directives and normatives:

- DLgs 79/99 and DLgs 87/2003 (Italian Parliament, 1999, 2003a), where the Green Certificates[7] (the same as the Renewable Obligation certificates in UK) have been introduced in the Italian Market;
- Reg. 1069/09 (European Parliament, 2009b) that gives a definition of the OFMSW when treated as “animal by-product”;
- DLgs 152/06 (Italian Parliament, 2006)that gives a definition of the OFMSW as “waste”;
- CER 09.06.04 that gives the definition of Quality Digestate, which allows the digestate to be used as fertilizer.

The financial incentives for Waste-to-Energy systems set up by the Italian Government are mainly based on the European Directive 2001/77/CE (European Parliament, 2001) and referrer to the electricity production from renewable energy (Italian Parliament, 2003b). Moreover, the Dlgs 28 of 03/03/2011 puts into effect the European Directive 2009/28/CE on the promotion of Renewable Energies and defines, along with the D. Mns 5 of 6.07.2012, the tariffs for energy production from Renewable sources. Those are (Rossi, 2011):

[7] Certificati Verdi (CV).

- Feed in Tariff for size plant below of 1 MWe: 28 €cent per kWh produced by OFMSW defined both as "animal by-product" or "waste";
- Green Certificates for size plant bigger than 1 MWe: the GC value has to be multiple by 1.8 if the OFMSW is defined as "animal by-product" and for 1.3 if the OFMSW is defined as "waste".

Micro-cogeneration

The trend of the final energy consumption sectors in Italy is characterized by the strong reduction in the industry sector in the last 4 years, mainly due to the economic crises (Figure 2.10). In 2010, the first sector for final energy consumption was the residential one.

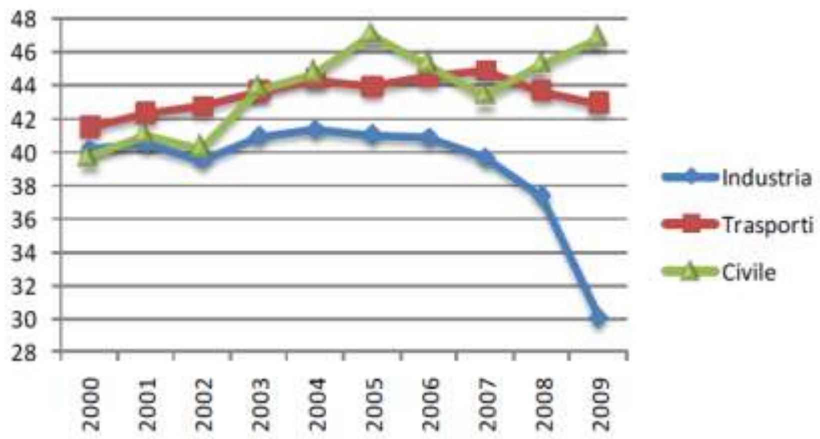


Figure 2.10. Final energy consumption in Italy in 2009 (ENEA, 2010)**.**

In 2009, the majority of final energy was produced from natural gas in the residential sector, as shown in Figure 2.11.

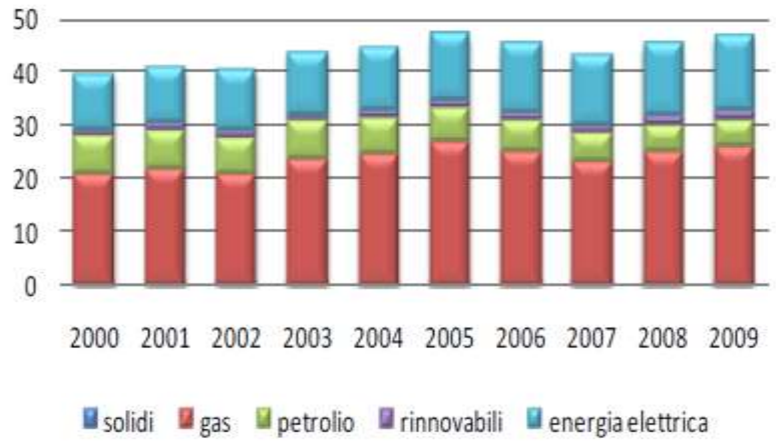


Figure 2.11. Final energy consumption for the residential sector, for each source (ENEA, 2010)

However, the primary energy produced from renewable sources has grown strongly in the last ten years in Italy, reaching the point where, at present, renewable sources alone could satisfy the pick primary energy demand during weekends, along with geothermal and hydrothermal resources (FIRE, 2007).

Micro generation technologies were introduced in Italy in 2005, with the 'TOTEM' model developed by FIAT, an internal combustion engine with 15 kWe and 35 kWth. The spread of this technology was very limited in Italy, due mainly to the opposition of the electricity and natural gas industries. The only successful story is the field trial of the municipality of Vicenza, which installed 31 machines in a residential area. Parts of them are still working thanks to maintenance operations. The installation allowed a 35 % of energy saving on a 5,200 hours per year of operation. Apart of that, very few field trials followed, resulting, nowadays, in a negligible contribution of the micro-cogeneration to the final energy production (FIRE, 2007).

The most relevant regulations for the micro-cogeneration in Italy are summarised below:

- The Italian government translated the European Directive 2004/8/CE in 2007, with the D.Lgs 20 of 8.02.2007 where the "high efficiency Cogeneration" is defined, based on the Primary Energy Saving (PES) index (as requested by the European Directive)[8] (Italian Parliament, 2007);

[8] The PES index identifies the quantity of primary energy saved with the co-generation of electricity and heat compared with the separated production of the same quantity of final energy.

- in the same law (n.d. D.LGs 20) the concept of “Guarantee of Origins” was introduced: a certification for the final producer that states that the energy was produced by a high efficiency cogeneration plant;
- the law 99 of 23.07.2009 defined the economic incentives for a period of 10 years, for the high efficiency cogeneration plants (Italian Parliament, 2009);
- the D. Lgs 28 of 3.03.2011 defines the financial instruments to incentive the production of energy from renewable resources to reach the national targets by 2020 (Italian Parliament, 2011b);
- the D. MSE 5.09.2011 set the new criteria for the definition of High Efficiency Cogeneration as a continuation of the D.Lgs 20 of 8.02.2007, introducing the White Certificates for cogeneration plants (Italian Government, 2011).

2.2 Distributed generation: definition and approach

In the last years, the new trend in power generation drives toward distributed power generation (Alanne & Saari, 2006). The term distributed generation (Ackermann et al., 2001) indicates that the energy conversion units are situated close to energy consumers using small power unit when compared to the usual plants used for large scale production. Several definitions of distributed generation power unit rating exist: from 1 W to 300 MW of capacity, depending of the number of units connected in a modular form (Elkhattam & Salama, 2004). *Distributed Generation* approach should be recognized as the new future power paradigm due to the economic, technical and environmental benefits it achieves (Manfren, Caputo & Costa, 2011). Application of distributed generation means that single urban districts could be, in the future, self-supported in terms of electricity, heat and cooling energy – see picture Figure 2.12.

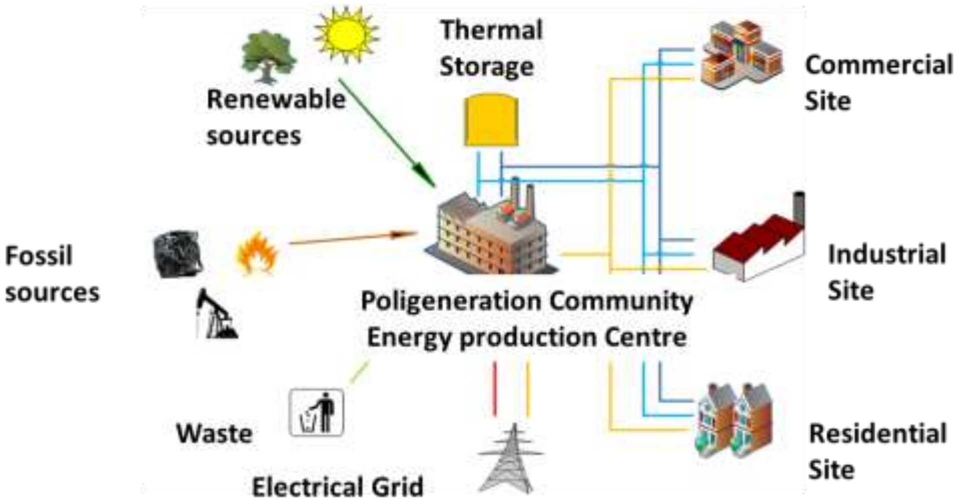


Figure 2.12. Concept schematic of Distributed Generation applied to an urban community.

This is in contrast with the facilities present to date, which concentrate the production of goods, thermal and electrical energy in large plants operating in a central location and connected with the final users via transmission and distribution networks. The centralized generation paradigm shows strong limitations due to the vulnerability of complex systems and the scarcity of the fossil fuels commonly used in large scale plants. On the other hand, small scale plant can advantageously use fuels locally available, often produced by Renewable Energy Sources, e.g. biogas from wastes and wood biomass, improving the sustainability of the power generation. The energy saving benefits of distributed generation paradigm compare with the centralised one are summarised in Figure 2.13 and Figure 2.14 elaborated using literature sources (Vourliotakis, Giannopoulos & Founti, 2011): centralized generation shows overall losses of 65% to the final user – where transmission losses will vary depending on the distance between the loan and generator, while an integration between conventional sources and biowaste coming directly from the urban settlement shows a 90% efficiency system.

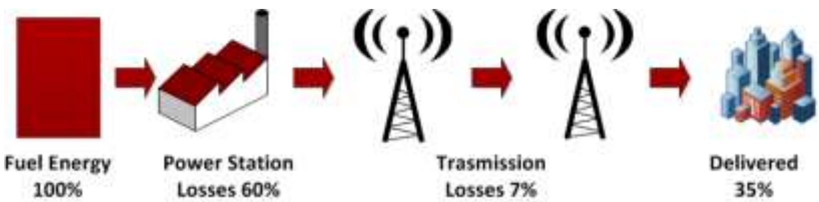


Figure 2.13. Centralised generation paradigm: energy efficiency.

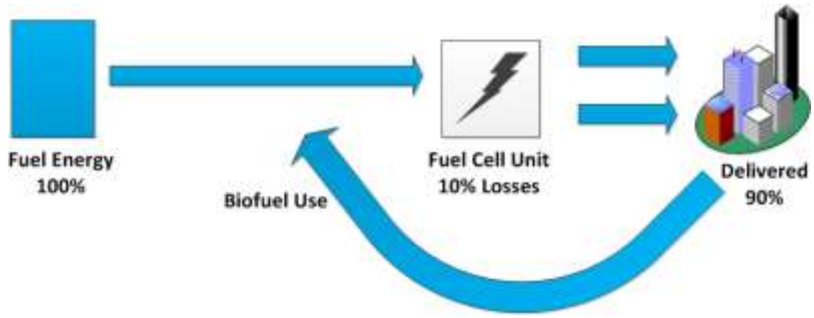


Figure 2.14. Distributed generation paradigm: energy efficiency.

In the European Union, the path toward future energy systems has been clearly underlined with several directives (European Parliament 2009; European Parliament 2006, European Parliament 2002), research initiatives, and short terms energy initiatives (cutting greenhouse gases emissions by 20%, 20% share of RES in UE energy consumption, cutting energy consumption through improved energy efficiency by 20%). The success of DG systems is strongly subjected to their ability to use the waste heat from electricity generation as a heat source, obtaining total system efficiencies up to 90% (Strachan & Farrell, 2006). These applications, commonly called Combined Heat and Power (CHP) can lead to significant reductions of CO_2 emissions.

2.3 Domestic Micro – CHP: concept and technology

2.3.1 Introduction

Micro Combined Hear and Power is the simultaneous production of heat and power in a single building (Pehnt, 2006). The EU CHP directive defines micro cogeneration *'a cogeneration unit with a maximum capacity below 50 kWe'* (European Parliament, 2004), while Pehnt restrict the definition to a maximum of 15 kWe. In the UK, the Section 82 of the Energy Act (2004) defined micro-generation as anything below 50 kWe (OFGEM, 2013), while in Italy the D.lgs 20/2007 (Governo Italiano, 2007) adapts the Italian definition with the European one. The heat produced is then used for space and water heating; the electricity is used within the building or fed into the grid. Figure 2.15 shows the concept of micro CHP applied to a dwelling.

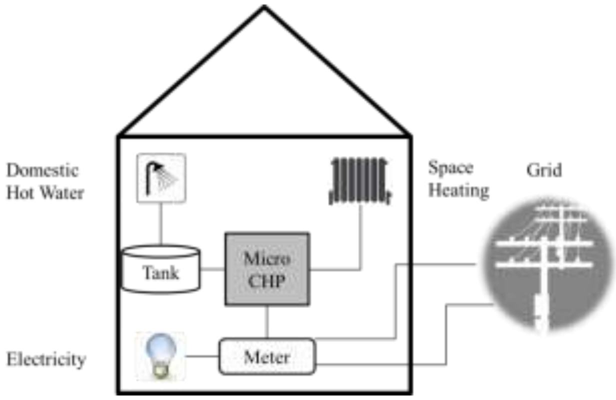


Figure 2.15. Schematic representation of a micro CHP unit for residential use (Hawkes et al. 2009).

Domestic-scale micro CHP embraces a range of technologies that are presently at varying stages of development and commercial availability. These can include internal combustion engine, fuel cells, micro gas turbine, Stirling engine, Organic Rankine Cycle (ORC) machines, thermo photovoltaic, thermoelectric devices, etc. Micro CHP systems belong to *micro generation* technology group where electricity and heat – at the same time or separately, are produced. Staffell (Staffell 2009) subdivided the micro generation technologies in four categories: *condensing boiler and furnaces*, where only heat is provided, *combined heat and power (CHP)* technologies, including fuel cells; small scale *renewables* such as solar panels and wind turbines, and *low carbon heating* with biomass or heat pumps. He identified for each category strengths and weaknesses as shown in Table 2.2.

Table 2.2. Straightness and weaknesses of micro generation systems (Staffell 2009)**.**

Technology category	Points of Straightness	Point of weaknesses
Condensing boilers and furnaces	✓ Low cost; ✓ Widely demonstrated technology;	○ Dependence on electricity grid; ○ High emissions;
Combined heat and power: Fuel cells, internal combustion and Stirling engine, micro gas turbine, etc,	✓ Displaced high carbon electricity; ✓ Relatively large CO2 reductions;	○ Technologies are emerging and currently too expensive;
Low carbon heating		
Biomass: heat from wood in the form of logs or pellets of compressed wood waste	✓ With sustainable forestry, net CO2 emissions are almost zero; ✓ Relatively common in north EU country, as UK;	○ Expensive feedstock and high running costs when compared with boiler; ○ Limited resources for growing wood in some areas;
Heat Pumps: electric heating	✓ Separates heating from fuel combustion, allowing renewable source to heat production; ✓ Can be low cost relatively to other micro cogeneration;	○ Increase demand for electricity; ○ Greed decarbonisation is needed to reach CO2 savings;
Renewables		
Solar Photovoltaic and Micro wind: electricity produced from sun or wind sources	✓ Displaced high carbon electricity;	○ High installation costs makes economic payback unlike especially in cold countries;
Solar Thermal: water or space heating from solar panels	✓ Widely demonstrated technology; ✓ Very simple and low cost technology;	○ Dependence on centrally generate electricity; ○ Requires auxiliary heating especially in winter time;

Micro CHP systems can offer significant advantages compare to the other categories: firstly electricity cost is 3-3.7 times higher than natural gas – as shown in Figure 2.16 and in Figure 2.17, so using natural gas as fuel to produce electricity can let households to save money; secondly, the total efficiency of a micro CHP system can rise up to 80-90%. Moreover micro CHP systems are seen from the public opinion as the most unobtrusive low carbon systems compared with large and centralized renewable or nuclear power plants (Staffell 2009).

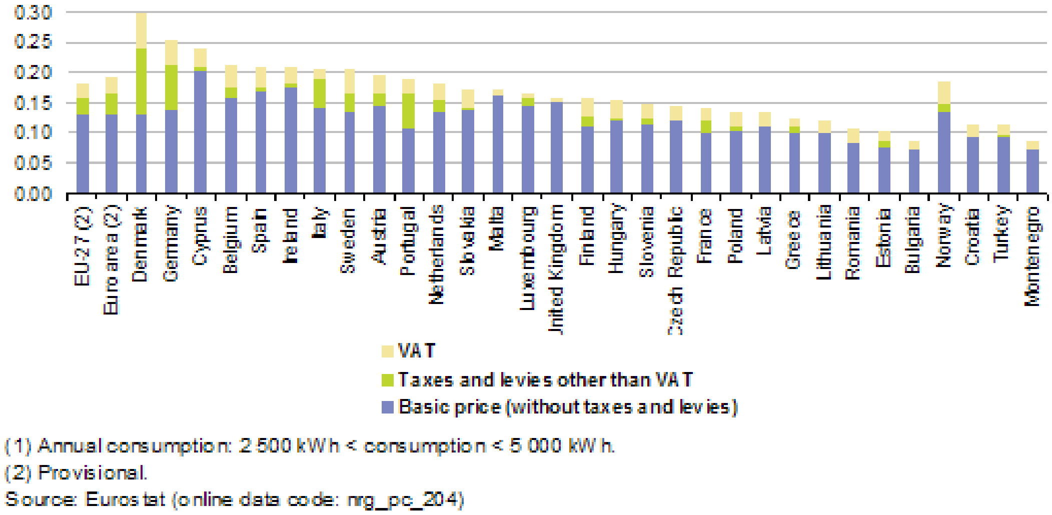


Figure 2.16. Electricity prices for household consumers in European Member States, second half 2011 (EUROSTAT, 2012d)**.**

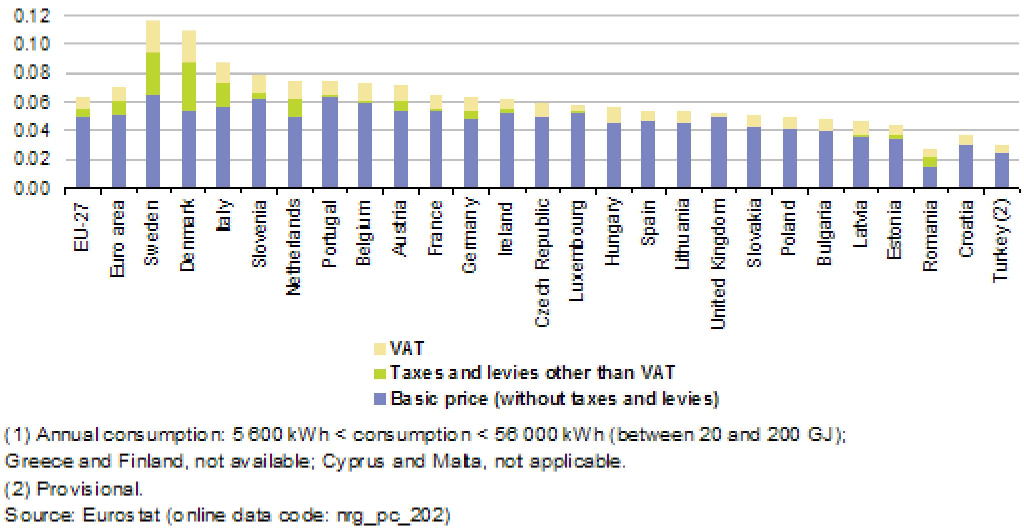


Figure 2.17. Natural gas prices for household consumers in European Member states, second half 2011 (EUROSTAT, 2012d)**.**

This work is focus on three micro-CHP technologies, i.e. Solide Oxide Fuel Cell unit, micro Gas Turbine and Stirling engine. These technologies are chosen because considered the most interesting in terms of future installations and possibility to improve the efficiency and the fields of application. In the following sections a brief introduction of the systems is given, trying to highlight points of strengthens and weakness for each technology.

2.3.2 Fuel cells

Fuel cells are electrochemical energy conversion devices that convert chemical energy content in the fuel directly into electricity and heat, without combustion process (Hawkes et al. 2009). A condensing way to look at a fuel cell is to consider it as a cross between a battery and a heat engine. This is why they are sometimes referred to as electrochemical engines. The main difference is that fuel cells are not limited by the amount of energy stored in the cell itself. Fuel cells are very promising energy conversion devices: they show a very high electrical efficiency, more than in traditional technologies, in which the energy content of the fuel is converted first to thermal energy and then to mechanical energy and finally to electricity.

Figure 2.18 shows a generic FC system for residential micro-CHP application. The main components are shown. However it is a simplistic representation and many other parts are missing (such as pressure valves, mass flow, sensors, etc.).

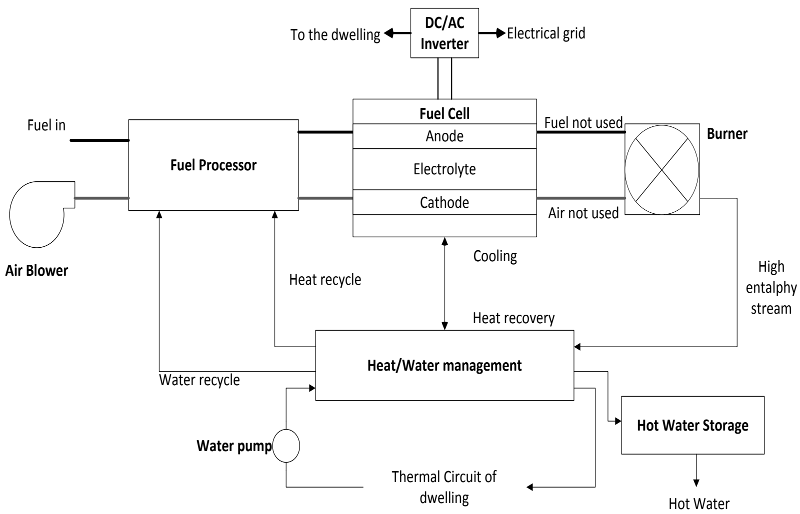


Figure 2.18. SOFC - micro CHP unit for residential application.

The components of a fuel cell system can be identified as (Coloccini, 2010):

- *Unit cells*, in which the electrochemical reactions take place. Unit cell is the core of a fuel cell. These devices convert the chemical energy electrochemically stored in a fuel into electrical energy.

- *Stacks*, in which individual cells are modularly combined by electrically connections to form unit with the desired output capacity.

- *Fuel Processor*. Except when pure fuels (such as pure hydrogen) are used, fuel preparation is required, usually involving the removal of impurities and thermal conditioning. In addition, many fuel cells that use fuels other than pure hydrogen require fuel processing, such as reforming, in which the fuel reacts with some oxidant (usually steam or air) to form a hydrogen-rich anode feed mixture. Different range of processes can be used to convert natural gas to hydrogen: steam reforming, partial oxidation and auto-thermal reforming. Steam reforming is in general the preferred method as it produces higher concentrations of hydrogen, and thus gives the highest operating efficiency (Staffell 2009). The drawbacks are that highly endothermic reactions and high operating temperatures (up to 800°C) required by the process make the start-up and transient performance of the FC difficult. The hydrogen rich stream leaving the reformer contains a proportion of CO and sulfurous compounds from the fuel source and added odorants. Both of these molecules are poisonous for fuel cell. Sulfur compounds must therefore be removed, usually by reacting with ZnO or adsorption using activated carbon. These units will need to be changed periodically, adding to the maintenance cost of the system. Other desulfurization techniques exist, but most are not suitable for such small scale applications (Piroonlerkgul, Assabumrungrat, Laosiripojana, *et al.*, 2008).

- *Air supply system*. In most practical fuel cell systems, this includes air compressors or blowers as well as air filters. The air blower is the main parasitic electrical load on the system, the power requirement scaling with the mass flow rate of air delivered. The main gas pressure that enters a home, and certainly the pressure of tanked storage, is sufficient to operate a micro-CHP system.

- *Heat Management systems.* The high temperature operating of SOFCs means that the fuel and air entering the stack needs to be pre-heated to a level that avoids thermal shock the ceramic component of the stack. Therefore, appropriately sized heat exchangers are required to heat the reactant streams and raise steam if a reformer is used. Fuel processor unit usually contains all the stages required to the fuel to be used in the FC, plus the heat management systems and steam generator to supply water vapour to the reformer and shifter.

- *Water Management systems*. High purity water is required for steam reforming. Heat exchanger condenser remove water from the exit of the FC, harvesting heat and supplying process water that can be used in steam reforming.

- *Afterburner.* The unreacted fuel can be combusted at the exit of the stack, in order to obtain additional heat. Thermal energy can be extracted from high temperature exhausted gas leaving the afterburner, raising the Heat to Power ratio of the FC. This heat is used to pre-warm the gas inlets to the stack and maintain the reformer temperature in the fuel processor. The excess is used as domestic thermal energy, though a condensing heat exchanger. A possible option is to allow a bypassing of the stack, and sending all the fuel in the afterburner, to thus operating in “boiler only” mode. However, this would oversize the burner compared with normal CHP operation, adding more costs (Hawkes et al. 2009).

- *Power conditioning.* The electricity produced by the stack goes through power conditioning where it is modified to match the load requirements of the electrical appliances and for export to the grid, in terms of voltage, power quantity and transients (Ang, Fraga, Brandon, *et al.*, 2011).

In addition to these units, a control system unit implements a strategy to control the system operating parameters, e.g. flow rates, temperature, pressure, etc. The level of complexity of the system changes whereas it is designed to respond to thermal loads or to electrical loads (varying dynamically) rather than operating at constant set point.

There are several fuel cells technologies under commercial and academic development, however only a select few are suitable for domestic micro-CHP. Table 2.3 summarises the typical parameters for each fuel cell stack.

Among the different types of fuel cells commercially available, low temperature Polymeric Electrolyte Membrane Fuel Cell (PEMFC) and high temperature Solid Oxide Fuel Cells (SOFC) are considered reliable to be applicable for DG – micro CHP (Calise 2011; Hawkes et al. 2009). Despite a very low capital cost, PEMFC requires pure hydrogen to be fed. If coupled with hydrocarbon fuels, i.e. biogas, it needs to be converted to H_2-rich gas before feeding the PEMFC, using reformer and CO removal (as showed in Table 2.3), dramatically increasing system complexity and decreasing the overall electrical efficiency. On the other hand, SOFC are very flexible in burning several types of fuels due to the high working temperatures that allow them to convert hydrocarbons into hydrogen internally, with an overall electrical efficiency of about 55-60% even when fed by methane. This is mainly due to the ceramic material of the first generation fuel cell[9] and its thermal expansion coefficients. In fact, the substitution of the ceramics material with lower-cost metals is another challenge for further improvements of SOFCs. This requires lowering the operating temperature, with a reduction to around 550°C. High temperature on the other hand, makes the FC less sensitive to impurities, thus enabling fuel flexibility. SOFCs are mainly intended for stationary use, as the high temperatures may be less suitable for transport.

[99] Second generation of SOFCs, metal-supported, are at the present under development (Hui, Yang, Wang, *et al.*, 2007).

Table 2.3. Overview of the main FC types and potential areas of use (Staffell 2009; Mathiesen 2008; Coloccini 2010)**.**

	PEMFC	AFC	PAFC	SOFC	MCFC
Electrolyte	Polymer exchange membrane	Alkaline (Potassium Hydroxide)	Immobilised liquid phosphoric acid	Solid oxide conducting ceramic	Immobilised Liquid Molten Carbonate
Operating temperature	30-100°C	60-200°C	180-250°C	500-1000°C	550-770°C
Fuels	Pure H_2	Perfectly pure H_2	Pure H_2	H_2, CO, NH_3, hydrocarbo ns, alcohols	H_2, CO, NH3, Hydrocarbo ns, alcohols
External reformer for Hydrocarbon fuels	Yes	Yes	Yes	No, for some fuels	No, for some fuels and cell designs
Intolerant to	CO, S, NH_3	CO, CO_2	CO, S, NH_3	S	S
Potential electrical efficiency (%)	40-55	60	45	60	60
Product Water management	Evaporativ e	Evaporative	Evaporative	Gaseous Product	Gaseous Product
Product Heat Management	Process Gas + Liquid Cooling Medium	Process Gas + Electrolyte Circulation	Process Gas + Liquid cooling medium or steam generation	Internal Reforming + Process Gas	Internal Reforming + Process Gas
Potential applications	Mobile units, micro-CHP	Mobile units space, military	Smaller CHP units	From larger to micro-CHP	Larger CHP units

The 2012 *Fuel Cell Today Industrial Report* shows three main FC micro CHP schemes active at the moment. The Japanese residential scheme accounts for the majority of shipments in this application, under the *Ene – Farm* program developed by Tokyo Gas L.t.d and Panasonic Corporation (Panasonic, 2013). By 2010, the scheme has brought micro CHP system to over 13,500 households, offering both SOFC than PEMFC model. In Germany, the *Callux field test* program have seen the installation of over 250 fuel cell systems from Baxi Innotech, Hexis and Vaillant brands, both with SOFC and PEMFC model. Ceramic Fuel Cell Limited is providing a number of units to the UK and Hamburg for the SOFT-PACT demonstration project with E.ON. (E.ON, 2012), providing SOFC units between 1 and 2 kW of electrical power. Since 2006, the South Korea scheme has seen increasing the number of micro-CHP units, supplied by domestic manufacturers GS Fuel Cell, Hyosung and Fuel Cell Power (all PEMFC model). In USA, by the end of 2011 Clear Edge had over 100 installation of its 5 kW PEMFC unit in California.

In general, SOFC shipments grew by more than 300% between 2010 and 2011 (Fuel Cells Today 2011). California's Bloom Energy is one of the leading companies for large stationary application of SOFC systems while at residential scale Ceramic Fuel Cells Limited is one of the most successfully company. The *Callux domestic demonstration program* has allowed German manufacturers Hexis and Vaillant to field test their SOFC micro-CHP systems with installations across Germany.

Figure 2.19. Sunfire ISM 1.7 kW system (Sunfire GmbH, n.d.)

Table 2.4. Sunfire ISM 1.7 kW system: main features (Sunfire GmbH, n.d.)**.**

Rated power output	1.7 kW
Maximum operating temperature	< 860 °C
Maximum fuel utilization	85%
Operating media	Steam reformate, CPOX reformate, hydrogen
Guarantee	5,000 h in operation, 20 thermal cycles

The SOFC micro-CHP system considered in this study is the ISM 1.7 kW Sunfire, currently used in Vaillant system for the Callux demonstration program. Figure 2.19 shows an example of the unit.

2.3.3 Micro Gas Turbine

Micro Gas Turbines are a specific type of gas turbines, with limited electrical power size compared with traditional gas turbine. They can offer several advantages compared with traditional technologies for small scale power generation, such as compact size and low-weight per unit power, a small number of moving parts, lower noise, multi-fuel capabilities as well as opportunities for lower emissions. Moreover, if compared with internal combustion engines, mGTs show few balancing problems due to the absence of reciprocating and friction components. For the scope of this thesis, two important features of mGTs are: the possibility to work with fuel other than natural gas, included diesel, biogas, syngas, and other bio-based liquids and gases (Pilavachi, 2002); the majority of the waste heat is contained in the high temperature exhausted gas making it simpler to capture compared to ICE. However, ICEs show a quicker respond to changes in output power requirement and higher efficiency, although efficiency of mGT is increasing. Figure 2.20 shows a conceptual scheme of a micro gas turbine system with recuperator, for CHP production.

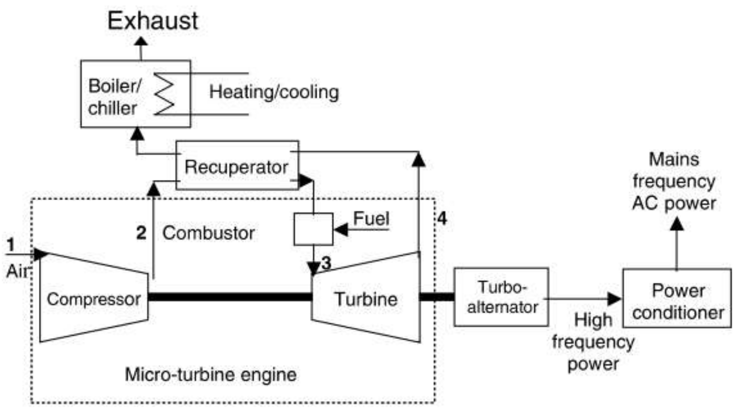


Figure 2.20. Schematic of a micro-turbine gas system (Pilavachi, 2002).

The air is entering the system (stage 1) where it is compressed, pre-heated in a heat exchanger (stage 2) and then used in the combustion chamber to oxidise the fuel (stage 3). The high temperature exhausted gas enters the turbine, producing electric power. After that, the gas leaving the turbine (stage 4) is sent to the recuperator (heat exchanger), where part of the heat is recovered to pre-heat the air before the combustion chamber. A second heat exchanger is then required to recover the waste heat form the exhausted gas, used to heat an external fluid, typically water, for the thermal demand. The mechanical power produced by the turbine is transformed in electric power by a generator connected with the turbo-compressor shuffle (Bruno, Ortega-López & Coronas, 2009).

The main fuel used in mGT systems is natural gas. However, several models allow using different fuel. For example Capstone realizes the C30, C65 and C200 models in double fuel version, allowing both natural gas and biogas. In the last case, the Low Heating Value of the fuel has to be in a defined range of values and to contain a level of H_2S less than 400 ppm. Apparently, the specific emissions do not change from one fuel to another for the C30 model and they are double for C65. When the mGT runs with biogas, the quantity of fuel required is bigger given the lower LHV of the biogas compared with natural gas. For this reason, the input manifolds are larger in the biogas fuelled – mGT turbine. Although the mGTs are more tolerant to H_2S than ICE, a desulphuriser unit is required before the fuel input the unit, as in the case of fuel cell. Rising of SO_2 emissions and corrosion problems can happen in this case, if no proper biogas cleaning unit is installed (Bianchi, Spina, Tomassetti, *et al.*, 2009).

Table 2.5 . Capstone C30 system: main features (Capstone Turbine Corporation, 2013)**.**

Electrical Power output	30 kW
Electrical efficiency	26%
Digester/Landfill gas (HHV)	13-32.6 MJ/m3
H2S content	< 70,000 ppmv
Inlet pressure (HHV dependent)	414 -483 kPa gauge
Net heat rate (LHV)	13.8 MJ/kWh
Exhausted gas temperature (after recuperator)	275 °C

Micro turbines are usually guaranteed for 6,000 – 8,000 h of operation per year, with only one stop for maintenance operations per year. The lifetime of a micro gas turbine is usually 10 years, but some of its components, i.e. the combustor chamber, required to be substituted more often (TURBEC, 2012). An important factor that limits the lifetime of a mGT system is the temperature (around 600°C) of the exhausted gas going out form the turbine and entering the recuperator (Bohn, 2005).

Capston Turbine Corporation (US) is one of the leading companies in micro gas turbine market. They offer a marketable model for micro CHP application of 15 or 30 kW of power (Capstone Turbine Corporation, 2013), run with biogas or natural gas. Example of installation is an array of 10 Capstone C65 micro turbines to supply heat and power to a 44,520 apartment complex in South Korea. Similar installations have been provided in Japan, while in Europe Capstone models are more used in public space, such as hospitals or universities. Another leading company in the field is the Swedish Turbec. However their smallest unit is the T100, a 100kW CHP unit for residential application. An under development product is the Ener Twin model of Micro Turbine Technology group, based in Netherland. They foresee to launch the 3 kW CHP unit in later 2013 (MTT 2013).

In this work, the Capstone C30 model has been chosen as micro CHP unit implementing micro gas turbine technology. This is because, for the author's knowledge, it is, presently, the smallest mGT model running with biogas available on the market. The characteristic of the C30 are shown in Table 2.5. Figure 2.21 shows a schematic drawing of a Capstone micro gas turbine.

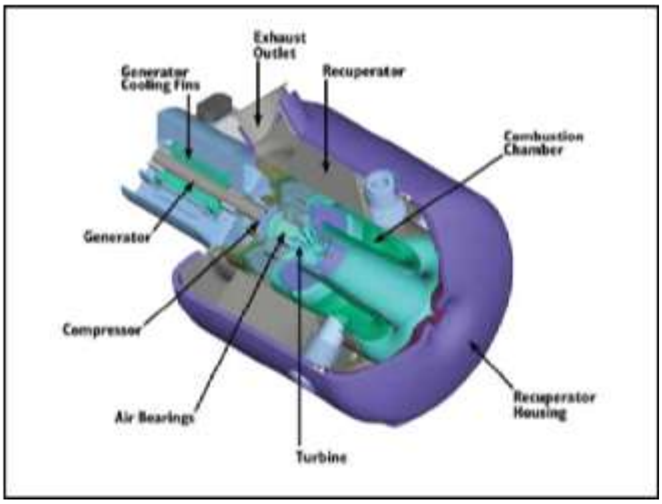


Figure 2.21 .Capstone Turbine (Capstone Turbine Corporation, 2013)**.**

2.3.4 Stirling engine

The Stirling engine patent was presented in 1916 by Robert Stirling. Since then different models have been developed for several applications. The main future of Stirling engine is that the heat source is external and this allows running with a variety of fuels, including biomass based fuels. Another important characteristic is that the combustion process can occur in steady state and therefore is easier to control. Moreover Stirling engines present low pollutant emissions, lower noise levels and long maintenance free operating periods (Ferreira, Nunes, Martins, *et al.*, 2012). Stirling engines have the potential of achieving higher efficiencies because they closely approach the Carnot cycle. Moreover they have a good capability to operate under part-load conditions. Their electrical efficiency, however, are only moderate, typically around 10 to 12% (Aoun, 2008).

In spite of a lot of effort in Stirling engines research during ’70 and ’80, Stirling engines never became commercially available, mainly due to the superiority of the internal combustion engine, in terms of efficiency and costs. Nowadays, with the scarcity of fossil fuel, much more aware of environmental issue and climate change problems related, Stirling engines are seen as an eco-friendly solution and new attention is put on this technology (Pourmovahed, Opperman & Lemke, 2011).

An elementary Stirling engine consists of an engine piston, an exchanger piston and three heat exchangers: a cooler, a regenerator and a heater. The piston converts gas pressure into mechanical power, whereas the exchanger piston is used to move the working gas between the hot and cold sources. Stirling engine drive methods are based on two distinct principles of operation: the kinematic drive and the free piston drive method. Kinematic Stirling engines use the mechanical elements to convert the reciprocal piston motion to a rotational output, say to drive a generator. The kinematic drives require special sealing to avoid leaks due to high pressures at which the working gas is subject. Free-piston Stirling engines move the reciprocating elements using the pressure variations in the space beneath the piston. As the linear alternator is tightly attached, the mechanical friction is minimized and, as a result, the leakage of the working gas is substantial reduced. So, the free piston engine does not require large maintenance costs, allowing a continuous power operation and a great potential for high efficiency (Boucher, Lanzetta & Nika, 2007). The working fluid consists generally of helium or hydrogen.

Both kinetic and free piston Stirling engine types are developed in a wide range of power capacity: while for the former existing unit the power capacity varies between 1.1 to 500 kW, for the latter the electrical capacity can be found in the range between 1 and 25 kW.

Several companies are developing Stirling units for CHP applications. WhisperTech (New Zeland) developed an alpha kinematic engine called WhisperGen with a capacity of up to 1.2 kW of electric power and a 12% and 80% electrical and thermal efficiency, respectively (WhisperTech Limited, 2013). Microgen unit, developed by BG Group from US, foresees a supplementary burner to meet the full thermal demand for larger homes. The unit is based on a free piston Stirling engine fuelled by natural gas. Cleaneregy (Sweden) offers two variants of small power plants: one for biogas and on for solar power. The unit size is 9 kW for electrical capacity, and the on run with biogas can produce 26 kW of thermal capacity. Infinia Corporation (US) launched recently the PowerDish unit that uses parabolic concentrator dish to concentrate the sun's energy onto the hot end of a free piston Stirling engine. Inspirti has developed a micro CHP unit based on a kinematic Stirling engine design. The electrical output is 3 kW and the thermal power is 15 kW. Another Stirling unit running with natural gas is the Stirling BioPower unit (US). The system is designed to run with natural gas, propane, alcohol, renewable energy such as biomass. The electrical efficiency can reach 27-28%, while the total efficiency achieves 75-80%. Finally, Siwgma Elektroteknisk (Norway) has developed a beta-type Stirling unit, with helium as working fluid, producing 1.5 kW of electric power and 9 kW of thermal power.

Table 2.6. Whispergen Stirling engine: main features (WhisperTech Limited, 2013)**.**

Electrical efficiency	12%
Thermal efficiency	80%
Electrical power	1.2 kW
Lifetime	50,000 h

The unit considered in this study is the Whispergen 1.2 kWe. This unit has been chosen for its widespread amongst the Stirling units produced that brings availability of data in the literature. Table 2.6 shows the main characteristics of the Stirling unit.

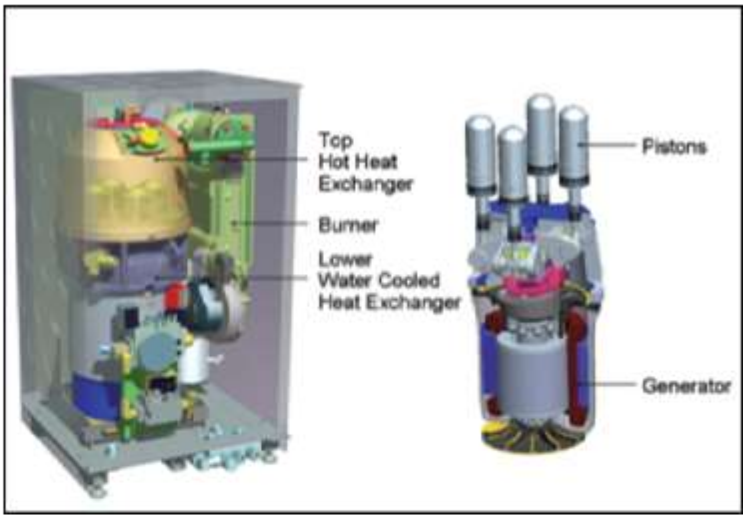


Figure 2.22. Whispergen unit (WhisperTech Limited, 2013)**.**

2.4 Heat to Power ratio: micro CHP and demand side

There are different ways to operate a micro-CHP system. Generally a thermally led operating strategy is chosen, especially for systems with high H to P ratio, such as Stirling engine. Comparing with large CHP plants, which are usually operating for more than 6,000 h per years and generally sized to meet base-load demand, micro-CHP systems are unlikely to operate for more than 3,500 h per year (Harrison, 2002). As noted by Harrison (2002), *'[for a micro-CHP system] is uneconomic to match base-load, impractical to meet average load and practically impossible to attempt load following*'. It is than necessary and fundamental that the micro CHP unit operates in parallel with the network, exporting surplus power to it.

For any electrical generator used in a CHP plant, there is a balance between the electrical power that can be generated and the heat that can be recovered for use on-site. This is generally referred to as the *Heat to Power ratio* and it is expressed as the quantity of heat recovered per unit of electricity generated, as shown by Equation 1 (Strachan & Farrell, 2006). The heat to power ratio is a useful way of assessing the suitability of a CHP generator for a particular site and compared it with the conventional paradigm of centralized electricity production and heat only boilers.

$$HtoP\_ratio = \frac{energy\_produced\_(consumed)\_as\_Heat}{energy\_produced\_(consumed)\_as\_Power} \quad (1)$$

When matching the energy provided by the mCHP system to the electricity and heat demand from a house, the temporal precision of the mCHP operating mode and demand profile chosen gets important (Hawkes & Leach 2005). This is especially true when optimization modelling is undertaken during the design of the mCHP. A typical house has a relatively low-level of energy consumption for the majority of the day and the electrical requirement reaches several kilowatts when high power devices are operated, while the thermal demand reaches high load when thermostatically controlled space heating is required. Things are even more complicated if two different geographical areas are taken as case studies, as in this study. UK and Italy have got very different climate and consequentially the energy demand of a typical Italian and UK house is very different. Figure 2.23 shows an example of energy profiles for a typical single family house of 80m^2 and 2.7 numbers of occupants.

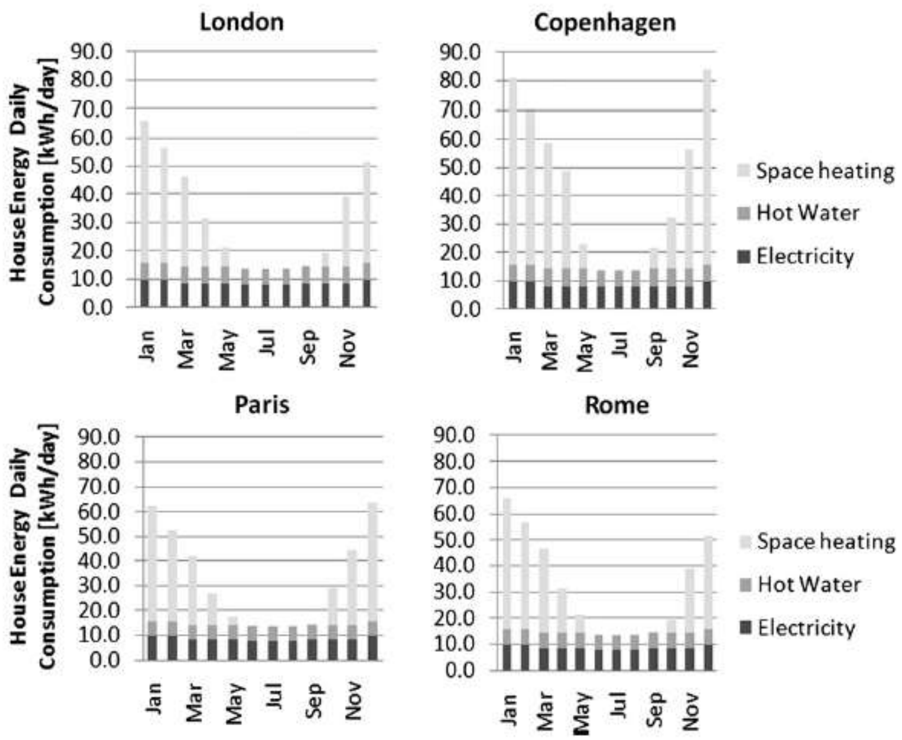


Figure 2.23. Energy demand profiles of a typical house in four different cities in EU (Liso, Zhao, Brandon, *et al.*, 2011)**.**

The micro-CHP system could be designed to meet the peak energy demands (as in condensing boiler): however it is not convenient from a cost and operational point of view. Some compromises are chosen by the manufacturer, depending on the technology, and the micro-CHP is usually designed to meet an average energy load (heat or electricity). Additionally, there are time when pick energy is supplied by the conventional systems, as for example condensing boiler and electricity from the grid. On the other hand, when the energy produced by the m-CHP system is not wanted, for example if electricity is demanded in summer when no heating is required, the surplus energy can be stored or sold to the grid. The production of electricity and heat is inseparably linked in a CHP system, and while there are several methods available to dynamically change the H to P ratio of the fuel cell, none of these have proven technically feasible in commercial models so far (Staffell 2009). Flexible and aggregation of loads therefore is necessary to address this important technical issue for the m-CHP systems of matching H to P of the energy supply system with the H to P of the load. The closer the match, the more efficient the system will be (Strachan & Farrell, 2006).

In this study, an average yearly based energy demand is chosen to assess the environmental performance on the three micro-CHP systems described in Section 2.3, in the geographical areas identified. Table 2.7 shows the average H to P ratio of the three systems compared with the demand of a typical house in UK and Italy.

Table 2.7. H to P ratio for the three technologies chosen compared to a typical dwelling in UK and IT and to a competitive centralised technology.

	SOFC 1.7 kW **Sunfire**	**Stirling Engine** **1.2 kW Whispertech**	**Micro Gas Turbines** **Capstone 30 kW**	**Internal Combusti on Engine 420 kW**	**UK Dwellin g**	**IT Dwellin g**
Investment Cost (€/kW)	3000 – 4000	2500-4000	2000	800 - 1400		
H to P	1.57	6.7	2	1.7	3.13	2.3
Status	demonstration	demonstration	Few commercial models	Mature technolog y		

The three technologies chosen differ for the size of the units. The mGT, in fact, has a power output of 30 kWe, while the SOFC and Stirling units have got 1.7 and 1.2 kWe respectively.

The installation of a *micro-grid* in the case of the micro Gas Turbine is assumed in order to compare the three DG scenarios. This will be discussed later in Section 4.9. Micro-grids are local distribution systems that contain generation plants and can run both in autonomous (stand-alone) and non-autonomous (grid-connected) mode. A micro-grid is operated by means of a control centre that monitors real-time energy demand and supply, optimizing the dispatch of dispersed generators, storage systems and acting on loads through *demand response management*[10]. This concept is interesting in particular for energy districts applications where hybrid integrated energy systems can be successfully introduced (Manfren, Caputo & Costa, 2011).

2.5 Techno-economic evaluation: comparison amongst the three chosen technologies

Although it is beyond the scope of this thesis to perform a techno-economic evaluation of micro-CHP systems, this section gives an overview of the costs associated with the installation and operation of a micro-CHP unit.

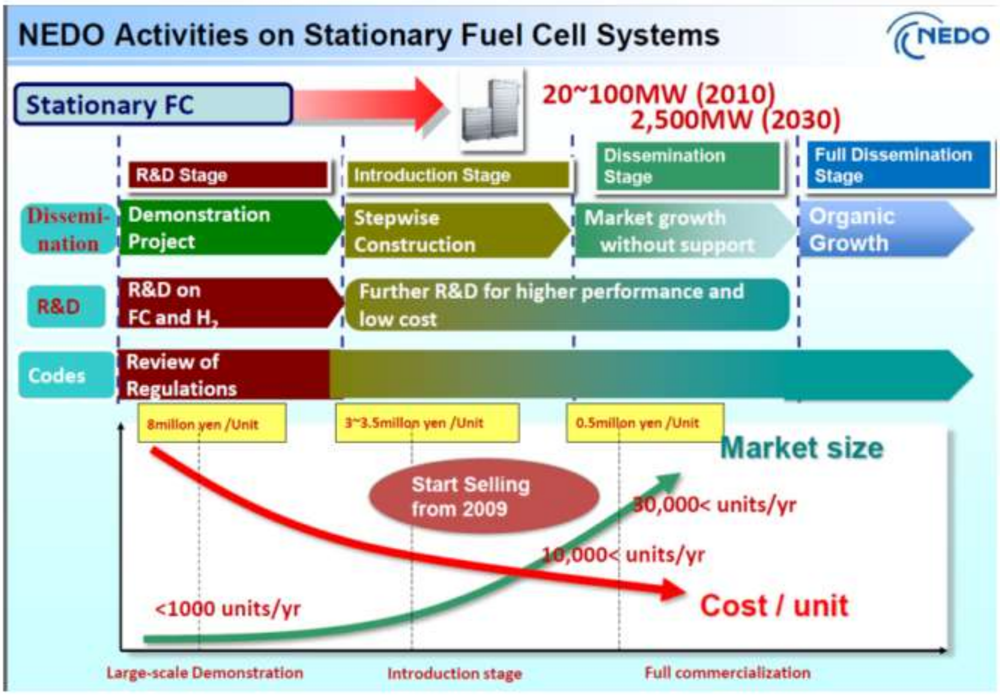
Table 2.7 gives an overview of the investment costs associated with each technology. Fuel cell appears to be the most expensive one, mainly due to the demonstration stage of the technology. To perform a techno-economic analysis of micro CHP systems not only the installation cost has to be taken into account, but even the potential operational cost/saving. In fact, the three technologies have different operational hours per year and moreover, being at different development stage, it is difficult to estimate the hours precisely. Finally, further uncertainties are related to the different H to P ratios of the three systems that determine the substitution of the reference scenario and the possible savings in terms of electricity imported from the grid and natural gas supplied to the dwelling. It is worth noting that the incentives schemes currently applied in each country considered here is fundamental in determining the final threshold to install or less the micro-CHP system.

As an example of what stated above, the analysis presented in the Report elaborated by the Carbon Trust for the UK government on the penetration of micro-cogeneration system in

[10] Demand Response Management is applied to reduce load in peak hours in response to real-time market prices and variable conditions, but it can be successfully interpreted as a more wide and general concept, involving other types of load, when transferred to local energy systems (Manfren, Caputo & Costa, 2011).

the UK market is reported (Carbon Trust, 2011). It is a techno-economic evaluation of a Stirling engine unit installed at domestic level. The Stirling is assumed to be thermally led. At market introduction, the marginal cost of the Stirling engine micro-CHP system over an equivalent condensing boiler is expected to be around £2,500. At this price, the value of the electricity generated by the micro- CHP systems in the field trial (n.d. conducted by the Carbon Trust during the project) is insufficient on its own to provide an attractive payback for the majority of consumers. Even taking account of the incentives available under the new system of Feed-in Tariffs, the payback is around 16 years for a typical larger household with an annual heat demand of 20,000 kWh. For small commercial micro-CHP, for which no such incentives are currently available, the payback period is estimated to be 20-25 years.

The cost of micro-CHP systems can then be reduced with economies of scale. Figure 2.24 shows an example of possible effects of economies of scale on the development of the fuel cells. However, Staffel and Green (2012) reported as '*even a heroic effort by industry is unlikely to reduce the price of small domestic-scale systems (fuel cells) to the $1000/kW mark. By aligning the scope and boundaries of cost estimates with the realities of domestic micro-generation systems, we show that a long-term target of $3000 and $5000 for 1 and 2kW systems is more realistic, and could feasibly be attained by 2020 at the current rate of progress*'.



(Source: Presentation by Japanese government agency NEDO – March 2011)

Figure 2.24. Example of the possible effect of economies of scale (FuelCellToday, 2012)**.**

2.6 Biogas production from anaerobic digestion of Organic Fraction of Municipal Solid Waste

2.6.1 Anaerobic digestion

Anaerobic Digestion encompasses a series of processes in which organic matter is converted into biogas (composed mainly by a mixture of methane and carbon dioxide) and digestate by micro-organisms in the absence of oxygen. AD has been widely used in sewage sludge treatment, however in the last few decades it has become increasingly widespread for treating food waste, farm waste and green garden waste. The main product of the AD is the biogas that can be used for energy generation or bio-fuel production, and the by-product is digestate (see Section 2.6.2) that under certain conditions can be used as an organic fertilizer on agricultural land. AD has the potential of reducing greenhouse gas emissions mainly by substituting energy from fossil fuels with biogas; it can contribute to carbon storage; it enables recovering nutrients such as nitrogen (N), phosphorus (P) and potassium (K) and replacing chemical fertilizers (Møller et al., 2009).

A lot of organic material can be processed with AD. The selected feedstock can include animal manures, agricultural crops, agri-food processing residues, food residues, the organic fraction of household waste, organic fractions of industrial wastes and by-products, sewage sludge, municipal solid waste, etc. The feedstock can be either a single input or a mixture of two or more feedstock types (in this case the process is called co-digestion). Most biogas plants use more than one substrate (Lukehurst, 2009).

There are a number of different processes falling under the definition of anaerobic digestion. They are usually distinguished on the basis of operating temperature (thermophilic plants operate at around 55°C and mesophilic at around 35°C) and the percentage of dry matter in the feedstock (dry systems with more than 20% dry matter, wet systems have less than 20% dry matter). In this work a mesophilic process is considered, operating in wet regime (12% of dry matter in the substrate).

The advantages of AD have been widely recognised in many countries. The reports from the International Energy Agency (IAE) provide the data on biogas plants in Europe. Germany has more than 7,000 of small- and large-scale biogas plants, Switzerland - 560, Sweden – 230. There are more than 200 biogas plants in UK with the total output of 178 MWe, with almost 75% of them being used for sewage sludge treatment (IEA, 2004).

The high degree of flexibility associated with AD is claimed to be one of the most important advantages of the method, since it can treat several types of waste, ranging from wet to dry and from clean organics to grey waste. AD of OFMSW has been commercially available for approximately 20 years and in that time, the heterogeneous and variable nature of the feedstock has given rise to a considerable number of different processes in operation in many different countries (Hogg, Favoino, Nielsen, *et al.*, 2002).

Figure 2.25 shows a schematic representation of AD process, highlighting the environmental benefits that can be reached with this treatment option.

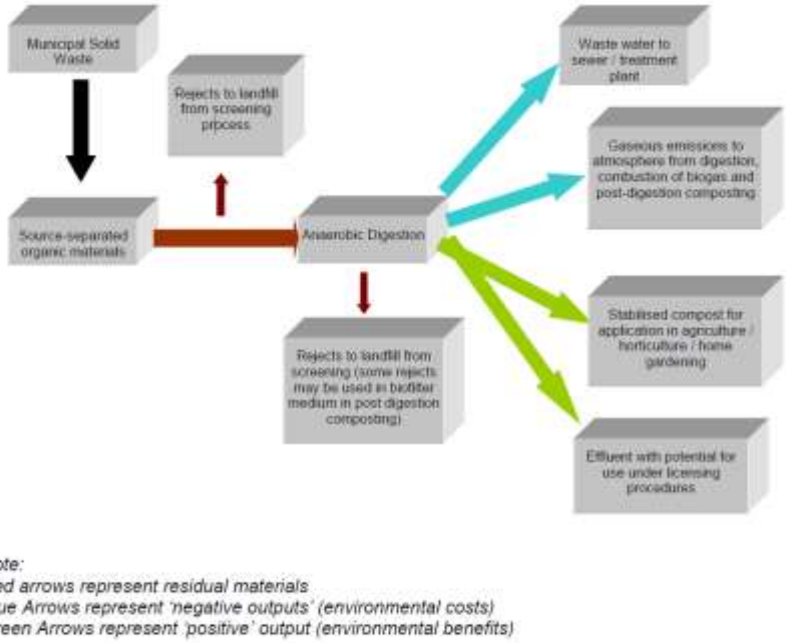


Figure 2.25. Schematic representation of AD inputs and outputs (Hogg, Favoino, Nielsen, *et al.*, 2002)**.**

2.6.2 Digestate use

Digestate is a by-product of AD process and can be used successfully as a substitute for commercial fertilisers. Although the fertilising values of the digestate depends on the nutrients content in the feedstock, its characteristics can vary between batches from the dame digester and are very specific to each digester tank (Lukehurst, 2009). During the biological process, the organic compounds in which the nutrients are present changed, enhancing their availability to crops. Table 2.8 shows an example of how a part of the organic nitrogen supplied with the feedstock is converted to ammonium, although the total nitrogen content in the digestate remains the same as in the feedstock.

Table 2.8. Example of the average nutrient composition over 52 weeks of feedstock (dairy cow slurry) and digestate in a mesophilic digester in Northern Ireland (Lukehurst, 2009)**.**

	Dry matter (g/kg)	Total N (g/kg fresh)	NH4-N (g/kg fresh)	NH4-N (% Total N)
Feedstock	72.2	3.5	2	67
Digestate	59.3	3.6	2.4	80.5
Change	-17.90%	2.80%	20%	

Digestate can be spread as a whole or separated mechanically in liquid and fibrous materials that need to be stored and handled separately. Currently the majority of AD facilities recycle the digestate to local agricultural land as an organic fertiliser (WRAP, 2011a). However the window for land application is limited to agricultural and crop requirements (WRAP, 2012a), and for large capacity AD plants, a substantial area of land is required to provide a secure and suitable market for the digestate. If application to agricultural land is not feasible, due to transport distances, legislative requirements or other restrictions, digestate can be used for land reclamation. As the use of AD increases the demand for agricultural land will also increase, potentially requiring plants to transport digestate further in search of suitable land. This is important for the increasing number of centralised AD facilities operating in urban areas. Digestate must therefore be carefully managed to ensure it is utilised as a resource and maximum benefit is achieved whilst avoiding excessive transportation costs.

The application of nitrogen in organic materials to agricultural land is regulated by the European Nitrates Directive (European Parliament, 1991). As a consequence the spreading of digestate to land is controlled (based on nitrogen content) and dependent on location and crop demand. Moreover the Nitrates Directive defines the Nitrogen Vulnerable Zones (NVZ) that are areas of land in each member state's territories affected by pollution or at risk of being so. This can result in digestate being transported greater distances to find suitable land-based markets and avoid over application; this will increase transport and operational costs. Furthermore land application is only appropriate during the growing season, requiring digestate to be stored for significant months of the year.

The importance of Nitrogen

Nitrogen (N) is a key nutrient in manipulating plant growth. Most nursery/floral producers use large quantities of N fertilizers in a "blanket" attempt to meet the needs of their crops. However a thorough understanding of N nutrition can be useful in optimizing both the concentration and form of N best suited for the plant species, stage of growth, and time of year and production objectives. Plants require N in relatively large quantities and in forms that are readily available.

Nitrogen metabolism is a well-studied and a vital aspect of plant growth. Nitrogen is one of the important building blocks in amino acids. When N is deficient in plants restricted growth of tops and roots and especially lateral shoots may occur. Plants also become spindly with a general chlorosis of entire plant to a light green and then a yellowing of older leaves. This condition may proceed toward younger leaves. Older leaves defoliate early. Plants can take up Nitrogen in four forms:

1. NH_4 Ammonium
2. NO_3 Nitrate
3. Organic Nitrogen
4. Molecular Nitrogen

Regardless of the N source (inorganic fertilizer, organic fertilizer, manure, etc.) plants can only take up N in these four forms. That means that some conversions must occur in the growing media/root zone before some sources of N can be taken up by the plant. All four forms of available N have unique characteristics that influence plant growth in different ways. Understanding these characteristics is very important in matching the best N fertilizer with plant species, stage of growth, time of year and production objectives (Benitez, Rosas, Barajas, *et al.*, 2011; Prapaspongsa, Poulsen, Hansen, *et al.*, 2010).

The fertilising value of nitrogen in digestate can be expressed as the 'utilisation ratio'. This is defined as the relative quantity of commercial fertiliser nitrogen necessary to obtain the same yield of crop as the quantity of total nitrogen supplied in digestate. The fertilising value of the digestate increases with increasing nutrient utilisation percentage. It is mainly the mineral nitrogen (ammonium nitrogen) component of digestate that is available to crops immediately after application. In theory, the utilisation percentage of N in manure and digestate should be equivalent to the share of ammonium. However, when digestate is applied to a field surface some ammonia volatilization will take place after application. As a result the utilisation percentage will decrease. As a consequence it is important to minimise the surface area of digestate that is exposed to air after application so as to minimise ammonia volatilisation. The expected utilisation percentage of nitrogen is greater for digestate than for slurry; for spring applications rather than applications in summer (Lukehurst, Frost & Al Seadi, 2010)

2.6.3 Biogas use

The biogas composition varies depending on several factors, such as the feedstock composition, the feeding rate of the digester, the organic matter load. Anyhow, its main components can be identified in CH_4 (45-65%) and CO_2 (35%-55%), plus traces of H_20, H_2S and N_2. Table 2.9 shows the average composition of biogas for different feedstock.

Table 2.9. Biogas composition based on feedstock characteristics (Bruijstens, Kadijk & Bleuanus, 2008)**.**

Components	Food waste	Wastewater treatment plants	Agricultural waste	Waste of agrifood industry
CH_4 (%vol)	50-60	60-75	60-75	68
CO_2 (%vol)	38-34	33-19	33-19	26
N_2 (%vol)	5-0	1-0	1-0	-
O_2 (%vol)	1-0	<0.5	<0.5	-
H_2O (%vol)	6	6	6	6
H_2S mg/m3	100-900	1000-4000	3000-10000	400
NH_3 mg/m3	-	-	50-100	-

Based on the data presented by EurObserv'ER, in 2011 the production of primary energy from biogas in the European Union was 10.1 Mtoe (Eurobser'ER, 2012), a value almost double compared with the production in 2007 (5.9 Mtoe). UK and Italy are the second and the third major biogas producers in EU, while the first place is covered by Germany. The biogas is used mainly in two different way: directly on site, through a combustion of the gas and production of electricity and/or heat; or, after an upgrading process, it exits the possibility to obtain a bio-methane, with a concentration of CH_4 of a minimum of 95-97% (depending on the national legislation) that can be directly injected into the natural gas grid or used as a fuel. In some countries, for example Sweden and Switzerland, the bio-methane obtained with upgrading of biogas is sold as biofuel for transport.

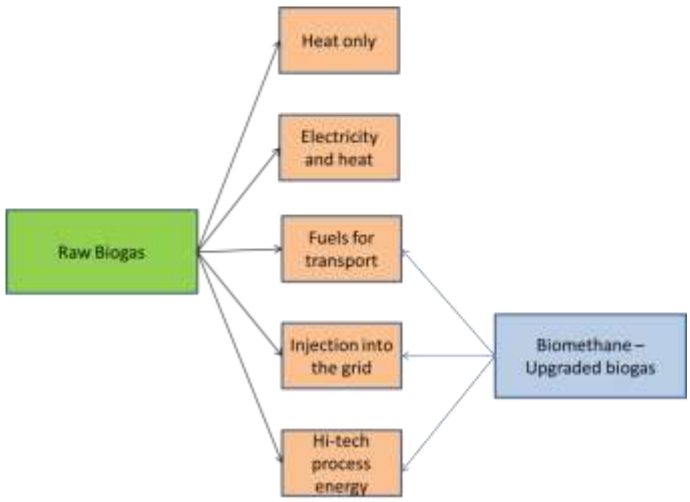


Figure 2.26. Schematic overview of different uses of biogas.

Figure 2.26 shows the different way to use biogas. Today, the most of the biogas is used in combined heat and power plants, driven by the subsidy schemes to stimulate renewable energy. In small scale installation, heat only is produced through boilers or transformed to steam. The CHP production is often located close to the digester plant and internal combustion engine are most commonly used, with an average electrical efficiency of 30-45% and 35-60% of thermal efficiency. Even in CHP or heat only installation, some unwanted substances, such as H_2S, H_2O, O_2, N_2, have to be removed. This is mainly due to prevent corrosion and mechanical wear of the energy recovery system. The water is removed by cooling, compression, adsorption or adsorption processes. After this, the H_2S has to be removed in desulphuriser systems; finally O_2, N_2 – if present, have to be removed with membranes or activated carbons processes.

Currently, bio-methane is produced in 177 plants in Europe, including 128 that feed into national natural gas distribution grids. The remaining plants use the bio-methane generated on the production site, primarily as a fuel. To produce bio-methane from biogas, an upgrading process is required, to remove the most of the CO_2 and to increase the energy density. To reach the specification for grid injection, different for each country, further step are required, such as drying to a certain water dew point, or adding other component such as LPG (Foreest, 2012).

2.7 Alternative treatment for the OFMSW: landfill

In some countries, the majority of municipal waste, and in particular the organic fraction, is still buried in landfill. The term 'landfill' is used to refer to a wide range of facilities across Member States, from open dumps to sites which are engineered specifically for the purpose (and sometimes, for specific wastes) (Hogg, Favoino, Nielsen, *et al.*, 2002).

The dumps are landfill where many different kinds of waste are disposed of with little or no benefit of an engineering plant. The waste is not compacted and no measures are taken to prevent methane and leachate emissions to the atmosphere and soil. On the other side are the engineered landfills, which include bioreactors[11], flushing-bio-reactors[12] and semi-aerobic landfills[13]. They implement measures typical of a conventional landfill – such as collection and management of the leachate and gas generated – plus additional technologies to enhance the waste degradation process, in order to make it faster and more efficient. This leads to high gas generation rates early in the lifetime of the landfill, with a more efficient gas collection system and utilization pathways for the gas, to produce electricity – mainly – or combined heat and power generation (Manfredi, Tonini, Christensen, *et al.*, 2009).

The degradation of biodegradable wastes under landfill conditions creates methane. Methane is a powerful greenhouse gas (25 times more powerful than carbon dioxide according to CML methodology) and the Landfill Directive is designed partly to address the issue of methane emissions from landfills. All municipal wastes can be accepted by landfill. These wastes generate different emissions depending upon their potential to degrade under landfill conditions, and this affects the impacts of landfilling (see Figure 2.27). Different materials also degrade at different rates, and the contribution of different fractions to leachate will vary. Leachate will quite possibly affect groundwater at some later

[11] *Bioreactor landfills* typically recirculate collected leachate through the waste mass; this keeps the waste moisture content close to field capacity and provides a continuous supply of moisture and nutrients, resulting in an enhancement of the microbial anaerobic environment (Manfredi et al, 2009).

[12] *Flushing-bio-reactor landfills* recirculate the leachate together with additional amounts of water in order to flush-out the soluble waste constituents (Manfredi et al, 2009).

[13] *Semi-aerobic landfills* rely on a hybrid anaerobic/aerobic degradation sequence. The anaerobic phase comes first and it is stopped by air injection when the methane yield becomes too low to justify LFG utilization (typically after 5 to 10 years). The aerobic degradation phase will then quickly stabilize the waste, blocking, at least in theory, residual methane generation (Manfredi et al, 2009).

date. Whether, and if so, when leachate will become a problem will be determined in part by the landfill lining and the geological characteristics of the site.

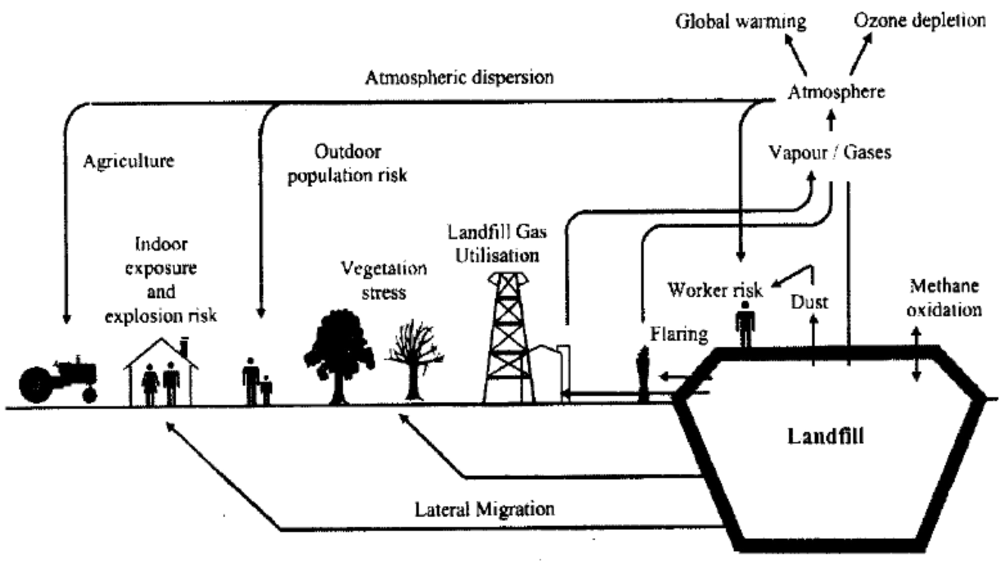


Figure 2.27. Generation pathways of emissions to atmosphere in landfill (Hogg, Favoino, Nielsen, *et al.*, 2002)

The only 'end product' for landfills is landfill gas, which is constituted by 50% of methane. If collected can be used to generate energy. The final residues in landfills consist of material which has not degraded (in landfill conditions) and the leachate residues which may be treated through various approaches. The former may have substantial carbon content. As such, to the extent that certain materials which might degrade under aerobic conditions do not do so in landfills, landfills may be considered to be a net sequester of carbon. This will be discussed later in Chapter 5.

3. Approach

3.1 Environmental system analysis tools: the Life Cycle Assessment approach

3.1.1 Introduction

As stated in Section 1.1, climate change and environmental threats came more into focus in the last years and currently they represent a big challenge for our society. This means that environmental considerations have to be integrated into a number of different decisions made by businesses, individuals, policymakers and public administrators (Finnveden et al. 2009). Environmental Systems analysis is the field that '*attempts to find supply technology systems solutions to these challenges*' (Chalmers University, 2012)

Many tools and indicators for assessing the environmental impacts of different systems have been developed in the last years, such as Life Cycle Assessment, Environmental Impact Assessment (EIA), Environmental Risk Assessment (ERA), Strategic Environmental Assessment (SEA), Cost-Benefit Analysis (CBA), Material Flow Analysis (MFA) and Ecological Footprint (Fruergaard 2010). This thesis is focused on the application of Life Cycle Assessment methodology.

3.1.2 Life Cycle Thinking and Assessment

Life Cycle Thinking is an approach for the evaluation of environmental burdens of products and services - that aims at identifying single steps as well as the whole picture of an entire product or a service system. In regards to a product, it starts with raw material extraction and processing, then considers transportation and manufacturing, distribution, use and ends with re-use, recycling or ultimate disposal. The overall idea of making a holistic evaluation of a system's impact can be defined as Life Cycle Thinking. The key aim of this approach is to avoid burdens shifting that means '*minimizing impacts at one stage of life cycle, or in a geographic region, or in a particular impact category, while helping to avoid increases elsewhere*' (JRC - IES, 2011).

Environmental concerns may be related to the long-term resources at the base of human societies, such as scarcity of fossil resources, or may be orientated towards health impacts or threats to the ecosystem. Whatever the reasons for people's environmental concerns, they result in different actions, policy decisions, or business investments. In order to determine which course of actions is more '*environmentally friendly*', a holistic and structured tools is needed to assess the environmental impacts of different choices (Baumann & Tillman, 2004). Life Cycle Assessment is one of such tools. In its *Communication on Integrated Product Policy* (European Commission, 2003), the European Commission concluded that Life Cycle Assessment provides the best framework for assessing the potential environmental impacts of products or services currently available. In LCA emissions, resources consumed and pressures on health and environment are assessed and attributed to different goods(s) or service(s), taking into account their entire life cycle, from 'cradle to grave'.

LCA approach seeks to quantify all the physical exchanges with the environment, such as inputs of natural resources and energy, and outputs in the form of emissions to air, water and soil. This information is organized in a balance sheet, or *life cycle inventory*, for the system under study. After the inventory is completed, emissions and resources are related to various environmental problems with different way of classification and characterization, based on indicators associated with different burdens, for example resources depletion, climate change, acidification, etc.. Finally, the different environmental impacts related to the life cycle may be put on the same scale through weighting (Baumann and Tillman, 2004).

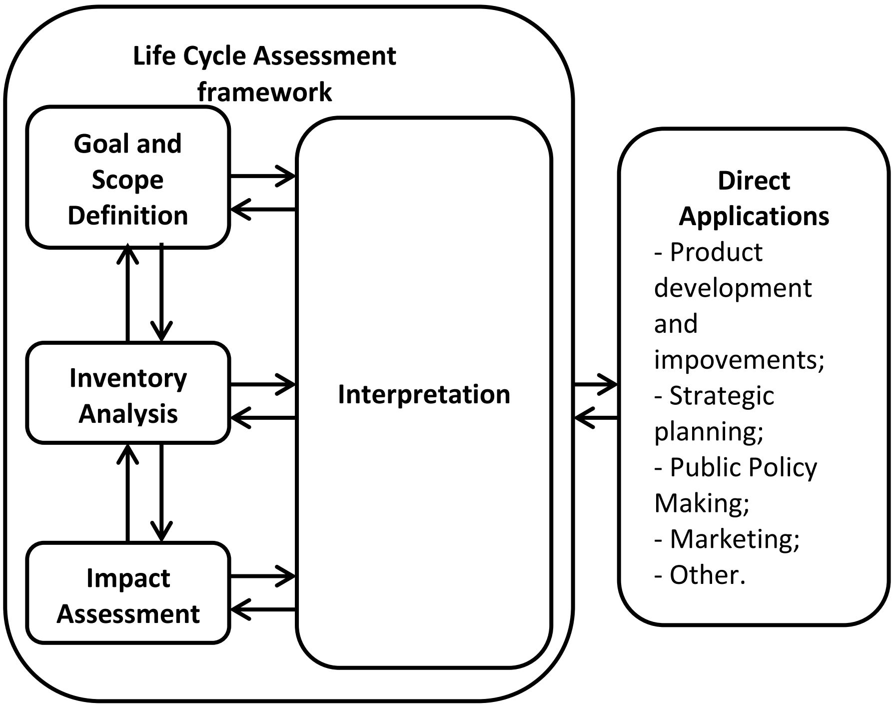


Figure 3.1. Phases of a LCA study (ISO, 2006)**.**

LCA is an international standardised method, and the International Standard Organization provides a rigorous approach for improving decision support in environmental management in the norm ISO 14040 (ISO, 2006). The LCA methodology consists of four stages, as showed in Figure 3.1:

- Goal and scope definition;
- Inventory analysis;
- Impact assessment;
- Interpretation.

Goal and scope definition

The first stage presents the purpose of the study. This includes:

- why the LCA is to be carried out and what decision maker is to be informed by the results;
- the description of system boundaries with the processes and operations which are to be included;
- The definition of the functional unit that will be the basis for comparison and common between all alternatives and limitations and assumptions of the study.

Other things that have to be defined at this stage are the types of environmental impacts that will be considered in the analysis and the level of detail in the study and, thus, the requirements on the data. There is a list of default impacts which are usually considered in a LCA study, such as Global Warming Potential (GWP), Abiotic Depletion (AD), Acidification Potential (AP), Eutrophication Potential (EP), and Nutrient Enrichment (NE). The impacts analysed determine the parameters that will be collected during the inventory phase. The standard stresses that the goal and scope of a LCA study must be clearly defined and consistent with the intended application.

Inventory analysis

In this phase a system model is built according to the requirements of the goal and scope definition. The system model is a flow model of a technical system, sometimes called *techno-sphere*, where the system boundaries are shown, and it represents the incomplete mass and energy balance of the overall system. It is 'incomplete' because only the relevant environmental flows are considered. The system model is usually represented by a flow chart, where the processes/activities included in the analysis are detailed, as well as the system boundaries. After that, the data are collected for each single process. Finally, the amounts of resources and emissions of the system are calculated based on the functional unit. This allows completing the inventory of all relevant *environmental interventions* occurring along the process, thus, '*every human intervention, physical, chemical or biological*', as defined by Guineé et al. (2001).

This phase is complicated when the process under analysis is a multifunctional process. In LCA, a *multifunctional process* is defined as an activity that fulfils more than one function, such as a waste management process dealing with waste and generating energy (Ekvall & Finnveden 2001). It is then necessary to find a rational basis for allocating the environmental burdens between the processes. The problem of allocation in LCA has been the topic of much debate (e.g. Clift et al. 2000; Heijungs & Guinée 2007). The ISO standards (ISO, 2006) recommend that the environmental benefits of recovered resources should be accounted for by broadening the system boundaries to include the avoided burdens of conventional production (Eriksson et al. 2007). The same approach is recommended for product labelling provided that it can be proved that the recovered material or energy is actually put to the use claimed (BSI, 2011). This approach, usually called *system expansion*, is also applied in this thesis.

Impact assessment

In this phase the environmental impacts related to the emissions and resources collected in the inventory phase are described and evaluated in aggregated parameters. The first step of Life Cycle Impact Assessment (LCIA) phase is *classification*, where the environmental interventions listed during the inventory phase are qualitatively classified according to the type of environmental impact they contribute to and assigned to a specific *impact category*. Then the environmental interventions are quantified based on a common unit specific for each category, allowing aggregation to a single score (the *category indicator*). This step is called *characterization.* Such calculations are based on scientific models of cause – effect chains in the natural systems using *characterization factors*[14] defined while modelling the cause-effect chain. The definition of characterization factors is based on the physical-chemical mechanism of how a different substance can contribute to different impact categories, based on a specific *characterization model*. The category indicator, the characterization model and the characterization factors derived from the model constitute the *characterization method* (Guinée, Gorrée, Heijungs, *et al.*, 2001).

An important part of an impact assessment phase is the choice of the characterization method applied to describe and quantify the environmental burdens. These methods are sometimes called *impact assessment methods.* They are based on scientific methods coming from chemistry, toxicology, ecology, etc. The general categories of environmental impacts under consideration in a LCA study are *resource use*, *human health* and *ecological consequences* (Baumann & Tilmann, 2004). They may be defined close to the intervention (the *midpoint* or *problem-oriented* approach) or, alternatively, they may be defined at the level of category endpoints (the *endpoint* or *damage* approach). For most impact categories in LCIA the category indicator describes events in the early stage of the cause-effect chain (*mid-point* characterization method). This means that the *potential* rather than the actual effects of the pollutant are described. In this way, the environmental problems of a specific geographical location are avoided and, as a matter of fact, global impacts are better dealt

[14] The terminology in literature to identify characterization factors is quite varied. It is possible to find *equivalent, potential,* or *category indicator* (ISO, 2006). In this thesis, it is referred as *characterization factor*.

with than local impacts in LCAs. Figure 3.2 shows an example of model for midpoint category indicators used in this study.

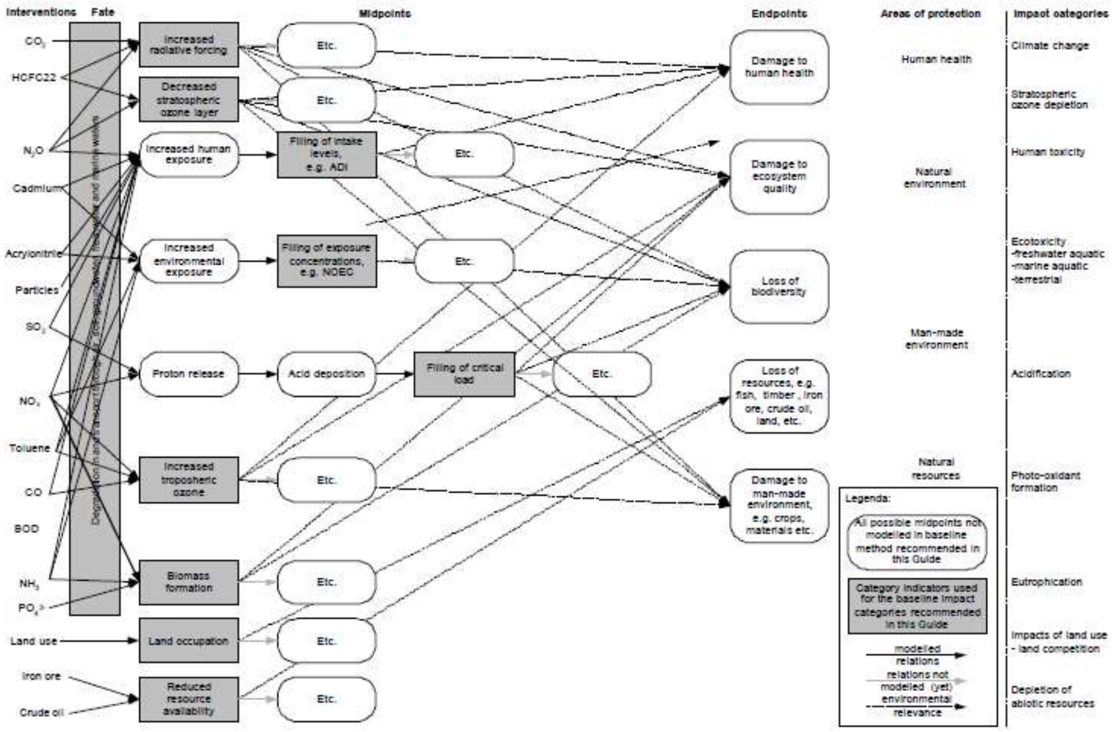


Figure 3.2. Model for midpoint category indicators for impact assessment (Guineè et al. 2001)**.**

Some *impact categories* can have different *characterization methods*, and others can have none. Several LCIA methods exit, where the environmental burden associated with a specific pollutant is indicated relatively to other pollutants and resources. The most famous are:

- Eco-indicator'99 (*www.pre.nl/eco-indicator99*), developed for product development applications with the aim to simplify the interpretation and weighting of results and based mainly on endpoints methodology (European Commission, Joint Research Centre & Institute for Environment and Sustainability, 2010);
- EPS 2000 (*http://eps.esa.chalmers.se/* and *http://www.cpm.chalmers.se/*), developed originally for the Volvo Car Corporation in Sweden, with a midpoint-endpoint structure. The last update took place in 2000 (European Commission, Joint Research Centre & Institute for Environment and Sustainability, 2010);

- EDIP97 (*http://www.dtu.dk/English/Service/Phonebook.aspx?lg=showcommon&id=166960*), supports the classic emission-related impact categories at a midpoint level as well as resources and working environment (European Commission, Joint Research Centre & Institute for Environment and Sustainability, 2010);
- CML (*http://www.leidenuniv.nl/cml/ssp/projects/lca2/lca2.html*), developed by Leiden University, provides best practice for midpoint indicators, with recommendations for normalization and implementing the ISO 14040 series of Standard (Guinée et al, 2001).

The results of the characterization phase can be aggregated into a limited number of impact categories in the *normalization* step. Sometime the results need a further aggregation and interpretation. This can be done in the *weighting* step, with formalized quantitative weighting procedures. While *classification* and *characterization* are compulsory steps in a LCA study (ISO 2006), *normalization* and *weighting* are optional.

Interpretation

Interpretation phase is defined by the standard as the '*phase of life cycle assessment in which the findings of either the inventory analysis or the impact assessment, or both, are combined consistent with the defined goal and scope in order to reach conclusions and recommendations*' (ISO, 2006). In this phase the results obtained in the previous phases are assessed and conclusions are drawn, based on the initial goal and scope of the analysis. An important step of this phase is to test the robustness of the model through *sensitivity analysis, uncertainty analysis,* and *data quality assessments.*

In this work, *sensitivity analysis* has been performed to test the robustness of the model by varying some *key* parameters, thus, those input parameters for which only a small change will lead to a reversal of the results (as the ranking of the different alternative scenarios).

Attributional VS consequential LCA

The LCA community agrees on recognizing two main types of LCA: *attributional* (or *accounting*) and *consequential* (or *change-oriented*). Attributional LCA focuses on the environmentally relevant physical flows to and from a life cycle system or its subsystems. This type of LCA answers questions such as "which environmental impacts can be associated with this product/service". Consequential LCA, on the other hand, focuses on how environmentally relevant flows will change in response to possible decisions (Finnveden et al. 2009). It answers questions as "What would happen if…?". Some authors argue that consequential LCA should be applied to decision-making, when uncertainties in decision outweigh the insights gained within the attributional LCA analysis (Lundie, 2005). Other authors prefer consequential study because of its relevance in understanding the product chain and the weak points in a wider picture (Weidema et al. 1999). Attributional and consequential LCA can both be applied for modelling of the future, past or current systems. Some authors have shown the difference in results when applying attributional and consequential analysis on the same product (Thomassen, Dalgaard, Heijungs, *et al.*, 2008). The choice of attributional or consequential LCA is reflected in different methodological choices of the analysis (Bauman & Tillman, 2004). Table 3.1 summarises the differences.

Table 3.1. Characteristics of attributional and consequential LCA studies (Baumann & Tillman, 2004)**.**

	Type of LCA	
Characteristics	**Attributional**	**Consequential**
System boundaries	Additivity Completeness	Parts of system affected
Allocation procedures	Reflecting causes of the system Partitioning	Reflecting effect of change System expansion
Choice of data	Average	Marginal (at least in part)
System subdivision	-	Foreground and Background

In regards to data type, *average data* represents the average environmental burdens for producing a unit of product/service in the system. *Marginal*[15] data represents the effects of a small change in the output of goods and/or services from a system on the environmental burdens of the same system.

[15] The marginal technology has been defined as '*the technology actually affected by a small change in demand'* (Weidema et al., 1999), and this definition originates from economics where the marginal cost is the cost of producing one more unit of a good. From a mathematical perspective the change is infinitesimal. Using the term 'marginal' therefore implies that the change is insignificant with respect to the affected system. In LCAs of waste management systems where a decision may involve hundred thousand tonnes of waste, it is relevant to ask whether the induced changes can be defined as marginal (Fruergaard 2010).

The decision to use marginal data can be significant when electricity production technologies are being modelled. This is because the technologies used in this field are under development due to the awareness on climate change and fossil fuel scarcity. This is related with temporal effects as well. Short term effects are changes in the utilization of the exiting production capacity in present production plants. Long term effects involve changes in the production capacity and/or technology (Weidema et al. 1999).

In this study, average and marginal data have been used for the electricity production technologies, considering short term effects. In fact, evaluation of long term effects specific for the two countries chosen as case studies would have been a much more time consuming action. Therefore, due to time limitations they have not been investigated.

Limitations of LCA

The main benefit of an LCA analysis is its holistic approach, while at the same time it is also its limitation. In fact, the scope of analyzing the entire life cycle of a product or service can only be achieved at the expense of simplifying other aspects. Briefly, the main limitations of LCA approach are the following (Guinée, Gorrée, Heijungs, *et al.*, 2001):

Local impacts. It is possible to scale down some of the results and to identify the regions in which certain emissions take place, but LCA is not a full local risk assessment study because of the impossibility to take into consideration the pollutants levels of a specific area.

Time aspects. LCA is a steady-state approach rather than dynamic. Future development or changing in the process can be taken into account by the researcher, especially considering marginal technologies rather than average.

Scale effects and linear approach. LCA does not consider market mechanisms or secondary effect on technology development. All the processes are evaluated as linear, both in the economy and in the environment.

Assumptions based. LCA is a science-based approach. However, several technical assumptions have to be made to model the processes, especially when *comparative* LCA analysis is performed which evaluates future scenarios. An important rule is to make these assumptions as transparent as possible.

Referring to the objectives of this study, the validity of the results have to be taken at a global level, or at least at a macro -region level, such as country level. Even if the input data comes from a specific area, the impact has to be considered global. Regarding scale effects, a medium scale is assumed for the digester plant, while the energy system is at a micro scale. Given the state of the development of technologies considered in this study, conclusions regarding scale effects are quite difficult to draw. Finally, a primary aim of this work is to present uncertainties related to technical assumptions and choice of values as clearly as possible, allowing other scientists to apply the approach followed in this study to different context and system's boundaries.

3.2 Previous studies

3.2.1 Life cycle assessment and waste management

Different assessment methods can be applied when evaluating waste management systems. Some of them focus on environmental performances, while other focus on economic aspects. Finnveden et al. (2011) provides an overview of various methods and suggests that '*LCA is an appropriate method for comparing environmental impacts from different waste management options'*. Reviews of the literatures have confirmed that LCA has been used as the prime tool in environmental assessment of waste management studies.

Several papers on application of LCA method to waste management have been analysed. The papers can be divided into two main topics: studies that focus on the methodological approaches in LCA on waste management systems, and studies that focus on the results of these methodologies to specific cases. This section provides summaries of the most significant studies reviewed and cited in this thesis.

Review of studies on methodology approach

In the definition of LCA the term 'product' does not only refer to a physical product system, it can also refer to a service system, for example waste management system. In the paper "*Methodological aspects of life cycle assessment of integrated solid waste management systems*" Finnveden (1999) identifies four main points, which characterize an LCA of waste management systems and that are still topics of much debate. The outlines of the topics are:

- *Upstream and downstream system boundaries.* LCA is usually referred to as 'cradle to grave' assessment, where the 'cradle' is the extraction of raw materials and the 'grave' is the disposal of the product after use. Things are different when waste treatment is one of the main functions of the process under study, where the 'cradle' is the disposal of a product, i.e. when a product enters the waste management system as waste (Fruergaard 2010). This is referred as *zero burden* approach, where all upstream burdens associated with the production of waste are omitted from the LCA (Clift, Doig & Finnveden, 2000). The 'grave' is when the waste is processed at waste treatment facilities and the outcomes are emissions, energy, or secondary materials (as digestate). This approach is applied in this work, thus, the downstream processes related to the production of materials that became waste are not considered. See Section 4.3.

- *Allocation problems.* A multifunctional process is defined as an activity that fulfils more than one function such as a waste management process dealing with waste and generating energy (Ekvall & Finnveden, 2001). It is then necessary to find a rational basis for allocating the environmental burdens between the processes. The problem of allocation in LCA has been the topic of much debate (Clift, Doig & Finnveden, 2000). The ISO standards recommend that the environmental benefits of recovered resources should be accounted for by broadening the system boundaries to include the avoided burdens of conventional production (Eriksson et al. 2007). A practical example made by Finnveden (1999) and reported here is the comparison between landfill and incineration of solid waste. The main function is the treatment of solid waste, but incineration plant can have a heat/electricity recovery unit, thus providing a second function. Since the two processes provide different functions, a direct comparison is not possible. If the system is expanded and *system expansion* is applied, an alternative method for producing the equivalent amount of heat/electricity is considered in the landfill scenario. It is then possible to compare the incineration system to the combined landfill and heat/electricity production system (see Figure 3.3). The so-called *avoided burdens* (or *avoided emissions*) associated with the avoided heat/electricity production can transform the environmental impact of incineration scenario from positive to negative values (see Figure 3.4). In particular, the broad perspective of an LCA makes it possible to take into account the significant environmental benefits that arise from different waste management processes. For example, waste incineration with energy recovery reduces the need for other energy sources; material from recycling processes replaces production of virgin material; biological treatment may reduce the need for production of commercial fertilisers and vehicle fuel; residues from waste incineration may be used for road constructions. System expansion is normally used in multifunctional system studies and it is recognized by the standard as a means to avoid allocation problems. However, discussions on the drawbacks of using system expansion are still on-going (Finnveden 1999; Heijungs & Guinée 2007).

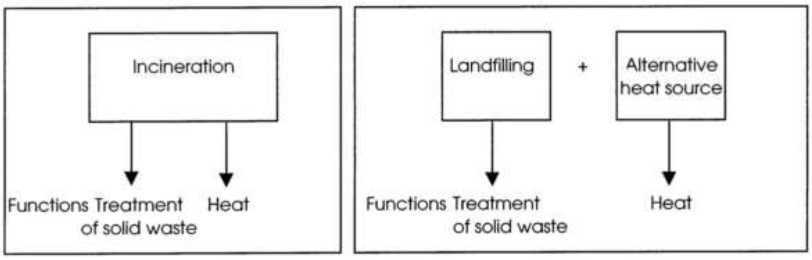


Figure 3.3. Application of system expansion to avoid allocation problems (Finnveden 1999)**.**

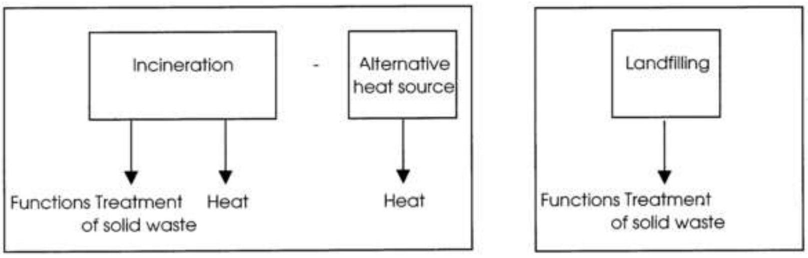


Figure 3.4. Alternative way of presenting system expansion (Finnveden 1999)**.**

- *Time aspects.* This becomes relevant when landfill is one of the options under study. Emissions from landfills may prevail for a very long time, often thousands of years or more. In order to make the potential emissions from landfilling comparable to other emissions during the life cycle, the potential emissions have to be integrated over a certain time-period. This will be discussed in Section 4.5.
- *Impact assessment*. When analysing emissions from landfill, the emissions will occur in a future situation. The emissions cannot be measured, they can only be predicted. As a consequence, only potential emissions rather than actual emissions can be included in the LCA for the landfilling processes. This can make the impact assessment more difficult because there are increased problems with modelling

background concentrations and other aspects which may be important for the impact assessment. The standard solution to this problem is to treat all emissions as if they occur at the same moment (Finnveden 1999). This will be applied in this study.

Clift et al. (2000) summarised the methodology for applying LCA to Integrated Waste Management of MSW developed for and used by the UK Environment Agency in their paper '*The application of life cycle assessment to Integrated Solid Waste Management – Part 1*'. Particular attention is given to a system definition leading to rational and clear compilation of the Life Cycle Inventory. The same definitions are applied in this study.

First of all, a pragmatic distinction between *Foreground* and *Background* is suggested, considering the first as '*the set of processes whose selection or mode of operation is affected directly by decisions based on the study*' and the second as '*all other processes which interact with the Foreground, usually by supplying or receiving material or energy'* (Clift at al., 2000). The principal distinction lies in the way the inventory data is compiled. The Foreground should be described by primary data based on the actual processes and their operating conditions if such data is available. The Background activities can be described by generic average industry data, for example taken from a reliable database of life cycle inventory data. Figure 3.5 shows a schematic concept of Foreground and Background systems.

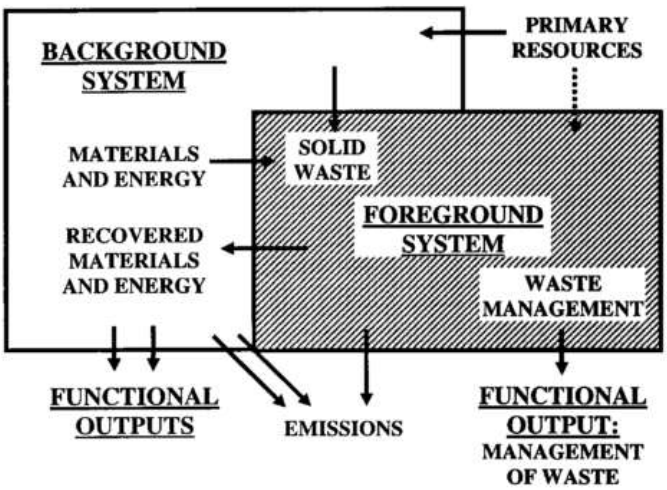


Figure 3.5. General representation of Foreground and Background systems (Clift et al. 2000).

Clift et al (2000) suggested evaluating the burdens considering the following three categories:

- *direct burdens* – those coming directly from the foreground system;
- *indirect burdens* – those due to upstream and downstream processes in the background system (such as energy provision for electricity or diesel for transportation) and
- *avoided burdens* – associated with the products that are replaced with those from the foreground system (such as energy or fertilizer).

The recommended way by Clift et al. (2000) to report the Life Cycle Inventory for a waste management system is to add *Direct Burdens* associated with the waste management operations themselves; add *Indirect Burdens* associated with providing materials and energy to the waste management operations; subtract *Avoided Burdens* associated with economic activities which are displaced by materials and/or energy recovered from waste. This approach is applied in this study.

Review of LCA studies applied to anaerobic digestion and waste treatment technologies

Several models exist in the literature that apply LCA methodology to the treatment of OFMSW via AD. A brief summary of the main characteristics of each method is given in Table 3.2.

Table 3.2. Summary of LCA models applied to Anaerobic Digestion process from OFMSW from the literature.

MODEL	DESCRIPTION
IFEU	German model based on *UMBERTO* software tool, initially developed for environmental assessment of waste systems. It allows a comparison of different treatment options for urban organic waste and the quantification of environmental effects from biological treatment is relatively detailed.
ORWARE	Swedish model developed in cooperation with different research institutes and universities. Initially developed for environmental assessment of biodegradable liquid and organic waste (including sludge), but it can also handle treatment of mixed waste.

MSW-DST	Model for environmental and economic assessment of integrated waste management systems developed by the Research Triangle Institute, North Carolina State University and the United States Environmental Protection Agency. It includes a module for composting of organic MSW, eventually pre-treated to reduce contamination.
WRATE	It is an English model for environmental assessment of waste management systems. It was jointly developed by Environmental Resources Management and Golder Associates on behalf of the Environment Agency for England and Wales. It includes a number of bio-treatment technology modules based on a consistent process input/output framework. It includes 4 type of anaerobic digestion, 10 types of composting and 15 types of MBT-plants and default data.
IWM2	It was developed by Procter & Gamble for environmental and economic assessment of waste management systems. Composting and anaerobic digestion are the options available in the module for biological treatment. Different technologies can be modelled in each sub-module by defining specific process data and parameters. Different pre-treatments are available prior to the bio-treatment.
WISARD	Energy and material consumptions, and the mass balance are calculated on a monthly basis and then converted to process specific data using time-based operational parameters of the facility (e.g. working days per month, tonnage treated per month). The anaerobic digestion module includes modelling of biogas production and utilization. Biogas production is calculated on the amount of putrescible material within the waste using biogas generation values provided by defaults in the database for the landfill gas generation. The composition of biogas (CO2, CH4, H2S, hydrocarbons) is defined on a mass basis (g kg 1 of biogas) and a fraction of its loss between the digester and the generator.
LCA-IWM	It is a model for assessing the environmental sustainability of municipal waste management planning. It was funded by the European Commission under the Fifth Framework Programme and developed by a cooperation of different institutions (Technical University of Darmstadt, Germany; University of Tarragona, Spain; novaTec, Luxemburg). Anaerobic digestion and composting are the available biological treatments in LCA-IWM. In both cases, mechanical pre-treatments are possible and waste characteristic is a user defined input. Default mass flows of the

	processes are provided by the model, but adjustments can be made by the user.
EASYWASTE	It supports LCA studies throughout the four above mentioned LCA phases. It includes modelling of anaerobic digestion, composting, combined anaerobic digestion and composting, Mechanical-biological-treatment (MBT) plants for treatment of mixed residual waste. Both source-separated and mixed waste can be treated in the biological treatment module.
	PROCESS DESCRIPTION
IFEU	Anaerobic digestion is modelled as 'wet one-step mesophilic digestion' and 'dry one-step thermophilic digestion'.
ORWARE	Anaerobic digestion is simplified by continuously stirred tank reactor (CSTR) one-step mesophilic digestion. Four pre-treatments are possible for the incoming waste: hygienisation (70°C) sterilization (130°C), maceration and separation of metal and plastic.
WRATE	The composition of bio-treated material is predefined in the model where four generic grades of bio-treated materials can be chosen. The biological process includes construction, maintenance and decommissioning data.
IWM2	The anaerobic digestion module calculates the amount of biogas and compost produced based on the mass of organics lost during the process, defined by the user.
WISARD	The facility (e.g. working days per month, tonnage treated per month). The anaerobic digestion module includes modelling of biogas production and utilization. Biogas production is calculated on the amount of putrescible material within the waste using biogas generation values provided by defaults in the database for the landfill gas generation. The composition of biogas (CO2, CH4, H2S, hydrocarbons) is defined on a mass basis (g kg 1 of biogas) and a fraction of its loss between the digester and the generator.
LCA-IWM	Anaerobic digestion is defined with a thermophilic dry 1-stage process. Wastewater, biogas, and digestate are the outputs of the process.
EASYWASTE	Wet, dry, and semidry as well as one and two steps processes. The main differences are energy consumption, emissions of unburned methane and energy

	production.
	DIGESTATE TREATMENT
IFEU	The digested organic waste is separated into a wet and dry fraction: the wet fraction led to wastewater treatment and the dry fraction stabilized by composting step.
ORWARE	The produced compost/digestate can be routed to a use-on-land module, where environmental consequences of spreading residuals to arable land can be modelled. The composition of bio-treated material is calculated based on the waste composition.
WRATE	The digested organic waste is separated into a wet and dry fraction: the wet fraction led to wastewater treatment and the dry fraction stabilized by composting step.
IWM2	Liquid residue from the digestion process is routed to wastewater treatment plant, but such process is not included in the model. The composition of digestate is predefined in the model. Both the compost and the eventual residue from screening operation can be routed to further treatments (incineration, landfill). Compost can as an alternative be sent out on the market; replacement of mineral fertilizers is modelled by means of default substitution processes.
WISARD	Flows of reject materials can be routed to further treatment or to disposal.
LCA-IWM	Bio-treated material and wastewater compositions are calculated based on the waste composition. Leaching coefficients are used to determine the distribution of several substances between wastewater and digestate. Wastewater can be routed (user defined) to a wastewater treatment plant, which includes phosphorous removal and sludge stabilisation. The solid residue produced during the digestion process is aerobically stabilized to produce marketable compost. The maturation phase is 4 weeks long and takes place in windrows in a rotting hall. No air cleaning system is installed.
EASYWASTE	The composition of bio-treated material is calculated based on material input composition by means of degradation ratios and transfer coefficients. The degradation of organic material (VS) for each waste material fraction is user defined. It is calculated based on the methane potential and methane yield in case

	of anaerobic digestion. The TS is distributed between the defined output fractions according to used defined transfer coefficients (TCs) that have to be defined for each input material fraction. The mass balancing approach is linear, as in the LCA context a process is treated as a "black box".
	BIOGAS USE
IFEU	The biogas is used for electricity delivered to the grid, and heat, primarily utilized at the biogas plant.
ORWARE	The production of biogas (CO2 + CH4) is proportional to the amount of organic matter degraded. The electricity consumption is estimated to be approximately 5% of the energy contained in the biogas, while the heat consumption is estimated taking into account the surface area of the reactors and the retention times. Utilisation of the biogas is modelled in a separate sub-module, with various options for the energy recovery (engine, boiler, buses, cars and trucks). When the produced biogas is combusted in a stationary engine, the energy recovery efficiency is 30% for electricity and 60% for heat.
WRATE	For anaerobic digestion systems, the quantity of electric energy produced is linearly correlated to the quantity of biogenic carbon in the incoming waste however it is not possible to see the quantity of methane produced by such a system.
IWM2	The amounts of energy used in the process and recovered from biogas are estimated by the user and entered in the model in terms of kWh Mg 1 wet weight (ww) input to the digester. The value regarding energy production covers a number of other parameters determining the methane yield and recovery, such as the methane potential of the waste, methane and energy contents of the biogas, and engine type and efficiency.
WISARD	Use of energy for transportation and spreading on land of the stabilized material is accounted for in the model. The quantity of electricity and heat recovered from biogas utilization is defined on a monthly basis. For the heat recovered, the substituted energy technology is defined by the user – coal, natural gas and oil are the options.
LCA-IWM	Biogas yield is calculated based on methane potential defined by the user in case taking into account the level of contamination in the waste. The produced biogas

	is used for energy production in a combustion unit with electricity and (optionally) heat recovery (CHP). The amount of the energy generated is linked to the amount and quality of the biogas, while air emissions of several substances generated during the combustion process are process specific and user defined (in mg m 3 of flue gas) and should take into consideration a possible installation of a flue gas treatment system. In the default data it is assumed that the CHP is equipped with an oxidative catalytic air cleaning unit. Energy consumptions occurring throughout all phases of the process are summed up to a unique ''overall energy consumption'' value, which is used in the calculation.
EASYWASTE	It considers that the methane generation is not proportional to the VS degradation rate, to avoid overestimating methane production. The biogas produced during anaerobic digestion can be flared, combusted in an engine or upgraded and used as fuel in motor vehicles. The amount of energy recovered from biogas is related to the energy content of CH4 from digestion process. It has a separate module called "Biogas Treatment". It has two parts, one to determine the energy recovery and the second to specify the process-specific emissions regarding the process for combustion of biogas.
EMISSIONS AND CYNETIC	
IFEU	Emission of unburned methane from combustion of the gas is quantified and included in the assessment. Detailed mass balances for carbon and nitrogen describe the fate of two components in the investigated systems. During the AD a fraction of the carbon is transformed into biogas or eventually lost to wastewater. The rest of the carbon is transferred to the composting step, where it is partly lost as emissions to air (CO2 or CH4) or as wastewater. A substantial part of the nitrogen contained in the wet fraction is lost as ammonia. During the composting stage nitrogen may be emitted to air (NH3, N2O, N2) or to wastewater. The remaining nitrogen is found in the treated organic waste.
ORWARE	The modelling is done using a modular approach and transfer coefficients are used to define the elemental distribution in each environmental compartment. Transfer coefficients are also used to model the degradation of organic persistent pollutants (CHX, AOX, PAH, phenols, PCB and dioxins). Biological treatments available are anaerobic digestion and composting. The degradation of organic matter is estimated taking into account the degradation potential of the substrate

	and the retention time in the digester. Mineralisation of organic nitrogen to ammonium and sulphur to hydrogen sulphide are proportional to the degradation ratio of proteins contained in the waste.
WRATE	The direct operating emissions are based on the typical emissions of existing plants (whether measured or estimated). These emissions are directly proportional to the composition (waste specific emissions) and the quantity of waste (process specific emissions) defined by the user. All the typical fugitive emissions (e.g. CH4, N2O, NH3) are defined for each waste management facility. N emissions are calculated based on the N content of the incoming waste.
IWM2	Air emissions are included in terms of CO2, generated both from the degradation process and the combustion of biogas. No other air emissions are considered.
WISARD	Emission factors for different material and processes (e.g. diesel combustion) are included by default, but they can be re-defined by the user.
LCA-IWM	Emissions to air are modelled using emissions factors describing distribution of degraded C- and N containing matter into different compounds (i.e. CO2, CH4, NMVOC, NH3, N2O, N2). It is assumed that no leaching of metals and nutrients occurs during the maturation phase. The produced compost can be further routed to a use on land module.
EASYWASTE	Emissions of CH4, NH3 and N2O are modelled as a function of the degradation of C- and N containing compounds. It is assumed that no emissions of nitrogen containing substances occur during the digestion process.
	REFERENCES
IFEU	(Ifu Hamburg GmbH, 2013)
ORWARE	(Bjorklund et al. 1999; Eriksson et al. 2002; Assefa et al. 2005; Dalemo 1996; Dalemo 1996; Dalemo et al. 1997; Eriksson et al. 2011)
MSW-DST	(Thorneloe et al. 2007)
WRATE	(Burnley & Coleman, 2012)
IWM2	(McDougall 2005; Thomas & McDougall 2005)
WISARD	(Ecobilan, 1997)

LCA-IWM	(den Boer et al. 2007)
EASYWASTE	(Manfredi 2009; Astrup et al. 2012; Turconi et al. 2011; Kirkeby et al. 2007; Manfredi et al. 2011)

Gentil et al., (2010) and Boldrin et al., (2011) presented reviews of published LCAs of AD treatment. It is difficult to draw conclusions from earlier studies on the comparison between different treatment technologies and scenarios because the studies differ in the assumptions made on system boundaries and methodology for evaluating Life Cycle Inventory burdens and often insufficient details are provided on the methodology used. Furthermore, most of the studies that tackle LCA of landfill treatments focus on general MSW (Arena et al. 2003; Cherubini et al. 2009; Manfredi et al. 2009; Manfredi 2009; Manfredi & Christensen 2009) or report insufficient details to enable a comparison with anaerobic digestion (Patterson et al. 2011; Nielsen & Hauschild 1998). To the authors' knowledge, only the study of Patterson et al., (2011) addresses the LCA of anaerobic digestion systems in the UK, but the study focuses on Wales alone. There are only a few studies with case studies for Italy and they investigate mainly landfill and incineration treatments, while no studies focus on AD process. This is because there are only a few operating AD plants treating OFMSW exist in Italy, as stated in Section 2.1.5.

3.2.2 Life cycle assessment and micro CHP systems

There are only a few studies at present that focus on LCA of SOFC implemented systems (Staffell & Green 2012; Giannopoulos & Founti 2011a; Pehnt 2001; Pöschl et al. 2010; Patterson et al. 2011; Lunghi et al. 2004). The majority of them focus on the use phase of the SOFC unit, considering only natural gas as a fuel. Staffell et al (2012) carried out a carbon footprint assessment of a SOFC based on domestic CHP with the current embedded technologies in the UK. They evaluated both the manufacturing and use phase of SOFC unit, concluding that the production of the fuel cell accounts only for 10% of total emissions. They considered natural gas as a fuel, two cases for the displaced energy production technology - UK grid mix (2009) and high efficiency Combined Cycle Gas Turbine plant- and they designed the fuel cell to give a least-cost operation[16]. The only experimental field trial reported up to now seems to be the 67 Kyocera systems operating in Japanese homes from 2009 to 2011. Penth (2008) investigated environmental impacts of several micro cogeneration systems. It considered natural gas as a fuel and optimized operational pattern to supply energy to a single German dwelling, different for each micro generation system analysed: fuel cells, Stirling engines and internal combustion engines. He concluded that the achievable reduction is low because in the optimized operation the micro cogeneration systems do not supply the whole energy demand and electricity from the grid and heat from conventional boiler are needed. Giannapoulis and Founti (2011) compared the environmental impact of a SOFC unit with an internal combustion engine, both run on natural gas and displacing electricity from the grid and condensing boiler. In UK, several authors investigate micro –CHP systems modelling for residential applications and environmental impacts of those systems (Cockroft & Kelly 2006; Hawkes et al. 2009; Hawkes et al. 2009a; Hawkes et al. 2009b; Hawkes & Leach 2005; Hawkes et al. 2011). Hawkes & Leach (2005) analysed the impact of temporal precision in optimization modelling of two systems: SOFC micro –CHP unit and Stirling engine. They concluded that considering 1h demand blocks in heat and power demand data '*leads to averaging effects results in misleading environmental and economic outcomes for micro-CHP systems*'. They found up to 40% of variation range between precisions analysed for energy generated and CO_2 emissions reduction. Cockroft and Kelly (Cockroft & Kelly, 2006) analysed the performance of four different micro-CHP technologies compared with a standard situation (condensing boiler and electricity grid), in order to identify the CO_2 emissions reduction that can be achieved. They concluded that air source heat pump achieves more CO_2 savings than any of the other technologies examined (Stirling engine, fuel cells, internal combustion engine).

To the author's knowledge, only three studies considered other kind of fuel for a FC micro-CHP unit. Lunghi and Desideri (2004) conducted an LCA of a Molten Carbonate Fuel cell system for landfill gas recovery. They compared biogas from landfill with natural gas from the grid. Patterson et al (2011) completed an LCA on potential biogas infrastructures on a regional scale in Wales. They analysed different scenario for the biogas utilization, including

[16] Least-cost operation profile means that the FC is designed to follow the maximum of local electrical and thermal loads, with the constraint of always running at a minimum of 20% power (Hawkes & Leach 2005).

upgrading to bio-methane, but not for FC applications. Another study dealing with different biogas utilization pathways – including fuel cell - is the one by Poschl et al (2010). They did not perform an LCA analysis, focusing instead on energy efficiency evaluation of various feedstock and biogas utilization.

In conclusion, from all studies analysed it is evident that the assumptions made on the energy technology displaced significantly influences environmental impact results.

3.3 Renewable carbon

Biogenic CO_2 emissions are defined as '*emissions from a stationary source directly resulting from the combustion or decomposition of biologically-based materials other than fossil fuels*' (US EPA, 2006). In this study, following the approach of Manfredi et al. (2011) and Christensen et al. (2009) in LCA of waste management studies, the biogenic CO_2 is considered neutral with respect to global warming because it is part of the renewable carbon cycle. Therefore, its characterisation factor is taken as zero throughout the study. This leads to treating biogenic carbon remaining in the landfill or in the digestate and soil after 100 years as sequestered and removed from the atmosphere; this is discussed further in Chapter 5.

Amongst LCA community, the biogenic carbon is a widely discussed issue and recent publications investigated the concept of 'bioenergy carbon debt' (US EPA, 2011; Cherubini, Bright & Strømman, 2012). As reported by Levasseur et al. (2011) '*the combustion of biomass causes more GHG emissions per unit of energy compared to the use of fossil fuels, creating a carbon debt. Then the debt is paid down as the biomass grows up and sequesters carbon from the atmosphere. However, by the time the biomass grows up, the additional amount of carbon released by the replacement of fossil fuels with bioenergy has an impact on climate, especially for wood, because forests often take decades to mature'.* In the same paper, Levasseur et al. provided a comparison amongst different LCA methodologies, analysing results with and without the biogenic carbon. They concluded that, apart from the lack of consensus on how to treat biogenic carbon, '*to do away with the paradigm of biogenic CO_2 carbon neutrality improves the decisions made with LCAs in several cases, but increases the need for reliable data*'.

In regards to the objectives of this work, considering biogenic CO_2 emissions would improve considerably the emissions reduction achievable with the AD treatment of the OFMSW. This is because (as shown later in Chapter 5) biogenic carbon in the OFMSW is a considerable fraction of the total carbon. For this reason and also due to the lifetime of technologies involved in the distributed generation scenarios, the biogenic CO_2 is not accounted for in this study.

The modelling of bio-treatment processes in different LCA studies in literature is also differentiated by the assumptions made concerning the biogenic carbon sequestration. As noted by Gentil et al. (2010) EASEWASTE calculates the quantity of biogenic carbon sequestered, estimated to be between 10% and 15%, depending on the soil type, over a 100 year period. EPIC/CSR includes carbon sequestration but only for paper recycling (e.g. increased forest sequestration due to reduced demand on virgin paper) and landfill, but exclude sequestration from bio-treatment processes. Carbon sequestration is not included in IWM2. In LCA–IWM sequestration is excluded from landfill but included in the soil application from compost (8.2%, over a 100 year period). It is possible to model carbon sequestration in ORWARE, which is set at different values for sequestration to soil (from land spreading), where 20% of the biogenic carbon is assumed to be bound in soil and 3–10% of the biogenic carbon is assumed to be sequestered in landfill. These values are currently been updated. In WRATE, sequestration is assumed to be 2% for soil application. These different assumptions will generate differences in the LCA results and need to be addressed when comparing waste LCA models.

4. LCA of waste to energy technologies for distributed generation

4.1 Introduction

In this chapter the whole system analysed in the work is presented. The structure of the chapter reflects the phases required by the ISO standard when a LCA analysis is performed. The system under analysis is a multifunctional system, thus from one side it provides energy and from the other it can be seen as a mean of waste treatment technology. For this reason the work has been carried out analysing two different subsystems, with two different functional units. The chapter follows this methodological choice.

4.2 Goal and Scope definition of the overall study

The goal of this LCA is to evaluate and compare the environmental impacts of different scenarios for the exploitation of the OFMSW for energy production, in the distributed generation paradigm. The scope is to investigate the potentiality to create a *waste - to - energy closed loop*, thus from the collection of the waste to the production and distribution of the energy to the single dwelling. Three *Distributed Generation* scenarios – lately called DG scenarios - are compared: starting from the production of biogas via AD, feed this bio-fuel to three different micro-CHP units: a SOFC unit, a micro-gas turbine unit and a Stirling engine unit.

Three different scenarios are investigated for the operating strategy of the micro-CHP systems, designing the energy unit to satisfy: the full thermal demand, the half thermal demand and the electricity demand - later identified as *FT*, *HT* and *EL* scenarios respectively - of the dwellings. Further scenarios come from considering both the use of raw and upgraded biogas (biomethane) to feed the SOFC micro-CHP unit.

To demonstrate the feasibility of the *waste to energy closed loop* system, two different case studies are analysed, in two geographical areas: the Royal Borough of Greenwich in London (UK) and the municipality of Livorno (IT). The two chosen countries show a similar behaviour in terms of waste management as national average treatments, and for both of them landfill is still the final destination for around 50% of the waste generated (see Section 2.1.).

The treatment of the waste via AD is compared with other two competitive and spread technologies: landfill plant with electricity production and incineration with energy recovery. Thus, the environmental impact associated with the alternative treatments of the same amount of OFMSW is evaluated as avoided burdens for the function 'waste management', in the different scenarios investigated – later indicated as *WM* scenarios.

Natural gas is considered a competitive fossil fuel when compared with biogas. For this reason, another scenario investigated is the natural gas from the grid running the same SOFC micro-CHP unit analysed in the DG scenarios.

A critical operational threshold is determined, over which the environmental burdens associated with the DG scenarios are lower than a *reference scenario*[17] in terms of functions provided: energy production and organic fertiliser. The AD plant in fact gives another by-product: the digestate, which, as stated in Section 2.6.2, can be substituted to commercial fertiliser. The avoided burdens can finally be considered through the application of *system expansion*.

The full list of the scenarios analysed is shown in Table 4.1.

Table 4.1 List of all scenarios analysed in the study.

	Scenarios	Fuel	Technology	Operating Strategy	Avoided Waste Treatment	Electricity Scenario	Case Study
1	DG_BIOGAS_FC_FT_AV_UK	Biogas	FC	Full Thermal	Landfill	Average	UK
2	DG_BIOGAS_FC_HT_AV_UK	Biogas	FC	Half Thermal	Landfill	Average	UK
3	DG_BIOGAS_FC_EL_AV_UK	Biogas	FC	El	Landfill	Average	UK
4	DG_BIOGAS_FC_FT__MARGINAL_U	Biogas	FC	Full Thermal	Landfill	Marginal	UK

[17] Some authors refer to it as *compensatory system* (Eriksson et al. 2007)

	Scenarios	Fuel	Technology	Operating Strategy	Avoided Waste Treatment	Electricity Scenario	Case Study
	K						
5	DG_BIOGAS_FC_FT_MARGINAL_UK	Biogas	FC	Half Thermal	Landfill	Marginal	UK
6	DG_BIOGAS_EL_FC_MARGINAL_UK	Biogas	FC	El	Landfill	Marginal	UK
7	DG_BIOMET_FT_FC_AVERAGE_UK	Biomethane	FC	Full Thermal	Landfill	Average	UK
8	DG_BIOMET_HT_FC_AVERAGE_UK	Biomethane	FC	Half Thermal	Landfill	Average	UK
9	DG_BIOMET_EL_FC_AVERAGE_UK	Biomethane	FC	El	Landfill	Average	UK
10	DG_BIOMET_FT_FC_MARGINAL_UK	Biomethane	FC	Full Thermal	Landfill	Marginal	UK
11	DG_BIOMET_HT_FC_MARGINAL_UK	Biomethane	FC	Half Thermal	Landfill	Marginal	UK
12	DG_BIOMET_EL_FC_MARGINAL_UK	Biomethane	FC	El	Landfill	Marginal	UK
13	DG_BIOGAS_mGT_FT_AVERAGE_UK	Biogas	micro GT	Full Thermal	Landfill	Average	UK
14	DG_BIOGAS_mGT_HT_AVERAGE_UK	Biogas	micro GT	Half Thermal	Landfill	Average	UK
15	DG_BIOGAS_mGT_EL_AVERAGE_UK	Biogas	micro GT	El	Landfill	Average	UK
16	DG_BIOGAS_mGT_FT_MARGINAL_UK	Biogas	micro GT	Full Thermal	Landfill	Marginal	UK
17	DG_BIOGAS_mGT_HT_MARGINAL_UK	Biogas	micro GT	Half Thermal	Landfill	Marginal	UK
18	DG_BIOGAS_mGT_EL_MARGINAL_UK	Biogas	micro GT	El	Landfill	Marginal	UK
19	DG_BIOGAS_SE_FT_AVERAGE_UK	Biogas	Stirling Engine	Full Thermal	Landfill	Average	UK
20	DG_BIOGAS_SE_HT_AVERAGE_UK	Biogas	Stirling Engine	Half Thermal	Landfill	Average	UK
21	DG_BIOGAS_SE_EL_AVERAGE_UK	Biogas	Stirling Engine	El	Landfill	Average	UK
22	DG_BIOGAS_SE_FT_MARGINAL_UK	Biogas	Stirling Engine	Full Thermal	Landfill	Marginal	UK
23	DG_BIOGAS_SE_HT_MARGINAL_UK	Biogas	Stirling Engine	Half Thermal	Landfill	Marginal	UK
24	DG_BIOGAS_SE_E	Biogas	Stirling	El	Landfill	Marginal	UK

	Scenarios	Fuel	Technology	Operating Strategy	Avoided Waste Treatment	Electricity Scenario	Case Study
	L_MARGINAL_UK		Engine				
25	DG_NG_FC_FT_AVERAGE_UK	Natural Gas	FC	Full Thermal	-	Average	UK
26	DG_NG_FC_HT_AVERAGE_UK	Natural Gas	FC	Half Thermal	-	Average	UK
27	DG_NG_FC_EL_AVERAGE_UK	Natural Gas	FC	El	-	Average	UK
28	DG_NG_FC_FT_MARGINALE_UK	Natural Gas	FC	Full Thermal	-	Marginal	UK
29	DG_NG_FC_HT_MARGINAL_UK	Natural Gas	FC	Half Thermal	-	Marginal	UK
30	DG_NG_FC_EL_MARGINAL_UK	Natural Gas	FC	El	-	Marginal	UK
31	CG_AVERAGE_LANDFILL_UK	Biogas	ICE	-	LANDFILL	AVERAGE	UK
32	CG_MARGINAL_LANDFILL_UK	Biogas	ICE	-	LANDFILL	MARGINAL	UK
33	CG_AVERAGE_INCINERATOR_UK	Biogas	ICE	-	INCINERATOR	AVERAGE	UK
34	CG_MARGINAL_INCINERATOR_UK	Biogas	ICE	-	INCINERATOR	MARGINAL	UK
35	DG_BIOGAS_FC_FT_AV_IT	Biogas	FC	Full Thermal	Landfill	Average	IT
36	DG_BIOGAS_FC_HT_AV_IT	Biogas	FC	Half Thermal	Landfill	Average	IT
37	DG_BIOGAS_FC_EL_AV_IT	Biogas	FC	El	Landfill	Average	IT
38	DG_BIOGAS_FC_FT__MARGINAL_IT	Biogas	FC	Full Thermal	Landfill	Marginal	IT
39	DG_BIOGAS_FC_FT_MARGINAL_IT	Biogas	FC	Half Thermal	Landfill	Marginal	IT
40	DG_BIOGAS_EL_FC_MARGINAL_IT	Biogas	FC	El	Landfill	Marginal	IT
41	DG_BIOMET_FT_FC_AVERAGE_IT	Biomethane	FC	Full Thermal	Landfill	Average	IT
42	DG_BIOMET_HT_FC_AVERAGE_IT	Biomethane	FC	Half Thermal	Landfill	Average	IT
43	DG_BIOMET_EL_FC_AVERAGE_IT	Biomethane	FC	El	Landfill	Average	IT
44	DG_BIOMET_FT_FC_MARGINAL_IT	Biomethane	FC	Full Thermal	Landfill	Marginal	IT
45	DG_BIOMET_HT_	Biomet	FC	Half	Landfill	Marginal	IT

	Scenarios	Fuel	Technology	Operating Strategy	Avoided Waste Treatment	Electricity Scenario	Case Study
	FC_MARGINAL_IT	hane		Thermal			
46	DG_BIOMET_EL_FC_MARGINAL_IT	Biomethane	FC	El	Landfill	Marginal	IT
47	DG_BIOGAS_mGT_FT_AVERAGE_IT	Biogas	micro GT	Full Thermal	Landfill	Average	IT
48	DG_BIOGAS_mGT_HT_AVERAGE_IT	Biogas	micro GT	Half Thermal	Landfill	Average	IT
49	DG_BIOGAS_mGT_EL_AVERAGE_IT	Biogas	micro GT	El	Landfill	Average	IT
50	DG_BIOGAS_mGT_FT_MARGINAL_IT	Biogas	micro GT	Full Thermal	Landfill	Marginal	IT
51	DG_BIOGAS_mGT_HT_MARGINAL_IT	Biogas	micro GT	Half Thermal	Landfill	Marginal	IT
52	DG_BIOGAS_mGT_EL_MARGINAL_IT	Biogas	micro GT	El	Landfill	Marginal	IT
53	DG_BIOGAS_SE_FT_AVERAGE_IT	Biogas	Stirling Engine	Full Thermal	Landfill	Average	IT
54	DG_BIOGAS_SE_HT_AVERAGE_IT	Biogas	Stirling Engine	Half Thermal	Landfill	Average	IT
55	DG_BIOGAS_SE_EL_AVERAGE_IT	Biogas	Stirling Engine	El	Landfill	Average	IT
56	DG_BIOGAS_SE_FT_MARGINAL_IT	Biogas	Stirling Engine	Full Thermal	Landfill	Marginal	IT
57	DG_BIOGAS_SE_HT_MARGINAL_IT	Biogas	Stirling Engine	Half Thermal	Landfill	Marginal	IT
58	DG_BIOGAS_SE_EL_MARGINAL_IT	Biogas	Stirling Engine	El	Landfill	Marginal	IT
59	DG_NG_FC_FT_AVERAGE_IT	Natural Gas	FC	Full Thermal	-	Average	IT
60	DG_NG_FC_HT_AVERAGE_IT	Natural Gas	FC	Half Thermal		Average	IT
61	DG_NG_FC_EL_AVERAGE_IT	Natural Gas	FC	El	-	Average	IT
62	DG_NG_FC_FT_MARGINALE_IT	Natural Gas	FC	Full Thermal	-	Marginal	IT
63	DG_NG_FC_HT_MARGINAL_IT	Natural Gas	FC	Half Thermal	-	Marginal	IT
64	DG_NG_FC_EL_MARGINAL_IT	Natural Gas	FC	El	-	Marginal	IT
65	CG_AVERAGE_LANDFILL_IT	Biogas	ICE	-	Landfill	Average	IT
66	CG_MARGINAL_LANDFILL_IT	Biogas	ICE	-	Landfill	Marginal	IT
67	CG_AVERAGE_INCINERATOR_IT	Biogas	ICE	-	Incinerator	Average	IT

	Scenarios	Fuel	Technology	Operating Strategy	Avoided Waste Treatment	Electricity Scenario	Case Study
68	CG_MARGINAL_INCINERATOR_IT	Biogas	ICE	-	Incinerator	Marginal	IT

Following the definition given in section 3.1.2, this study can be defined as an *attributional LCA analysis with system expansion*. In fact, even if different scenarios are analysed and compared amongst them, they have not been elaborated with a consequential approach: present perspective is assumed for the identification of waste treatment options and the same methodology has been applied for the definition of the reference scenario. Although average data are assumed for the inventory phase, a *marginal* perspective is assumed for the electricity production technologies in the reference scenario (see Chapter 5). This is considered in the interpretation phase, along with the sensitivity analysis performed on key parameters.

4.3 System boundaries: Background and Foreground

Figure 4.1 shows the systems analysed in this study. The study is conducted considering two different subsystems: the WM scenarios and DG scenarios. The *WM scenarios* include the analysis of different waste management treatments, compared with the foreground process (AD). The *DG scenarios* include the comparison amongst three different micro CHP systems in the DG paradigm, and the comparison with the use of an alternative fuel (NG). The two sub-systems are described in section 4.9 and 4.10 respectively.

As stated in section 2.6, the waste considered in this study is supposed to be separated at the source: it means that is separated by the consumers directly after use. It is then collected from the kerbside and transported to the Transfer Station. The *Waste Transfer Stations* are facilities where municipal solid waste is unloaded from collection vehicles and briefly held while it is reloaded onto larger long-distance transport vehicles for shipment to landfills or other treatment or disposal facilities. This is a way for the community to save money, by combining the loads of several individual waste collection trucks into a single shipment (US EPA, 2012).

In this study only transportation from the transfer station to the waste management facility is considered. This is a comparative LCA study and therefore the processes that are identical in each alternative (such as transportation from the houses where the waste is generated to the Transfer Station) have been omitted as they will not affect the overall results (Finnveden et al. 2011).

All the scenarios analysed provide energy as output. This is evaluated as avoided burden with the expansion of the boundaries of the system, using marginal and average technologies.

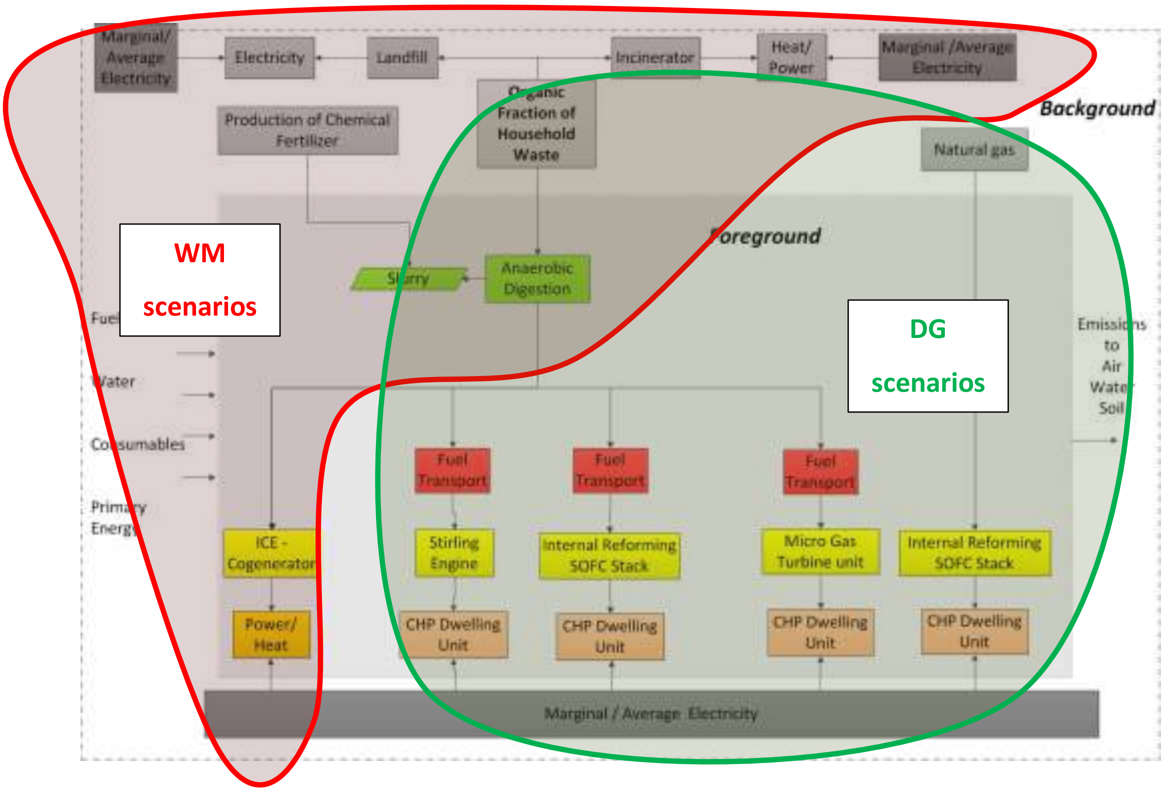


Figure 4.1. System boundaries and scenarios considered in this study.

4.4 Geographical scope

Two different countries are chosen as case studies, to verify the feasibility of the waste to energy closed loop approach: UK and Italy. The following sections present briefly a description of the areas and the main characteristics.

The two areas have been chosen for the sake of simplicity: one is the home town of the author and the other a borough of the city where she is living in at the present. In addition, it is interesting to compare two different Countries in terms of services organization and normative structure.

4.4.1 UK case study

As UK case study, the Royal Borough of Greenwich in London area is considered. This is one of the 32 boroughs of London and it is located in the South East area of the city – see Figure 4.2.

Figure 4.2. Greater London, Inner area. In black the Royal Borough of Greenwich (UK Government, 2012)**.**

In 2009, London produced 3,822,000 tons of MSW, defined by the Greater London as '*waste in the control of a waste collection or waste disposal authority'* (Greater London Authority, 2011). 79% of this waste was from household, while the 21% from small and medium sized business in the London area.

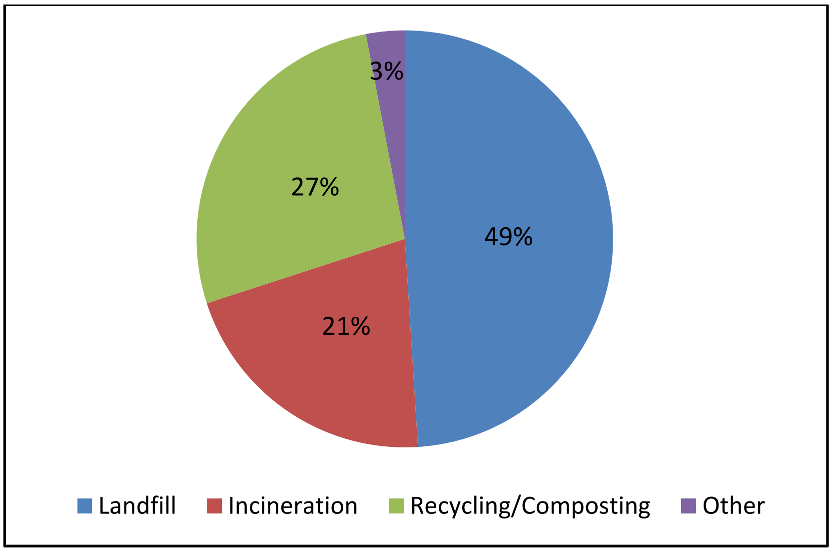


Figure 4.3. London's Municipal Waste treatments in 2010[1](DEFRA, 2010b)**.**

Note: (1) 'Other' is waste material sent for some form of pre-treatment or unknown destination. Recycling or composting includes organic waste sent for anaerobic digestion. Less than one per cent of London's municipal waste is treated using anaerobic digestion.

In 2011, the Greater London Authority published a report entitled *'London's Wasted Resource - The Mayor's Municipal Waste Management Strategy'*. Here, climate change is recognized as one of the key drivers for this strategy. It estimates that the municipal waste that London sends to landfill generates approximately 460,000 tonnes of CO_2 equivalent each year. The strategy aims at bringing emissions saving with a combination of avoiding the emissions from diverting waste to landfill and sending it to AD plants, plus rising recycling rate of virgin materials and substituting energy from coal or gas with energy produced directly with waste. The saving that can be achieved is set up to 1.5 millions of tons of CO_2 eq per year, *'equivalent to avoiding the emissions associated with powering London's Underground Network each year, plus avoiding emissions from all of London's registered taxis'* (Major of London, 2010). Each year around 460,000 tonnes of municipal food waste is sent to landfill in the London area. This waste could instead be used to generate renewable, low carbon energy using anaerobic digestion, potentially providing enough electricity for about 24,000 homes and heat for 6,750 homes.

Currently, there are 12 Local Authority responsible for both collection and disposal of the waste. The other 30 borough are responsible for the collection of their waste but the disposal is arranged across four waste disposal's authority. There is a high variability in the recycling and compost rate across the boroughs. Greenwich Local Authority is responsible for both collection and disposal of the waste and in 2010 it showed a recycling a composting rate for household waste slightly higher than the city's average: 35% compared with 32% (DEFRA, 2010b).

The population of Greenwich is approximately 245,000 (Greenwich Council, 2012). MSW from this borough is presently transported by truck to a Waste Transfer Station. Here it is assumed to follow the average London's waste flows, Figure 4.4.

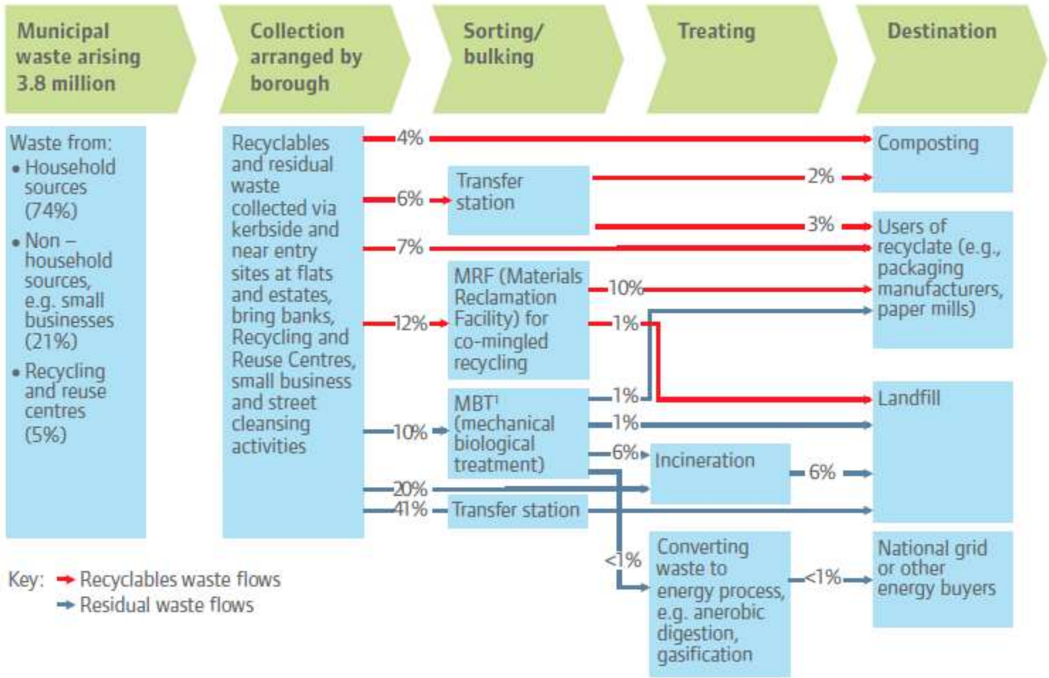


Figure 4.4 London's Municipal waste flows in 2010 (Greater London Authority, 2011).

It is reported that in 2010, 557 kg of household waste were collected per head of population (Environment Agency, 2011). The fraction of food waste varies slightly but the organic fraction is on average approximately 33% of the total amount of MSW produced (DEFRA, 2009). Therefore, the total annual production of food waste in the bough of Greenwich is estimated in 45,033 tons/year. Considering an average of 2.4 persons per family for a typical household in UK, the average food waste production per single household in the borough of Greenwich is 441 kg per year, while the number of households is 102,083 (National Statistics, 2012).

The locations of the waste treatment plants considered in the analysis are:

- South East London Combined Heat and Power (SELCHP) incineration plant located in South Bermondsey (Al-Salem, Mechleri, Papageorgeu, *et al.*, 2012), which is about 15 km from the farthest part of the Borough;
- South Ockendon landfill site (Greater London Authority, 2011), 30 km far from the farthest point of the borough;
- The AD plant is not existed at the moment. The distance considered in this study is 50 km between the TS and the plant;

- Average distance between the AD plant and the farmland for the spreading and use of the digestate is assumed 30 km.

Energy demand for a typical UK dwelling

Energy demand of a single-family house consists usually of the demands of electricity, hot water and space heating. In this study, only electricity and space heating are considered. The data are taken from Hawkes et al. 2007: the dwelling considered is an average UK house, with 85 m^2 floor area, with the heating system providing a resultant temperature of 25°C during heating hours. The average number of people was 2.4. Hawkes et al. defined a specific schedule for the heating time along months and days in a year, as well as for the occupancy schedules for every room of the house, considering specific appliances' use. The heating system is supposed to work from beginning of September to end of April. For a full description please refer to Hawkes et al. 2007.

The resulting annual space heating demand for a single dwelling is then 13,610 kWhth, while the annual electricity demand is 4,350kWhe.

4.4.2 IT case study

As Italian case study, the city of Livorno has been considered. This is one of the 10 Provinces of Toscana Region, in central Italy.

Figure 4.5. Toscana region map (Municipality of Pisa, 2012).

In 2010, in Toscana 2,457,412 tons of MSW have been produced. The pro-capite production is 670 kg of waste per person per year (ISPRA, 2012). The waste management system in Toscana is organized in three different areas. Livorno is included in the Costa Environmental Authority (ATO COSTA), see Section 2.1.5), together with other four provinces. This has been re-organized recently and there is the urgency to build new plants to satisfy the national and European targets in terms of landfill directive and recycling rate (ATO Costa, 2013). New biological treatments, such as AD and compost plants, are planned for the future, in order to meet the target of 65% of recycling rate by 2020.

In the ATO Costa, the municipal solid waste produced was 890,000 tons in 2011. The recycling rate was 38%, of which 23% is FORSU. Livorno shows the lowest level, with 14% of FORSU up the total recycled waste. Presently, only the 25% of the total FORSU in the ATO Costa is collected at house level; the majority of that is taken directly from the street collection. Figure 4.6 shows the waste flows in ATO Costa in 2011.

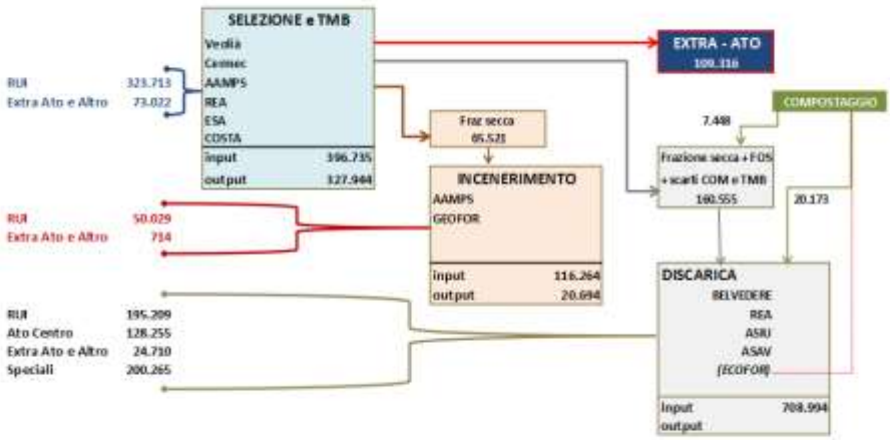


Figure 4.6. ATO Costa municipal waste flows, 2011 (ATO Costa, 2013)**.**

Livorno has a population of 156,779 inhabitants (ISTAT, 2013), with a number of households equal to 71,608. The organic waste, representing the 35% of the total MSW generated (ISPRA, 2012) is equal to 513 kg/year per household.

The distance considered between the WTS and the single plants are:

- Vallin dell'Aquila landfill plant, 16 km far from the farthest point of the city;
- Picchianti incineration plant, 18 km far from the farthest point of the city;

- Scapigliato AD plant, 25 km far from the farthest point of the city. The plant has been partly built, but for technical problems the construction is suspended presently. In this thesis, the same location is assumed;
- The farmland for the spreading and use of the digestate is assumed to be 10 km far from the AD plant.

Energy demand for a typical IT dwelling

For the IT case study, a typical house is considered to estimate the space heating and electricity demand. The data here are taken from Liso et al. 2011. They simulated a single-family house with $80m^2$ heated area. The temperature set-point is 21°C (then 4°C less than the UK case study), while the average number of occupants is 2.7. They considered a specific pattern for lighting and appliances, in accordance to IEA Annex 42, identical for the 4 European cities taken as case study (Beausoleil-Morrison, 2008). They simulated the space heating demand as well, resulting in an average number.

The space heating and electricity demand for a typical house in Livorno (IT) result then in 7,000 kWhth and 3,000 kWhe respectively.

4.5 Time perspective

Several time perspectives can be assumed in a LCA study. From a Global Warming potential point of view, usually the impacts can be assessed over 20, 100 or 500 years. Based on this, different characterization factors are used, to simulate how the various gasses concentration decays over time in the atmosphere. Here a GWP on 100 years is assumed.

Time horizon in landfill

An important concern when dealing with LCA of landfill system is the consideration about the degradation of the material during time. The majority of the materials is stored in the landfills and emissions from this can continue thousands of years (Fruergaard 2010; Eriksson et al. 2002).

Finnveden et al. (2000) defined the '*survayable time period'* as '*the time until the most active process in the landfill has ended and the landfill has reached a pseudo steady state*'. This time is usually considered equal to 100 years. The same approach is used by other authors. A dilemma is how to compare the emissions from the landfill with the instant emissions from the other processes considered in the study. Over the years in general three different phases for landfills can be distinguished:

- the *short- time phase,* characterised by the emplacement of waste and active and passive maintenance. The time horizon amounts to decades;
- the *medium-term phase*, characterized by no more active maintenance. External factors, environmental impacts on the landfill body are more or less constant. The time horizon comes to centuries. These two phases can be concluded as '*survayable*';
- the *long-term phase*, where the time horizon reaches 104–105 years, the external factors change and developments are not foreseeable in detail (Obersteiner, Binner, Mostbauer, *et al.*, 2007).

Landfill models are included in the packages presented in Table 3.2 about LCA of anaerobic digestion of waste. The time horizon is handled in different ways. In ORWARE (see Chapter 3) two time horizons are considered: emissions of the first 100 years, based mainly on monitoring and the remaining emissions that will potentially be emitted in the future (Eriksson et al. 2002; Dalemo et al. 1997; Bjorklund et al. 1999). In other types of models, a 100-year time horizon has been chosen (EPIC/CSR, LCA–IWM and MSW-DST), although MSW-DST also allows a 20 and 500 year time horizon for landfill leachate. In some models (WISARD and WRATE), it is assumed that the long-term impacts should consider ''infinity'' to encompass more than 90% of the emissions. In WRATE, 20,000 years have been considered as ''infinite'' for the modelling of leachate emissions, which is suggested to correspond to about 95% of the potential emissions(Hall, Plimmer & Thomas, 2006). EASEWASTE allows the user to define the time horizon which provides the greatest flexibility. IWM2 provides a different approach where the time horizon is not defined, instead the typical amount of landfill gas and leachate generated produced per tonne of waste landfilled is defined. Finally, the Society of Environmental Toxicology and Chemistry (SETAC) recommends that the emissions should be integrated over an infinite time period; if this is not possible, a time interval of 100 years should be applied.

In this study, a 100 years period is considered to evaluate the impact of the landfill. Given that this is compared with other waste treatment options, it is assumed that the emissions

– that would occur in time in the reality – are occurring at the same time, as for incineration or anaerobic digestion plant.

4.6 Technological scope

The technological scope is highly related to the time horizon and the geographical scope, as the technology data should reflect the time period as well as the location of the assessment. Waste technologies have been subject to a significant improvement in the last years, especially for emissions control and energy efficiency. This is even more important when the focus is on the energy recovery systems investigated in this study. Data used are strongly characterized by uncertainty on energy efficiency and lifetime. Giannopoulos and Founti 2011 have showed how the results of the LCA can changed when the electrical efficiency of the fuel cell rises from 25% to 35%.

4.6.1 Micro-CHP operating strategies

In order to assess the waste to energy closed loop approach three different operating strategies are investigated for the micro-CHP technologies.

- Full Thermal demand (FT): the micro-CHP system is designed to fully satisfy the space heating demand of the dwellings over one year period. Given that the systems are able to produce electricity and heat at the same time, based on the H to P ratio of the specific system an amount of electricity is produced in this mode. This energy is supplied to the dwellings to satisfy the electricity demand. Two cases are here possible: if surplus electricity is produced, it is sold to the grid; if the electricity produced by the micro-HCP is not enough to satisfy the electricity demand of the dwelling, electricity is supplied directly from the grid (considering the average mix of technologies).
- Half Thermal demand (HT): the micro-CHP system is designed to satisfy at 50% the space heating demand of the dwellings over one year period. The rest is supplied by the conventional technology, thus the natural gas condensing boiler. Even in this case, based on the H to P of each system, electricity is produced and the two possible cases explained above can occur.
- Electricity demand (EL): the micro-CHP system is designed to fully satisfy the electricity demand of the dwellings. The heat produced is supplied to the dwelling either, up to satisfy its space heating demand. When surplus heat is produced, a

district heating grid is assumed to receive the excess thermal energy. If the heat is not enough to satisfy the space heating demand, thermal energy from condensing boiler is supplied.

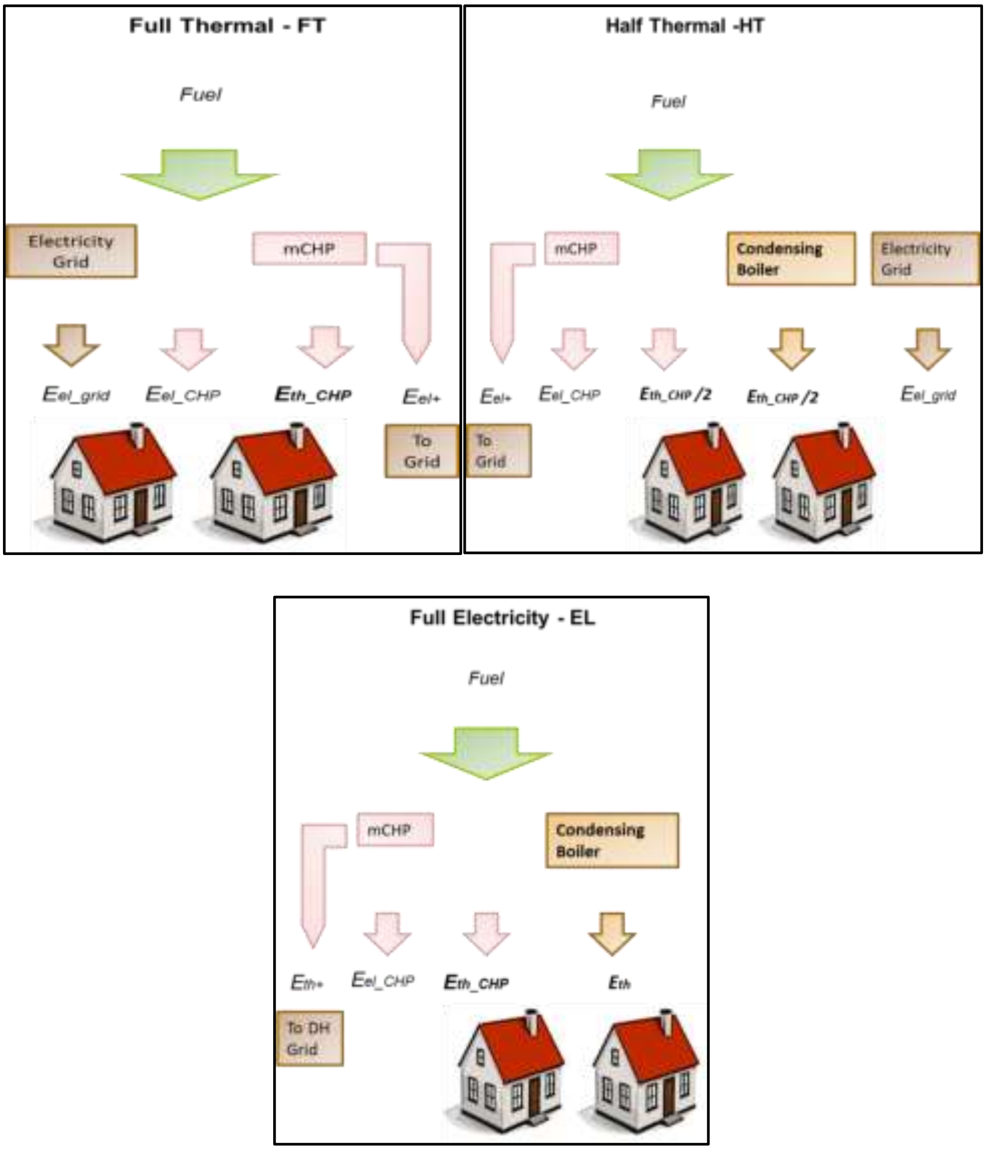


Figure 4.7. Concept schematics of the three operating strategies considered in the study.

The timeframe for the micro-CHP system analysis is set accordingly to the estimated operation lifetime of the reference scenario. It means that 20 years is considered. Due to that, more than one stack is required throughout the SOFC unit lifecycle. Emissions from landfill are assumed to occur all at the same time, for 20 years.

4.7 Life cycle impact assessment

Two organizations have been mainly working on the definition of standardised impacts for the Impact Assessment phase: the Society of Environmental Toxicology and Chemistry and the International Standard Organization. The ISO describes procedures rather than specific methodologies or models for LCIA, while the SETAC – Europe Working group on Impact Assessment seeks to define a recommended list of impact categories complete with category indicators and characterization factors (Guinée et al. 2001).

In this study the problem-oriented approach (or *midpoint*) is applied. Guineé et al. defined a list of categories, based on the SETAC working group studies. It distinguishes three different groups:

- Baseline impact categories: for which characterization methods are scientifically defined and used in almost all LCA studies.
- Study-specific impact categories: with categories that may be included, depending on the specific Goal and Scope definition of the LCA study and whether data are available.
- Other impact categories: categories which require further elaboration before they can be used in LCA studies, with research still in progress.

The categories included in the Baseline group are listed in Table 4.2.

The study takes into consideration the following impact categories: Abiotic Resources Depletion (ARD), Global Warming Potential (GWP) as an indicator of greenhouse effect, Acidification Potential (AP) as an indicator of acid rain phenomenon, Photochemical Ozone Creation Potential (POCP) as an indicator of photo-smog creation and as defined by CML (Guinèe, Gorrée, Heijungs, *et al.*, 2001), and Nutrient Enrichment (NE) as an indicator of surface water eutrophication and as defined by EDIP97 method (Wenzel, Hauschild & Alting, 1997). These categories have been chosen given their environmental relevance and the fact that they are internationally accepted in accordance with ISO 14040 recommendations (ISO, 2006). The impact categories listed above are based upon a distinct identifiable environmental mechanism and they ensure that the results are robust enough to form a basis for further consideration or decisions (European Commission, Joint Research

Centre & Institute for Environment and Sustainability, 2011; Stranddorf, Hoffmann, Schmidt, *et al.*, 2005).

Table 4.2 Baseline Impact Categories, adapted from Guineé et al. 2001.

Baseline Impact Categories
Depletion of abiotic resources
Impact of land use
Land Competition
Climate Change
Stratospheric ozone depletion
Human Toxicity
Ecotoxicity
Freshwater acquatic ecotoxicity
Marine acquatic ecotoxicity
Terrestrial ecotoxicity
Photo-oxidant formation
Acidification
Eutrophication

Abiotic resource depletion is one of the most frequently discussed impact categories and there are consequently a wide variety of methods available for characterizing contributions to this category (Finnveden et al. 2009). The method adopted here is taken from Guineè et al. (2001) and it is related to some measure of available resources or reserves and extraction rates (Guinèe et al., 2001). Abiotic resources are those considered as non-living resources such as iron ore, crude oil and wind energy.

The *Global Warming Potential* is the main indicator for the Climate Change category. This is defined as the impact of human emissions on the radioactive forcing of the atmosphere (Guinée et al. 2001). This may have adverse impacts on ecosystem health, human health and material welfare. Most of these emissions enhance radioactive forcing, causing the temperature at the earth's surface to rise: this is commonly referred to '*greenhouse effect'*. The GWP indicator used in this study does not take into account the CO_2 emissions associated with the biogenic carbon (see Section 3.3).

Acidification Potential is measured in kg of SO_2 equivalent and it is the main indicators of the Acidification impact category. Acidifying pollutants have a wide variety of impacts on soil, groundwater, surface water, biological organisms, ecosystems and materials. Examples include fish mortality, forest decline and crumbling of building materials. The major acidifying pollutants are SO2, NOx and NHx.

Photo-oxidant facilitates the formation of reactive chemical compounds such as ozone by the action of sunlight on certain primary air pollutants. These reactive compounds may be injurious to human health and ecosystems and may also damage crops. Photo-oxidant may be formed in the troposphere under the influence of ultraviolet light, through photochemical oxidation of Volatile Organic Compounds (VOCs) and carbon monoxide (CO) in the presence of nitrogen oxides (NOx). The impact associated with this category is the *PhotoChemical Ozone Creation Potenital*, measerd in kg of ethylene equivalent.

Finally Eutrophication covers all potential impacts of excessively high environmental levels of macronutrients, the most important of which are nitrogen (N), and phosphorus (P). Nutrient enrichment may cause an undesirable shift in species composition and elevated biomass production in both aquatic and terrestrial ecosystems. High nutrient concentrations may also render the surface waters unacceptable as a source of drinking water. The indicator representatives of this category in the *Nutrient Enrichment Potential*, measured in kg of PO_4 equivalent.

4.8 GaBi 5

At the moment, about more than thirty software packages exist to perform LCA analysis. The mainly differ in scope and capacity: some are specific for certain application, others have been directly developed by industrial organization (European Commission, Centre & Institute for Environment and Sustainability, 2012). The main advantages in using a software package are: it provides ready available LCI information (that can be more or less wide depending on the databases included); large number of calculation can be done rapidly; very useful in the interpretation phase with tools that help in sensitivity analysis, Monte Carlo simulation, contribution analysis, etc. However some disadvantages are present. The main drawback is that the calculations are not always transparent and the practitioner may have less control on the system boundaries (what is excluded/included in the life cycle) (Basson, 2012). The most commonly used software packages are SimaPro (www.simapro.com.uk/), GaBi (www.gabi-software.com/), Umberto (www.umberto.de.en/), Team (www.ecobilan.com/uk_team/). They differ in databases and modelling approach.

In this study GaBi 5 (and the new version Gabi 6) has been used (PE International, 2012). GaBi is software based on a mass and energy balance approach that calculates life cycle inventories and associated life cycle impacts. It uses a process flow-sheet type representation of the system and it has been developed with an object orientated programming approach: Figure 4.8 shows one of the scenario models with GaBi 5.

GaBi 5 (and its new version GaBi 6) contains databases directly developed by PE International and it may contain also as industry organizations' database (Plastic Europe, Alluminium, etc) and National databases (Ecoinvent, Japan database, US database, etc). This work has been carried out using only database directly developed by PE International (Baitz, Colodel, Kupfer, *et al.*, 2012).

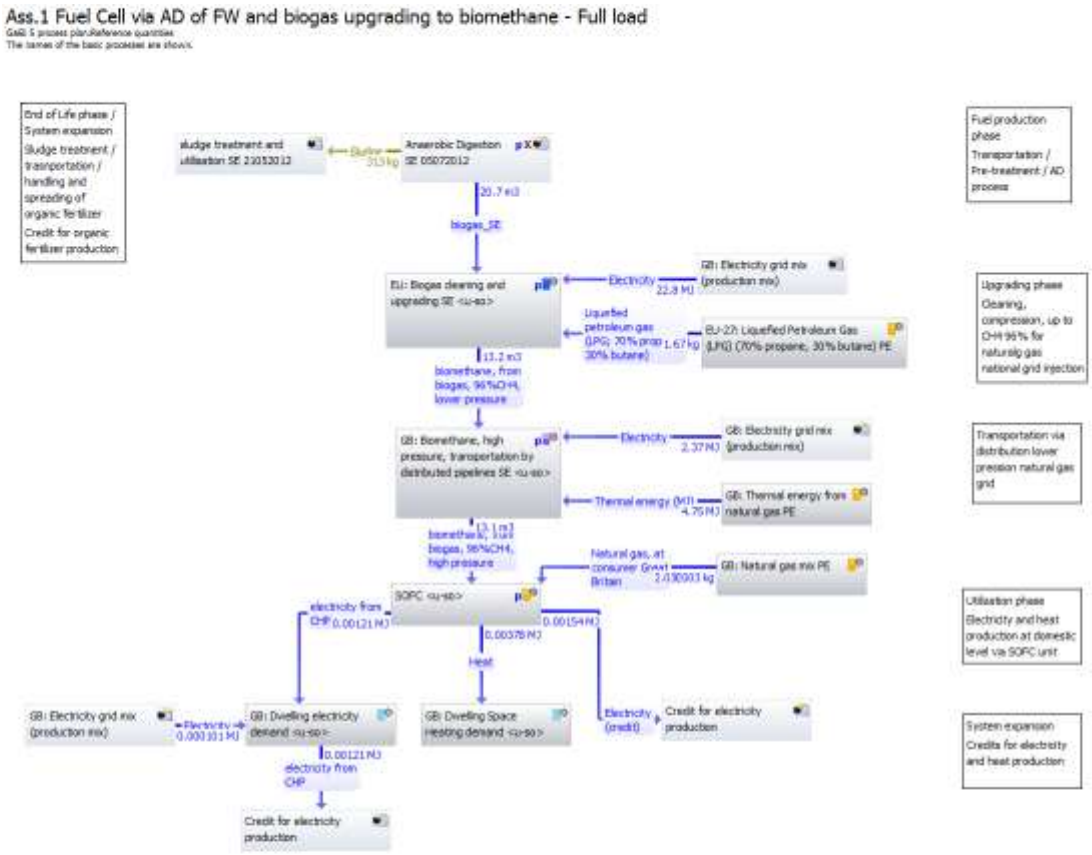


Figure 4.8. Flow sheet of DG_BIOMET_FT_FC_AVERAGE_UK scenario represented in GaBi 5.

4.9 Waste management options

4.9.1 Goal and scope definition

The primary aim of this part of the work is to compare three different alternatives for the treatment of OFMSW introduced in Section 4.2: landfill with gas recovery for electricity generation, incineration with energy recovery by CHP and anaerobic digestion with CHP and organic fertilizer production, for both the case study presented in Section 4.3. Those correspond to the scenarios 31-34 (UK based) and 65-68 (Italy based) in Table 4.1. The objective of this part is to evaluate and compare the environmental impacts of these alternatives. Initially, the system is modelled using baseline parameters from various literature sources. The assessment includes a 'hot-spot' analysis to identify the key parameters which contribute most to the environmental impacts in the foreground system. Some parameters are then modified in order to perform sensitivity analysis and identify their influence on the overall results, as shown in Section 5.4. As explained in Section 4.3, to allow for recovery of materials and energy from the waste, the system is expanded to include affected background processes outside the immediate waste treatment system: inorganic fertilizer production displaced through treatment and use of the digestate and energy production displaced through energy recovery from the waste.

A secondary objective of this part of the study is to compare different landfill models in order to find a common methodological approach in terms of system boundaries and burdens evaluation that allows a fair comparison with the foreground system, i.e. anaerobic digestion. In this way the aim of this part is to contribute to the development of a robust methodology for comparative life cycle analysis in the waste management sector.

4.9.2 Functional unit and system boundaries

In a multifunctional process analysis, different functional units can be used depending on the scope of the LCA. The Functional Unit (FU) used for this part of the study is the total amount of OFMSW produced in the borough of Greenwich (for the UK case study) and in the city of Livorno (for the Italian case study). Table 4.3 summarises the data used to determine the Functional unit for the two case studies.

Table 4.3. Main data to assess the Functional Unit.

	Borough of Greenwich (UK)	Livorno (IT)
Number of households in the area	102,083	71,608
OFMSW per household	441 kg/year	513 kg/year
*Total OFMSW produced by the households – **Functional Unit***	45,033 tons/year	36,765 tons/year

Considering the FUs, they correspond to medium size plants across Europe (Baere & Mattheeuws, 2012). The composition of the OFMSW is assumed to be constant during the year.

Figure 4.9 illustrates the processes considered; the scenarios simulated are summarized in Table 4.4. The Foreground system consists of the putative AD process including waste pre-treatment, CHP production, and the treatment and spreading of the digestate on agricultural land as an inorganic fertilizer substitute. The Background system includes the displaced production of electricity, heat and organic fertilizer. The two waste management alternatives - landfill with gas recovery and electricity generation and incineration with recovery of electricity and heat – are also treated as possible background processes which may be displaced by the AD treatment. In the first part of this work, a generic landfill plant is modelled without accounting for the avoided burdens due to carbon returned to the soil and sequestered. The avoided burdens are strongly dependent on the compensatory systems chosen for heat, electricity and commercial fertiliser production.

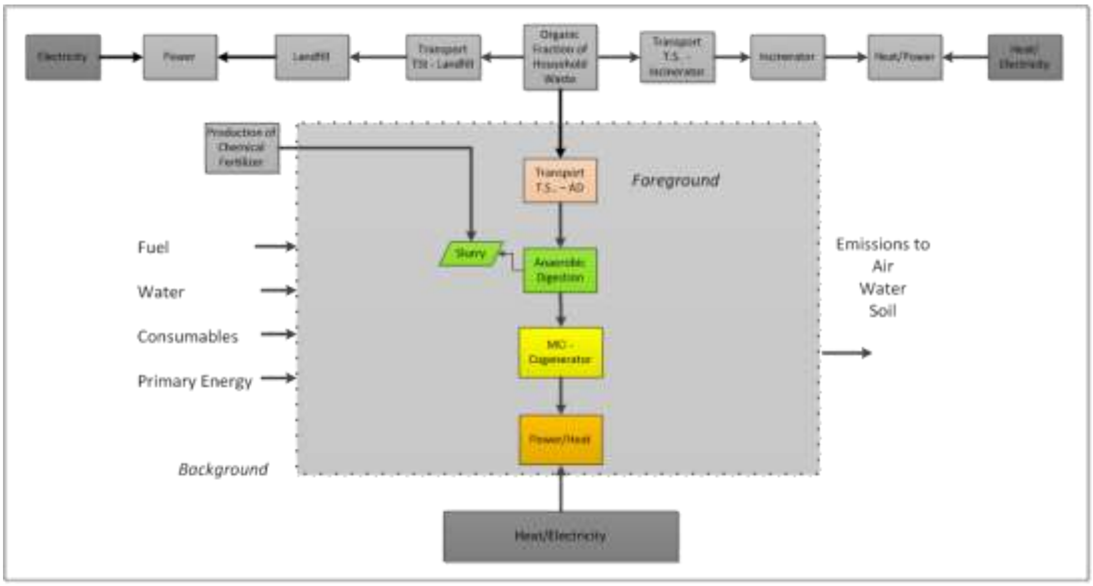


Figure 4.9. System boundaries analysed in WM scenarios study.

Table 4.4. Scenarios analysed in the WM scenarios.

Waste treatment Process	Energy recovery	Avoided burdens
Incineration	CHP	Electricity /Heat
Landfill	Electricity	Electricity
Anaerobic Digestion	CHP	Electricity/Heat/Inorganic Fertiliser

4.10 Distributed generation

4.10.1 Goal and scope definition

The goal of this second part of the study is to evaluate the environmental impact of the DG scenarios comprising micro-CHP systems fed by biogas produced by OFMSW, to supply energy for a group of dwellings, in the distributed generation paradigm. The two areas presented in Section 4.3 are considered as case studies. The system is design to create a “waste to energy closed loop”, where the total amount of food waste produced at residential level in the borough/city is used to satisfy the total energy demand of the dwellings in the same area. This is compared, through the system expansion, to conventional processes - *reference scenario*, where the energy is supplied through conventional technologies and the waste is treated in a landfill plant.

The three different operating strategies described in Section 4.5 are analysed here.

Table 4.5. Summary of the main characteristics of the three Micro-CHP technologies analysed.

	SOFC 1.7 kW Sunfire	**Stirling Engine 1.2 kW Whispertech**	**Micro Gas Turbines Capstone 30 kW**
Electrical efficiency (%)	35	12	26
Thermal efficiency (%)	55	80	52
Lifetime	40,000	50,000	50,000
Status	demonstration	demonstration	Few commercial models

4.10.2 Functional unit and system boundaries

Figure 4.10 shows the systems analysed in the DG scenarios part. For the foreground, DG scenarios with 3 different technologies for micro-CHP systems are considered run with the biogas produced with an anaerobic digestion plant using the OFMSW as feedstock. The three technologies, presented in Section 2.1, are: SOFC-micro CHP unit, micro Gas Turbine and Stirling engine. Two different pathways for biogas use are investigated: SOFC, mGT and SE units directly run with raw biogas and upgrading of the biogas to biomethane before the injection in the SOFC-micro CHP unit. Both pathways provide the transportation of the biogas from the production site to the single dwelling. An alternative and competing fuel use is evaluated, in order to feed the same SOFC micro-CHP system: natural gas.

The system under analysis is a multifunctional system, providing at the same time two functions: waste treatment and energy production. A mutli-functional unit is used in order to answer to the scope of this LCA analysis. It is composed by the amount of OFMSW produced in the area under study by the population living in there and the total energy demand for the same quantity of people (assuming that an average family is leaving in a single dwelling). In order to compare the DG scenarios with natural gas scenario, the burdens associated with the landfilling of the same amount of organic waste treated in AD plant are considered.

The total FU used for the DG scenarios is shown in Table 4.6.

Table 4.6. FU used in the DG scenarios study.

	Borough of Greenwich (UK)	**Livorno (IT)**
Number of households in the area	102,083	71,608
Total OFMSW produced by the households	45,033 tons/year	36,765 tons/year
Energy demand for the total households	Electricity: 444 GWhel/year Space Heating: 1,389 GWhth/year	Electricity: 215 Gwhel/year Space Heating: 501 GWhth/year

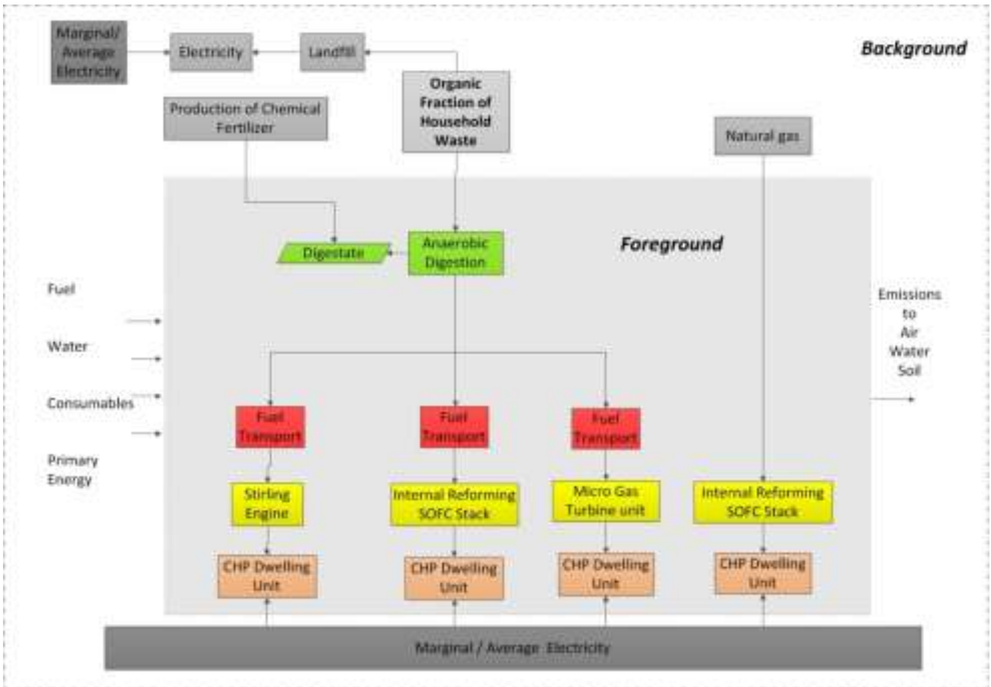


Figure 4.10. System boundaries analysed in the DG scenarios study.

System expansion is applied to evaluate the substitution of energy and fertiliser produced via conventional process.

4.11 Inventory analysis

The inventory for all the processes involved in the analysis is here presented. Where possible, site-specific data are used for foreground processes; otherwise average data from the literature and specific datasets are used. In the literature it is possible to find many studies that investigate the environmental impact of the anaerobic digestion process. The author of this thesis has reviewed part of them.

Given that '*LCA studies tend to produce quite diverging and even conflicting results*', all the assumptions will be described as clearly as possible to allow others to replicate the study in other contexts (Heijungs & Guinée, 2007).

4.11.1 System expansion: the reference scenario

The reference scenario is shown in Table 4.7.

Table 4.7. Reference scenario considered in this study.

	Compensatory systems	References
Heat	8 kW Condensing Boiler using natural gas, 50,000 h lifetime	(Staffell et al. 2011; Liso et al. 2011; A. Hawkes & M. Leach 2005; Strachan & Farrell 2006)
Electricity	Average Electricity Production: country based mix of technologies (UK, IT)	(Staffell et al. 2011; Liso et al. 2011; A. Hawkes & M. Leach 2005; Strachan & Farrell 2006)
Inorganic Fertiliser	Average industrial production	(Møller, Boldrin & Christensen, 2009; Svensson, Odlare & Pell, 2004; Wood & Cowie, 2004)

When applying system expansion in a multifunctional process, it is necessary to identify the type and quantity of product – digestate or energy in this particular case – which is replaced by the process itself (Fruergaard & Astrup, 2011). Attributional (or 'accounting') LCA studies, which describe a specific current or proposed process, for example for product labelling, generally use average technology data to evaluate the avoided burdens associated with system expansion (BSI, 2011). By contrast, consequential (or 'prospective') studies, which aim to explore the consequences of broader or policy changes, use marginal technology data (Fruergaard et al. 2009). This study is concerned with a specific, albeit putative, process; therefore it is a type of attributional analysis and average data for energy and commercial fertiliser production are used.

The avoided burdens associated with electricity exported to the national grid are evaluated for the average UK/IT generating mix, described by data from PE International (2011). The avoided burdens for heat are evaluated for a natural gas-fired condensing boiler with an efficiency of 82-89% (Staffell et al., 2012). The results are shown in Table 4.8 where the avoided burdens per MJ of energy produced (electricity and heat) are reported in terms of Abiotic Depletion, Global Warming Potential, Acidification Potential, Photo-Ozone Creation Potential and Nutrient Enrichment.

Table 4.8. Avoided burdens for electricity and heat production in the reference scenarios, per MJ of energy produced.

Impact Category	Electricity grid mix UK	Thermal energy from natural gas UK	Electricity grid mix IT	Thermal energy from natural gas IT	Thermal energy from condensing boiler with natural gas (EU)
AD (MJ)	2.55	1.05	2.12	1.13	1.14
GWP (kg CO_2 eq)	0.155	0.0625	0.1608	0.071	0.0693
AP (kg SO_2 eq)	0.00053	0.000031	0.000408	0.000062	0.000043
POCP (kg ethane eq)	0.000030	0.0000047	0.000033	0.000013	0.00001
NE (kg NO_3 eq)	0.00046	0.000059	0.00028	0.00007	0.000029

All the impacts associated with the production of 1 MJ of thermal energy with a natural gas Italian mix result higher than the UK one. This is mainly due to the different mix of natural gas in the two countries. Figure 4.11 and Figure 4.12 show the sources in the two cases. As it is possible to see, in the UK more than half of the natural gas comes from national resources, while only the 10% of the total natural gas supplied in Italy comes from internal supplier. This brings a higher environmental impact associated with 1 MJ of natural gas in Italy compared with the same amount in UK.

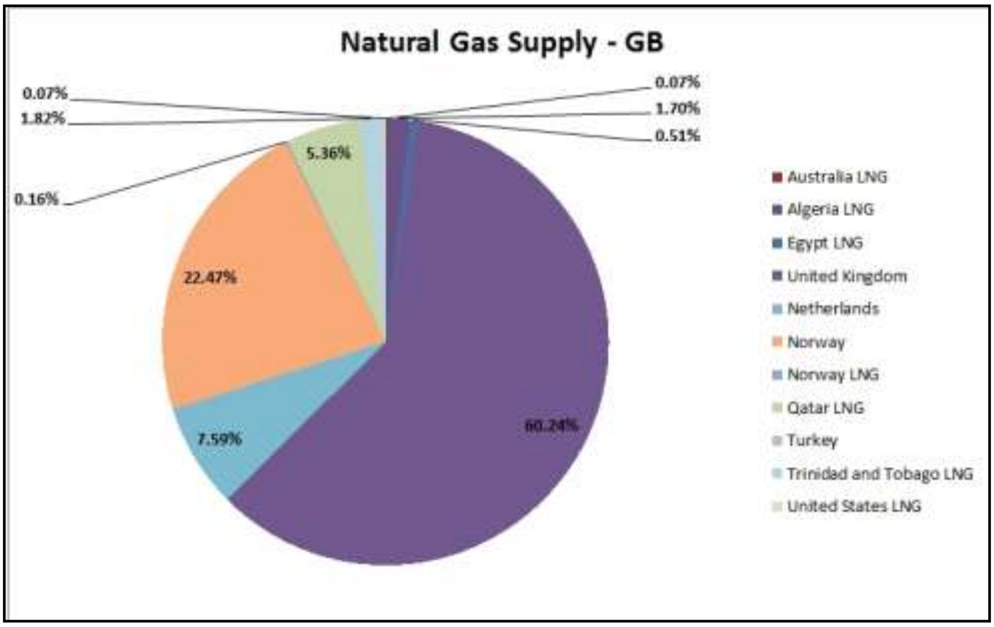


Figure 4.11. Natural gas supply mix, UK (GaBi 6, 2013)**.**

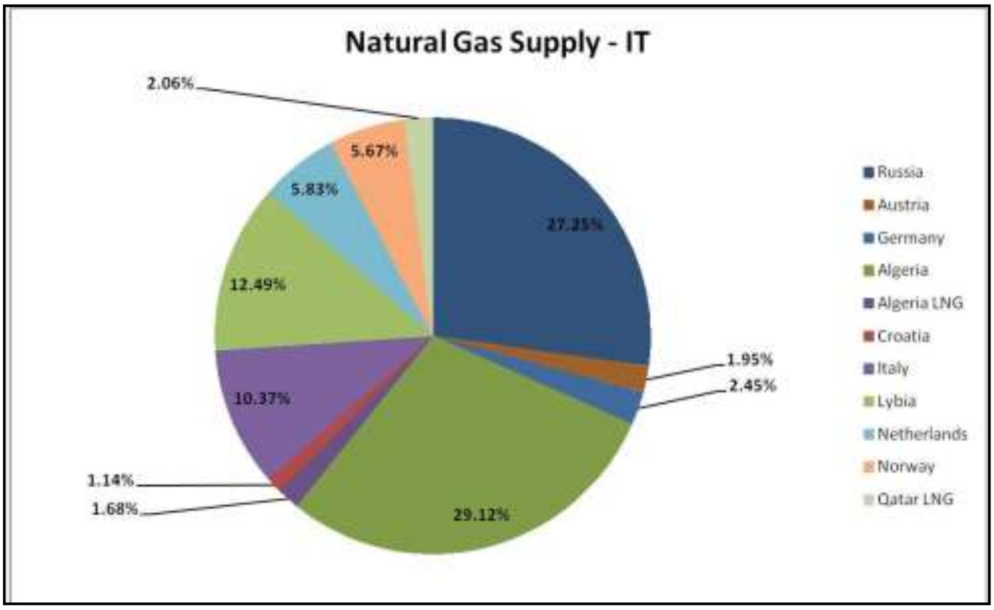


Figure 4.12. Natural gas supply mix, IT (GaBi 6, 2013)**.**

Avoided emissions from the substitution of the commercial fertilisers can be estimated on the basis of the nutrient content in the digestate in connection with inventories of fertiliser production. The substitution of the fertiliser is a much-discussed topic, as discussed in Section 2.5.3 (Møller et al., 2009; Bernstad et al., 2011). Here we use the average value suggested by DEFRA (Hill et al., 2011) for the production of N, P and K fertilisers as shown in

Table 4.9. Only the total CO_2 eq is considered due to availability of data. Consequently, the only impact affected is the Global Warming Potential.

Table 4.9. Avoided emissions from the substitution of commercial fertilisers.

Commercial Fertiliser	**kg CO_2 eq/kg fertiliser** (Hill et al. 2001 from William et al. 2006)	**kg CO_2 eq/kg fertiliser** (Møller, Boldrin & Christensen, 2009).
Nitrogen (N)	6.8	8.9
Phosphorous (P)	1.2	1.8
Potassium (K)	0.5	0.96

4.11.2 Organic Fraction of Municipal solid waste

The waste flow considered is the total amount of source-separated OFMSW produced in the reference areas. Food wastes can be highly variable depending on their sources. Some characteristics of food wastes that have been reported in the literature indicate moisture content of 74–90%, volatile solids to total solids ratio (VS/TS) of 80–97%, and carbon to nitrogen ratio (C/N) of 14.7–36.4 (Zhang, El-Mashad, Hartman, *et al.*, 2007). The physical and chemical characteristics of the organic waste are important information for designing and operating anaerobic digesters, because they affect biogas production and process stability during anaerobic digestion. They include, but not limited to, moisture content, Volatile Solids content (VS), nutrient contents, particle size, and biodegradability. The biodegradability of a feedstock is indicated by biogas or methane yield and percentage of solids (Total Solids or Volatile Solids) that are destroyed in the anaerobic digestion.

The OFMSW is assumed to come from a separated collection, so that no mechanical separation is considered before the pre-treatment section. This is in line with the recommendations of the EU: in the '*Communication on bio-waste management in the EU'* (European Commission, 2010), separate collection use for managing bio-waste is strongly encouraged. Experimental data (Cavinato, 2011) show that a higher Specific Gas Potential (SGP) can be reached with this kind of OFMSW in terms of cubic meters of biogas generated per kg of total volatile solids (VS) sent to the AD plant: a value of about 0.7 m3/kg VS can be realized for OFMSW, compared with 0.3 m3/kg VS for mechanically separated organic waste.

The physico–chemical characteristics assumed for the waste in the two case study areas are summarised in Table 4.10, based on results of the European project Valorgas (www.valorgas.soton.ac.uk), led by the University of Southampton (MTT, 2010). They investigate the potentiality of food waste for energy generation, characterizing the feedstock in different European countries including Italy and UK. The values in Table 4.10 lie within the range of values used in similar studies (Berglund & Borjesson, 2006; Boldrin, Neidel, Damgaard, *et al.*, 2011; Bernstad & la Cour Jansen, 2011; Patterson, Esteves, Dinsdale, *et al.*, 2011; Møller, Boldrin & Christensen, 2009; Pöschl, Ward & Owende, 2010; Fruergaard & Astrup, 2011). An average biogas production of 600 Nm3/ton of Total Volatile Solid (VS) is assumed for both the sites, with a methane content of 63% v/v (Møller et al., 2009).

Table 4.10. Assumed composition of Organic Fraction of Municipal Solid Waste.

Parameter	Unit	OFMSW UK	OFMSW IT
TS[1]	%	26	27
VS[2]	% TS	91	87
C	% TS	51,3	47
K	mg/kg $_{TS}$	12,900	10,000
N	mg/kg $_{TS}$	31,300	25,000
P	mg/kg $_{TS}$	4,870	3,470

Note. [1]Total solid concentration, expressed as a fraction of the wet mass of the prepared feedstock. [2]Volatil Solid concentration, expressed as a fraction of the total solid mass.

4.11.3 Transport

It was noted that source-separated waste is considered; therefore only transportation from the transfer station to the waste management facility is considered. This is a comparative LCA study and therefore the processes that are identical in each alternative (such as transportation from the houses where the waste is generated to the Transfer Station) have been omitted as they will not affect the overall results (Finnveden et al. 2009).

Diesel consumption for transportation of waste from the transfer station to the specific plant is depending on several factors, such as the truck type, the speed of the truck and the collection area. Fuel consumption is modelled in GaBi, depending on the weight of the collected waste and the average distance travelled considering a full and empty load. The assumptions used for the average distance travelled have been already presented in Section 4.4. Table 4.11 summarises the vehicles and fuel consumption assumed in this study, and based on GaBi database. Data is valid for both the case study areas.

Table 4.11. Data used for transportation distances, type of trucks, fuel type and consumption (PE International, 2012)**.**

Scenario	Transport	Vehicle	Fuel type and consumption
Landfill	Transfer station to Landfill plant	Transportation vehicle, 22 ton (EURO 3), 80% loading	Diesel 0.02 l/(km*ton)
Incineration	Transfer station to incineration plant	Transportation vehicle, 12 ton (EURO 3), 80%	Diesel 0.03 l/(km*ton)
Anaerobic Digestion	Transfer station to AD plant	Transportation vehicle, 25 ton (EURO 3), 80%	Diesel, 0.02 l/(km*ton)
	AD plant to farmland	Transportation vehicle, 12 ton (EURO 3), 50%	Diesel, 0.03 l/(km*ton)

Transport Data from GaBi Database is referred to specific locations, such as Germany, Austria and Switzerland. However, the models are representative for the entire Europe due to the similarity of the vehicle structure and the same emissions limit values. The road categories and the utilisation behaviour affect the precision of the results, although an adaptation can be carried out by setting the driving share in highway, urban and rural roads, as well as the utilisation ratio and sulphur content in the fuel, for individual conditions. The reference year of the dataset is 2011 (Baitz, Colodel, Kupfer, *et al.*, 2012)

4.11.4 Landfill with electricity recovery

The landfill model used for this first part of the study represents a typical municipal waste landfill with surface and basic sealing that satisfies European limits for emissions. The model includes treatment of landfill gas and leachate and deposition of sludge. The total electricity produced by the landfill gas is 345 MJ per ton of OFMSW (Baitz et al.2012), and the 50% of the generated landfill gas is used for energy production. Credits for electricity production are discussed in Section 4.11.1. As for the incineration scenario, the analysis considers only the emissions due to the organic fraction, despite the waste being usually processed unsorted.

4.11.5 Incineration with energy recovery

Waste incineration is modelled according to average data for European waste-to-energy plants taken from the database of GaBi 5.0 software. 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 database represent averages over a number of European incinerators. The system includes generation of steam to produce electricity and heat. A total of 495 MJ of electricity and 1280 MJ of thermal energy per ton of OFMSW are produced. The approach to assessing the avoided burdens for electricity and heat generation is explained in Section 4.12.1.

The data for the incineration and landfill models are based on the GaBi database. Both the processes are considered '*aggregated*' in the software: it means that the emissions are listed all together, with no distinction between indirect and direct burdens.

4.11.6 Biogas plant

The energy requirements for the pre-treatment and operation stages in the AD route can vary significantly depending on the operating conditions of the plant, such as temperature, digester type and retention time. In this study, a continuous, single-stage, mixed tank mesophilic reactor operating at a temperature of 35°C is assumed. AD processes can be classified according to the dry matter content of the substrate in the reactor (Khalid, Arshad, Anjum, *et al.*, 2011). A wet regime process with the substrate containing 12% dry matter is considered in this study, with the composition controlled by adding fresh water. This technology is chosen given its broad application, although the dry regime is likely to be preferred for future AD plants (Baere & Mattheeuws, 2012).

In the model, data on the energy required for pre-treatment and operation are taken from Berglund et al., (2006) and shown in Table 6. The electricity demand is affected by variations in maceration, pumping and mixing requirements. In our study, no dewatering of the digestate is considered. Before entering the digester the waste has to be heated to 70^{o}C for 1 hour for hygienization. A continuous heat demand is required to maintain the digester temperature constant at 35^{o}C (Borello, Tortora & Evangelisti, 2012).

Table 4.12. Energy required for the pre-treatment and operation of AD plant (Berglund & Borjesson, 2006)**.**

Parameter	Unit	Value
Electricity required[1]	% of biogas produced MJ	11
Extra Electricity required for dilution	MJ/ton raw material	33
Substrate Dry Matter content	%	12[2]
Average Heat required	% of biogas produced MJ	13

1 The figures refer to a continuous, single stage, tank reactors, operating at mesophile temperature.

2 Value taken from Dalemo at al. 1997, in line with the value used by Laraia et al. 2002 and Poschl et al. 2010.

There are large variations in specific gas production and biogas composition, depending on the process used and waste characteristics. As noted in Section 4.11.2, this study assumes a value of 600 Nm3 of biogas per ton of VS (MTT et al., 2010) with a methane content of 63% (Møller et al., 2009), corresponding to a net calorific value of 23 MJ/Nm^3. Methane losses from the digestion process, primarily fugitive emissions, are assumed to be 2% of the total methane generated (Dalemo et al., 1997; Berglund et al., 2006; Fruergaard et al., 2011; Boldrin et al., 2011). Methane losses are difficult to measure; they vary significantly from one plant to another (Møller et al., 2009) and can be as high as 7% (Patterson et al., 2011). The significance of this is discussed in Section 5.4.

4.11.7 Digestate use

In this work, the digestate is considered as a by-product of the anaerobic digestion process which can be applied to agricultural soil, substituting commercial fertilisers and acting as a means of carbon storage. It is assumed here that the whole digestate is managed and spread as commercial fertiliser. Berglund et al. (2006) reported the primary energy demand for the diesel for spreading the digestate. The same value is considered here as 4.7 MJ per ton of digestate (Berglund et al., 2006).

Digestate behaves differently from commercial fertilisers. The quality of digestate is strongly related to the characteristics of the organic waste used to produce it and the effects of its application to the soil are '*complex, interacting and greatly dependent on local conditions'* (Hansen et al. 2006). In this study, it is assumed that no nutrients are lost during the anaerobic digestion process; therefore, the total nutrient content in the digestate is the same as that in the waste, following the approach of Møller et al. (2009).

Several agricultural models have been developed to simulate the processes in the soil, for example (Bruun, Hansen, Christensen, *et al.*, 2006). Sub-models for land application of treated organic waste in models for environmental assessment of waste management systems constitute only a small part of them, as presented in Table 3.2.

Table 4.13. Contributions to environmental impacts from land application of digestate in some existed models from the literature (Hansen, Christensen & Schmidt, 2006)**.**

	MW-DST	LCA-IWM	IFEU	ORWARE	EASEWASTE
Run-off to surface waters	Fixed amount of BOD, N, P per kg digestate	-	-	$K_1*N_{org}+N_{pool}$, eventually	K_1*N_{tot}
Leaching to ground water		-	-	-	K_2*N_{tot}
Loss of ammonia	-	-	$K_1*N_{org}+k_2*N_{am}$	K_2*N_{am}	K_3*N_{am}
Formation of nitrus oxide	-	-	K_3*N_{tot}	$K_3*N_{leaching}$	K_4*N_{tot}
Carbon binding	-	-	Peat substitution	-	K_5*C_{tot}
Commercial fertiliser production	-	Avoided air emissions included	Substitution (defined ratio)	Substitution (defined ratio)	Substitution (defined ratio)
Commercial fertiliser use	-	-	Avoided heavy metal input to soil included	Calculated emissions represent additional emissions	Emissions coefficients represent additional emissions

The EASEWASTE model's approach is mainly applied in this study. The loss of ammonia is determined as a fraction of the ammonia nitrogen in the waste. For digestate application, 7.5% of ammonia is assumed lost, while for commercial fertiliser no significant ammonia lost is assumed. Leaching of nitrate to surface and ground water are defined as a fraction of the nitrogen availability of the crop. Finally, nitrous oxide formation is determined as a fraction of the total nitrogen in the waste. Emissions of phosphorus to ground and surface water are not included in the model, since they depend '*strongly on soil properties and actual phosphorus content and less on the amount of phosphorous fertiliser applied to the land'* (Hansen et al. 2006). The amount of substituted commercial fertiliser is determined by the utilisation ratios representing the fraction of crop-availability of the nutrients in the digestate compared to commercial fertilisers. The utilisation ratio describes the cumulative effect over time, since the impact from digestate does not only occur during the first year of spreading. Moreover, specific national legislation can influence the amount of commercial fertilisers which can be substituted. The EASEWASTE model is referred to Danish conditions for type of soil and legislation.

The Waste, Resources and Action Programme (WRAP) is leading the '*Digestate and compost in agriculture'* project in UK, to investigate the use of quality anaerobic digestate and compost in agricultural soils. This includes the following objectives (WRAP 2011a WRAP 2011b; WRAP 2012a):

- To qualify and quantify the characteristics of the nutrients, especially nitrogen, and to estimate the proportion available to crops. This will help farmers to use smaller amounts of non-renewable commercial fertiliser.
- To quantify the effects of digestate applications on different soil and crop quality. Soil quality particularly affects the behaviour of nitrogen and its emission to the environment (air, soil and water).
- To quantify the emissions to air, soil and water from the application of digestate.

Results from this project have been used in this study, for both the case studies. Based on outcomes obtained by WRAP, the substitution ratio assumed here are 48% for nitrogen, 50% for phosphorus and 80% for potassium (WRAP, 2011a, 2011b). A sandy soil is assumed for the case study area (WRAP, 2012a) both in UK and Italy. The nitrogen in the organic fertilizer is contained in different compounds, as described in Section 2.6.2. The WRAP project has demonstrated that whole digestate from food waste contains about 80% of Ready Available Nitrogen (RAN) (WRAP, 2012b). Furthermore, in this study it is assumed that the quality and spreading of the digestate complies with national UK and IT regulations with regards to the use of fertilizers from organic sources (see Section 2.6.2).

The use in agriculture of fertilisers with high available nitrogen content, i.e. digestate, is likely to be affected by restrictions on the use of nitrogen in Nitrogen Vulnerable Zones (European Parliament, 1991). In UK, a limit of 250 kg N/ha per year is applied for NVZ, while in Italy is 170 kg N/ha/year. This has to be evaluated when the availability of land to receive the digestate is considered.

Part of the carbon in the waste is not released as biogas but remains in the digestate, while part of the carbon in the digestate is sequestered in the soil and not released to the atmosphere as CO_2 during the timeframe considered (Møller et al., 2009). This because the application to land of the digestate is seen as part of '*a changed farming practice resulting in generally increased carbon level in the soil, representing an actual removal of carbon dioxide from the atmosphere and therefore a negative contribution to the global warming impact*' (Hansen et al. 2006).

Table 4.14. Model parameters for digestate use on agricultural land with 100% sandy soil (Boldrin et al. 2011)**.**

Emissions	**Unit**	**Value used in this study**	**Values adopted by Bernstad and la Cour Jansen, 2011**
NO_3-N runoff	% of Applied N	25	25
NO_3-N leaching	% of Applied N	22	45
NH_3-N content	% Total N	13	-
NH_3-N evaporation	% of NH3	7.5	5 (of total N)
N_2O formation	% total N	1.4	1.25
C binding sequestration	% total C	13	-

The modelling parameters used in this study are summarised in Table 4.14 (Bruun et al., 2006 and Boldrin et al., 2011); they reflect the difference between use of inorganic fertilisers and digestate. This aspect has been explored by sensitivity analysis: see section 5.4.

4.11.8 Upgrading of biogas to biomethane

The biogas composition can change based on the composition of the feedstock and on the digestion process. In this work the following mix is assumed: 63% of CH_4, 30% of CO_2, 1% of N2, 6% of H_2O, 600 ppm of H_2S (Zhang, El-Mashad, Hartman, *et al.*, 2007). Whether the biogas is directly use in the energy recovery system or is upgraded to biomethane, a desulphurisation unit is needed to keep the H_2S under the level of 1 ppm. The most attractive and convenient method to remove H_2S from biogas is the use of an activated carbon bed, usually with a ZnO catalyst, very effective for hydrogen sulphur removal. In this study the desulphuriser unit is assumed to be placed at the digester plant, where the biogas is produced.

Currently, biomethane is produced in 177 plants in Europe, including 128 that feed into national natural gas distribution grids. The remaining plants use the biomethane generated on the production site, primarily as a fuel. To produce biomethane from biogas, an upgrading process is required, to remove the most of the CO_2 and to increase the energy density. To reach the specification for grid injection, different for each country, further step are required, such as drying to a certain water dew point, or adding other component such as LPG (Foreest, 2012). In this study, environmental impact associated with the upgrading of the biogas to 97% of CH_4 and calorific adjustments to reach the same value of natural gas are considered. The methodology considered to remove the CO_2 is the Pressure Swing Adsorption (PSA). With this technique, carbon dioxide is separated from the biogas by adsorption on a surface under elevated pressure. A methane loss of 3% is considering in the upgrading process (Patterson et al, 2011). The calorific adjustment is done with the addition of 0.03705 m3 Liquid Petroleum Gas (LPG) per m^3 of upgraded biomethane (Patterson et al., 2010). The electricity consumption considered is 1.1 MJ/m^3 biogas, based on the study of Poschl et al., 2010.

4.11.9 Biogas/biomethane transportation

In both the scenarios with biogas and biomethane, the fuel is assumed to be transported from the digester plant to the single dwelling. An average distance of 15 km is assumed, considering the Greenwich area and 25 km for the Livorno area. The energy demand for heating and compressing the gas to 1.6 MPa is 0.36 MJ/Nm^3 of gas and 0.18 MJ/Nm^3 of gas, respectively (Patterson et al. 2011; Poschl et al. 2010). A gas volume loss of 0.7% is assumed to occur every 10 km (Halliday, Ruddell, Powell, *et al.*, 2005).

4.11.10 Energy recovery system-Centralised scenario

Biogas can be used to produce energy through different conversion systems. In this scenario, it is assumed that biogas is used to produce electricity and heat with a CHP – Internal Combustion Engine (ICE) unit. Key parameters used to model the CHP plant include an electrical conversion efficiency of 32% and a thermal conversion efficiency of 50%, as suggested by Patterson et al., (2011). Emissions from the combustion of biogas are based on Fruergaard et al., (2011) and are shown in Table 4.15.

Table 4.15. Emissions associated with combustion of biogas in a CHP internal combustion engine (Fruergaard et al., 2011).

Parameter	Unit	Value
CO	g/MJ biogas (63% CH_4)	0.115
NOx	g/MJ biogas (63% CH_4)	0.148
CH_4 (for unburned CH_4)	g CH_4/MJ biogas (63% CH_4)	0.465
NMVOC	g/MJ biogas (63% CH_4)	0.105

4.11.11 Energy recovery system – Distributed generation scenarios

For the purpose of this study, life cycle inventory data for the micro-CHP systems are obtained from system manufacturers and additional data sources. Where possible, operational data from experimental tests are used (Farhad, Hamdullahpur & Yoo, 2010; Shiratori, Ijichi, Oshima, *et al.*, 2010). The reference year chosen for the comparison is 2012. This is particularly important for the on-going development technologies, such as SOFC unit or Stirling engine. Table 4.16 summarises the inventory data related with the micro-CHP units considered.

Table 4.16. Inventory data for micro-CHP units here considered.

	SOFC 1.7 kW Sunfire	**Stirling Engine 1.2 kW Whispertech**	**Micro Gas Turbines Capstone 30 kW**
Electrical efficiency (%)	35	12	26
Thermal efficiency (%)	55	80	52
Lifetime (h)	40,000	50,000	50,000
H to P	1.57	6.7	2
Status	demonstration	demonstration	Few commercial models
Parassitic loads (% of the total electricity produced)	2% (Halliday, Ruddell, Powell, *et al.*, 2005)	2% (Pöschl, Ward & Owende, 2010)	10% (Pöschl, Ward & Owende, 2010)
Emissions (g/Nm^3 of fuel as input)	NOx:0.003 (Pehnt, 2008)	NOx: 0.08 (Pehnt, 2008)	NOx: 0.03 (Pinelli, 2004) CO: 0.03 (Pinelli, 2004)

The operational strategy for the micro-CHP units has already been defined in Section 4.6. To supply space heating for the dwellings, a thermal storage is required. Following the approach of Giannoupoulis and Founti, 2011, 15% thermal losses are then considered, which means that the micro-CHP units are designed to meet the 115% of the total space heating demand. Consequently, considering that the H to P ratio of the dwelling is different from the one of the micro-CHP units, surplus electricity is sold to the grid and considered as avoided burdens. Moreover, an assumption is made on the annual electric import from grid needed in every scenario by the dwelling: 10% of the total annual electric load. This is due to thermally driven operation strategy, and to the limitation in terms of thermal cycles for the micro-CHP units (especially for the SOFC).

As stated in Section 4.6, the timeframe for the DG scenarios is set accordingly to the operational framework of the reference scenario. For the FT and HT scenarios, this is the condensing boiler, which is assumed to have a lifetime of 50,000 hours or 20 years. More than one SOFC stacks per year, per single dwelling is than necessary. Here the case of the SOFC-micro CHP unit in the UK case study is described (for IT would be the same). The SOFC-micro CHP unit has to operate for 5798 h/year, considering: space heating demand of a single dwelling equal to 15,654 kWh per year (gross, including heat losses), assuming no lifetime limitation for the fuel cell, and given its thermal output of 2.7 kWth. Unfortunately, a lifetime operational hour is one of the weakest points of fuel cells at the present and it cannot be neglected. Although in this study is assumed no limitation in terms of thermal cycles, 3.7 stacks are required per year per single dwelling to fully satisfy the space heating demand (FT scenario). The results are halved for the HT scenario, as showed in Table 4.17. The SE works the same hours of condensing boiler, given that the thermal power output is the same for the two systems. The evaluation is presented even for the mGT, even if the size of the unit requires the installation of a micro grid to supply energy to more than one dwelling with the same unit. The value for the mGT has to be referred to the portion of one unit that is dedicated to a single dwelling. The calculations are performed for the EL scenario too. Here the technology in the reference scenario is the electricity grid. To evaluate the number of stacks for SOFC and SE a lifetime of 20 years is taken into account, considering the average lifetime of a standard power generator.

A constant number of units are considered for all the three operating strategy scenarios, for each technology. For SE and SOFC technologies the number of units is equal to the number of dwellings. For the mGT, given that the size of the unit is larger than the other two, the following procedure has been followed to determine the number of units:

1. The operating hours per year are calculated ($h_{\text{as\_single\_unit}}$), assuming that only one unit of micro Gas Turbine has to satisfy the total energy demand of one dwelling;
2. the number mGT units are calculated, assuming the lifetime of the micro Gas Turbine ($h_{lifetime}$=50,000 h) and the lifetime of 20 years for a typical power generator ($y_{reference}$):

$$n_{mGT\_units} = \frac{h_{as\_single\_unit}}{\dfrac{h_{lifetime}}{y_{reference}}}$$

The results are shown in Table 4.17.

Table 4.17. Number of stacks required and operating hours for the three technologies for single dwelling.

		UK		IT	
		Number of stacks	**Hours of operation per year**	**Number of stacks**	**Hours of operation per year**
FT	***Condensing Boiler***	1.00	1,956.44	1.00	1,006.25
	SOFC	3.70	5,796.85	3.70	2,981.48
	Stirling Engine	1.00	1,956.44	1.00	1,006.25
	Micro Gas Turbine	0.13[1]	260.86	0.13[1]	134.17
HT	***Condensing Boiler***	1.00	978.22	1.00	503.13
	SOFC	1.85	2,898.43	1.85	1,490.74
	Stirling Engine	0.50[1]	978.22	0.50[1]	503.13
	Micro Gas Turbine	0.07[1]	130.43	0.07[1]	67.08
EL	***SOFC***	1.28	2,558.82	1.28	1,764.71
	Stirling Engine	1.45	3,625.00	1.45	2,500.00
	Micro Gas Turbine	0.06[1]	145.00	0.06[1]	100.00

[1]Number of stacks minus than 1 has not a correspondence with the reality and it is just an indication.

5. Results

5.1 Introduction

The layout of the results presented here is a mirror of the approach followed in Chapter 4, dividing the study in two parts. The impacts associated with the waste management processes (WM scenarios) are described in Section 5.2, for both the case studies. First the UK case study is analysed, then the Italian ones. After, Section 5.3 presents the results for the DG scenarios, in the three operating strategies investigated. Section 5.4 shows a sensitivity analysis for key parameters associated with the two sub-system models. Finally, Section 5.5 presents a sensitivity analysis on the landfill model in LCA.

5.2 Waste management scenarios

The results presented here refer to the five impact categories introduced in Section 4.7: Abiotic Depletion, Global Warming Potential, Acidification Potential, Photochemical Ozone Creation Potential, and Nutrient Enrichment. Figure 5.1-5.6 show the results obtained for the three WM scenarios (landfill, incineration and AD), dividing the burdens in *direct*, *indirect* and *avoided*. The burdens associated with the reference scenario are showed as avoided burdens; therefore they have a negative contribution – thus they represent a reduction in terms of emissions - to the final environmental impact. Furthermore, the indirect burdens for incineration and landfill scenarios account only for the emissions associated with the transport of the waste from the Transfer Station to the plant and emissions due to the production of the diesel, given that the model used to represent the two processes is aggregated.

UK scenarios

Figure 5.1 shows the environmental impact considering the electricity grid as average production technology in the reference scenario. In terms of Abiotic Depletion potential, anaerobic digestion shows the best impact. This category is strictly linked to the amount of energy produced and the substituted technology. In this particular case, the energy produced with the incineration plant accounts only for the energy allocated with the organic fraction of the MSW, which is usually not convenient for combustion due to its moisture content. It is evident that AD is the most favourable alternative in terms of Global Warming Potential, the most popular category indicator for climate change impact (Guinée et al. 2001), with a total of -6,520 tons of CO_2 eq per functional unit, compared with an impact of +31,800 and -5,180 tons of CO_2 eq for the landfill and incineration scenarios, respectively. The negative values for AD and incineration scenarios represent a saving of emissions, arising from the avoided burdens of energy production and, for AD, commercial fertiliser production.

Looking at the impact category Acidification Potential, AD is again the most favourable option, as shown in Table 5.1. Despite this, the relative difference with the landfill scenario is smaller compared with the GWP impact. The main reason for that lies in the emissions of NO_x from the CHP unit (see Figure 5.2). In this category, the pollutants' potential for acidification is measured by its capacity to form H+ ions; however, actual acidification varies depending on where the acidification pollutants are deposited (Guinée et al. 2001).

Table 5.1. Environmental impacts for the Waste Management scenarios, UK case study, average electricity production technology. The lowest value for each category is highlighted in red.

	Incineration	Landfill	Anaerobic Digestion
Abiotic Depletion (ADP fossil) [MJ] X 10,000	-9.39E+03	1.10E+03	-1.34E+04
Global Warming Potential, excl biogenic carbon (GWP 100 years) [kg CO_2-Equiv.] X 1000	-5.18E+03	3.18E+04	-6.52E+03
Acidification Potential (AP) [kg SO_2-Equiv.]	6.63E+03	5.79E+03	-1.26E+03
Photochem. Ozone Creation Potential (POCP) [kg Ethene-Equiv.]	-1.17E+02	1.04E+04	4.44E+03
Nutrient enrichment potential [kg NO_3-Equiv.] x 10	3.01E+03	4.55E+04	1.45E+04

The comparison changes when the focus is on the Photochemical Ozone Creation Potential and the Nutrient Enrichment potential. For these two impact categories, AD turns out to be the second option (4.44 tons of ethene eq and 14.7 tons of NO_3 eq, respectively), while incineration appears to be the most environmentally friendly solution, showing a negative impact in terms of POCP (-0.117 tons of ethene eq) and an impact of 3.01 tons of NO_3 eq in terms of Nutrient Enrichment. Photo-oxidant refers to formation of reactive chemical compounds by the action of sunlight on certain primary air pollutants, such as VOCs, CO and NO_x, causing injuries to human health, ecosystems and crops (Guinée et al. 2001). For POCP, combustion of biogas with consequent production of Non Methane Volatile Organic Compounds (NMVOC) in the CHP unit contributes more than 70% of the total ethene eq. (see Figure 5.2). Nutrient Enrichment refers to the environmental impact of nutrients that leads to shift in species composition and increased biological productivity, for example algae blooms (Baumann & Tillman, 2004). The burdens associated with the use of digestate on agricultural soil contributes more than 73% to the total NO_3 eq in the anaerobic digestion scenario (Figure 5.2) while the CHP unit gives another 16% through NO_x emissions to air. It is important to remember that the soil quality, the waste composition and the precipitation pattern specific for the area can significantly influence the performance of this process.

Incineration with energy recovery is the best option when GWP, POCP and NE are compared with the landfill scenario, mainly due to the bigger amount of heat and electricity produced in the former respect to the electricity produced in the latter. However, AP for landfill with gas recovery shows a slightly lower value (5.79 compared with 6.63 ton SO_2eq), while the high value of NE potential obtained for the landfill scenario (45.5 ton of NO_3 eq) is mainly due to leakage of nutrients into water and soil.

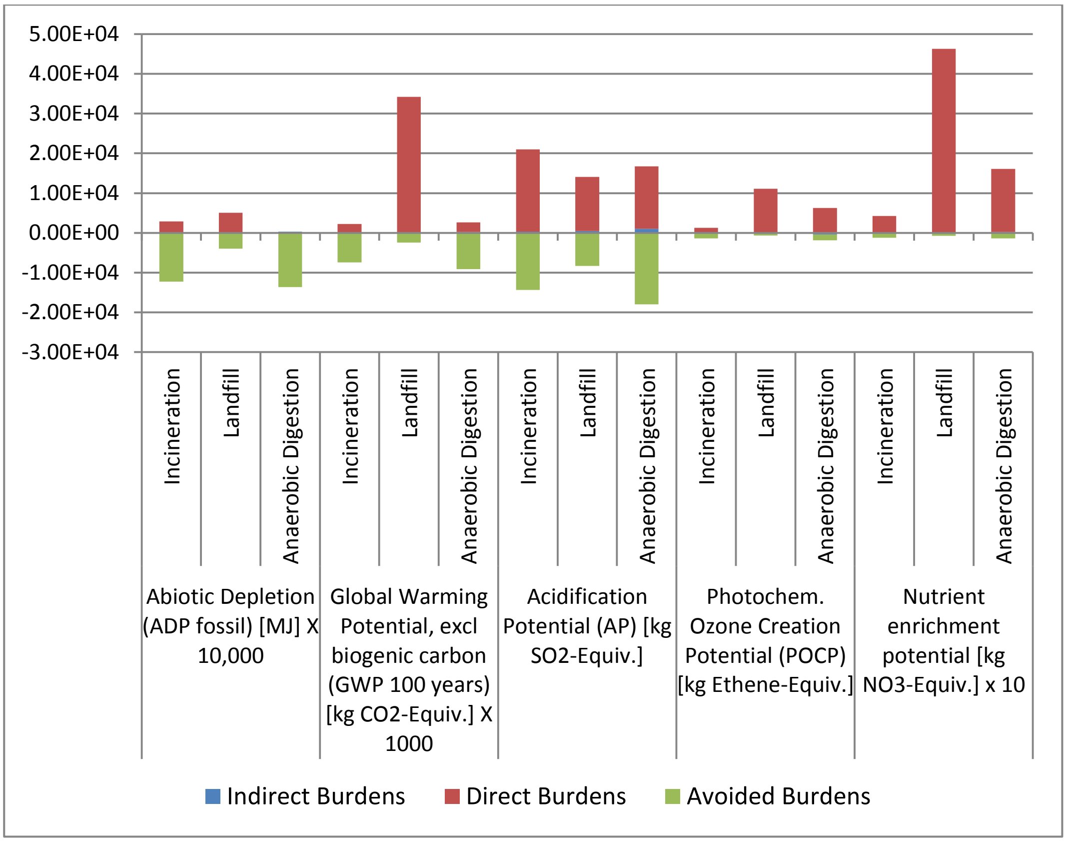


Figure 5.1. Environmental impacts for the WM scenarios, UK case study, considering average electricity production technology (FU = 45,033 tons of OFMSW per year).

Looking at the results, it is worth nothing that the indirect burdens contribute with a small impact to the final values. The emissions accounted for in this category are related with the transport of the OFMSW from the Transfer Station to the specific plant and they consider both the production of the diesel and the emissions associated with the trucks. The values reflect the assumptions made in the three scenarios on the distance between the Transfer Station and the specific plant. Summarising what has been already described in Chapter 4, the assumptions are: 50 km from the TS to the AD plant; 15 km from the TS to the incineration plant and 30 km from the TS to the landfill site.

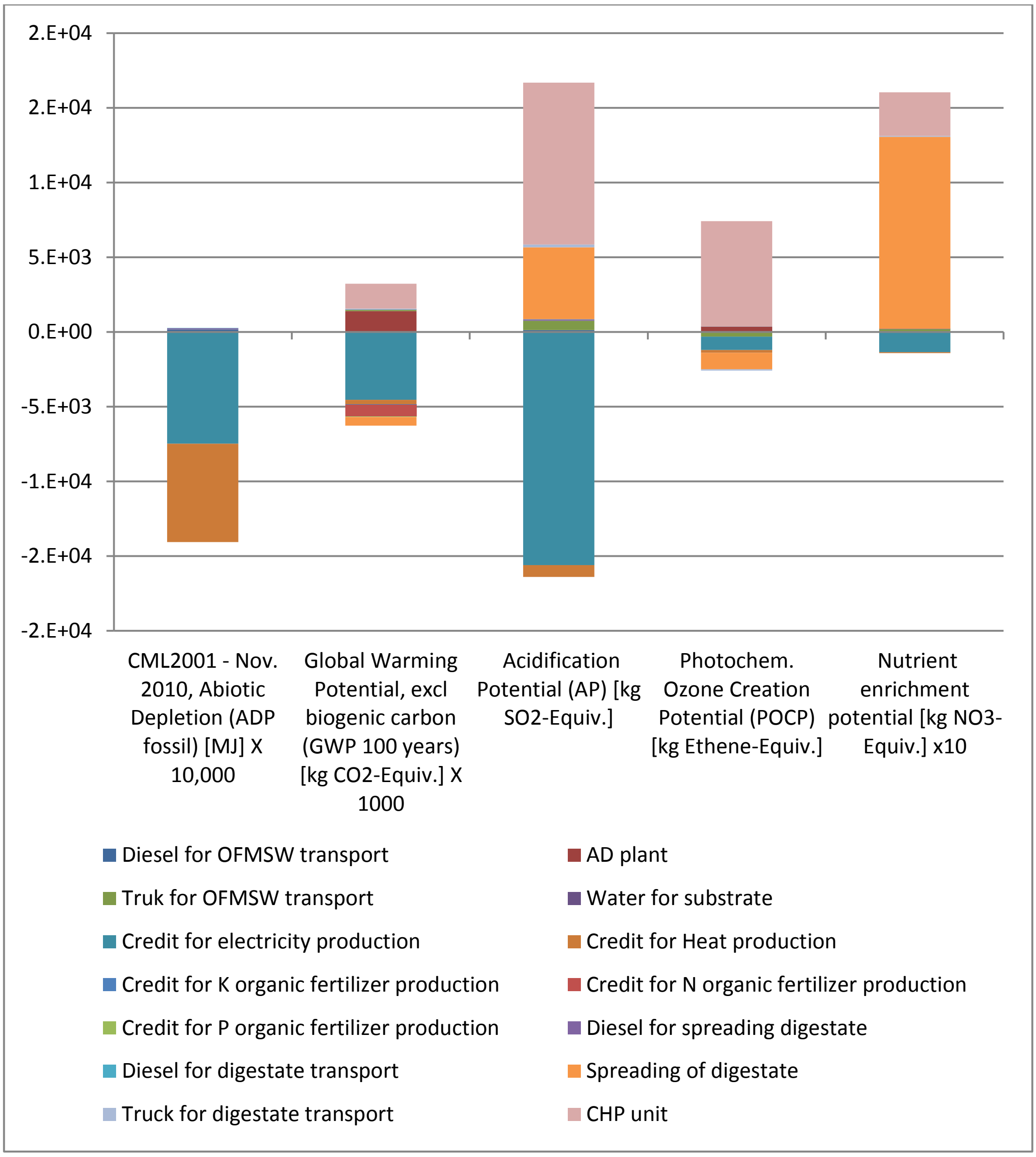


Figure 5.2. Hot spot analysis, WM – AD scenario, average electricity production technology, UK.

Hot spot analysis is used to identify which activities or process steps cause the greatest environmental impact (Baumann & Tillman, 2004). Figure 5.2 shows the hotspot analysis for the anaerobic digestion scenario for the five impact categories considered. The process phases that contribute most to the total GWP are the CHP unit (13%), where the production of energy occurs, and the AD plant (16%). For this category, the burdens are mainly due to methane leakage from the plant and to emissions of unburned methane at the exit of the gas engine. Looking at AP, it has been already said that the process phase that contributes most is the combustion of the biogas. The spreading of the digestate has instead the highest impact in terms of kg of NO_3 eq, due to the different nitrogen compounds in the organic digestate compared against commercial fertiliser.

The importance of considering a specific substitution technology when evaluating the avoided burdens is well established (e.g. Fruergaard et al. 2009; Turconi et al. 2011). Results from previous studies on anaerobic digestion processes employed as an alternative waste treatment options show that the technology substituted by energy production can change the final ranking between different treatment alternatives (Bernstad & La Cour Jansen, 2011). The difference is highly significant for policy decisions and technology development. Over a long time frame (more than the 20 years corresponding to the typical life on an AD plant), the system for the production of energy in the background may change completely, favouring newer technologies which may not currently be available, and changing completely the final impact results.

Figure 5.3 and Figure 5.4 show the WM scenarios when marginal technologies for the electricity production are considered. *Marginal data* represents the production technologies actually affected by the induced change to the system, in this case the energy generation displaced by energy recovery from waste (Fruergaard et al. 2009). Generally one single technology is assumed to be the "marginal" energy production plant. The best way to identify the correct technology would be through an energy system analysis (Münster & Lund, 2010). However, this is beyond the scope of this thesis and approaches from the literature have been evaluated to identify the proper technology.

Table 5.2. Environmental impacts of marginal electricity production technologies in UK, per MJ of electricity produced (PE International, 2013)**.**

Environmental impact	Unit	Coal plant	CCGT plant
AD	MJ/MJ	2.9	2.11
GWP	kg CO_2 eq/MJ	0.278	0.125
AP	kg SO_2 eq/MJ	0.0013	0.000006
POCP	kg ethane eq/MJ	0.00007	0.000009
NE	kg NO_3 eq/MJ	0.00116	0.000012

Various approaches have been proposed to define the marginal technology for electricity production in UK (Fisher, 2006; Weidema, Frees & Nielsen, 1999). Staffell et al. (2011) suggest that the marginal energy in the UK is provided by coal and CCGT plants which are *''marginal' or 'peaking' plants (which) respond to instantaneous changes in the nationwide energy demand by varying their output'*. Table 5.2 shows the environmental impacts of these technologies (from PE database, 2012, UK).

If the marginal generation displaced by electricity from biogas is entirely produced by coal-fired plant, as shown in Figure 5.3, the environmental profile the anaerobic digestion plant improves considerably, along with the ones of the other two WM scenarios. However the ranking amongst the different treatment remain the same for all the categories, except for AP. The GHG emissions of the Anaerobic Digestion plant decrease by 80% and the acidification potential shows the biggest benefits. In fact, in this category, the Incineration plant – which showed the worst behaviour in the Figure 5.1 – has a lower impact compared with the landfill plant.

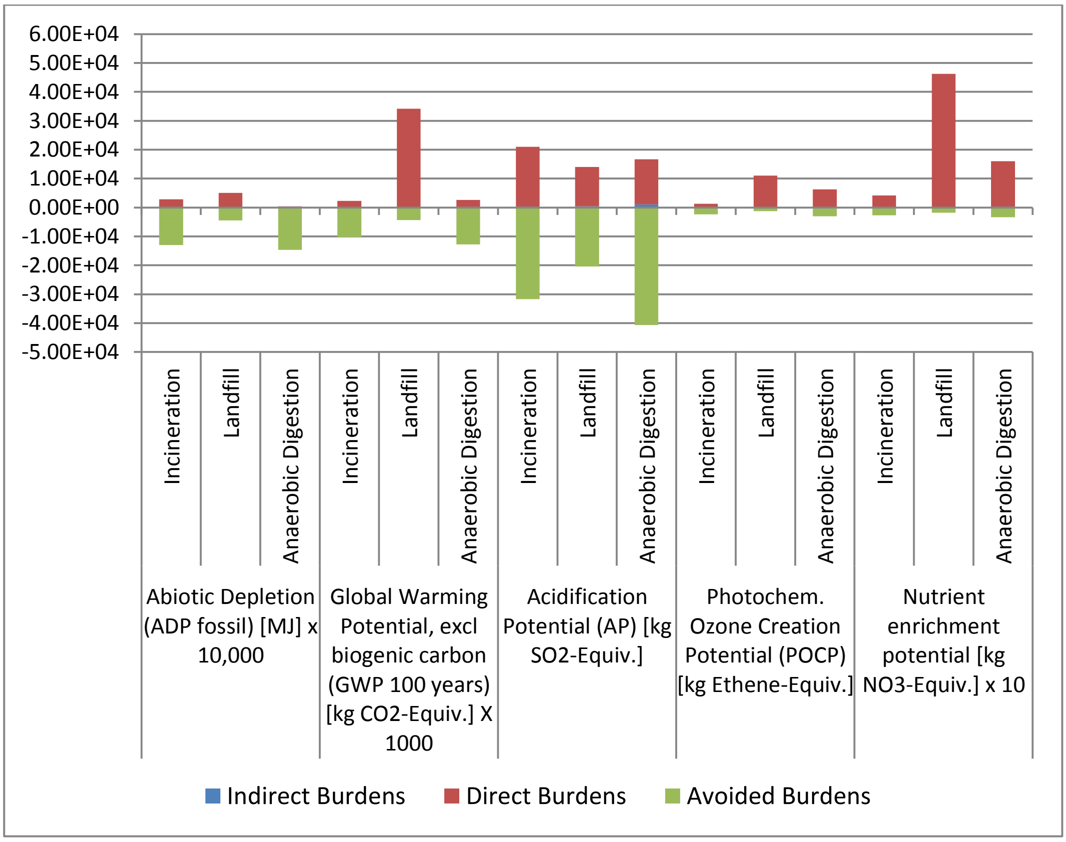


Figure 5.3. Environmental impacts for the WM scenarios, UK case study, considering Coal plant - Marginal electricity production technology (FU = 45,033 tons of OFMSW per year).

Comparing the result of Table 5.1 with the ones in Table 5.3, the AP for the Incineration scenario decreases by 260%, the impact of the landfill scenario decreases by 209% and the AD scenarios decreases by twenty times. This results in a negative value of the three WM scenarios. This is mainly due to the impact in terms of Acidification Potential of the Coal plant technology which is almost double compared with the average UK mix for the electricity production.

Table 5.3. Environmental impacts for the WM scenarios considering coal plant as marginal electricity production technology for UK. The lowest value for each categoriy is highlighted in red.

	Incineration	Landfill	Anaerobic Digestion
Abiotic Depletion (ADP fossil) [MJ] x 10,000	-1.02E+04	5.57E+02	-1.44E+04
Global Warming Potential, excl biogenic carbon (GWP 100 years) [kg CO_2-Equiv.] x 1,000	-7.92E+03	2.99E+04	-1.01E+04
Acidification Potential (AP) [kg SO_2-Equiv.]	-1.07E+04	-6.29E+03	-2.40E+04
Photochem. Ozone Creation Potential (POCP) [kg Ethene-Equiv.]	-1.03E+03	9.80E+03	3.24E+03
Nutrient enrichment potential [kg NO_3-Equiv.] x 10	1.47E+03	4.44E+04	1.26E+04

If the electricity displaced is entirely from natural gas CCGT plants (Figure 5.4), energy-from-waste by AD appears worse in all impact categories. While, in terms of GWP, the production of electricity via biogas from AD still represents a more environmentally friendly option compared with CCGT plants, it leads to an increase of the environmental burdens in terms of AP (positive value). Moreover all the scenarios appear closer in terms of impact and although a ranking amongst them is still possible, it is more difficult to justify a new investment in terms of environmental benefits.

Table 5.4. Environmental impacts for the WM scenarios considering CCGT plant as marginal electricity production technology for UK. The lowest value for each category is highlighted in red.

	Incineration	Landfill	Anaerobic Digestion
Abiotic Depletion (ADP fossil) [MJ] x 10,000	-8.40E+03	1.79E+03	-1.21E+04
Global Warming Potential, excl biogenic carbon (GWP 100 years) [kg CO_2-Equiv.] x 1,000	-4.52E+03	3.23E+04	-5.64E+03
Acidification Potential (AP) [kg SO_2-Equiv.]	1.71E+04	1.31E+04	1.25E+04
Photochem. Ozone Creation Potential (POCP) [kg Ethene-Equiv.]	3.52E+02	1.08E+04	5.06E+03
Nutrient enrichment potential [kg NO_3-Equiv.] x 10	3.79E+03	4.60E+04	1.57E+04

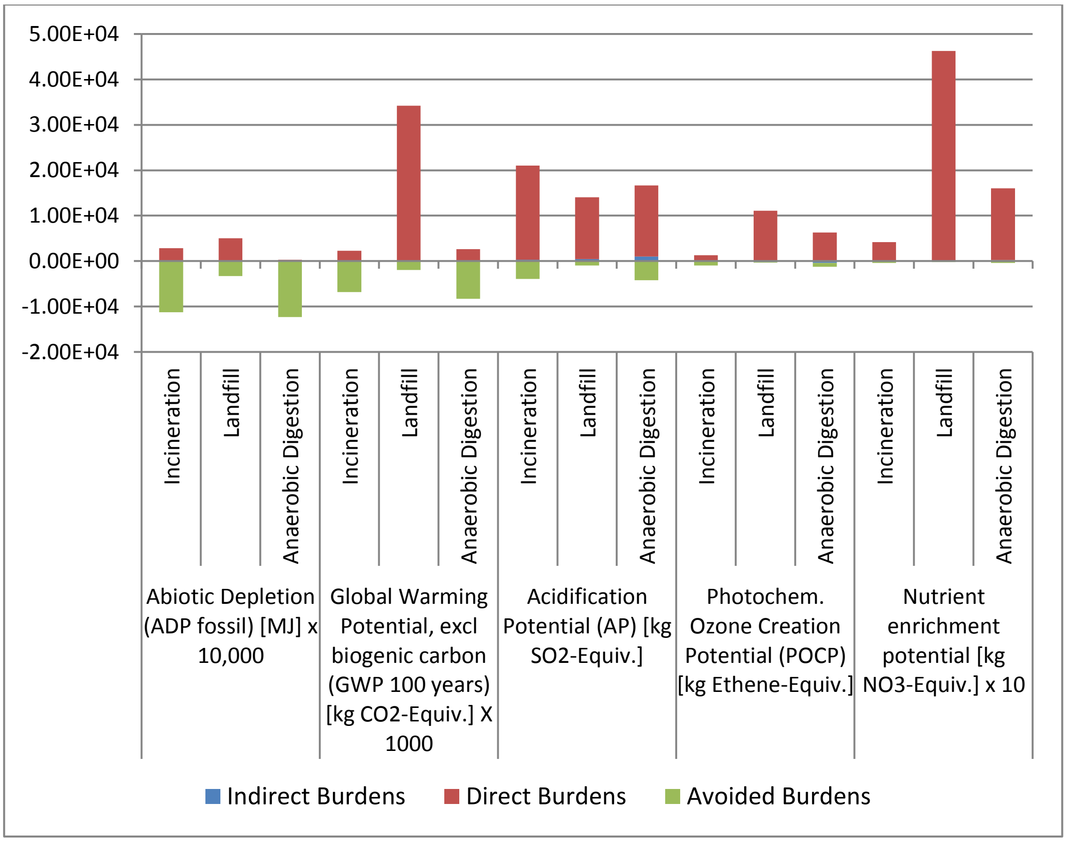


Figure 5.4. Environmental impacts for the WM scenarios, UK case study, considering CCGT plant - Marginal electricity production technology (FU = 45,033 tons of OFMSW per year).

IT scenarios

The results for the Italian case study are slightly different. Although the rankings amongst the three waste treatments are the same for all the impact considered, the relative values are different. This is especially true for POCP, AP and NE. In fact, comparing the results of Table 5.5 with the results presented in Table 5.2 is possible to highlight that:

- AP impact turns positive for AD in the Italian scenario, while it was negative in the UK case. This is mainly due to the different impacts of the energy production, here considered as avoided burdens, between the two countries (see Table 4.8 and Section 4.12.1).;
- in terms of Abiotic Depletion, the UK scenarios show a lower impact. Again this is due to the different avoided burdens associated with the energy production technologies considered in the reference scenario;
- for both the case studies, the indirect burdens, mainly associated with the transport of the waste from the TS to the plant and of the digestate from the AD plant to the arable land, is negligible when compared with the direct and avoided burdens. The only exception is the GWP in the Italian case study, where the indirect and direct burdens are of the same order of magnitude.

Table 5.5. Environmental impacts for the WM scenarios, IT case study, average electricity production technology. The lowest value for each category is highlighted in red.

	Incineration	Landfill	AD
Abiotic Depletion (ADP fossil) [MJ] x 10,000	-6.87E+03	1.40E+03	-9.95E+03
Global Warming Potential, excl. biogenic carbon (GWP 100 years) [kg CO2-Equiv.] x 1000	-4.33E+03	2.59E+04	-5.53E+03
Acidification Potential (AP) [kg SO2-Equiv.]	7.72E+03	6.15E+03	7.32E+02
Photochem. Ozone Creation Potential (POCP) [kg Ethene-Equiv.]	-1.65E+02	8.56E+03	3.92E+03
Nutrient enrichment potential [kg NO3-Equiv.] x 10	2.80E+03	3.73E+04	1.03E+04

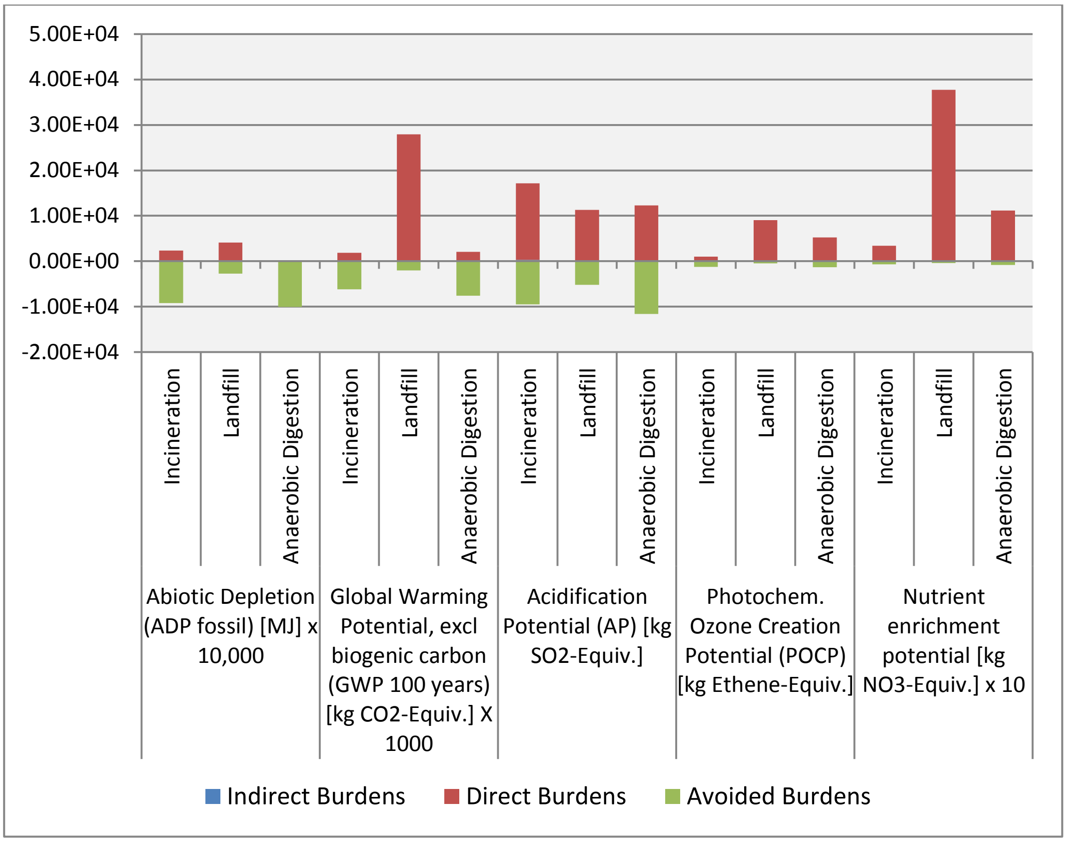


Figure 5.5. Environmental impacts for the WM scenarios, case study, considering average electricity production technology (FU = 36,765 tons of OFMSW per year).

Looking at the hot spot analysis for the AD scenario in the IT context, we can see that the main difference compared with the UK one is the avoided emissions associated with the substitution of electricity and heat from the reference scenario. In fact, as shown in Table 4.8, the emissions associated with the production of 1 MJ of electricity in Italy are higher than the ones in UK. This is due to the different technologies mix used to produce electricity in the specific country. UK, in fact, produces the 20% of electricity from nuclear plant which is a quite green technology[18] , while the largest amount – i.e. 48% - is produced from natural gas and 30% comes from hard coal. In Italy, on the other hand, the main source is again natural gas – i.e. 55%, followed by the hydro power, which represents the 20% and the hard coal that contributes for the 15%. Finally, a 10% is produced by heavy fuel oil plant (PE International, 2013).

[18] Whether to consider nuclear energy a green technology or not is a much discuss topic and it depends on if we include the full life cycle – from cradle (the extraction of raw material) to grave (the disposal of nuclear waste), or only the direct burdens related to the energy production process only.

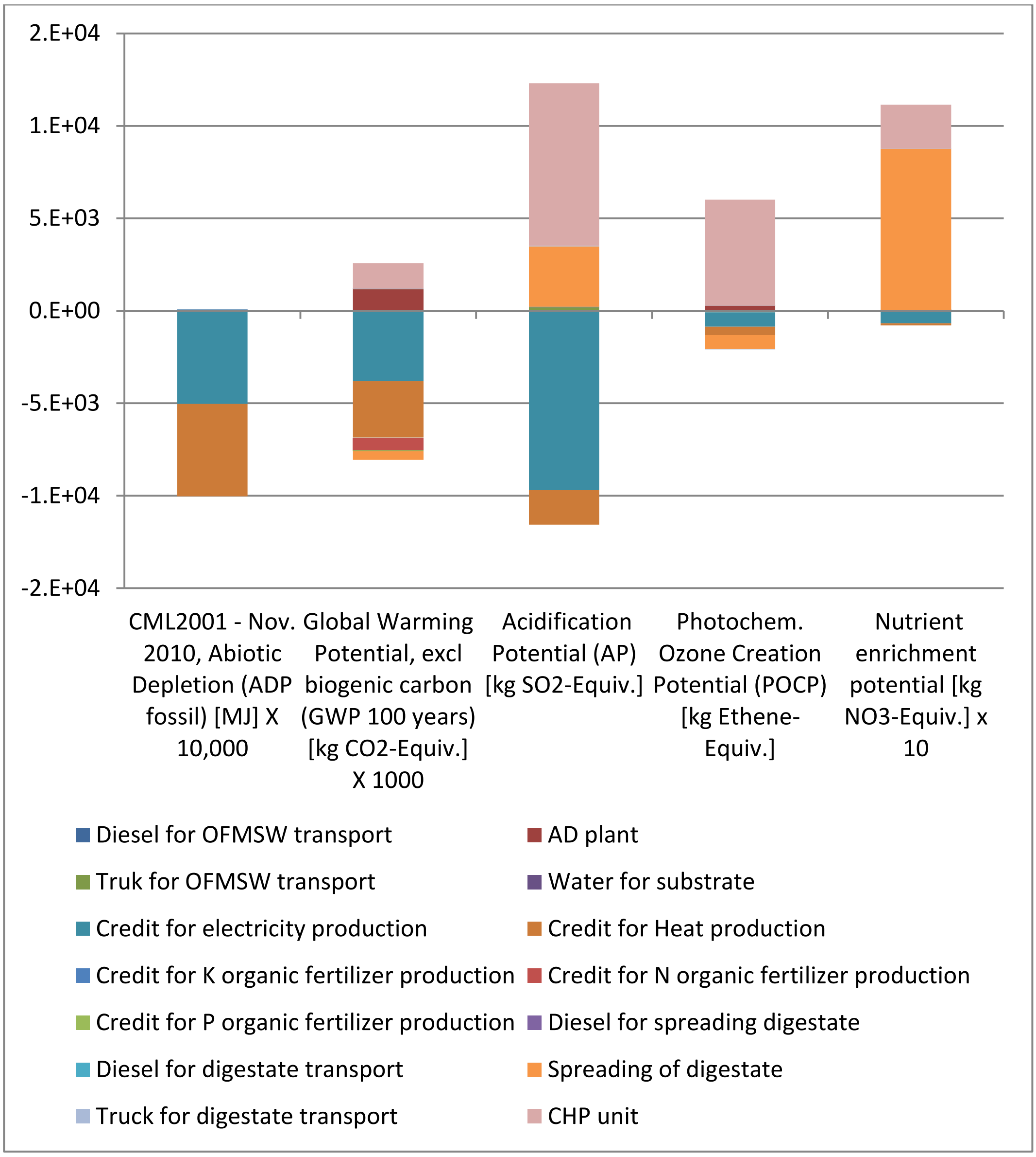


Figure 5.6. Hot spot analysis, WM – AD scenario, average electricity production technology, IT.

The work of Turconi et al. (2011) has been taken as reference to identify the marginal technologies for electricity production in Italy. Electricity production has been steadily increasing over the last decade, mainly based on increased natural gas consumption. Simultaneously oil consumption had decreased at a similar rate. Within the same period, coal played a less significant role, contributing to a minor share of the total production. In a growing electricity market, such as the Italian, the responding technology meeting this increased energy demand is also the one which will be affected by increased electricity production from waste. For Italy, this means that increased electricity production from waste will most likely reduce the need for additional electricity production from natural gas (Turconi, Butera, Boldrin, *et al.*, 2011).

Table 5.6 shows the burdens associated with the production of 1 MJ of electricity when the two marginal technologies are considered.

Table 5.6. Environmental impacts of electricity production technologies for IT, to produce 1 MJ of electricity (GaBi, 2013).

Environmental impact	Unit	Coal plant	CCGT plant
Abiotic Depletion	MJ/MJ	3.044	2.32
Global Warming Potential, excl. biogenic carbon	kg CO_2 eq/MJ	0.30920	0.14561
Acidification Potential	kg SO_2 eq/MJ	0.00090	0.000128
Photochem. Ozone Creation Potential	kg ethane eq/MJ	0.000053	0.0000026
Nutrient enrichment potential	kg NO_3 eq/MJ	0.00066	0.000015

Figure 5.8 and Figure 5.7 show the results of the system when marginal electricity production technologies (coal plant and CCGT plant respectively) are considered.

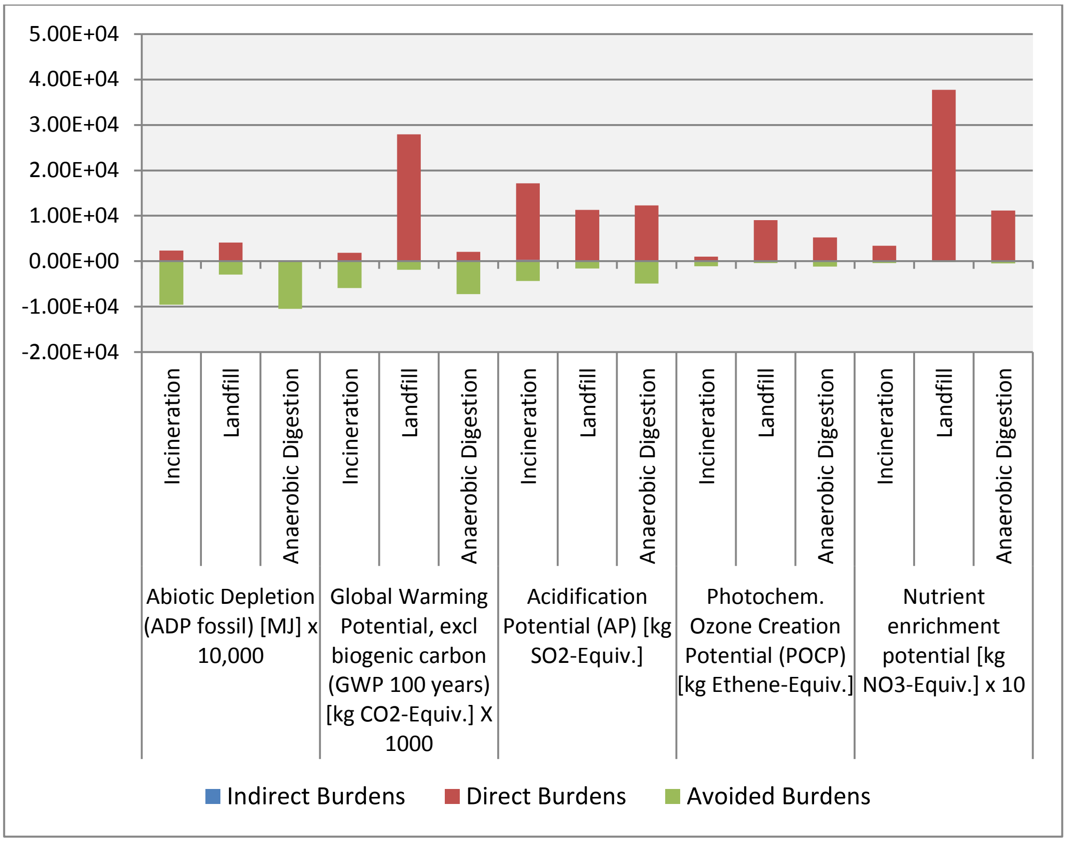


Figure 5.7. Environmental impacts for the WM scenarios, IT case study, considering CCGT plant - Marginal electricity production technology (FU = 36,765 tons of OFMSW per year).

Table 5.7. Environmental impacts for WM scenarios for Italy case study, considering CCGT plant as marginal electricity production technology. The lowest value for each category is highlighted in red.

	Incineration	Landfill	AD
Abiotic Depletion (ADP fossil) [MJ] x 10,000	-7.23E+03	1.15E+03	-1.04E+04
Global Warming Potential, excl. biogenic carbon (GWP 100 years) [kg CO2-Equiv.] x 1000	-4.05E+03	2.61E+04	-5.17E+03
Acidification Potential (AP) [kg SO2-Equiv.]	1.28E+04	9.70E+03	7.38E+03
Photochem. Ozone Creation Potential (POCP) [kg Ethene-Equiv.]	-3.30E+01	8.65E+03	4.09E+03
Nutrient enrichment potential [kg NO3-Equiv.] x 10	2.99E+03	3.75E+04	1.07E+04

Table 5.7 shows the environmental impacts of the WM scenarios when CCGT plant is considered as marginal production technology for electricity production. The ranking amongst the three treatments remains the same, and in general all the categories worsen.

To complete the IT case study, a coal plant is considered as in the UK scenarios. The results reflect what have been obtained in the previous section, with all the categories getting better due to the higher avoided burdens associated with the electricity produced from biogas.

Table 5.8. Environmental impacts for the WM scenarios for Italy case study, considering Coal pant as marginal electricity production technology. The lowest value for each category is highlighted in red.

	Incineration	Landfill	AD
Abiotic Depletion (ADP fossil) [MJ] x 10,000	-8.54E+03	2.39E+02	-1.21E+04
Global Warming Potential, excl. biogenic carbon (GWP 100 years) [kg CO2-Equiv.] x 1000	-7.03E+03	2.40E+04	-9.05E+03
Acidification Potential (AP) [kg SO2-Equiv.]	-1.35E+03	-1.34E+02	-1.10E+04
Photochem. Ozone Creation Potential (POCP) [kg Ethene-Equiv.]	-5.33E+02	8.30E+03	3.44E+03
Nutrient enrichment potential [kg NO3-Equiv.] x 10	2.05E+03	3.69E+04	9.45E+03

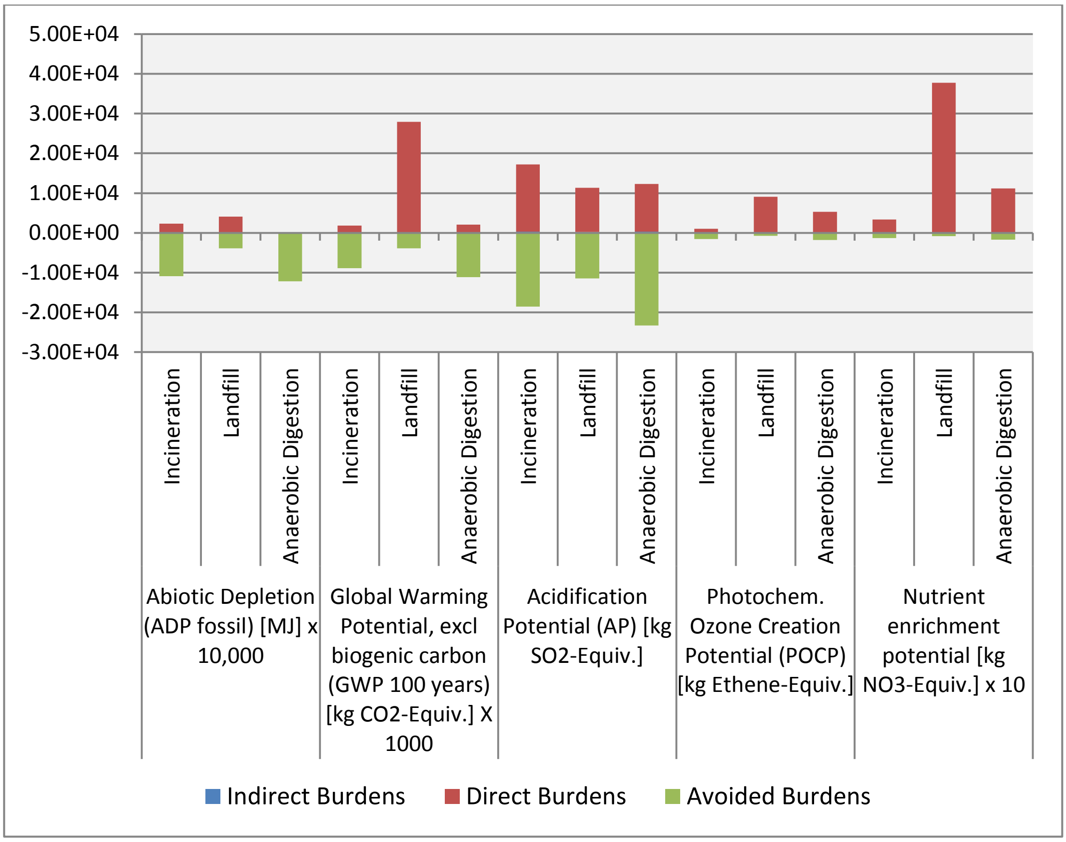


Figure 5.8. Environmental impacts for the WM scenarios, IT case study, considering Coal Plant - Marginal electricity production technology (FU = 36,765 tons of OFMSW per year).

Comparison between the two geographical areas

Comparing the results between the UK and IT scenarios, it is necessary to point out, first of all, that the functional units used in the two studies are different, reflecting the different amount of OFMSW generating in the area. Although the Livorno municipality has a higher rate of OFMSW production per household (513 kg/year) compared with the Royal Borough of Greenwich (441 kg/year), the total amount of organic waste considered here is higher in the latter, due to the larger number of families living in the borough. Normalization is then needed to compare the results. The FU considered here is the amount of waste produced per household in the specific country: therefore the quantity is different for the two areas, but the function performed by the two scenarios is the same. The comparison is shown in Figure 5.9 below.

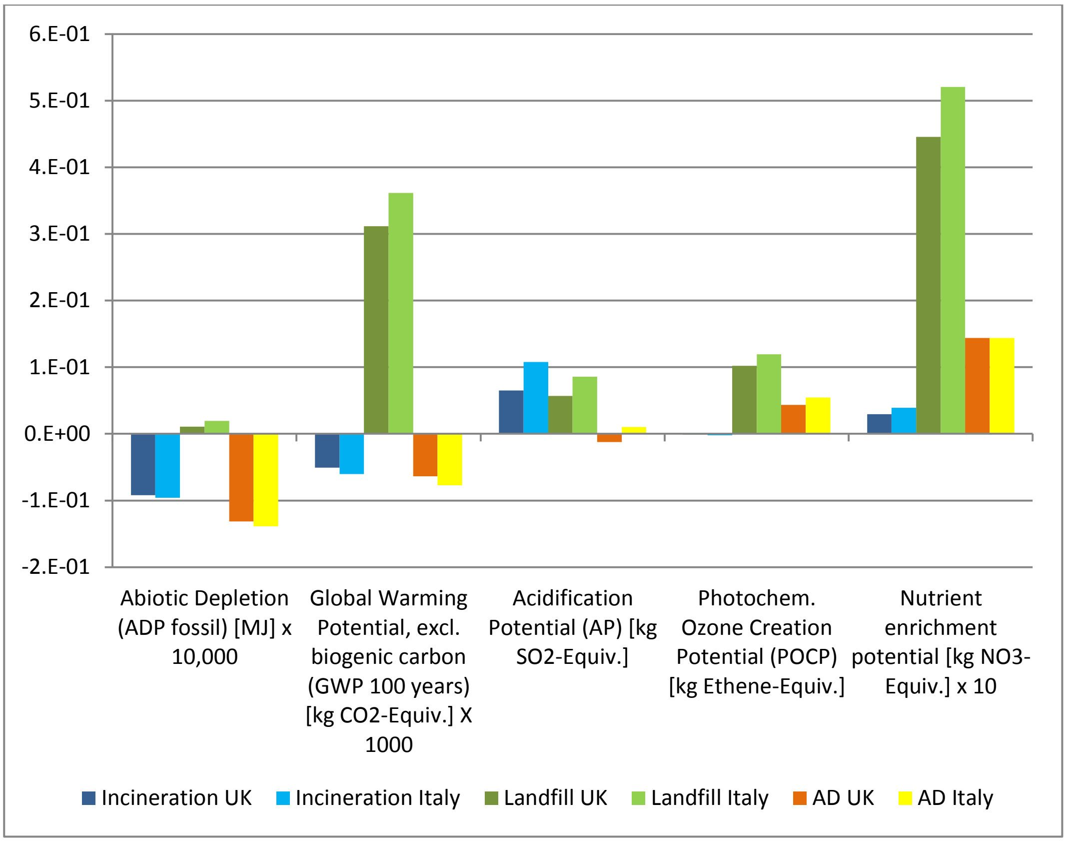


Figure 5.9. Comparison between the two geographical areas, considering as FU the amount of OFMSW produced per household.

The impacts in the Italian case study are higher for almost all the categories. This is in line with the fact that the kg of OFMSW treated per household is bigger in Italy compared to UK. The only exception is the Nutrient Enrichment potential for the AD scenario. Here, in fact, the values are the same. This is mainly due to two reasons: firstly the transport distances assumed in Section 4.4 between the Transfer Station and the plants are different (generally double in the UK scenario compared with the IT one); secondly the quantities of nutrients assumed in the organic waste are higher in the UK case than in the IT one (see Section 4.12.2). The assumptions are based on a European funded project, carried out in several countries in Europe (UK and Italy included), on the elemental composition of the organic fraction of MSW (MTT, 2010).

Figure 5.10 shows the comparison when the Functional Unit is equal to 1 kg of OFMSW, in order to observe the influence of the type of electricity production technology that is assumed as avoided burdens. The higher values of the IT scenarios in the Abiotic Depletion and Acidification Potential impact categories, for example, reflects the higher impact of the average electricity production technology in UK, that is considered as avoided burden and then subtracted to the total impact.

It is worth highlighting the different impacts in terms of AP for the AD scenarios in the two geographical contexts. For UK, in fact, it results -2.28 x 10^ (-5) kg SO_2 eq per kg of OFMSW treated, while for IT is 1.99 x 10^(-5) kg SO_2 eq. The different mix of technologies that are considered as average electricity production technology plays then a fundamental role, resulting in a shifting from saving to burden. Decision makers have to consider this, when evaluating different technologies for investments in the waste-to-energy sector.

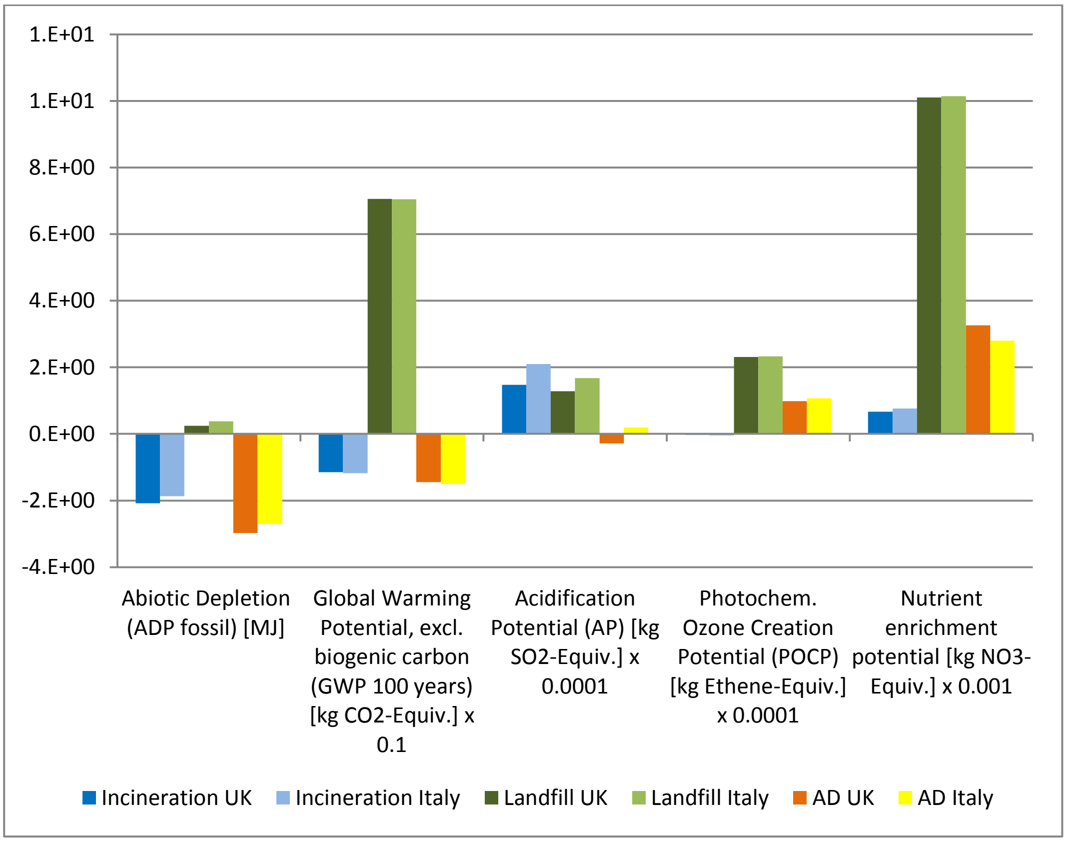


Figure 5.10. Comparison between the two geographical areas, considering as FU 1 kg of OFMSW.

5.3 Distributed Generation scenarios

This section is focused on the Distributed Generation scenarios presented in Figure 4.10, i.e. OFMSW exploitation via biogas production and energy recovery in the distributed generation paradigm (biogas and biomethane), with three micro CHP technologies, alternative use of natural gas to run the SOFC- micro CHP units and the reference scenario where electricity is supplied by the grid, heat is supplied via condensing boiler and OFMSW is treated in a landfill plant.

The burdens associated with the Waste treatment options (AD and Landfill plant) are taken from the results of the previous Section 5.2. The AD and Landfill units are considered as *aggregated processes* in the DG scenarios. Therefore no distinction between direct burdens – associated with the process - and indirect burdens – associated with transport and provision of energy and materials - is made for these processes, but all the emissions associated with the waste management treatment are considered as direct burdens.

The three micro-CHP technologies investigated are: 1.7 kWe Solid Oxide Fuel Cell, 30 kWe micro Gas Turbine and 1.2 kWe Stirling Engine. The analysis has been conducted considering three operating strategies for the design of the micro-CHP units: full thermal energy demand, half thermal energy demand and electrical demand for the dwellings.

All the results are presented per functional unit (as showed in Table 4.6), thus based on the input of 45,033 tons of OFMSW produced by the household and collected in the area and on the energy output of 444 GWh_e and 1,388 GWh_{th} for the UK scenarios which correspond to the total electricity and space heating demand for the same amount of household, and on 36,765 tons of OFMSW and on the energy output of 215 GWh_e and 501 GWh_{th} for the IT scenarios.

UK scenarios

This section presents the results for the case study based on the Royal Borough of Greenwich, UK.

As shown in Table 5.9, in order to meet the energy output target, natural gas has to be supplied to the micro CHP units because the biogas produced with the waste collected from the same dwellings is only the 1%-3% of the total fuel needed to satisfy their energy demand.

The approach followed to evaluate the environmental impacts of the scenarios analysed is the same used for the WM scenarios:

- *direct burdens* from the foreground system,
- plus *indirect burdens* from the background system,
- minus *avoided burdens* due to system expansion.

Table 5.9. Percentage of natural gas which has to be supplied to the micro CHP units in the UK scenarios (% of the total fuel supply to the system).

Scenarios	Full thermal	Half thermal	El
Biogas_FC	99%	97%	98%
Biomet_FC	99%	97%	98%
Biogas_MGT	99%	97%	98%
Biogas_SE	98%	97%	99%

The results are presented for each impact category, for the three operating strategies considered.

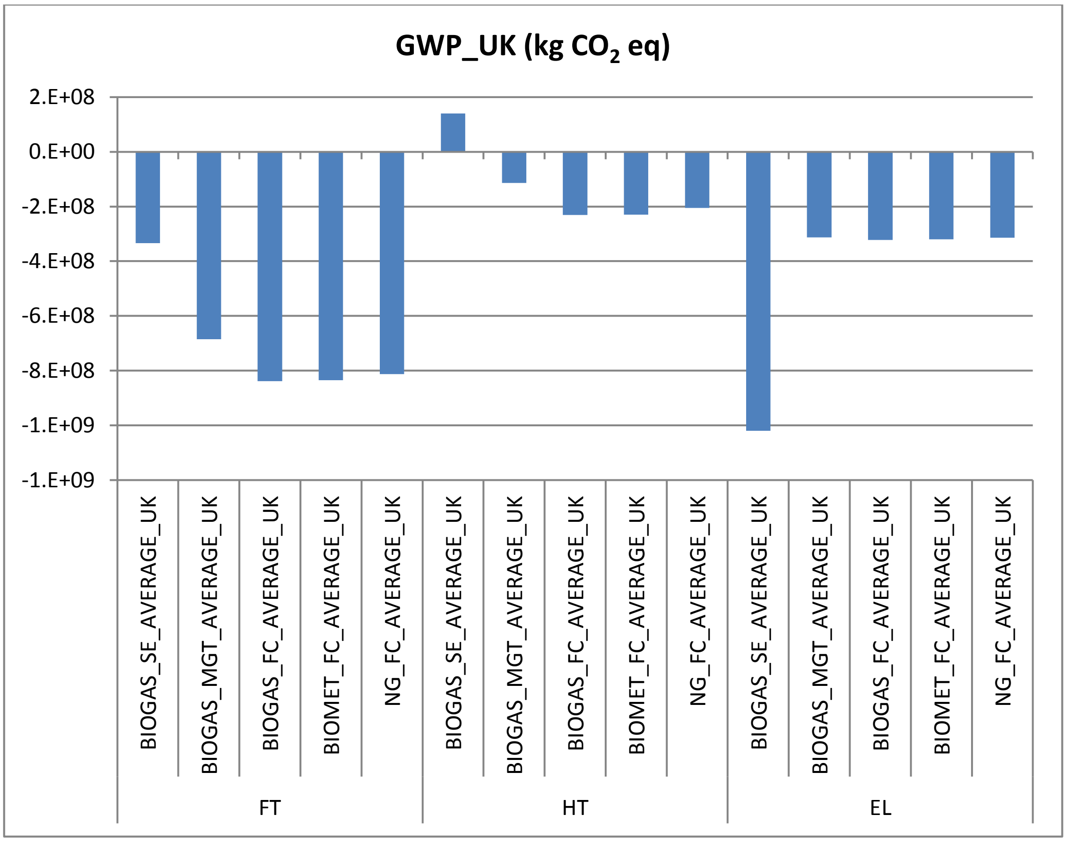


Figure 5.11. Global Warming Potential for the micro CHP scenarios with biogas/biomethane and Natural gas scenario, in the UK case study.

It is evident that all the scenarios – except one - represent a savings in terms of emissions compared with the reference scenario (here represented as avoided burdens) in all impact categories. The exception is the Stirling Engine scenario in the Half Thermal operating strategy which has a positive impact of 1.4 x 10^5 tons of CO_2 eq (as shown later in Table 5.11).

Considering the Full Thermal energy demand – operating strategy (shown as FT in Figure 5.11), the internal reforming SOFC micro-CHP unit fed with raw biogas shows the best behaviour in terms of GHG emissions. The total impacts are in fact: -8.39 x 10^5 tons of CO_2 eq for biogas+SOFC units, -8.35 x 10^5 tons of CO*2* eq for biomethane+SOFC units, -6.85 x 10^5 tons of CO*2* eq for micro gas turbine, -3.33 x 10^5 tons of CO_2 eq for Stirling engine, and -8.12 x 10^5 tons of CO_2 eq for natural gas+SOFC units. The impact due to the production of the equal amount of electricity from the grid is considered with a negative value, in order to quantify the avoided burdens related with the electricity surplus sold to the grid. The results reflect the different H to P ratios of the three units and the greatest contribution in the SOFC unit scenarios is represented by the surplus electricity produced when the unit is designed to follow the total thermal energy demand of the dwellings.

The results of the Half Thermal scenarios mirror the previous ones, except for the BIOGAS_SE scenario, as stated above. In the Electricity demand - operating strategy scenarios the BIOGAS_SE impact changes completely, being the most environmentally friendly solution. The greatest contribution is due to the surplus heat exported to the grid, assuming that a district heating grid is installed in the borough. This is, again, due to the H to P ratio of the Stirling Engine unit which allows a large production of heat when the unit is electricity-led.

Although the waste treatment assumed in the natural gas running SOFC-micro CHP scenario (NG_FC) is the landfill plant, the total impact is very similar to the one with AD (BIOGAS_FC and BIOME_FC). This is a consequence of what stated at the beginning of this section: the biogas produced via AD is between 1-3% of the total fuel needed by the micro-CHP and consequently the three scenarios differ only for this small amount of renewable fuel in terms of natural gas use.

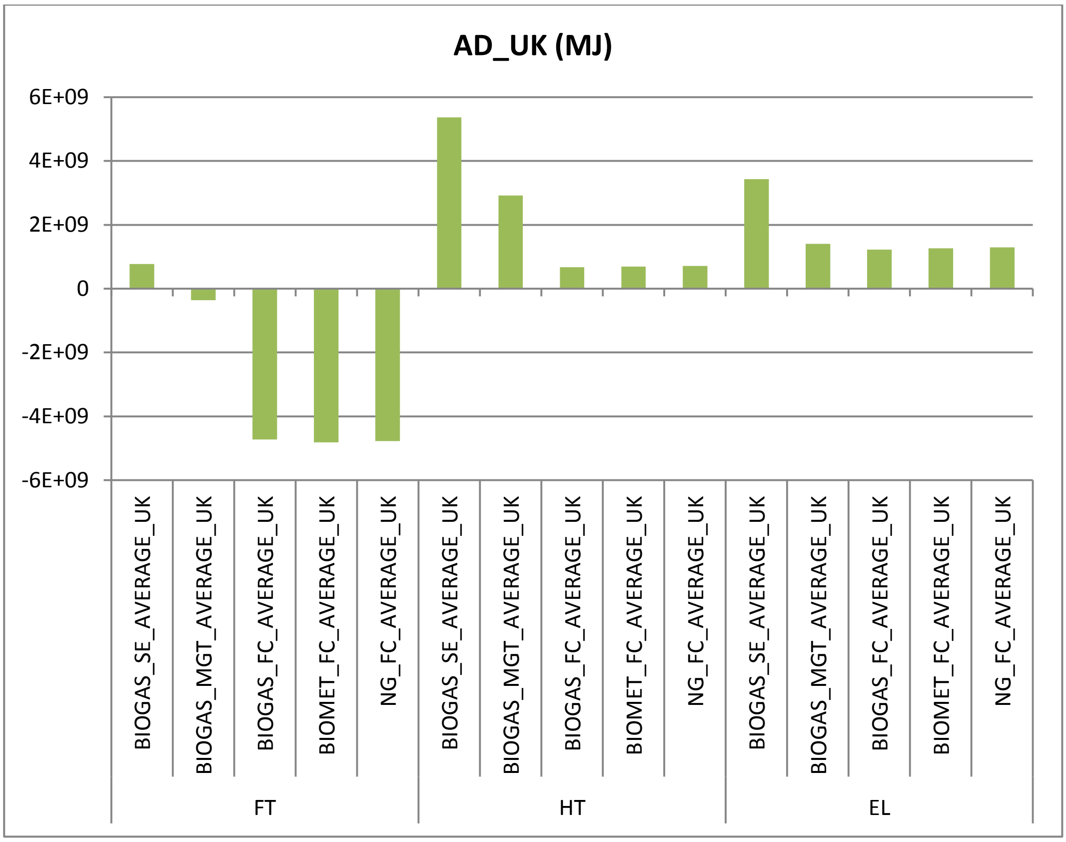


Figure 5.12. Abiotic Depletion potential for the micro CHP scenarios with biogas/biomethane and Natural gas scenarios, in the UK case study.

The results in Figure 5.12 for the FT operating strategy show that the SOFC unit scenarios represent the best solution in terms of fossil resources' use. In fact, it is possible to offset the use of fossil resources, thanks to the surplus electricity sold to the grid. Again the NG_FC scenarios show similar results to the BIOGAS/BIOM_FC ones, with a slightly higher value due to the increase in natural gas demand. In the HT and EL scenarios, instead, all the three technologies show a positive impact in terms of fossil resources depletion, despite the avoided burdens in term of energy production. This is mainly due to the natural gas needed by the micro-CHP units to fully satisfy the energy demands of the dwellings.

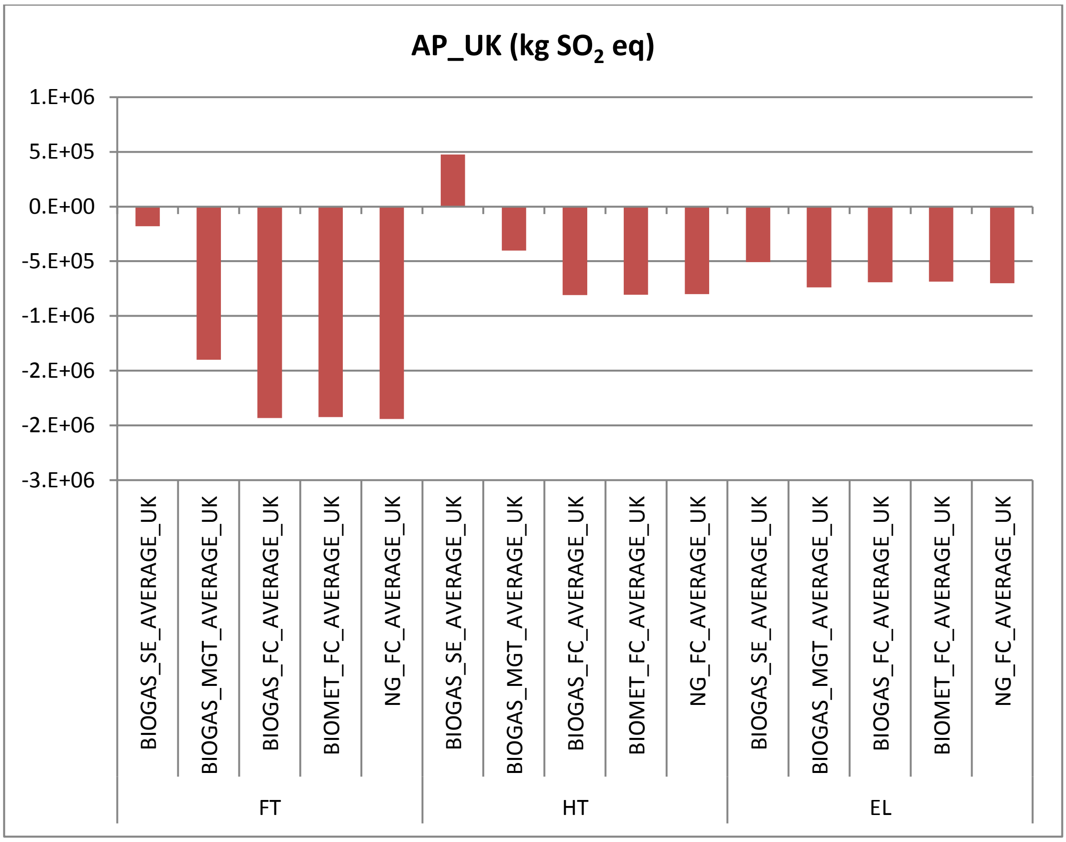


Figure 5.13. Acidification Potential for the micro CHP scenarios with biogas/biomethane and Natural gas scenarios, in the UK case study.

Figure 5.13 shows the results for the Acidification Potential impact categories. This category is very important at local scale, where the effects of acidification are most significant. For the FT and HT scenarios the ranking amongst the micro-CHP technologies is the same of the GWP, with the FC representing the best solution in terms of environmental impact. The things are different if we look at the EL scenarios. Here the most environmentally friendly solution is represented by the micro Gas Turbine. This is due to the different impact in terms of GWP and AP of the natural gas and the avoided burdens associated with electricity and thermal energy production. In fact, as discussed later, the micro Gas Turbine produces more thermal energy than the Fuel Cell when the micro-CHP units are electrically-led. This is again due to the H to P ratio. This is despite the fact that the Stirling Engine produces the biggest amount of thermal energy. In this particular impact category, the impact related with the micro-CHP unit emissions is then significant. Moreover, the saving in emissions associated with the avoided burdens in the BIOGAS_MGT scenario is bigger (with a negative value) than the one related with the indirect burdens (mainly natural gas supply) and direct burdens (mainly micro-CHP emissions), resulting in -7.40 x 10^3 tons of SO_2 eq in total.

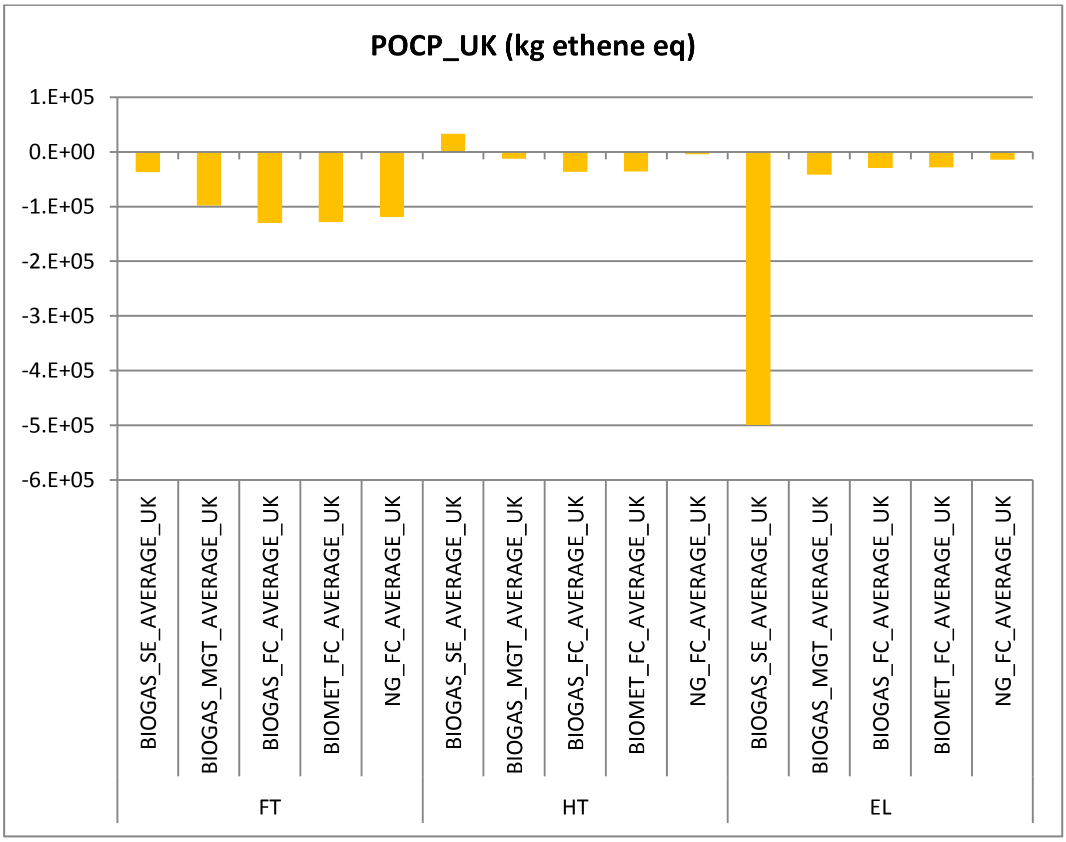


Figure 5.14. Photochemical Ozone Depletion Potential for the micro CHP scenarios with biogas/biomethane and Natural gas scenarios, in the UK case study.

The results for the POCP impact category are shown in Figure 5.14. The BIOGAS_FC appears to be the best scenario in the FT and HT operating strategies, while the Sterling Engine shows the lowest impact when the micro-CHP are electrically-led. All the micro-CHP scenarios have a negative impact, thus represent a savings in terms of emissions, apart for the BIOGAS_SE in the HT operating strategy. In general the ranking amongst the technologies is similar to the AP results. The bigger difference between the NG_FC and BIOGAS_FC is mainly due to the impact associated with the treatment of the organic waste in the landfill plant, which is one order of magnitude bigger than the AD.

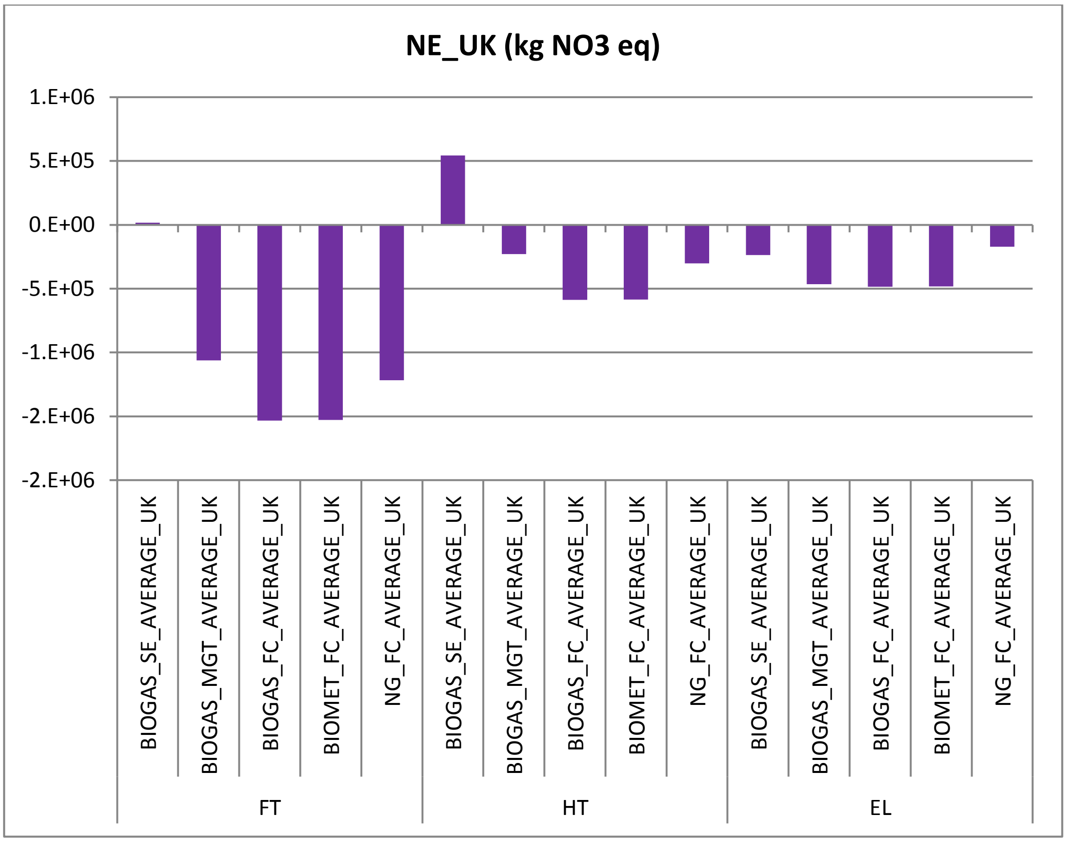


Figure 5.15. Nutrient Enrichment Potential for the micro CHP scenarios with biogas/biomethane and Natural gas scenarios, in the UK case study.

The Figure 5.15 represents the results for the Nutrient Enrichment potential impact category. The results again mirror the ranking obtained for the other impact categories. The main differences are: the positive impact of the BIOGAS_SE in the FT operating strategy and the relative bigger difference in this category respect to the previous one of the NG_FC scenarios compared with the BIGAS/BIOM_FC ones, due - again - to the impact associated with the landfill plant compared with the biological treatment.

IT scenarios

In this section, all the results obtained for the case study based on Livorno municipality (IT) are presented and discussed in comparison with the results of the UK case study.

Table 5.10 shows the percentage of natural gas which has to be supplied to the micro-CHP units in the IT scenarios. The amount is slightly lower compared with the UK case study – due to the amount of total energy demand of the dwellings that has to be satisfied - but still more than 94-98% of the total fuel needed.

Table 5.10. Percentage of natural gas which has to be supplied to the micro CHP units in the IT scenarios (% of the total fuel supply to the system).

Scenarios	Full thermal	Half thermal	El
Biogas_FC	97%	94%	95%
Biomet_FC	98%	96%	97%
Biogas_MGT	97%	94%	97%
Biogas_SE	96%	94%	98%

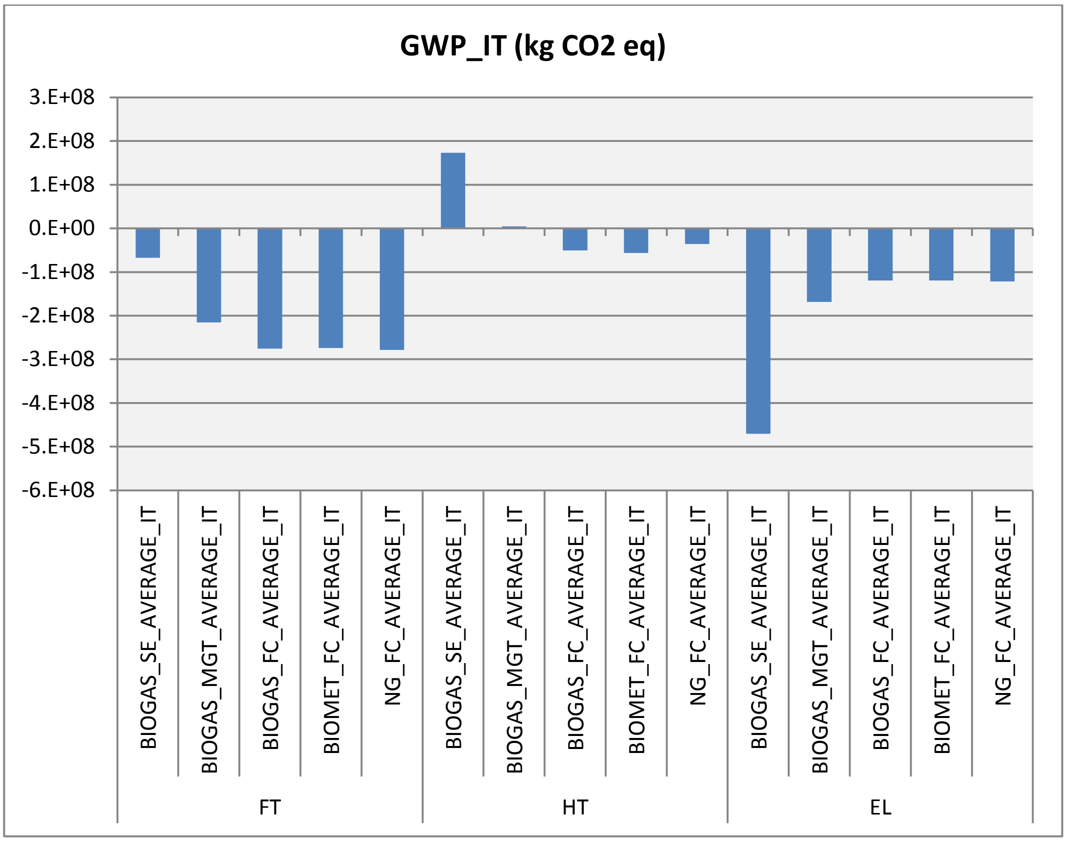


Figure 5.16. Global Warming Potential for the micro CHP scenarios with biogas/biomethane and Natural gas scenarios, in the IT case study.

The results shown in Figure 5.16 reflect the trend of the UK case study, with the BIOGAS_FC scenarios being the most environmentally friendly solution in the FT and HT scenarios, whereas in the EL operating strategy is the BIOGAS_SE that represents the best environmental solution. The only difference with the UK case is the positive impact of the BIOGAS_MGT scenario in the HT operating strategy. This is due to the different amount of energy demand that has to be satisfied in the two case studies that brings to a different amount of avoided burdens. This will be discussed later in the next section. However, all the scenarios represent a saving compared with the reference case, where electricity is supplied by the grid and space heating is supplied by the condensing boiler, except for the abovementioned mGT and SE scenarios in the HT operating strategy.

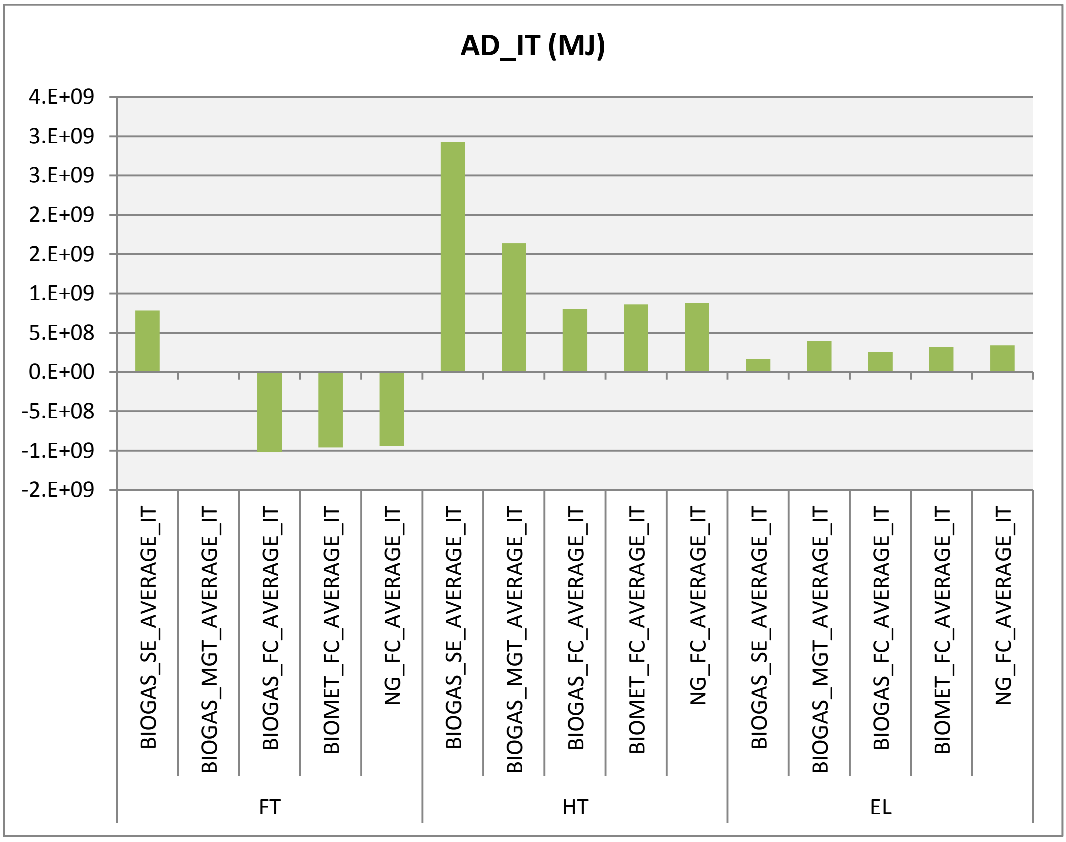


Figure 5.17. Abiotic Depletion Potential for the micro CHP scenarios with biogas/biomethane and Natural gas scenarios, in the IT case study.

Figure 5.17 shows the results for the Abiotic Depletion potential. Compared with the UK case study, the trend is very similar. The main difference is in the BIOGAS_SE scenario when is electrically - led. In the UK case study, in fact, this was the scenario with the biggest impact, while here is the one with the lowest. This is mainly due to the different energy demand of the two case studies related to the H to P ratio of the micro-CHP technologies considered.

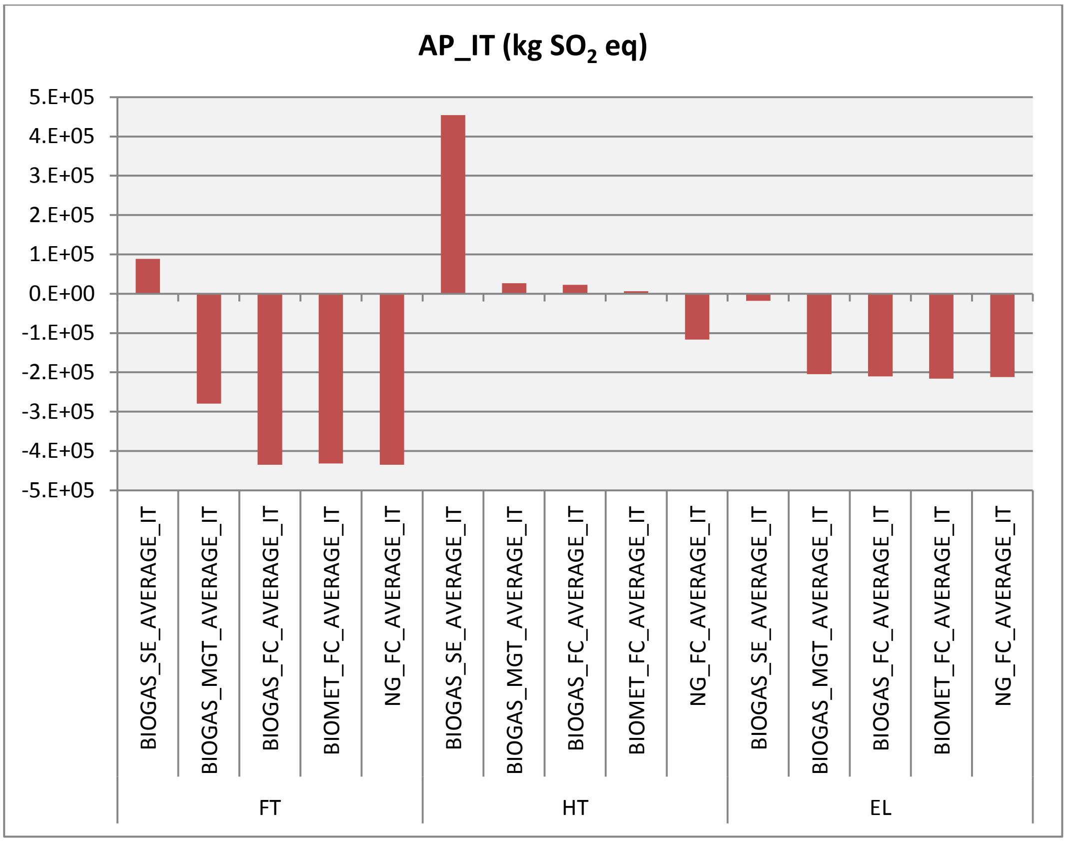


Figure 5.18. Acidification Potential for the micro CHP scenarios with biogas/biomethane and Natural gas scenarios, in the IT case study.

The Acidification Potential trend for the DG scenarios in the IT case study is similar to the UK ones for the FT and EL operating strategies, but it looks different for the HT one. The BIOGAS_SE scenario is still the one with the greatest impact. However the BIOGAS/BIOMET_FC has a positive impact here, while in the UK case study they showed a negative value. Moreover, the NG_FC is here the best environmentally friendly solution for the HT operating strategy, while in the UK case study it was the BIOGAS_FC scenario. This is mainly due to the different amount in terms of indirect and direct contributions in the two case studies - related mainly with the burdens associated with landfill and natural gas supplied - and the different H to P ratio of a typical dwellings in Italy and UK (as discussed later in the next section).

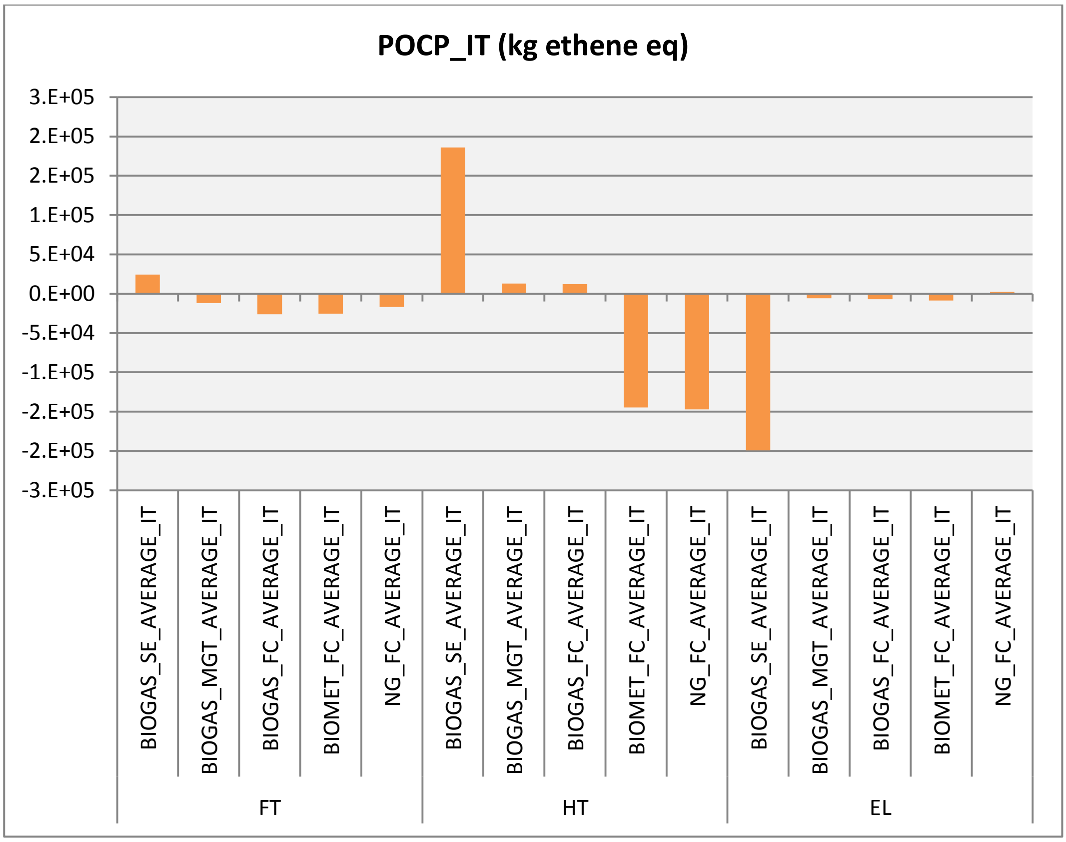


Figure 5.19. Photochemical Ozone Depletion Potential for the micro CHP scenarios with biogas/biomethane and Natural gas scenarios, in the IT case study.

The results for the Photochemical Ozone Depletion Potential impact category are shown in Figure 5.19. Again the trend is similar to the UK case study except for the HT operating strategy scenarios. In fact, here the most environmentally friendly solution is once more the NG_FC scenario, while in the UK case study was the BIOGAS_FC that shows a positive impact in the IT case. The reasons are the same as stated for the AP impact category. Another difference is the positive impact of the NG_FC and BIOGAS_SE scenarios in the EL and FT operating strategies, respectively, while it was negative in the UK case study for both.

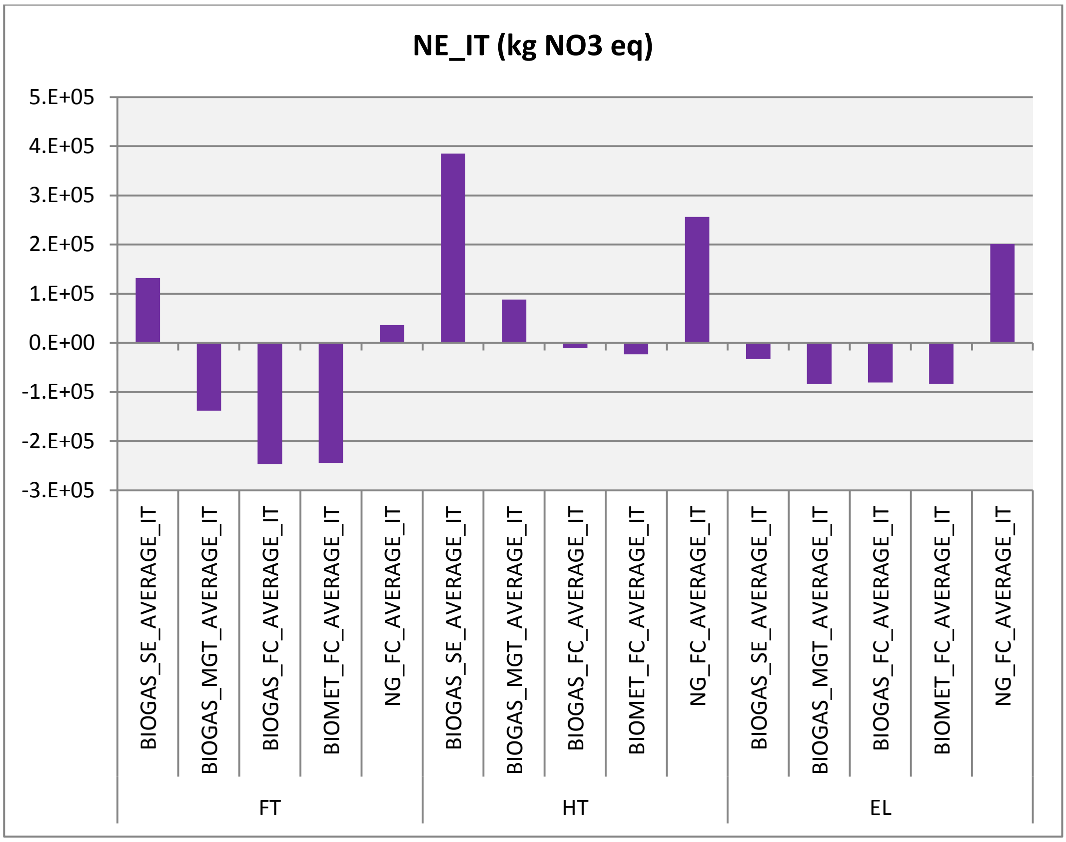


Figure 5.20. Nutrient Enrichment Potential for the micro CHP scenarios with biogas/biomethane and Natural gas scenarios, in the IT case study.

The results shown in Figure 5.20 are related to the Nutrient Enrichment potential for the IT case study. This mirrors the trend shown in the UK case in terms of ranking amongst the different technologies. However, in the IT case, the NG_FC scenarios turn to have a positive impact for all the three operating strategies investigated. The same is for the BIOGAS_MGT when is Half Thermally-led.

Comparison between the two geographical areas

In this section a comparison between the results obtained in the two case studies is performed.

As stated more than one time in the previous section, the differences in the results obtained for the two areas and for the three operating strategy are mainly due to the H to P ratio of the three micro-CHP technologies and a typical dwellings in UK and IT.

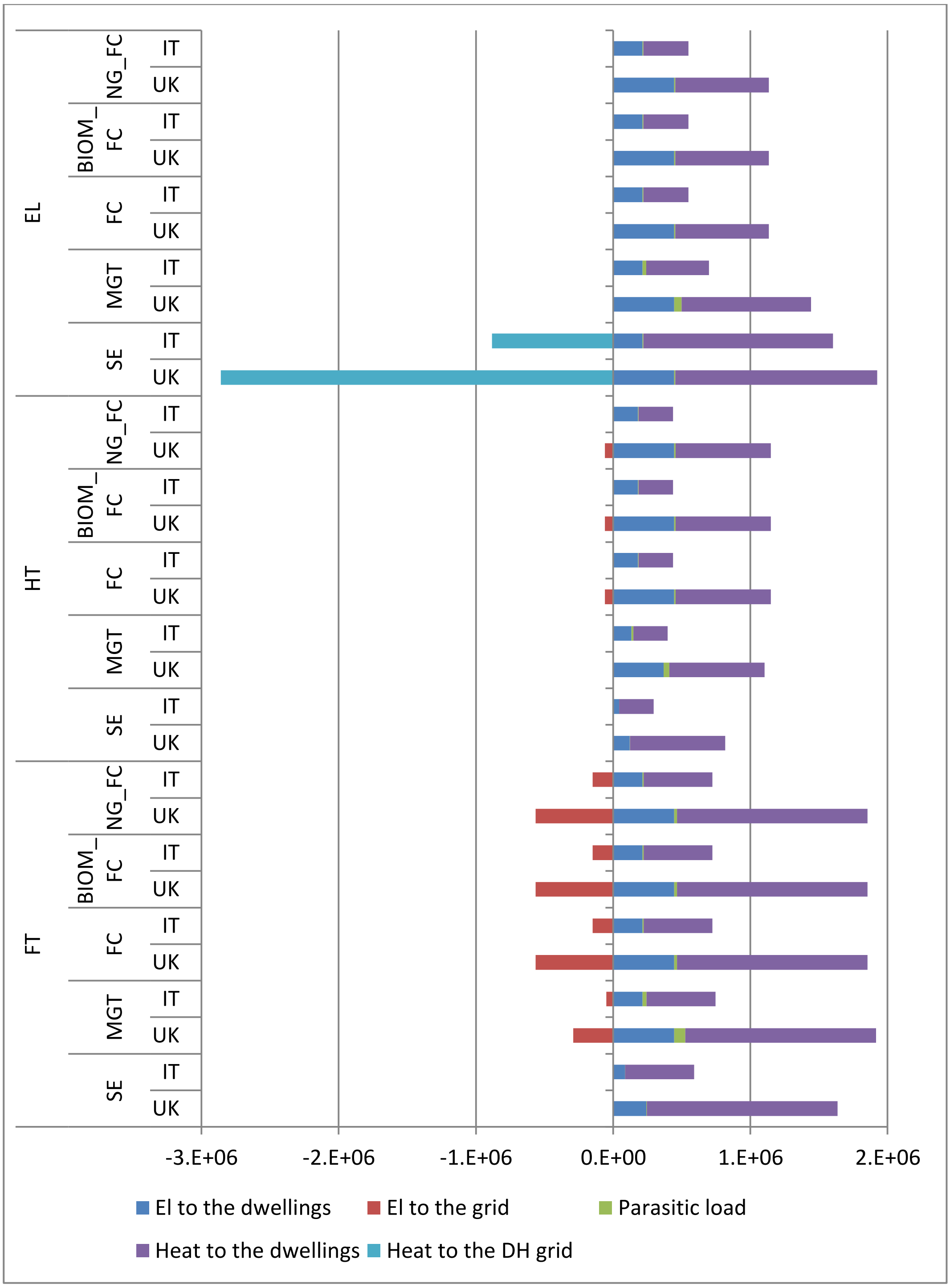


Figure 5.21. Energy balance of the three micro-CHP units for each operating strategy considered and for the two case studies (the values are referred to the overall energy demands for the total number of households and expressed in MWh per year).

Figure 5.21 shows the energy balance of the three micro-CHPs in each operating strategy considered, for the two cases. In the FT scenarios, the mGT and the FC produce in both the cases more electricity than the electrical demand of the dwellings, allowing an exporting of surplus electricity to the grid. Given the different energy demands in the two geographical areas, the quantity of energy exported is bigger in the UK case study and, as a consequence, the overall environmental impacts are different.

In the HT scenarios, only the fuel Cell units in the UK study produce more electricity than the amount demanded by the dwellings, allowing a large amount of avoided burdens in terms of electricity from the grid. That explains the difference in the HT scenarios in terms of environmental impacts highlighted in the previous section.

Finally, in the EL operating strategy scenarios, the Stirling engine units allow an exporting of thermal energy in both the case studies.

Moreover, from the Figure 5.21 is possible to highlight the importance of the parasitic loads in the three micro-CHP technologies. Only in the mGT scenarios they appear to be relevant – especially in the UK case, while for the others technologies the contribution is almost negligible.

Figure 5.22 shows a hot spot analysis for the GWP impact category only, referred to the FC units, in the full thermal demand operating strategy, for the two cases. As it is possible to observe, the contribution of the cleaning process of the biogas in the BIOM_FC scenarios is very limited and almost negligible in the IT case study. If we look at the direct and indirect burdens – the avoided burdens are due to the production of energy and exporting to the grid and they have been already discussed above – the main contributions arise from:

- the supply of natural gas to satisfy the energy demand of the dwellings, as assumed at the beginning of this study (see Section 4.10). This is almost the same in every scenarios considered in Figure 5.22. This means that the 'waste – to - energy closed loop' approach is not possible if only the Organic Fraction of Municipal Solid Waste is considered. However, the environmental benefits here are due to the use of natural gas as preferred fuel for micro cogeneration, enabling to replace other fuels such as heating oil in the heat market or coal in the electricity market;
- the electricity supplied to the dwelling in every scenario, equal to the 10% of the total energy produced by the micro CHP units and supplied to the dwelling, and assumed to cover the stopover of the micro-CHP systems;

- the landfill plant in the NG scenarios, considered as alternative Waste Management treatment compared with the Anaerobic Digestion. The impact associated with the landfill option is different in the two case studies. This is due to the different amount of organic waste treated in the specific area.

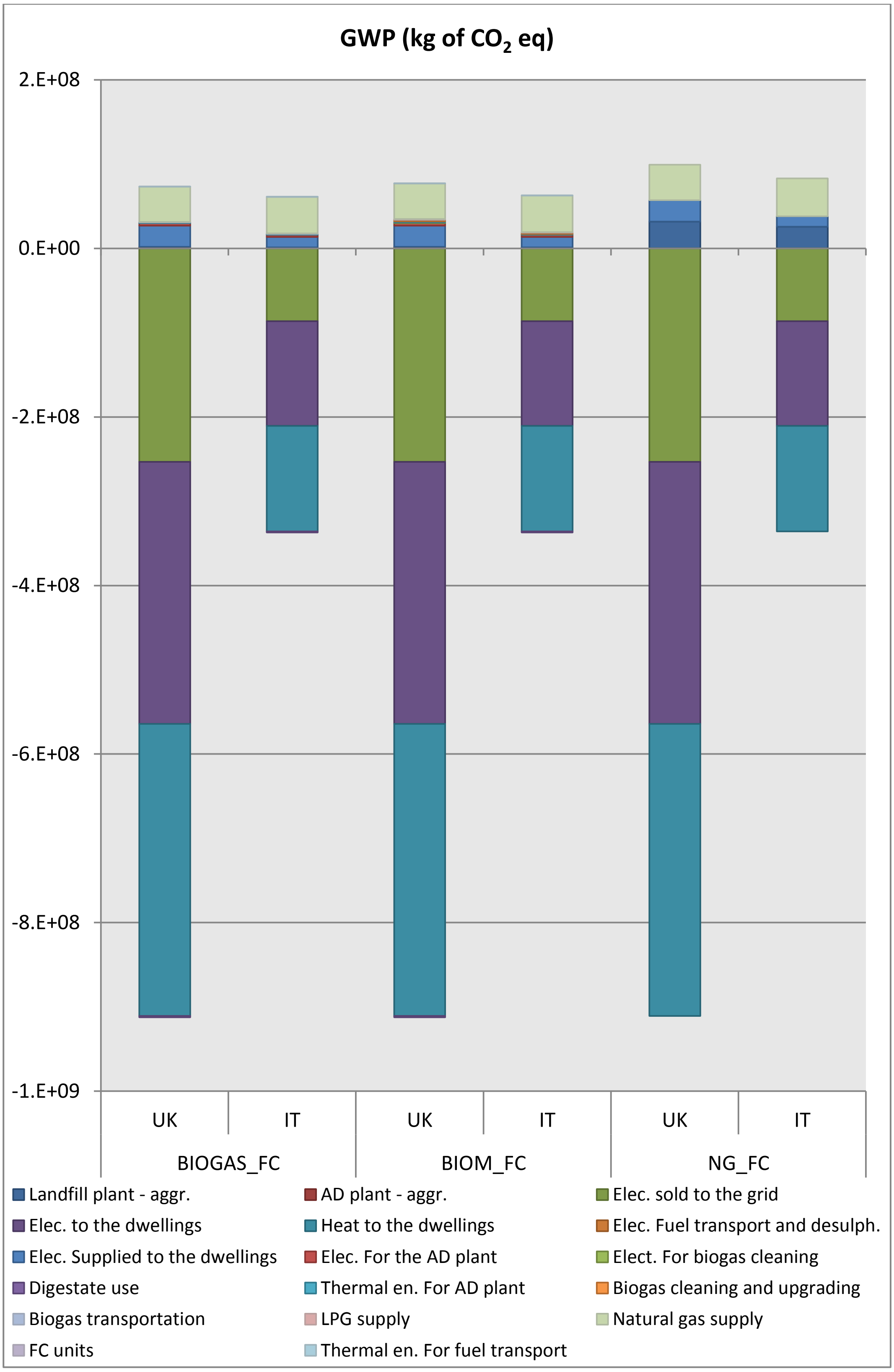


Figure 5.22. Hot spot analysis for the FC micro – CHP systems in the Full Thermal operating strategy scenario, for the two case studies.

Finally, Table 5.11 summarises the results of the environmental impacts analysis for each category considered for the DG scenarios. The best environmentally friendly solution for each category, each operating strategy and each country is highlighted in red in the table.

Table 5.11. Overall results for the DG scenarios.

		FT					HT					EL				
SCENARIOS		**BIOGAS_SE_AVERAGE**	**BIOGAS_MGT_AVERAGE**	**BIOGAS_FC_AVERAGE**	**BIOMET_FC_AVERAGE**	**NG_FC_AVERAGE**	**BIOGAS_SE_AVERAGE**	**BIOGAS_MGT_AVERAGE**	**BIOGAS_FC_AVERAGE**	**BIOMET_FC_AVERAGE**	**NG_FC_AVERAGE**	**BIOGAS_SE_AVERAGE**	**BIOGAS_MGT_AVERAGE**	**BIOGAS_FC_AVERAGE**	**BIOMET_FC_AVERAGE**	**NG_FC_AVERAGE**
IT	***NE (kg NO3 eq)***	1.32E+05	-1.38E+05	-2.47E+05	-2.44E+05	3.62E+04	3.85E+05	8.84E+04	-1.08E+04	-2.34E+04	2.56E+05	-3.29E+04	-8.37E+04	-8.08E+04	-8.30E+04	2.01E+05
	POCP (kg ethene eq)	2.43E+04	-1.19E+04	-2.61E+04	-2.54E+04	-1.67E+04	1.86E+05	1.31E+04	1.19E+04	-1.45E+05	-1.47E+05	-1.99E+05	-5.76E+03	-7.17E+03	-8.47E+03	2.13E+03
	AP (kg SO2 eq)	8.82E+04	-2.79E+05	-4.35E+05	-4.32E+05	-4.35E+05	4.54E+05	2.65E+04	2.24E+04	6.41E+03	-1.17E+05	-1.78E+04	-2.05E+05	-2.11E+05	-2.16E+05	-2.12E+05
	AD (MJ)	7.83E+08	-4.51E+06	-1.02E+09	-9.61E+08	-9.40E+08	2.93E+09	1.64E+09	7.98E+08	8.59E+08	8.80E+08	1.70E+08	3.94E+08	2.59E+08	3.17E+08	3.38E+08
	GWP (kg CO2 eq)	-6.74E+07	-2.16E+08	-2.76E+08	-2.74E+08	-2.78E+08	1.73E+08	4.62E+06	-5.07E+07	-5.61E+07	-3.58E+07	-4.70E+08	-1.68E+08	-1.19E+08	-1.19E+08	-1.22E+08
UK	***NE (kg NO3 eq)***	1.53E+04	-1.06E+06	-1.53E+06	-1.53E+06	-1.22E+06	5.43E+05	-2.30E+05	-5.87E+05	-5.87E+05	-3.03E+05	-2.36E+05	-4.65E+05	-4.86E+05	-4.83E+05	-1.72E+05
	POCP (kg ethene eq)	-3.68E+04	-9.78E+04	-1.30E+05	-1.29E+05	-1.19E+05	3.32E+04	-1.27E+04	-3.65E+04	-3.58E+04	-4.10E+03	-4.99E+05	-4.20E+04	-2.93E+04	-2.84E+04	-1.43E+04
	AP (kg SO2 eq)	-1.81E+05	-1.40E+06	-1.93E+06	-1.92E+06	-1.94E+06	4.76E+05	-4.02E+05	-8.07E+05	-8.06E+05	-7.98E+05	-5.06E+05	-7.40E+05	-6.90E+05	-6.86E+05	-7.01E+05
	AD (MJ)	7.76E+08	-3.58E+08	-4.73E+09	-4.82E+09	-4.77E+09	5.37E+09	2.92E+09	6.70E+08	6.92E+08	7.12E+08	3.43E+09	1.41E+09	1.23E+09	1.26E+09	1.29E+09
	GWP (kg CO2 eq)	-3.33E+08	-6.85E+08	-8.39E+08	-8.35E+08	-8.12E+08	1.40E+08	-1.14E+08	-2.31E+08	-2.29E+08	-2.05E+08	-1.02E+09	-3.12E+08	-3.23E+08	-3.20E+08	-3.14E+08

Figure 5.23 shows the results of the Global Warming Potential impact category when marginal technologies for the production of electricity are considered. Following the assumptions made in Section 5.2, the figure below shows the results for the UK case study: the marginal technologies considered are then Coal plant and CCGT plant. It is worth nothing that the same considerations could be made for the IT case. The fuel cell micro-CHP units are the most environmentally friendly solution even when the marginal electricity production technologies are considered as substitute to the energy produced by the foreground scenarios. This is mainly due to the surplus electricity produced with the SOFC and sold to the grid because of its high electrical efficiency. The same is for the other technologies in the three operating strategys considered. Things are different only for the SE-HT scenario, where the low electrical efficiency of the Stirling does not allow satisfying the electricity demand of the dwellings and almost half of it has to be supplied by the grid mix.

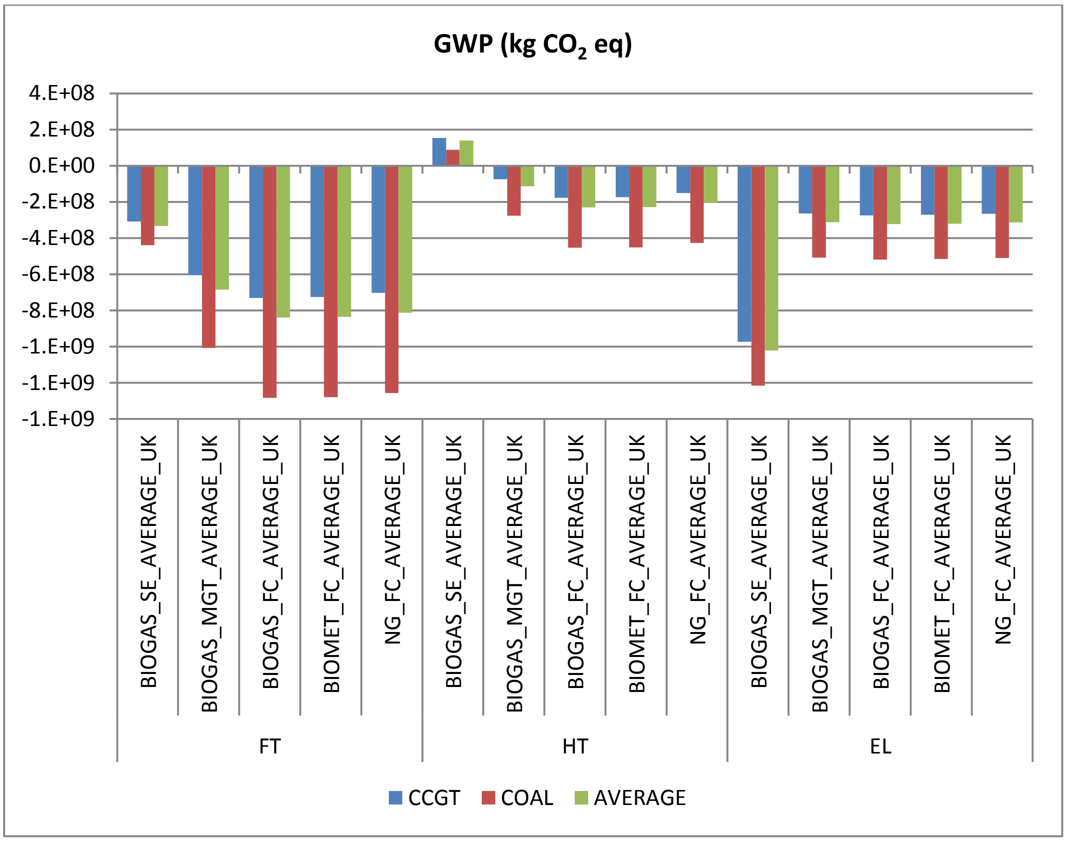


Figure 5.23. Global Warming Potential for the three micro CHP scenarios in the UK case study, comparing CCGT plant, coal plant and electricity from the grid as substituted technology.

5.4 Sensitivity analysis

Life cycle assessments are always associated with uncertainties, especially when data are not obtained directly from a specific plant but, as in this study, they represent an average situation (Clift, 2013). Cleary (2009) reviewed about 20 peer-reviewed papers which investigate the LCA of municipal solid waste management systems. He concluded that only four of them undertook sensitivity analysis, varying both parameters related to input data (i.e. MSW transportation distances, sorting efficiency, waste composition, etc.) and model parameters (such as waste treatment capacity, recovery rates of recyclable materials, etc.).

In this section, key parameters with potentially large impact on models for the AD and landfill technologies and on the overall results are investigated. The results are referred to the WM scenarios, and they are shown for the UK case study, but they would be the same for the IT one. The key parameters investigate are:

- Input data regarding the performance of the AD plant, where literature data show wide variation;
- Assumptions about degradation of organic material in landfills.

5.4.1 Anaerobic digestion plant

The variability of key parameters in the assessment of the environmental impacts of anaerobic digestion plant is discussed in several studies (Bernstad & La Cour Jansen, 2011; Patterson, Esteves, Dinsdale, *et al.*, 2011; Fruergaard & Astrup, 2011). In this work, four parameters are investigated to assess their influence on the final results:

a. Methane losses occurring during the production of biogas;

b. Efficiency of the Internal Combustion Engine running with biogas (CHP unit);

c. Emissions from digestate use;

d. Carbon sequestration in the digestate.

Methane losses

Fugitive emissions of methane can occur throughout the AD plant, from pipes, valves, over-pressure of the system, and the storage facilities for waste and biogas. It is very difficult to estimate these emissions due to their variability from one site to another (Møller, Boldrin &

Christensen, 2009). Here, an average value of 2% is initially assumed, as suggested by Dalemo et al. (1997), Berglund et al. (2006), Fruergaard et al. (2011) and Boldrin et al. (2011). In this section, a range of different values are analysed to assess the influence on the overall GWP; see Table 5.12.

Table 5.12. Literature values for methane losses.

References	*Methane losses range of values (% of the methane within the biogas produced)*	*Values used in the sensitivity analysis*	*Scenarios name in the sensitivity analysis*
Møller et al, 2009	0-3%	1%	a1
Patterson et al, 2011	3-5%	5%	a2
Borjesson and Berglund, 2006	1-15%	15%	a3

In the sensitivity analysis, three different values for methane losses are considered, expressed as percentages of the total methane in the biogas:

- 1% (scenario *a1*);
- 5% (scenario *a2*),
- and 15% (scenario *a3*).

Figure 5.24 shows the results only for GWP and POCP, since methane losses affect only these two indicators amongst the impacts considered in this work.

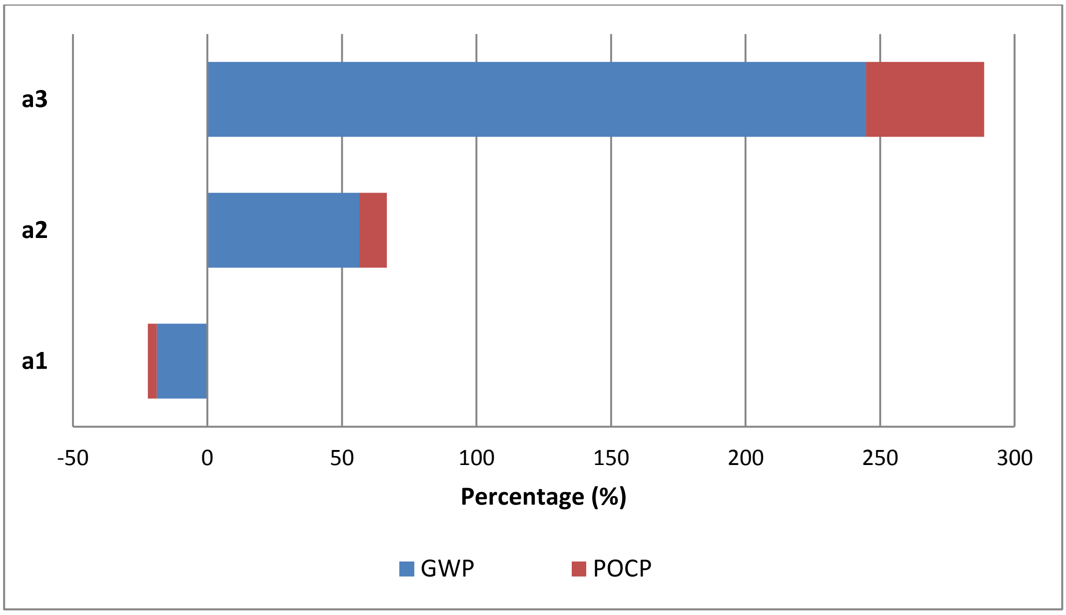


Figure 5.24. Effect of methane losses on environmental impacts: scenarios a1, a2 and a3, compared to baseline case (Section 5.2).

Methane losses during the formation of biogas contribute to the total GWP shown in Figure 5.24, based on 2% losses of biogas from the AD plant, as assumed in Section 4.11.6. Halving this value (*Scenario a1*) results in a reduction of the total CO_2 equivalent by 20%. On the other hand, increasing the percentage of methane losses to 5% (*scenario a2*) increases the GWP by more than 55%, while if the losses are increased to 15% (*scenario a3*) the GWP changes from a net reduction to a net contribution to GWP (more than double in absolute value). This clearly shows the importance of emission monitoring and controlling biogas production, since they have a great impact on the overall environmental performance of a biogas production system.

The POCP is affected less: considering *scenario a3*, where the methane losses are more than seven times the base case, the total ethane eq increases by 45% when compared to the basic scenarios.

CHP unit efficiency

The efficiency of internal combustion engines running on biogas can vary significantly. This study assumes an electrical efficiency of 32% and a thermal efficiency of 50% as baseline parameters, as described in Section 4.11.9. Table 5.13 shows the data used in similar studies. Sensitivity analysis is conducted on this parameter, considering values ranging from 30 to 41% and 45 to 60% for electrical and thermal efficiency respectively. Two different scenarios are then considered: *scenario b1*, with an electrical efficiency of 30% and thermal efficiency of 60%; and *scenario b2* with 41% electrical efficiency and 45% thermal efficiency. The results are shown in Figure 5.25.

Table 5.13. Summary of assumptions by other authors on CHP unit efficiency.

Parameter	**References**						
	This Study (Patterson et al., 2011)	Møller et al., 2009	Fruergaards et al., 2011[2]	Dalemo et al., 1996[1]	Boldrin et al., 2001	Polsch et al., 2010	Bernstad et al., 2011
CHP unit Electrical efficiency (%)	32	36	41	30	39	40	36
CHP unit Thermal Efficiency (%)	50	44	45	60	46	48	49

[1]Reference for scenario b1; [2]Reference for scenario b2.

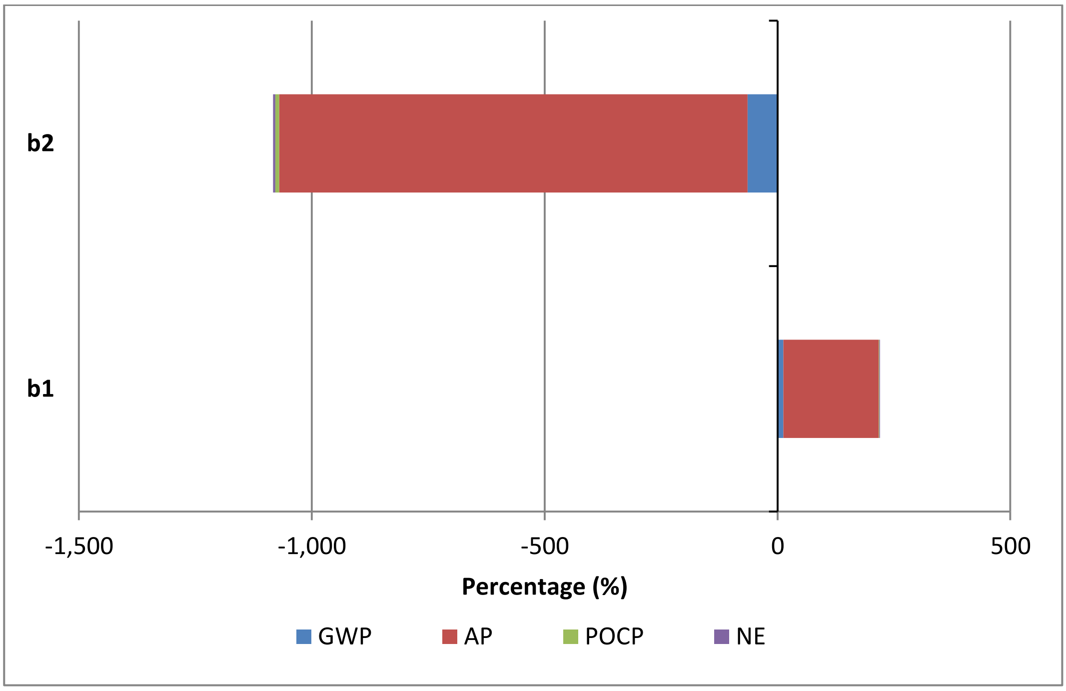


Figure 5.25. Effects of varying the electrical and thermal efficiency of the CHP unit: scenarios b1 and b2 compared to baseline case (Section 5.2).

In *scenario b1*, the total GWP increases by 12% compared with the baseline but the strongest change is in the AP (total SO_2 eq). On the other hand, when the electrical efficiency increases by 30% and the thermal one decreases by 10%, as in *scenario b2*, the GWP is 60% less while the total SO_2 eq is ten times smaller. It is important to note that these results depend significantly on the type of technologies used to substitute the energy produced. It is worth emphasizing that, for a CHP plant, the environmental impact of electricity production is more significant than heat generation, as shown by the comparison between *scenarios b1* and *b2*. These highlight that the burdens related to the production of 1 MJ of electricity (taken as the average electricity mix) are greater than the environmental impacts of 1 MJ of heat produced (condensing boiler fuelled by natural gas); therefore the reduction in impact relative to the other waste treatment routes is larger for *scenario b2*.

Emissions from digestate use

As stated above, emissions associated with the use and spreading of the digestate are subject to great variability, due to soil and weather conditions, spreading practice and crop types. Moreover, these emissions are related to the parameters used to model the behaviour of nutrients in the digestate, especially the extent to which nitrogen compounds avoid the use of commercial fertilisers. The sensitivity analysis assesses the influence of the division of nitrogen between N-org, NH_4 and NO_3 (Bernstad & la Cour Jansen, 2011); the range considered is summarised in Table 4.14, see *scenario c1*. These values can affect acidification potential and nutrient enrichment, as the main variation is in NO_3 leaching that contributes mainly to the enrichment of pollutants in the water (see *scenario c1* in Figure 5.26).

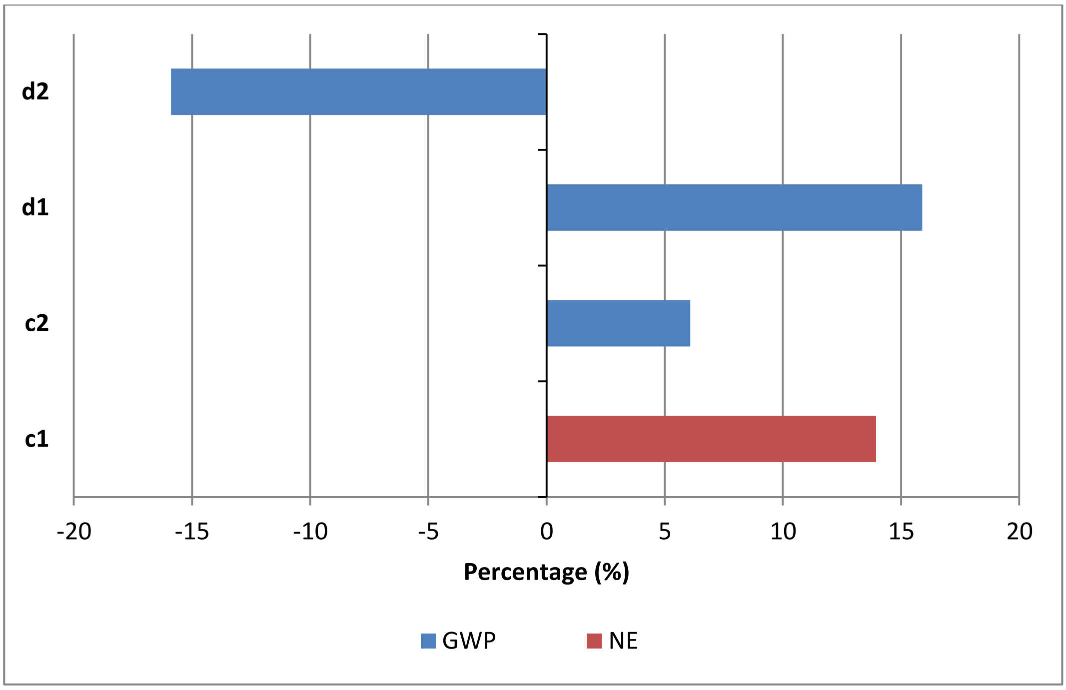


Figure 5.26. Alteration of GWP and NE impacts respect to model parameters: nutrient leaching into water (c1); emissions from inorganic fertilizer production (c2); carbon binding: 0% (d1) and 20% (d2).

Furthermore, to test the robustness of the model, the environmental impact of production of commercial fertilisers was assessed in *scenario c2* by comparing the results obtained using the values from Møller et al., (2009) with values suggested by DEFRA (Hill et al., 2011 from William et al., 2011) from a UK study which reports 6.8 kg CO_2 eq, 1.2 kg CO_2 eq and 0.5 kg CO_2 eq for the production of 1 kg of, respectively, N, P and K commercial fertiliser (*scenario c2* in Figure 5.26).

It is worth nothing that *scenario c1* affects only NE potential, due to an increase of NO_3 leaching to water. From the results in Figure 5.26, doubling the percentage of total N converted to leachable NO_3 increases the total NO_3 eq. by 13%. On the other hand, the model proposed by Hill et al., (2011) suggests that the burdens associated with production of commercial fertiliser have only a trivial effect on the GHG balance.

Carbon sequestration factor

Gentil et al., (2010) reviewed a number of LCA models developed for the anaerobic digestion process. Most of the models available in the literature do not consider a sequestration or 'binding' factor which describes the proportion of the carbon in the digestate which is retained in the soil. Values for the carbon binding factor can vary between 0 and 20% (Dalemo et al. 1997) this affects the GWP values but not the other impacts. In this work, two scenarios are considered where the carbon binding factor is taken as 0% (d1) and 20% (d2). From the results shown in Figure 5.26, the GWP in scenarios d1 and d2 differs by +/-15% of the total CO_2 eq.

5.5 Landfill models

Landfilling technologies have developed strongly during the last few decades and new technologies have been introduced, although not in every part of the world. Landfills range from dumps to highly engineered facilities as bioreactor landfills, flushing-bioreactor landfills and semi-aerobic landfills (Manfredi & Christensen 2009). The engineered landfills may have a range of landfill gas utilization and control systems leading to dramatically reduced emissions of methane and recovery of energy. These technologies adopt active measures to enhance the waste degradation process, in order to make it faster and more efficient. This leads to high gas generation rates early in the life of the landfill (higher than experienced in conventional landfills), which makes it more valuable to ensure an efficient gas collection system and undertake gas utilization schemes, such as electricity or combined heat and power (CHP) generation (Manfredi, Tonini, Christensen, *et al.*, 2009).

In UK the majority of the landfill plant are engineered landfill, thus they are supplied with a gas recovery system to produce mainly electricity. Despite this, it is difficult to define a specific value for the efficiency of the landfill gas recovery system in order to represent an average engineered landfill plant. The estimates of the efficiency of the LFG recovery system for closed parts of a landfill vary widely and in practice figures can be found from 10% to more than 90%. The extraction efficiency depends on several factors, such as well-spacing, attention of the landfill owner to the system (control of suction pressure on wells), design-capacity of the extraction system and utilization and the type and thickness of the cover. It also makes a difference whether a landfill gas project is designed and operated to extract a renewable energy source or whether the minimization of emissions is the objective (Hogg, Ballinger & Oonk, 2011). Nowadays, landfill plant permit conditions require to the operators to aim for 85% collection efficiency for cells or areas served by gas collection systems and the requirements to design and operate landfills to minimise gas escape have strengthened considerably since the 1990s. Overall, it is believed that a 75% collection efficiency for methane as an average over the gas-producing life of modern landfills is feasible, given industry and regulator experience, but further measurements are being pursued to improve confidence in this key factor (Brown, Cardenas, MacCarthy, *et al.*, 2012).

In Italy the amount of methane recovery in landfills has increased as a result of the implementation of the European Directive on the landfill of waste (European Parliament, 1999). The amounts of methane recovered and flared have been estimated in the *Italian Greenhouse Gases Inventory of 2011*, elaborated by ISPRA. It took into account the amount of energy produced, the energy efficiency of the methane recovered, the capitation efficiency and the efficiency in recovering methane for energy purposes assuming that the rest of methane captured is flared. The amount of recovered methane increased from 60% of the total, in 1998, to 70% since 2002 (ISPRA, 2011).

Given the strong uncertainties related with the environmental impact of landfill plant, different models from the literature are investigated to select the key factors in a landfill model.

The environmental impacts associated with landfill processes have been widely studied (Manfredi & Christensen, 2009; Barlaz, 1998; Perugini, Mastellone & Arena, 2003; Jambeck, Weitz, Solo-Gabriele, *et al.*, 2007; Gentil, Damgaard, Hauschild, *et al.*, 2010). However, the behaviour of material in landfills remains the most uncertain part of the LCA model, due to uncertainties associated with the emissions produced over time and the challenge of modelling the decomposition of different kind of materials (Gentil et al. 2010). Every landfill is a 'world in itself', dependent on the quality and quantity of the material landfilled as well as the geological and climatic conditions.

Different materials degrade at different rates and the final residues consist of non-degraded materials, usually with high carbon content. This carbon storage would not normally occur under natural conditions (virtually all of the biodegradable material would degraded to CO_2, completely the photosynthesis/respiration cycle), the landfill is modelled as a *carbon sink*, i.e. as a net sequester of carbon (Hogg, Favoino, Nielsen, *et al.*, 2002; Manfredi, Tonini, Christensen, *et al.*, 2009; Manfredi & Christensen, 2009; US EPA, 2006).

Carbon dioxide is produce in the initial aerobic stage and in the anaerobic acid stage of decomposition. However there is a luck of data to quantify the emissions during the aerobic stage, and they are in generally assumed to be a small proportion of total organic carbon inputs (less than 1%). Then the methanogenic stage starts, and landfill gas is generated, with a composition of approximately 50% CO_2 and 50% CH_4. The collected landfill gas has a higher CH_4 content, because some of the CO_2 is dissolved in the leachate (US EPA, 2006; Jambeck, Weitz, Solo-Gabriele, *et al.*, 2007).

In mass and energy balance terms, carbon storage can be determined as the carbon that remains in the landfill after accounting for the carbon in the landfill gas and the carbon in the leachate. Based on the results of Barlaz 1998, organic fraction of municipal solid waste contains about 50% of cellulose, 7% of hemicellulose, and 10% of lignin. While the degradation of cellulose and hemicellulose is well documented, lignin does not degraded so a significant extent in anaerobic degradation. Landfill stores part of the cellulose and hemicellulose carbon and all the carbon of the lignin that is in the waste initially. The amount of carbon stored varies with some environmental conditions of the landfill, such as pH and moisture contents.

The carbon sequestration in landfill is calculated in a different way in the LCA models applied to Waste Management systems. As noted by Gentil et al. (2009), it is calculated '*in EASEWASTE as the difference between the total amount of biogenic carbon entering the landfill site and the biogenic carbon emitted over a 100 years horizon. According to Manfredi and Christensen (2009), the calculated amount of sequestrated carbon is about 50% of the total incoming carbon. A distinction is made between biogenic and fossil carbon sequestration in term of the contribution to global warming potential* (Christensen, Gentil, Boldrin, *et al.*, 2009)*. Biogenic carbon is attributed a beneficial impact, while the sequestration of fossil carbon has no impact, nor benefits on climate change. About 50% carbon sequestration is also assumed in WRATE but no specific LCIA characterisation factor has been included for carbon sequestration. Other substances are also assumed to be sequestrated in the landfill due to various vitrification and fossilisation processes in WRATE. In the EPIC/CSR model, a sequestration factor is applied for the paper based waste only. This sequestrated carbon is removed from the carbon cycle and therefore subtracted from the inventories. LCA–IWM have specifically excluded carbon sequestration potential from the landfill model but included carbon sequestration in the compost module through the fixation of carbon in the com- post. In WISARD, carbon sequestration is currently not calculated. This may be reviewed depending on discussions at national level'.*

Landfill technologies have been developing in recent years and different types of sites can be found around the world. The plant assumed here is representative of UK practice: i.e. an engineered landfill with energy recovery and emission control (Brown, Cardenas, MacCarthy, *et al.*, 2012; Yassin, Lettieri, Simons, *et al.*, 2009). Four different landfill models have been compared with the baseline scenario described in Section 4.11.4 and Section 5.2 to show the dependency of the overall results on key landfill model parameters. Only the impact related to GHG is evaluated.

The details of the models are given in Table 5.14. The key parameters distinguishing the models are:

- carbon sequestration factor greater than zero;
- efficiency of landfill gas collection system; i.e. the proportion of gas collected over the time frame considered;
- oxidation factor; i.e. the fraction of methane not collected but oxidized in the surface layers of the landfill;
- proportion of initial carbon in waste which is converted to CH_4.

The analysis is carried out considering the UK case study only. According to the Annual report on the *UK Greenhouse Gases Inventory*, the UK average gas collection efficiency and oxidation factor are 75% and 10% respectively (Brown, Cardenas, MacCarthy, *et al.*, 2012). Then scenario *L4* is the most representative for this study.

Figure 5.27 shows the results of the four scenarios in terms of kg of CO_2 eq emitted per ton of organic waste. The total GHG emissions almost double between scenarios L1 and L2 compared with L3 and L4, due to the different oxidation factors. Increases of 20% and 30% in the energy efficiency in scenarios L3 and L4, corresponding to increases in both landfill gas collection efficiency and power plant efficiency, lead to improvements of 180% and 230% in avoided emissions when compared against scenarios L1 and L2.

Table 5.14. Summary of the assumptions for landfill scenario models analysed in the sensitivity analysis.

Scenario	Oxidation factor	Yield of CH_4 as % of initial carbon (in wet waste)	Biogenic carbon stored (% of total wet waste)	Landfill Gas Collection Efficiency	Power Plant efficiency	References
L1	40%	34%	4%	50%	95%	(Manfredi, 2009; Manfredi, Tonini, Christensen, *et al.*, 2009)
L2	40%	37%	5%	50%	95%	(Manfredi, Tonini & Christensen, 2010)
L3	10%	42%	2.4%	75%	85%	(Barlaz 1998; US EPA 2006; Themelis & Ulloa 2007)
L4	10%	42%	2.4%	75%	100%	(Barlaz 1998; US EPA 2006; Themelis & Ulloa 2007)

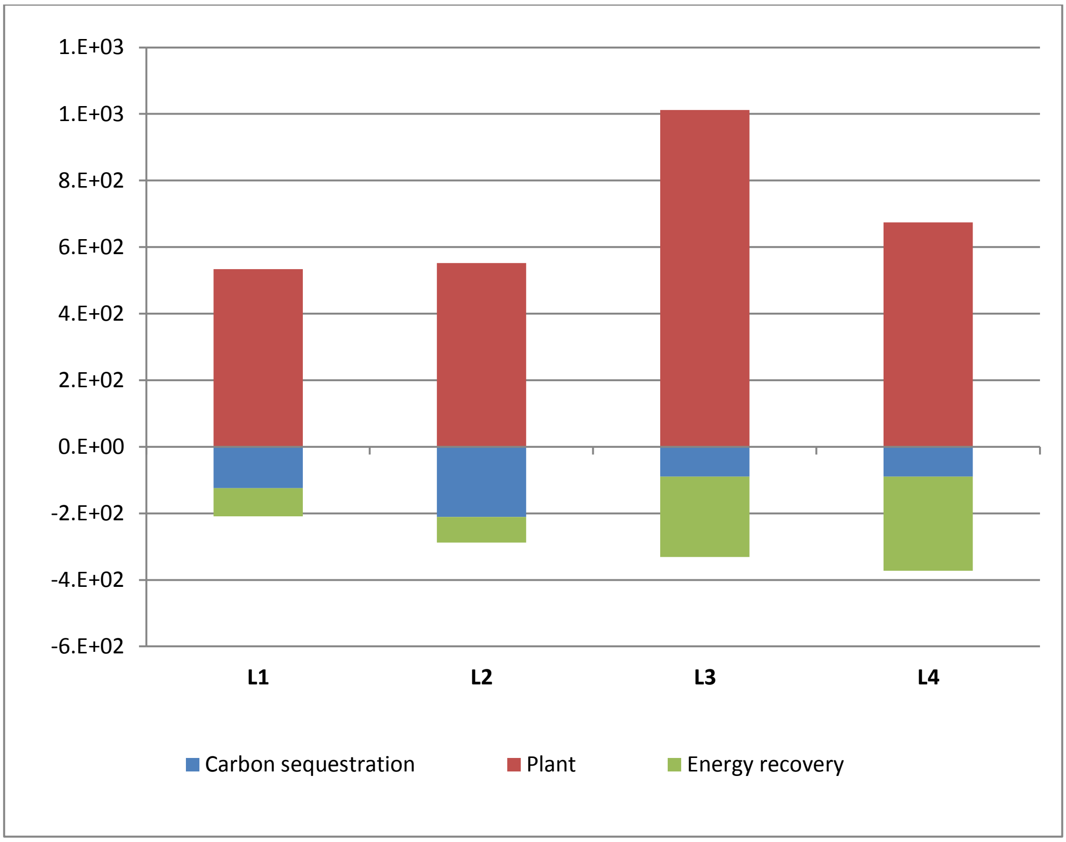


Figure 5.27. Global Warming Potential in kg of CO_2 eq per ton of Organic Waste for 4 different landfill scenarios (L1, L2, L3 and L4).

Comparing the results from the sensitivity analysis with those in section 5.2, the landfill treatment does not appear to have such a high environmental burden as might be expected. The results in Figure 5.27, and that in Figure 5.3 referring to the baseline landfill model, highlight that the comparison between different options is sensitive to the detailed assumptions in the models.

Table 5.15 summarises the results. Note that the value of 888 kg CO_2 eq per ton of OFMSW obtained for the baseline landfill scenario (see Figure 5.3) includes the burdens associated with the transport of waste, while in Figure 5.27 this is not considered. However, as expected (Gentil et al., 2010) the influence of the transportation distance is negligible on the final results. The environmental impact of the landfill plant varies between 302.55 to 888 kg CO_2 eq per kg of organic waste treated. This means a variation of -67% compared to the impact calculated as baseline.

Table 5.15. Summary of the results obtained for the landfill models in the sensitivity analysis.

	Baseline	L1	L2	L3	L4
GWP (kg CO_2 eq/ton OFMSW)	888	324.79	264.58	680.91	302.55

The results in Figure 5.27 are a simplification of the reality. In fact, the methane is not generating constantly during the 100 years period. The rate at which a specific material decays in landfill, under anaerobic condition, influences the landfill collection efficiency. Studies of La Cruz, de and Barlaz (2010) showed that different materials degrade at different rates. The rate at which methane emissions are generated from decaying material in a landfill depends mainly upon the type of material placed in the landfill and the moisture conditions of the landfill (La Cruz, de & Barlaz 2010).

Consequently, the landfill gas is not produced constantly. Usually only a small percentage of the gas soon after the waste is buried is collected, while almost all of the gas produced is collected once a final cover is installed. Figure 5.28 shows the landfill gas generation and extraction over a 100 years period. In the comparison, an average value over the timeframe is considered for the collection of landfill gas.

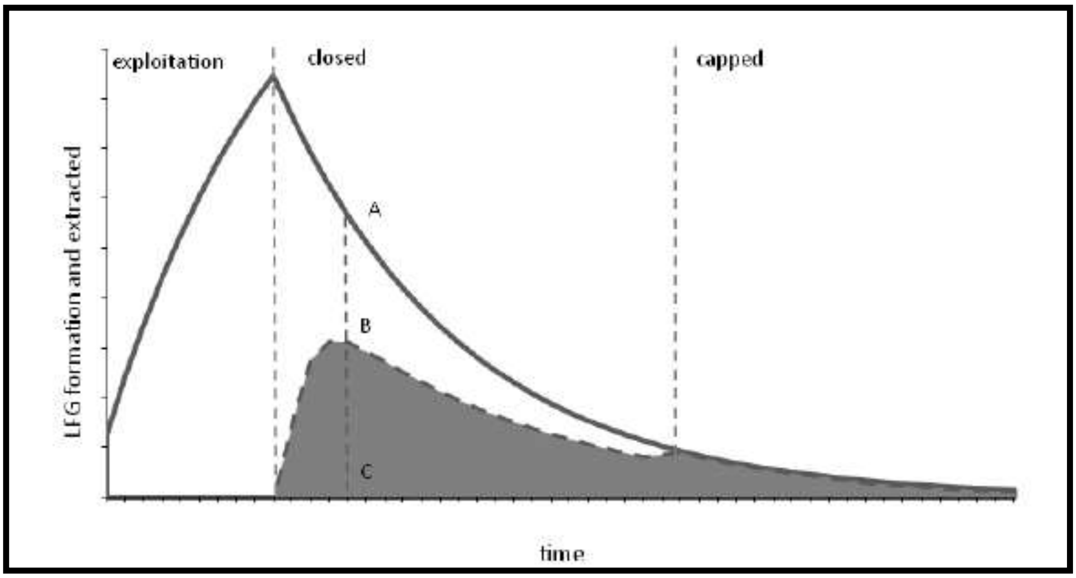


Figure 5.28. Landfill gas generation and extraction in time for a typical landfill (Hogg, Ballinger & Oonk, 2011)

6. Discussion

6.1 Introduction

In this section a critical analysis of the results is carried out, following the division of the two sub-systems presented in Chapter 4 and Chapter 5: Waste Management scenarios and Distributed Generation scenarios. The results are compared with the ones from similar studies and the main differences in the approach are highlighted.

6.2 Waste Management scenarios

Anaerobic digestion is investigated as a foreground process, representing a relative new treatment option for the organic fraction, strongly recommended by the UK and IT governments (see Sections 2.1.4 and 2.1.5). As reported by Baere and Mattheewus (2012), in the last two decades of waste management technologies development, the introduction of anaerobic digestion represents one of the most successful and innovative treatment despite the fact that major progresses were made in all areas of waste management. In particular, the biological treatment in anaerobic condition has become the first preferred and proven method for the treatment of biodegradable phase of the Organic Fraction of Municipal Solid Waste. Notwithstanding other alternative treatment technologies, such as gasification, pyrolysis, plasma, biological drying, have reached continued progresses, anaerobic digestion has reached a widespread implementation difficult to overtake. Baere and Mattheewus (2012) reported 244 installations of anaerobic digestion plants in Europe (already built or to be constructed by 2014) dealing with the Organic Fraction of Municipal Solid Waste as the main feedstock[19]. The cumulative capacity of all these plants is estimated to be 7,750,000 tons of Organic Fraction of MSW per year. If we assume 300 kg[20] of organic fraction of MSW generated per person in one year, this capacity represents

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[19] In (Baere & Mattheeuws, 2012) the authors pointed out that this number does not consider the plants not in operation. However the estimates of the plants which have to be constructed is low and compensate the total number.

[20] The authors considered the Municipal Solid Waste generation in the EU-27 to be around 520 kg/capita and they assumed that about 60 % of that waste is organic (Baere & Mattheeuws, 2012).

about the 5% of the total biodegradable waste generated in Europe by 550 million inhabitants (considering EU-27).

The primary aim of this part of the work (indicated as WM scenario) has been to compare three different alternatives for the treatment of OFMSW introduced in Section 4.2: landfill with gas recovery for electricity generation, incineration with energy recovery by Combined Heat and Power (CHP) and anaerobic digestion with CHP and organic fertilizer production, for both the case study presented in Section 4.4. The objective of this part has been then to evaluate and compare the environmental impacts of these alternatives.

Critical analysis of the results of the LCA and sensitivity analysis for the WM scenarios have shown that none of the scenarios investigated represents the best solution in terms of waste treatment for all the impact categories considered. In the WM scenarios only CHP production from biogas has been investigated; other energy recovery options have been then analysed in the DG scenarios. The sensitivity analysis has finally shown the robustness of the model with respect to some key parameters.

Fugitive emissions of methane during the production of biogas are highly uncertain; they have a large impact on the environmental contribution of anaerobic digestion and can influence the final ranking of the different treatment options, to the point of making the biological process appear worse than incineration in terms of GWP (see Section 5.2 and Section 5.4.1). As reported by Møller et al. 2009, the fugitive losses of methane are difficult to establish by measurements and probably highly variable from facility to facility. The IPCC gives ranges between 0 and 10% of the produced methane, but also states that '*where technical standards for biogas plants ensure that unintentional CH_4 emissions are flared, CH_4 emissions are likely to be close to zero*' (IPCC, 2006). In the sensitivity analysis in Section 5.4.1 a range of values between 0 and 15% have been taken into consideration, showing that in the worst case the GWP increase by 2.5 times, assuming a positive (thus with negative effects) impact.

Section 5.2 and section 5.4.1 show that the most important factor overall is the quantity and quality of energy produced and substituted. As a first assumption, an average electricity mix is considered as the technology substituted by electricity from biogas; in this case, a renewable energy source (organic waste) replaces 100% of the mix of renewable and non-renewable sources. This is clearly a simplification of reality. In a dynamic system such as the energy system, an increase in energy demand will likely affect both base-load and peak-load production technologies (Fruergaard, Astrup & Ekvall, 2009). When anaerobic digestion is compared against two well proven technologies, landfill and incineration in this study, it results in savings in total GHG emissions due to the energy produced and the use of the digestate as an organic fertilizer. However, the results in Section 5.4 show that different assumptions on the marginal technology for electricity production can change the ranking amongst the different process options. Uncertainties in defining the correct marginal technology depend not only on technology developments but also on future policy decisions, i.e. whereas (when and how) renewable energy targets have to be achieved. This can change the future electricity scenario to compare which the waste to energy technologies (Finnveden 2008). The time horizon is another important factor for the identification of the marginal technology, as extensively reported by several authors (Fruergaard et al. 2009; Fruergaard 2010; Fruergaard & Astrup 2011). Fruergaard referred to a *short-term perspective* (5-10 years) and a *long-term perspective* to identify the most suitable marginal technology. She considered that in a *short–term perspective* the capital investments are generally not expected to be affected and the production capacities are not altered. She defined, therefore, the marginal short-term technology as '*an existing technology capable of responding to a change in demand by adjusting its output'*. On the other hand, in a long-term perspective, capital investments can be expected to be affected, thus she defined the long term marginal technology as '*the affected production capacity (that is, whether or not new facilities are built or old plants decommissioned)'*. Following this approach, in this study the marginal production technology has been defined in a '*short-term*' perspective and without considering a dynamic modelling of the energy system.

The results obtained for the foreground process are consistent with other existed studies, as shown in Figure 6.1. First of all it is necessary to point out that, for the author's knowledge, the majority of the existed studies found in the literature refer to north European countries' case study, rather than Italy or UK. Few studies that investigate AD process from a Life Cycle Assessment perspective have been found to be settled in UK (Patterson, Esteves, Dinsdale, *et al.*, 2011), and even less in Italy (Cherubini & Strømman, 2011). Bernstad et al., (2011) showed that anaerobic digestion with energy recovery and the use of digestate as a substitute for commercial fertiliser result in greater avoidance of global warming and photochemical ozone when compared with composting or incineration, although the net contributions to nutrient enrichment and acidification potential are increased. By contrast, Fruergaard and Astrup (2011) concluded that incineration with energy recovery is a more beneficial option than anaerobic digestion for organic waste. The difference is attributable to the fact that their analysis is based on Danish conditions; with a different energy production system compared to the UK and IT with lower carbon intensity, and it does not consider the burdens avoided by inorganic fertilizer substitution.

Figure 6.2 shows a comparison of the results obtained in this study in term of Nutrient Enrichment potential with other existed studies. The value estimated by Bernstad et al., (2011) for nutrient enrichment impact category presents a higher impact compared with the results of this study, possibly due to the different parameters used in modelling the released of nitrogen to soil, air and water when digestate is spread on agricultural soil. I have already stated in Section 4.11.7 the complexity of modelling the emissions associated with the use of digestate produced by anaerobic digestion. Here the values used in the model are taken from literature sources and specific project results (Hogg, Gibbs, Favoino, *et al.*, 2007). For the author knowledge there is a general lack of information, especially for the Italian case study, about the application of national legislation in this field. This is mainly due to the relatively new process of application of digestate as fertiliser substitution in the two countries.

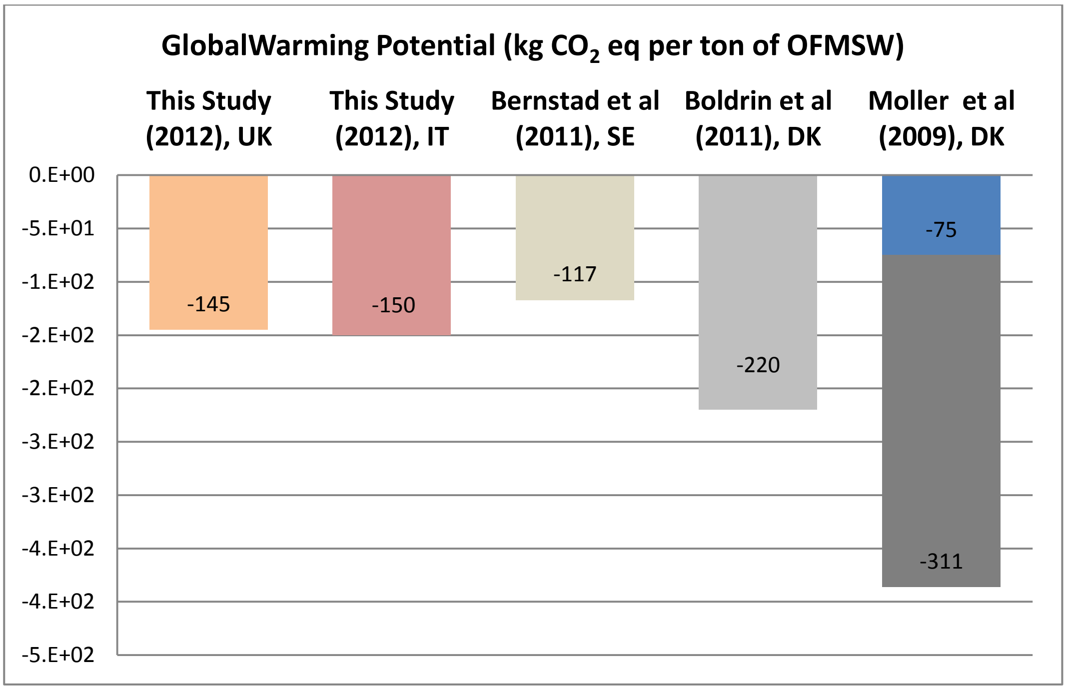

Figure 6.1. Results of the Global Warming Potential impact categories obtained in this study and compared with existed studies in the literature. The values are referred per ton of OFMSW.

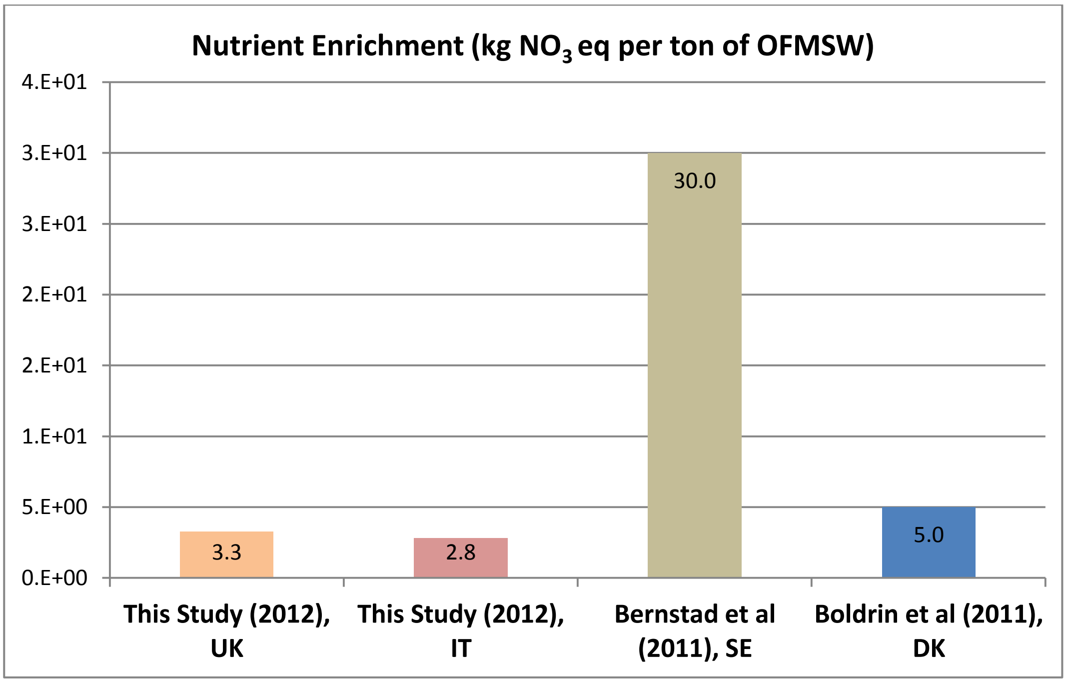

Figure 6.2. Results of the Nutrient Enrichment Potential impact categories obtained in this study and compared with existed studies in the literature. The values are referred per ton of OFMSW.

The models used to determine the environmental impacts of the landfill with electricity recovery and the incineration plant with energy production are aggregated processes based on the PE database (PE International). They are representative of typical European plants and not specific for the two countries analysed. In order to test the robustness of the model respect to this assumption, four different models for the landfill plant have been analysed in Section 5.5. The results have demonstrated that decomposition factors, LFG collection efficiency and oxidation are uncertain but crucial parameters whose values significantly affect the results. According to current estimates, about 69% of the methane generated in all UK landfills is recovered; this includes old sites without gas collection (Brown, Cardenas, MacCarthy, *et al.*, 2012). The methane that remains in the landfill is available for oxidation. The IPCC suggests that, in the absence of site-specific data, 10% of the total methane remaining in the landfill - after gas collection - should be treated as oxidised, and then not considered in the account of GWP. However this could be a pessimistic assumption, following results reported in the literature (Brown et al. 2012). The Environmental Protection Agency of the United State has conducted several studies on this issue, thanks to the precious contribution of Dr Barlaz (Milke & Barlaz, 2012.; Levis & Barlaz, 2012.; Barlaz, 1998). As highlighted in the EPA's report '*Landfilling*' (2012), the most important weakness point in the analysis of the GHG emissions and storage associated with landfilling is that most of the results are based on a single set of laboratory experiments conducted by Dr Barlaz. The results obtained in the sensitivity analysis (see Figure 5.27) show a range of values between 265 and 888 kg CO_2 eq per ton of OFMSW sent to landfill, for the Global Warming Potential impact category. This means variability up to 67% compared with the impact obtained in the baseline scenario. Given that the landfill plant is assumed to be the waste treatment technology in the reference scenario in the Distributed Generation sub-system, the assumption made on the landfill model can have a strong influence even on the DG scenarios results.

Although the GHG emissions associated with the landfill plant and calculated in the present work range from 265 to 888 kg of CO_2 eq per kg of organic waste, they are less than half the values suggested by DEFRA for the UK case study (DEFRA, 2011). In part, this is because the values used by DEFRA do not account for electricity production and therefore do not allow for the avoided burdens.

The analysis presented here shows the environmental benefit of increasing CH_4 recovery from landfill, although this is opposed to the present Waste Management strategy which aims to divert OFMSW from landfill that is the main contributor to CH_4 production. As noted by Clarke & Alibardi (2010), the investments in anaerobic digestion infrastructure in Europe are mainly driven, at present, by landfill directives that incentives the diversion of the biodegradable waste from landfill. At the base of these incentives are technologies to measure and capture the fugitive emissions from landfill plants. As noted by the two authors in the editorial note of the Volume 30, 2010, of the Waste Management Journal, '*reliable direct measurement technologies are urgently needed to verify emission estimation models. The first order decay model for emissions currently approved by the Inter-governmental Panel on Climate Change provides estimates of waste decay and concomitant methane production that typically exceed recorded gas capture rates and allowances for methane oxidation in soil covers. The potential emission liabilities for landfill owners, managers and designers under the IPCC model can therefore be significant*'.

Concluding this section on the critical review of the results obtained for the WM scenarios, a comment is needed on the carbon sequestration factor assumed in this thesis for the AD model. As reported by Gentil et al. (2010) in the review of LCA models dealing with waste management systems, these differ in the modelling of the biological process (n.d. anaerobic digestion) by the assumptions made on the biogenic carbon sequestration. In the sensitivity analysis (see Section 5.4.1) the carbon sequestration factor varies between 0 and 20% showing a variability of the GWP impact category of +/- 15% respect to the baseline (which considered 10% as carbon binding factor). This is in line with the EASEWASTE model, one of the most popular model in the literature for the Life cycle assessment of waste management systems – which assumes a factor between 10 and 15%, depending on the soil type, over a 100 year period (Boldrin, Neidel, Damgaard, *et al.*, 2011; Manfredi & Christensen, 2009; Kirkeby, Birgisdottir, Bhander, *et al.*, 2007).

6.3 Distributed Generation scenarios

This section analyses the results obtained for the Distributed Generation scenarios. The goal of this part of the study has been to evaluate the environmental impact of the DG scenarios comprising micro - CHP systems fed by biogas produced by OFMSW, to supply energy for a group of dwellings, in the distributed generation paradigm. The system has been designed to create a *waste – to - energy closed loop*, where the total amount of food waste produced at residential level in the borough/city is used to satisfy the total energy demand of the dwellings in the same area. This has been compared, through the system expansion, to conventional processes - *reference scenario*, where the energy is supplied through conventional technologies and the waste is treated in a landfill plant.

Distributed Generation could play an important role in the future energy systems, through effective reduction of greenhouse gas emissions from energy supply and possible diversification of primary sources of energy where alternative fuels are utilised. As noted by (Hawkes & Leach 2009) '*there are a wide variety of benefits of Distributed Generation, relating to each of the three pillars of energy policy; economics, environment and security*'.

Anlysing the results obtained in Section 5.3 it is possible to underline some critical aspects, due to uncertainties related to the technologies investigated, the assumption on thermal efficiencies, the methodological approach, the quality and quantity of the energy that is substituted.

Technologies investigated. All the technologies investigated are in their very early stage of being marketable products (micro Gas Turbine), or still in development (Fuel Cell and Stirling Engine). Especially for the fuel cells, their features could change rapidly in the next years and the efficiency could be higher than the one considered in this study. An important point is the efficiency of every system when fuelled by biogas. All the three manufacturers of the systems considered here state no change in efficiency parameters when the micro-CHP unit is coupled with biogas. That can be guaranteed for the Stirling engine, due to its nature of external combustion engine, but experimental results have demonstrated that there is a reduction for SOFC and micro GT efficiencies.

Furthermore the emissions due to micro-CHP systems are still under evaluation, given that few field trials have been carried out to determine the environmental impact of micro cogeneration systems when installed in a real dwelling. The main problem is related to the emissions of NO_x. In the past few decades, emissions of NO_x have generally decreased significantly, particularly due to measures taken regarding power plants and vehicles. However, all the exhaust emissions from micro-CHP systems are of particular importance in an environmental protection perspective. Nitrogen oxides are mainly formed during the combustion process (which is present in all the three technologies considered[21]) and contribute to many environmental impact categories. In this study the same levels of NO_x emissions have been assumed for the Stirling engine and micro Gas Turbine (0.08 g NO_x/Nm^3), while a lower value (0.003 g NO_x/Nm^3) has been assumed for the Fuel Cell. Pehnt (2008) observed a range of NO_x emissions for the Stirling engine of 0.020-0.110 g NO_x/Nm^3. Stirling engines can, in fact, lead to a very low level of emissions, due to their continuous combustion and the possibility to apply modern burner technology. The Fuel cells as well can lead to a very low impact, due to small contribution of the combustion process involved in the after burner, where the unused fuel – already clean from impurities - is burned.

The micro Gas Turbine size is larger in terms of power unit (30 kWe), compared to the other two (1.7 kW for the Fuel cell and 1.2 kW for the Stirling Engine). This is due to technology limitation of the gas turbine itself that allows commercial units only above 30 kWe. The problem here is overcome considering the installation of a micro - grid that allows supplying electricity to more than one dwelling, with one micro Gas Turbine units.

Thermal efficiency. It depends strongly on the application context, operating temperatures, and thermal cycle's limitation. The recovered of the heat is, theoretically, possible in every cogeneration system. Practically, it is more an economic evaluation than an environmental consideration. This is especially true in the FC scenarios where the waste heat is diffusively produced and it is more difficult to collect compared with the Stirling Engine or the micro Gas Turbine. Here a constant operation of the micro-CHPs is assumed. As shown by Staffell et al. (2011) actual demonstration of fuel cells in domestic buildings have shown that their operation is highly intermittent with utilisations of around 50%, and that efficiencies change with operating power and age of the system.

Methodological aspects. The system approach followed in the study has been the '*waste - to -energy closed loop*': thus the amount of OFMSW generated in a specific area has been considered as feedstock to produce a biofuel, used to run micro-CHP systems that generate energy to cover the energy demand of the households previously considered. From the author's knowledge, no other study in the literature has investigated the same system from

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[21] The combustion process is different for the three technologies considered. For the fuel cell, in particular, the combustion occurs in the afterburner, after the production of electricity.

a life cycle perspective. The results have showed that the biogas produced with only OFMSW generated locally is not enough to cover the fuel demand of the micro-CHP units and to finally supply the energy to the dwellings. In order to do that, natural gas has to be added and fossil resources are then utilized in a more sustainable manner: the micro CHP units show in fact a lower impact compared with the reference scenario. The amount of biogas/biomethane produced by the 102,000 households living in the Borough of Greenwich provides only 2% of the energy demand (total OFMSW produced equal to 45,033 tons/year). Based on this, the amount of organic waste needed to supply the total energy demand to the whole Borough would be 2,040 ktons per year. If we considered a large AD plant size of 120 ktons per year of OFMSW, 120 AD plants would be needed to satisfy the biogas demand. Hence, further investigation is required to explore alternative organic waste streams which can be made available in the Borough. A real possibility to increase the quantity of organic waste exists, considering MSW and food manufacturers waste treated in a single plant. The *Holsworthy* biogas plant in Devon already does this, being one of the first largest AD plant in UK, treating 100,000 tons of organic waste per year (Lukehurst, 2009; Monnet, 2003; Fisher, Collins, Aumonier, *et al.*, 2006).

Despite the solution investigated in this study does not result feasible considering only OFMSW, the possibility for micro-CHP applications fuelled by bioenergy resources do not end there. As reported by the JDS Associated report (JDS Associates, 2011), as bioenergy resources become more ample, micro-CHP using biomass or biogas would be a *'unique technology among domestic micro generation systems in offering the advantage of simultaneously generating carbon neutral electricity and heat*', potentially providing, by 2050, 15 GWe per year in UK.

Energy substitution. The electricity sold to the grid and supply to the dwelling is assumed to substitute at 100% the grid mix of technologies in UK and Italy, as already discussed in the previous section (Section 6.2). This is a simplification of the reality, because the micro-CHP unit is thermally - led and it will produce electricity at maximum in periods of high space heating demand, which are often also times of high electricity demand (Pehnt 2008). In a dynamic system, such as energy systems, an increased in energy demand will likely affect both base-load and peak-load production technologies. The accurate estimation of the environmental benefit of the displaced electricity from the grid is quite difficult, since it depends on the kind of generations that takes place at the specific time the micro-CHP is exporting to the grid (Giannopoulos & Founti 2011b).

For the DG scenarios, comparing the results obtained in this study with literature studies is not an easy task, due to the different approach followed in the literature studies. As cogeneration systems produced both thermal energy and electricity simultaneously, a comparison with no cogeneration system where only electricity or heat are produced is not possible based on the functional unit '1 kWhe' or '1 kWhth', but both products need to be taken into accounts at the same time. The environmental impacts obtained in Chapter 5 for the DG scenarios are compared with the reference scenario in order to estimate the annual saving that is possible to obtain in terms of kg CO_2 eq.

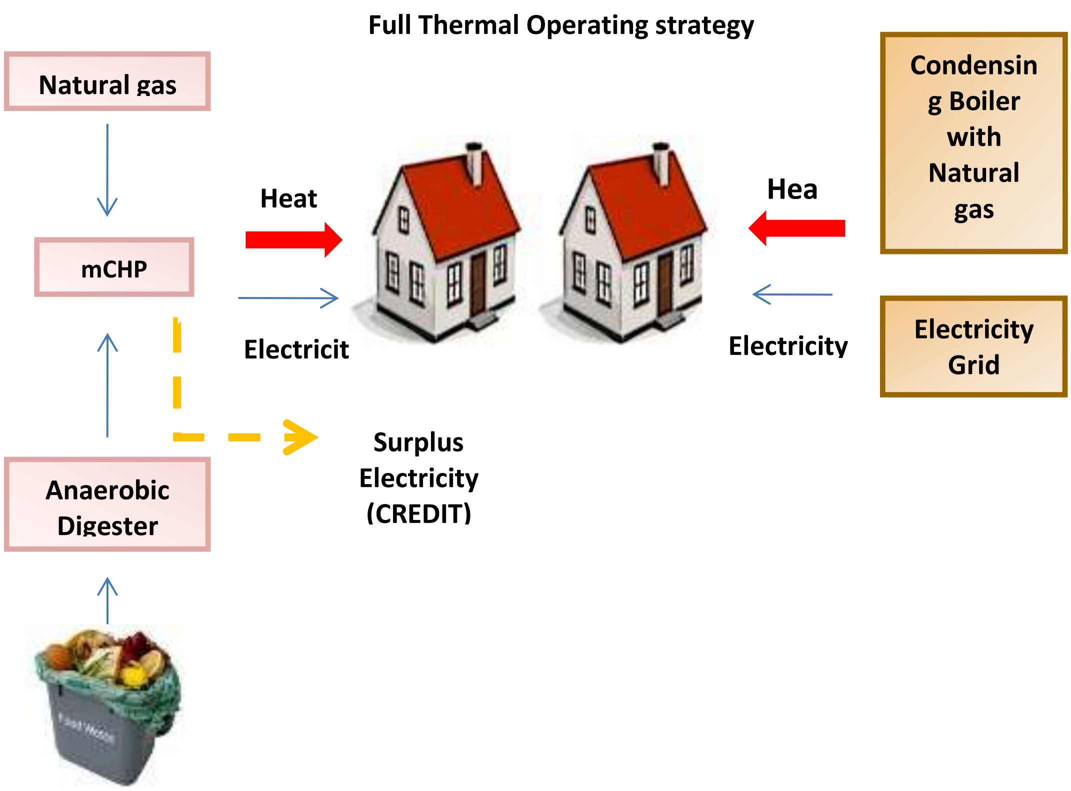

Figure 6.3. Distributed Generation scenario (on the left) and Reference Scenario (on the right) considered in the Full Thermal Operating strategy.

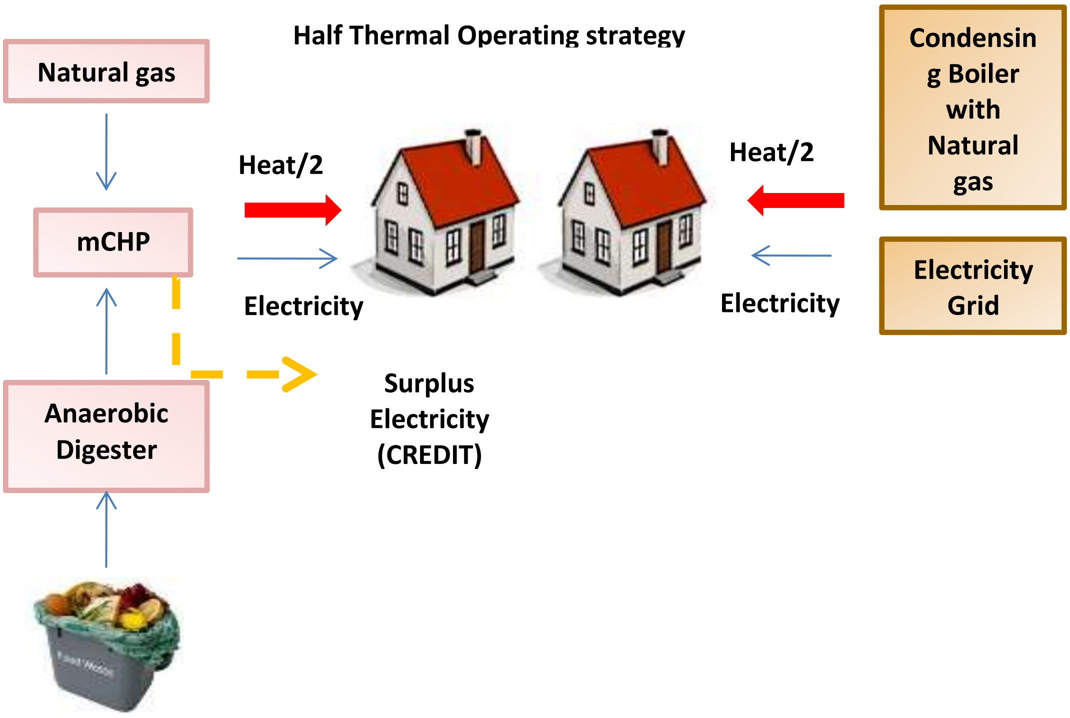

Figure 6.4. Distributed Generation scenario and Reference scenario in the Half Thermal Operating strategy.

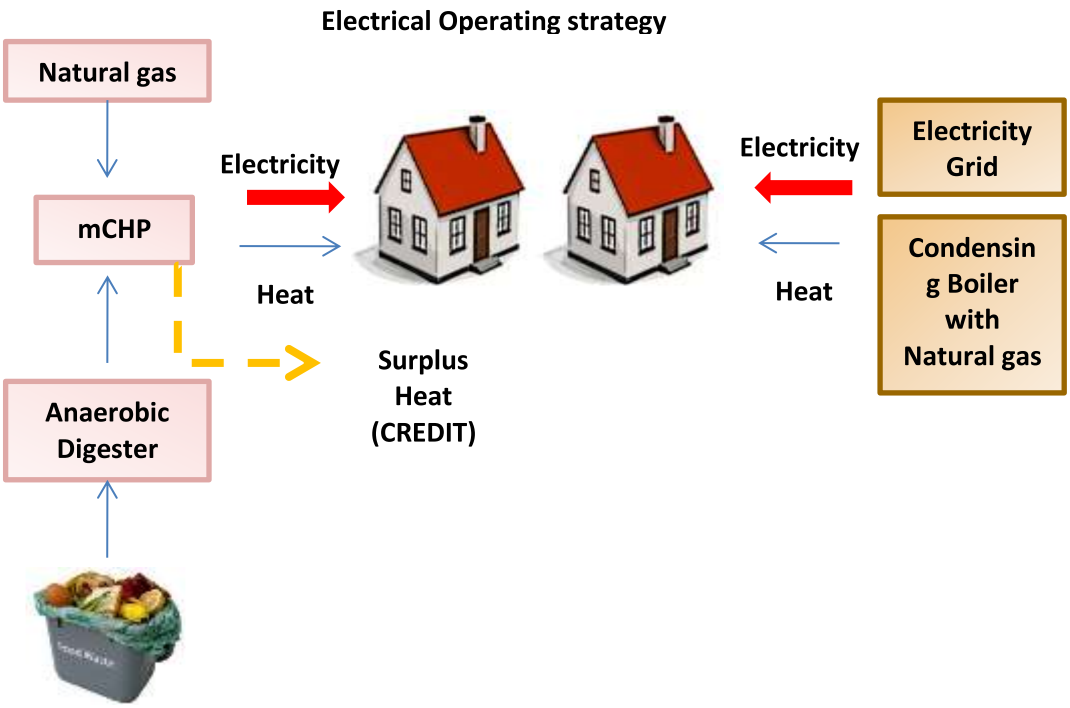

Figure 6.5. Distributed Generation scenario and Reference scenario in the Electrical Operating strategy.

Figure 6.3, Figure 6.4 and Figure 6.5 show the systems considered for the analysis. In the Full operating strategy, emissions accounted in the DG scenario are due to the production of the total thermal energy demanded by the dwelling. Electricity is produced at the same time, given the cogeneration nature of the micro-CHP systems. Moreover, for some of the technologies considered, surplus electricity is produced and it is considered as an avoided burdens. On the other hand, in the Reference scenario emissions are due to the production of the same amount of thermal energy from condensing boiler plus emissions due to the production of the electricity from the grid for the same amount produced in the DG scenario. The same approach is applied to the Half thermal and Electrical operating strategies. This allows obtaining an impact based on the production of the energy demand of 1 dwelling, and the results are shown in Figure 6.6.

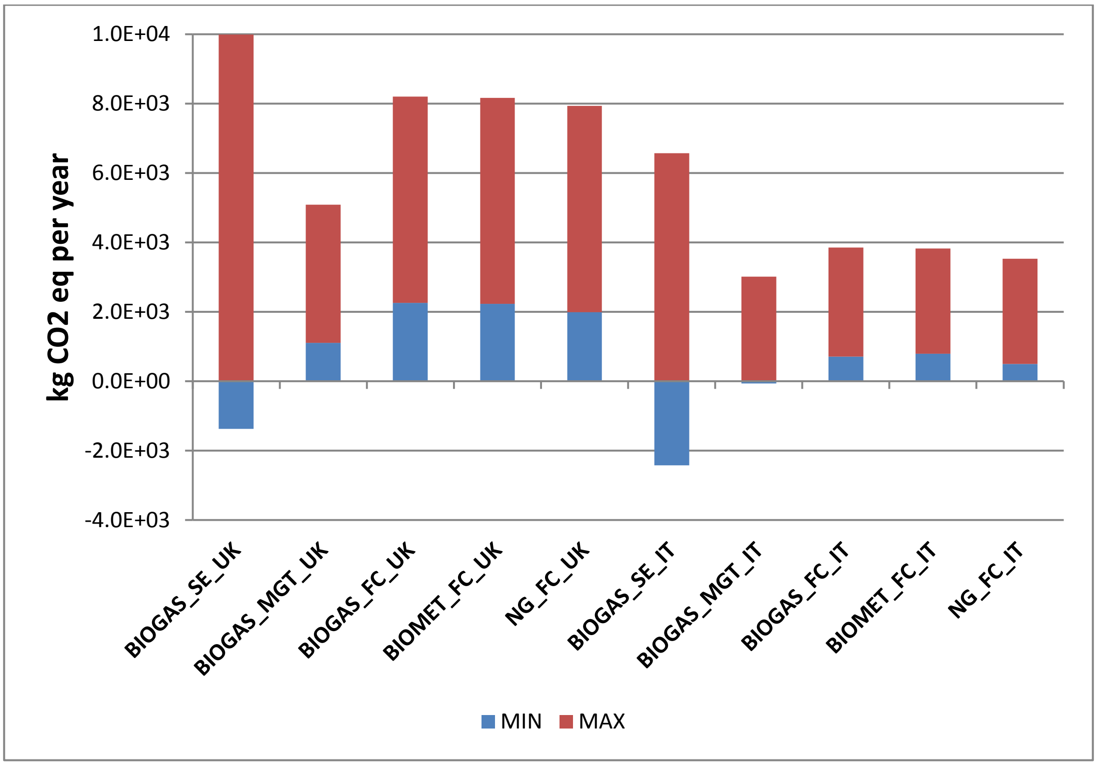

Figure 6.6. Maximum and minimum annual saving in terms of kg CO_2 eq per dwelling, considering the three operating strategies.

As stated in Section 3.2.2, fewer studies have been published on the evaluation of the environmental impact of micro CHP units at residential level, and even none – to the authors' knowledge – investigated the full cycle from the waste to energy in the distributed generation paradigm. Hawkes has published a series of papers on the economic - mainly –

and environmental impacts[22] of micro-CHP systems, especially fuel cells (Hawkes & Leach 2005; Hawkes et al. 2007; Hawkes et al. 2009a; Hawkes et al. 2009b; Hawkes et al. 2011; Hawkes et al. 2009; Hawkes 2010; Hawkes & Leach 2009). Hawkes et al. (2007) estimated a CO_2 emissions reduction of the order of 800 kg per year over the reference scenario, represented by a condensing boiler with natural gas and electricity from the grid for a typical UK dwelling with 1 kWe SOFC-micro CHP system installed. This value does not consider the avoided burdens due to the surplus electricity produced by the micro-CHP units and sold to the grid. In the abovementioned work, the authors considered a least cost operating strategy for the fuel cell. If we compare this with the results obtained in this study, the latter are bigger. In fact, the savings pass from being around 1 ton CO2 eq for the BIOGAS_FC_IT scenario, to be around 8 tons CO2 eq in the BIOGAS_FC_UK scenario.

Pehnt (2008) investigated the environmental impact of different micro-cogeneration units, analyzing, amongst all, SOFC-micro CHP unit and a Stirling Engine of the same size of the one used here. The study was based in Germany, and compared with the provision of the same amount of electricity and heat from the average grid production technology and condensing boiler with natural gas, respectively. The reductions in terms of Global Warming Potential per kWhe produced that he obtained are: 36% for the SOFC unit, and between 50 and 75% for the Stirling engine. The results regarding the FC units are lower to the one found in this study for the Electrical operating strategy, while the findings related with the Stirling engines are higher. Moreover, in the same work, Pehnt analysed the impact reductions when a typical single family in Germany is considered. All the considerations done above about the different energy profile are valid here. However, he found out a reductions of 10% for the Fuel cell unit and 20% for the Stirling engine (he analysed even other micro-cogeneration technologies, but they are regardless for the scope of this study), assuming a 100% full thermal demand for the latter and a 60% thermal demand for the fuel cell. If we compared this with the results shown in Table 6.1 the reductions obtained for the FT are much bigger here and this is mainly due to different operating strategies and demand profiles.

[22] In the published studies of Hawkes only a carbon footprint is performed, and not a full life cycle assessment.

Table 6.1. Reductions in terms of Global Warming Potential that can be achieved in the DG scenarios per single dwellings compared with the reference scenario (the values for UK and IT are referred to two different energy demand profiles).

	UK	Reduction	IT	Reduction
	BIOGAS_SE_AVERAGE_UK	69%	BIOGAS_SE_AVERAGE_IT	38%
	BIOGAS_MGT_AVERAGE_UK	87%	BIOGAS_MGT_AVERAGE_IT	86%
FT	BIOGAS_FC_AVERAGE_UK	141%	BIOGAS_FC_AVERAGE_IT	111%
	BIOMET_FC_AVERAGE_UK	140%	BIOMET_FC_AVERAGE_IT	110%
	NG_FC_AVERAGE_UK	136%	NG_FC_AVERAGE_IT	101%
	BIOGAS_SE_AVERAGE_UK	-58%	BIOGAS_SE_AVERAGE_IT	-198%
	BIOGAS_MGT_AVERAGE_UK	30%	BIOGAS_MGT_AVERAGE_IT	-3%
HT	BIOGAS_FC_AVERAGE_UK	55%	BIOGAS_FC_AVERAGE_IT	30%
	BIOMET_FC_AVERAGE_UK	54%	BIOMET_FC_AVERAGE_IT	34%
	NG_FC_AVERAGE_UK	48%	NG_FC_AVERAGE_IT	21%
	BIOGAS_SE_AVERAGE_UK	166%	BIOGAS_SE_AVERAGE_IT	100%
	BIOGAS_MGT_AVERAGE_UK	88%	BIOGAS_MGT_AVERAGE_IT	71%
EL	BIOGAS_FC_AVERAGE_UK	61%	BIOGAS_FC_AVERAGE_IT	58%
	BIOMET_FC_AVERAGE_UK	61%	BIOMET_FC_AVERAGE_IT	58%
	NG_FC_AVERAGE_UK	59%	NG_FC_AVERAGE_IT	59%

In their investigation on environmental impact of micro CHP system for residential use, Vooorspools and D'haeseleer (2002) concluded that because most micro CHP are thermally - led, they will operate less during the summer time when there is little or no heat demand. Therefore micro-CHP may indirectly cause an increase in GHG, because of older less efficient generation will operate during summer and no new investment will occur if many CHP units are installed. However, as suggested by Hawkes et al. (2007) if coupled with thermal storage, the SOFC micro-CHP system has a low heat to power ratio, and can provide an optimal solution even in summer period.

Staffell carried out a life cycle assessment of an alkaline fuel cell manufacturing phase and a carbon footprint of a SOFC installed in a dwelling unit, considering a thermal energy storage and different operating strategies (Staffell & Ingram 2010; Staffell et al. 2011). He noted that '*many authors, due to the lack of empirical data, have simulated the operation of SOFC systems using the patterns of energy demand from domestic houses. These agree with the Japanese experience, predicting 15-30% reductions in CO_2 emissions, however some authors suggest negligible or even negative impacts where the SOFC produces more CO_2 than the* *technologies it replaces'*. This confirms the difficulties in comparing the results obtained for a specific case study with the findings from other studies.

Staffell et al. (2011) pointed out that the assumptions made about the displaced technologies are the primary determinant of environmental outcomes: '*in countries which primarily burn coal and gas for electricity (e.g. the UK, USA, Japan) micro-CHP offers a distinct advantage as it can displace around 600 g of CO_2 for each kWh of electricity produced; however, in cleaner countries such as France or Brazil, electricity has a lower carbon intensity than natural gas, meaning it would be impossible for fuel cell micro-CHP to reduce emissions until lower carbon fuels become viable'*. In his PhD dissertation, he assumed a least cost operating strategy and a lifetime of the SOFC unit of 10 years. He found out a carbon footprint reduction for a typical house is UK between 0.75 to 4.7 tons of CO_2 eq per year, depending on the type of electricity displaced. He then evaluated the carbon payback time in the region of 0.5-1.5 years when displacing CCGTs. Overall, he concluded that the carbon intensity of electricity generation from an average SOFC is around 355 g/kWh when the construction of the stack and the system is included. He suggested that this is a modest improvement, estimating a saving compare with a CCGT plant of 15-20%.

Figure 6.7 shows the annual savings obtained in this thesis when two marginal electricity production technologies are considered. In particular it is possible to observe that the savings are all reduced when CCGT plant is considered as marginal technology, due to the lower emissions associated with the process. Although the general trend does not change for the scenarios analysed, the micro Gas Turbine in the Italian case study can increase the emission of the dwelling if it is not properly designed.

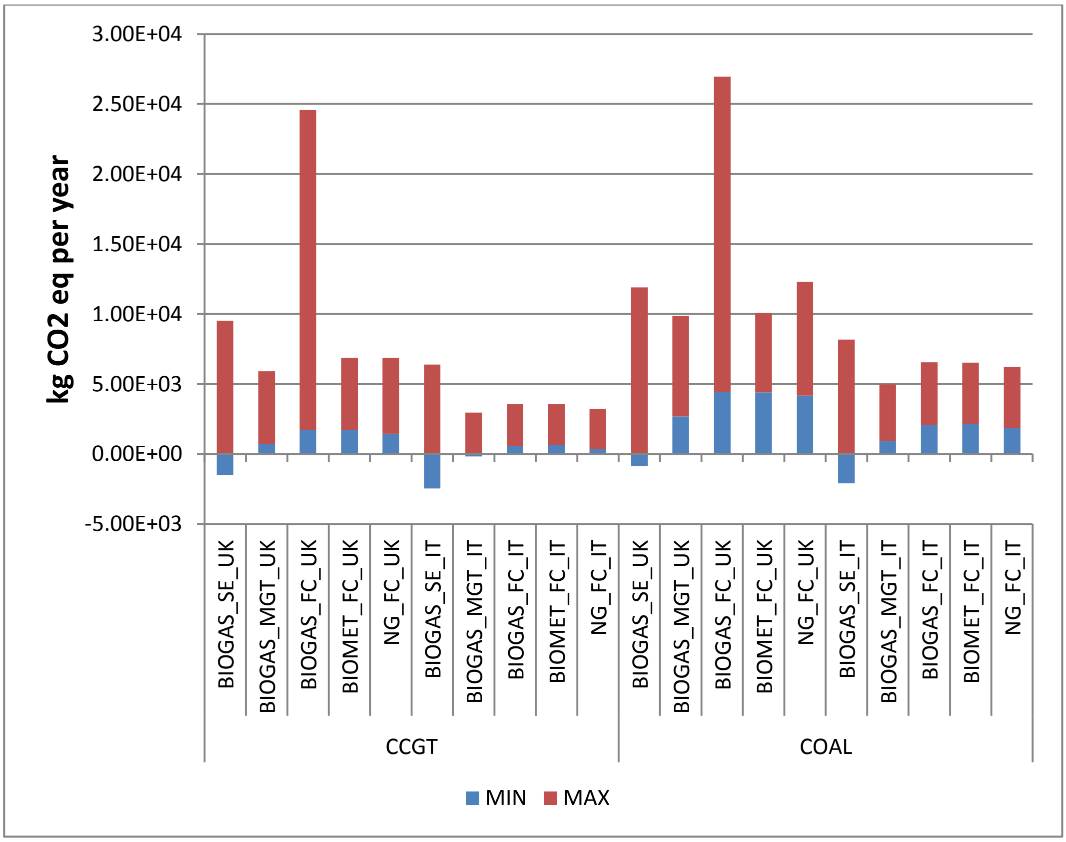

Figure 6.7. Maximum and minimum annual saving in terms of kg CO_2 eq per dwelling, considering the three operating strategies. CCGT and coal plants are assumed as the displaced technologies for electricity production.

In another work (Staffell 2009) he reviewed some literature studies on the issue. A summary of them is shown in Table 6.2. This shows one more time that conclusions are really difficult to draw, about the general savings obtainable for a single dwelling with the introduction of micro-CHP technologies.

Table 6.2. Summary of literature studies on carbon footprinting of Fuel cell – micro CHP units (Staffell 2009)**.**

Fuel cell technology	Displaced technologies	Situation	Reduction in CO_2	Reduction in NRPE
175 ENEFARM 0.7-1kW PEMFC systems η_{el}: 26.0%, η_{tot}: 63.1% (HHV)	Heat from a standard gas boiler (78% HHV, 236g per kWh of heat produced) Marginal electricity from the Japanese grid (36.9% HHV, 690g)	Average savings over 12 months, calculated from field trials in Japanese homes. Systems installed in 2005 (top), 2006 (middle) and 2007 (bottom).	28.0% 846kg	15.3% 2003kWh
777 ENEFARM 0.7-1kW PEMFC systems η_{el}: 26.4%, η_{tot}: 63.2% (HHV)			28.0% 792kg	15.8% 1920kWh
930 ENEFARM 0.7-1kW PEMFC systems η_{el}: 27.7%, η_{tot}: 64.8% (HHV)			30.8% 901kg	18.5% 2310kWh
27 Kyocera 0.7kW SOFC systems η_{el}: 34.1%, η_{tot}: 71.3% (HHV)		Average savings over 12 months (top) or 4 months (bottom) calculated from field trials in Japanese houses. Systems installed in 2007 (top) and 2008 (bottom).	34.2% 1135kg	15.3% 2220kWh
35 Kyocera 0.7kW SOFC systems η_{el}: 36.1%, η_{tot}: 74.0% (HHV)			37.2% 1404kg	18.7% 3027kWh
1kW PEMFC system 26%, 85%	High efficiency gas boiler and average UK electricity grid mix (430g)	Simulated operating in a detached house (top number for each) and a terraced house (bottom for each)	1430kg 1040kg	
1kW SOFC system 40%, 80%			1410kg 1320kg	
Generic 1kW fuel cell η_{el}: 50%, η_{tot}: 90% (LHV)	Heat from a 90% efficient condensing boiler (200g), average UK electricity mix (430g)	Simulated in a large English house.	16% 892kg	
Baxi Beta 1.5 Plus (1.5kW PEMFC) η_{el}: 32%, η_{tot}: 85% (LHV)	Standard (low temperature) boiler and average German electricity grid mix.	Simulated operating for 5000-6000 hours per year in a low-energy home		56%
Hexis Galileo 1000N (1kW SOFC) η_{el}: 30%, η_{tot}: 90% (LHV)	Condensing gas boiler and average German electricity mix (614g).	Simulation in German households	16%	
1kW SOFC η_{el}: 32%, η_{tot}: 85% (LHV)	Condensing gas boiler (97%) and average German electricity mix.	Simulated operating for 4786 hours per year (full load) in a German house	36%	
4kW PEMFC following heat demand η_{el}: 25%, η_{tot}: 80% (LHV)	Condensing gas boiler (107%), and electricity from: The average Belgian mix (top, 37%, 272g); CCGT (middle, 50%, 404g); Fossil mix (bottom, 42%, 617g).	Simulated in a detached house in Belgium with 4 occupants.	-6%	29%
			12%	12%
			37%	22%
2kW PEMFC, following heat demand η_{el}: 24%, η_{tot}: 84% (LHV)	Gas heating (241g) and average German electricity mix (600g)	Simulated in a German detached (top) and terraced house (bottom) with 4 occupants	18% 2000kg	
			17.5%	
SOFC of unknown capacity η_{el}: 45%, η_{tot}: 85% (LHV)	Heat from 91% efficient condensing boiler, average UK electricity mix (420g). Exported electricity was given no credit.	Simulated in an apartment (top), terraced (middle) and semi-detached house (bottom) in the UK	17kg	
			-139kg	
			-108kg	
1kW SOFC η_{el}: 31%, η_{tot}: 96% (LHV)	Condensing gas boiler (108%), and electricity from the European average (top, 29.7%, 554g); or Swiss average (bottom, 150g)	Simulated in an average Swiss house.	13% -11%	21%

Finally, a comparison between biogas and biomethane use with fuel cell reveals that only a 700 tons of CO_2 per year can be saved if raw biogas is used. The burdens associated with the upgrading of the biogas are mainly due to the energy demand for the PSA and the emissions of CH_4 that occur during the process. The environmental impact of this phase compensates the benefits in terms of higher calorific value of the biomethane compared with the biogas. This is true if the assumption made about the constant efficiency of the SOFC, when fuelled by biomethane or biogas, is valid. Although some alterations are required when the fuel is biogas, some authors investigated this feasibility. Farhad et al. (2010) investigated three biogas fuelled SOFC micro-combined heat and power systems for application in residential dwellings through computer simulation. They concluded that biogas is a suitable fuel for residential applications of the SOFC, and fewer reforming agent is needed to prevent carbon deposition over the anode cell. Shiratori et al. (2010) analysed the experimental behaviour of an internal reforming SOFC unit when fed with biogas from garbage and animal manure. They investigated the presence of H_2S in the biogas and its influence on the final energy output, an important issue when biogas fuel is used. They concluded that H_2 contamination in biogas is fatal for fuel cell operations depending on the operating temperature. The results indicated that the maintenance of desulfurizer is very important in the operation and higher-grade desulfurization is required for the lower operating temperatures. Lanzini and Leone (2010) analysed the feasibility of a biogas feeding option both with a modelling and experimental approach. Amongst all, they concluded that direct feeding of biogas to a SOFC is possible without any significant degradation of the cell, provided that a correct amount of oxidant is added to the fuel main-stream gas.

7. Conclusions

This work has aimed at evaluating the environmental impact of a waste – to- energy system in a distributed generation paradigm, the so called *waste – to – energy closed loop.* The system investigated has considered the organic fraction of the Municipal Solid Waste generated by a number of households living in a specific geographic area. The OFMSW has been the starting feedstock used to produce biogas through an anaerobic digestion process plant installed in the same area. The biogas produced has been then fed to several micro – CHP units installed directly in the dwellings. A secondary objective of the work has been to demonstrate the feasibility of the closed loop. This has been achieved with a specific design approach. The energy systems under analysis have been assumed to satisfy the energy demand of the same amount of dwellings which generated the OFMSW considered as feedstock. Two different case studies have been considered: the Royal Borough of Greenwich, in the Greater London area (UK) and the municipality of Livorno, in Toscany region (IT).

Following the research questions presented in Section 1.2 of this thesis, some conclusions are drawn. The layout of the chapter mirrors the structure of the guide questions of this research work.

7.1 How much CO_2 it is possible to save when bio-waste is diverted from landfill and sent to an anaerobic digestion plant.

A life cycle assessment has been undertaken to investigate the environmental burdens associated with different treatment options for the organic fraction of municipal solid waste, the so called *Waste Management scenarios*. The analysis has been carried out in the two geographical areas chosen as case studies. Anaerobic digestion has shown to be the best treatment option in terms of total GHG emissions – the most popular impact category in LCA studies - and acidification when energy and organic fertiliser obtained from the waste substitute non-renewable electricity, heat and commercial fertiliser. The total impacts in terms of CO_2 eq and SO_2 eq are both negative, meaning that they represent emission savings compared with the avoided processes. The results are different when the focus is on photochemical ozone and nutrient enrichment potentials; for these two indicators, AD is the second best option while incinerator appears to be the most environmentally friendly solution. This is mainly due to the emissions from the combustion of biogas, accounting for NMVOC emitted to air, and from spreading of the digestate on agricultural soil. The behaviour of macronutrients from organic fertilisers compared with a commercial fertiliser is a subject of current research; the results are strongly dependant on soil quality, waste composition and meteorological conditions specific to the area.

The results achieved have showed that, when normalised per kg of OFMSW produced in the two geographical contexts, 0.851 and 0.856 kg CO_2 eq can be saved in UK and Italy respectively, when AD is considered as alternative waste treatment compared to the landfill plant with energy recovery. If an incineration plant with electricity and heat production is considered as displaced process, the emissions saved are reduced to 0.030 and 0.035 kg CO_2 eq per kg of OFMSW treated in UK and Italy.

A second objective achieved during the investigation of the Waste Management scenarios has been to define a methodological approach to compare the environmental impact of a landfill plant with an anaerobic digestion plant. The results show that any conclusion on the best treatment option is model-dependant. Life cycle assessment is always associated with uncertainties; therefore the robustness of the model and the results has been investigated using sensitivity analysis on the key parameters. The most important assumption concerns the quantity and quality of the energy substituted by that produced from biogas. In this study, electricity and heat production have been modelled by expanding the system to evaluate the burdens avoided by substituting generation in the background system.

7.2 What it is a reliable alternative for the use of bio-waste for micro CHP applications.

In order to answer to this question, three different micro-CHP technologies have been investigated and compared through a LCA approach, the so called Distributed Generation scenarios: internal reforming SOFC unit, micro gas turbine; Stirling engine. Three different operating strategies have been assumed for the three units: full thermal – where the units have been designed to satisfy the full space heating demand of the dwellings; half thermal – where the units have been designed to satisfy the half space heating demands of the dwellings; electrical – where the units have been designed to satisfy the full electrical demand of the dwellings.

The three technologies investigated are at different stages of the product development scale and this has an influence to the final efficiency achievable and, consequently, to the final environmental impacts. Despite the fact that the impacts of part-load efficiency, voltage degradation and unutilised energy have not been taken into account in this work, they are important factors related to micro – CHP efficiency in real installations. It is therefore suggested that future studies incorporate a more specific analysis of the real performance of micro – CHP systems, including those dynamic effects that reduce the final efficiency. A key factor is the operability of the micro – CHP technologies when fed by raw biogas. This field is an on-going research at the moment and further work has to be made before to have reliable units running with biogas. In this work, two different ways for the biogas - fed FC units have been investigated: the first possibility has foreseen that the SOFC is directly fed by the biogas coming out from the AD plant, after a desulphuration unit; the second possibility has provided an upgrading and cleaning step process to transform the biogas to biomethane before the micro – CHP units. The results have shown that the impacts associated with the two ways are almost the same, and a slightly larger impact is associated with the biomethane production compared with the raw biogas.

It has been already stated in this thesis that the yearly base energy demand profile of the dwellings chosen to design the micro – CHP system as time – frame is not enough to estimate the performances of the energy units. Other authors suggested a minute base energy demand profile. However, for the objectives of this thesis – one of which has been to investigate reliable alternative for the use of bio-waste in CHP application and consequently estimate the potential emission savings in micro-cogeneration installations – the three yearly base energy demand profiles inquired have been considered satisfactory.

The weak aspect of the proposed Distributed Generation has turned out to be the amount of biogas produced by the OFMSW generated in the area: it is not enough to satisfy the energy demand of the dwellings in all the three operating strategies considered. To obtain this, natural gas has to be added (still fossil fuel) up to 98% (UK) of the total fuel demand. The results have then shown that other kind of organic streams need to be taken into account if we want to achieve the *waste to energy closed loop*.

7.3 How relevant is the geographical context when we talk about waste management and energy strategy in terms of environmental impact.

The two geographical contexts investigated presented differences in terms of amount of OFMSW produced per household, number of households living in the area, energy demands, and H to P ratio of the dwellings. Comparing the two cases, it is possible to highlight:

- UK scenario has a lowest amount of OFMSW production per household compared with the IT one. Despite this, the total amount of OFMSW available in the borough is larger, due to a larger amount of households living in there.
- The distances between the Transfer Station and the specific plants assumed in this study are higher in the UK scenario, compared with the Livorno case study. Although the surface of the area considered is bigger in Italy than in UK, the morphological composition of the area around the borough of Greenwich does not allow an easy installation for an AD plant. However results have shown that the impacts associated with the transport phases in the scenarios analysed are negligible.
- The Functional Units chosen for the study show how it has been privileged the *quality* of the functions carried out by the systems investigated, rather than the *quantity*. For the WM scenarios, in fact, it has been taken as FU the total amount of OFMSW generated in the case study area that turned out to be larger in the UK rather than IT. On the other hand, for the DG scenarios, the FU chosen has been multi to better represents the two aims of the system: to provide waste treatment option for the organic fraction and to supply energy to the local community. The energy demand profiles for the two case studies have been different, as a consequence of the different climate and different H to P requirements of the dwellings. Normalization of the results have been shown to express the potentially impact reduction on a single dwelling base.

7.4 How much CO_2 it is possible to save with fuel cells fed with biogas in UK and Italy.

The three micro–CHP scenarios have been compared with two competitive cases: a 100% natural gas running the same internal reforming SOFC unit in the three operating strategies considered; *reference scenario*, where the annual electric and thermal loads of the dwellings are covered by average grid electricity and gas boiler. Moreover, marginal technologies have been considered for the electricity production and other two scenarios have been then analysed: the electricity has been assumed to be produced by a CCGT plant, while the thermal load has assumed to be covered by condensing boiler; the electricity has been assumed to be produced by coal plant as second marginal technology.

Fuel cell micro – CHP could clearly reduce the impact of UK and IT homes compared with their current average. There is however great difficulty in estimate a precise value on the magnitude of these reductions for mainly three reasons:

- Estimates are based on the H to P ratio of the dwelling, typical for the specific geographic area and different from country to country[23]. Comparing the results obtained in this work with the one of Pehnt (2008), the first show a higher potential savings. If we compare the results on a kWhe production base, they are on the same order of magnitude;
- The energy profile chosen to design the micro – CHP unit (Full Thermal, Half Thermal or Electrical as in this study) determines if the new installation ends up in a saving or in a burdens compared with the reference scenario. The FC+Biogas scenario have resulted to be the most environmentally friendly solution for all the impact categories considered, in both the two countries, when the micro – CHP units are Full Thermal energy demand led. The things are different for the Half Thermal operating strategy: FC+Biogas is still the best solution for UK, but not for IT – where the less impact system is FC+Biomethane for the GWP impact category. Finally, in the Electricity operating strategy scenarios, the Stirling engine is the most environmentally friendly solution in both the countries, but only from a GHG saving point of view.

[23] Although studies have shown the strong variability of energy demands even amongst dwellings placed in the same borough (Staffell, 2009).

- The technologies chosen to represent the reference scenario have always to be clarified and to be taken in consideration. Many studies from the literature have pointed out the uncertainties in estimating the technology that would be offsetting in the reality if micro – CHP installations take place. Based on the scenarios analysed, a different mix of technologies have been considered in the two cases. This brought to different potential achievable reductions. The highest reduction is achieved for the FC+Biogas scenario in the FT operating strategy, UK case and equals to 91%. In the same scenario, in IT the reduction achieves 89%. In both cases, the average electricity mix has been assumed as production technology in the reference scenario. In general, the reductions are higher in the FT operating strategy, for the case with higher H to P ratio. This is in line with other studies, which concluded that the best situation is when the micro-CHP unit is thermally led.

Given that the displaced technology has such a profound impact on the emissions savings from fuel cells and other micro CHP systems and also from other large-scale renewables and demand reduction measures as well, further research on the types of plant that are displaces dynamically is highly recommended.

The approach of this thesis work has been based on the *waste – to – energy closed loop* system. This technical approach can be defined as '*system innovation*', following the definition of Smith et al. (2010): *'Systems innovation refers to the renewal of a whole set of networked supply chains, patterns of use and consumption, infrastructures, regulations, etc., that constitute the socio-technical systems which provide basic services such as energy, food, mobility or housing*'. This innovation not only involved the technical system per se, but even the social behaviour of the community involved. In this specific case, the results have shown that the amount of OFMSW is not enough at all to produce the biogas needed by the micro-CHP systems to supply the required energy. This brings to two main actions that can be followed to reach the *waste-to-energy closed loop*: from a demand side, a reduction of the energy requested by the dwellings that can be achieved only if a different behaviour of the users is undertaken. From a waste generation side, the amount of OFMSW that can be recovered at household level can be increased if different behaviour in waste disposal is assumed. Both the things are very difficult to get, especially because are related to the behaviour of the human being, normally difficult to change.

Two key issues need to be considered in the development and deployment of future anaerobic digestion plants: maximizing the electricity produced by the CHP unit fired by biogas, given that electricity is a more valuable product in terms of environmental benefit compared to heat, and defining the future energy scenario in which the process will be embedded. The former is related to technology development and research in this field is dealing with this topic, while the latter depends on macro-level national and European developments. Thus these factors highlight the importance of a holistic approach to inform decision-makers on the best solution for waste management treatments.

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