Life Cycle Analysis of Renewable Hydrogen And Methane as Fuel Vectors, and a Critical Analysis of their Future Development in the UK

Tim Patterson BSc (Hons).

Thesis submitted for the degree of Doctor of Philosophy to the University of Glamorgan, Wales

May 2013

Supervisors:

Dr Sandra Esteves (Director of Studies)
Prof. Richard Dinsdale
Prof. Alan Guwy

Wales Centre of Excellence for Anaerobic Digestion
Sustainable Environment Research Centre
University of Glamorgan
Wales, UK

UNIVERSITY OF GLAMORGAN
PRIFYSGOL MORGANNWG

(This form should be typewritten)

Candidate's Declaration Form

Name of Candidate: Tim Patterson

Degree for which thesis is submitted: Doctor of Philosophy

1. Statement of advanced studies undertaken in connection with the programme of research

Module PG4 D002 'Research Methods' was completed prior to commencement of this research degree.

2. Concurrent registration for two or more academic awards

I declare that while registered as a candidate for the University of Glamorgan's research degree, I have not been a registered candidate or enrolled student for another award of the University or other academic or professional institution.

3. Material submitted for another award

I declare that no material contained in the thesis has been used in any other submission for any academic award.

Signature of candidate: .... Date:..................................

ABSTRACT

Concerns over environmental impacts and long term availability of liquid fossil fuels means that sourcing alternative, renewable transport fuels has increased in importance. To date, implemented approaches have concentrated on the production of liquid biofuels biodiesel and bioethanol from crops. Even though technology for implementation is readily available in the form of biogas production and upgrading, gaseous fuels have been largely overlooked in the UK. Research completed showed that if produced from indigenous crops using currently viable technology, it is energetically more favourable to produce gaseous fuels rather than biodiesel or bioethanol with gaseous fuels also delivering some emission benefits at end use. To date, the subsidy system supporting biofuel production has not functioned well. Research showed that if the subsidies approached the maximum allowable value, and when produced from waste materials, the production of gaseous fuels can be economic compared to liquid biofuels. Life cycle assessment has showed that utilising biomethane as a vehicle fuel could be an environmentally appropriate approach if the conventional use for biogas of combusting in a combined heat and power plant cannot utilise the majority of the excess heat produced. A two stage process to produce a hydrogen / methane blend was shown to be energetically favourable when utilising wheat feed, although hydrogen production was low. The process was not energetically favourable when food waste was utilised, indicating the importance of optimising process according to feedstock characteristics. Life cycle assessment of electrolytic hydrogen production using a range of energy sources found that electrolysis driven by renewable energy was a valid option for future deployment. However, given current feedstock availability, indigenous biofuel production, regardless of the fuel produced, could only make minor contributions to overall fuel requirements. As such, a range of fuel vectors, or a significantly greater commitment of land resources to fuel production, will be required in the future.

ACKNOWLEDGEMENTS

I would like to thank the efforts of my research supervisors. Sandra, for the time and effort that you have contributed even though you have had little of either to spare. Richard, for your timely pearls of wisdom that never failed to put things into perspective. Alan, for steering a course through the chaos. All of your efforts are very much appreciated. I also thank Des Devlin, Matt Reilly, Ben Jobling-Purser, Shee Meng Thai and Philemon Kumi for their guidance during my unexpected appearance in the laboratory. If not for them I would probably still be there now, and I definitely would have broken much more stuff.

DEDICATION

This thesis, and all of the effort that it embodies, is dedicated to my family. Mum and Dad, your love and support have never gone unnoticed or unappreciated, if often unacknowledged. Keith and Ceri, knowing that I have a big brother out there somewhere has always been a comfort and a reassurance, and the long term benefits of epic nights out should never be underestimated. My wonderful children Lewis and Robin for being there and for being themselves – I promise to keep doing my best. Jo, my best friend, my other half in every sense, my past and my future, I cannot possibly thank you enough.

Now, what shall I do next……..

FUNDING

The author would like to acknowledge the financial support from:

- INTERREG IIIC WIRHEP, SUPERGEN SHEC (EP/E040071/1 and GR/S26965/01) and TSEC BIOSYS projects during the preparation of the study reported in Chapter 3 specifically.
- European Regional Development Fund (ERDF) and the Welsh Government (WG) through their funding for the Wales Centre of Excellence for Anaerobic Digestion and specifically for the delivery of the studies reported in Chapters 4, 5 and 6.
- ERDF in support of the LCRI H2Wales project for the preparation of the study presented in Chapters 6 and 7.

The results generated, the interpretation and the views contained here are those of the author and do not necessarily reflect the opinion or views of the supporting organisations.

TABLE OF CONTENTS

FIGURES AND TABLES

FIGURES

TABLES

UNITS AND ABBREVIATIONS

AD	Anaerobic Digestion
DBERR	Department for Business, Enterprise and Regulatory Reform
BSi	British Standards Institute
°C	Degrees Celsius
CCS	Carbon Capture and Storage
CH_4	Methane
CHP	Combined Heat and Power
CO	Carbon Monoxide
CO_2	Carbon Dioxide
CO_2 eq	Carbon Dioxide Equivalent
CONCAWE	Conservation of Clean Air and Water in Europe
CNG	Compressed Natural Gas
DALYs	Disability Adjusted Life Years
DDGS	Dried Distillers Grain and Solubles
DEA	Diethanolamines
DECC	Department of Energy and Climate Change
DEFRA	Department for the Environment, Food and Rural Affairs
DF	Dark Fermentation
DfT	Department for Transport
DGA	Diglycolamines
DM	Dry Matter
E95	Ethanol Fuel (95% Ethanol, 5% Ignition Improvers)
EC	European Commission
EJ	Exajoules
ETBE	Ethyl Tertiary Butyl Ether
EU	European Union
€	Euro
FAO	Food and Agricultural Organisation
FC	Fuel Cell
FIT	Feed In Tariff

FM	Fresh Matter
FQD	Fuel Quality Directive
FW	Food Waste
g	Gram
GHG	Greenhouse Gas
GWh	Gigawatt Hours
GWP	Global Warming Potential
H_2	Hydrogen
ha	Hectare
H_2O	Water
hr	Hour
H_2S	Hydrogen Sulphide
IA	Inventory Analysis
ILUC	Indirect Land Use Change
IPCC	Intergovernmental Panel on Climate Change
IRR	Internal Rate of Return
ISO	International Standards Organisation
km	Kilometre
ktoe	Kilotonnes of oil equivalent
l	Litre
LCA	Life Cycle Assessment / Analysis
LCIA	Life Cycle Impact Assessment
$LiMn2O_4$	Lithium ion battery
LPG	Liquefied Petroleum Gas
m^3	Cubic Metre
max.	Maximum
min.	Minimum
MEA	Monoethanolamine
mg	Milligram
MJ	Mega Joules
mPt	Milli Point (or Milli Eco-indicator Point)

MTBE	Methyl Tertiary Butyl Ether
MW	Mega Watt
MWh	Mega Watt Hour
N	Nitrogen
N/A	Not Applicable
NaOH	Sodium Hydroxide
ND	No Data
NGL	Natural Gas Liquids
NGV	Natural Gas Vehicle
Nm^3	Normal Cubic Metre
NMHC	Non Methane Hydrocarbons
NNFCC	National Non Food Crop Centre
NOx	Mono Nitrogen Oxides
NSCA	National Society for Clean Air and Environmental Protection
OLR	Organic Loading Rate
p	Pence
PAS	Publicly Available Specification
PDF	Potentially Disappeared Fraction
PEM	Proton Exchange Membrane
%	Percent
PHA	Polyhyroxyalkanoate
PHB	Polyhydroxybutyrate
PSA	Pressure Swing Adsorption
Pt	Point (or Eco Indicator Point)
PV	Photovoltaic
£	Pounds Sterling
RED	Renewable Energy Directive
RO	Reverse Osmosis
ROC	Renewable Obligation Certificate
RTFC	Renewable Transport Fuel Certificate
RTFO	Renewable Transport Fuel Obligation

Sb eq	Antinomy Equivalent
SMR	Steam Methane Reforming
SSFW	Source Segregated Food Waste
t	Metric tonne (1,000 kg)
tkm	Tonne kilometre
tpa	Tonnes per Annum
THC	Total Hydrocarbons
TS	Total Solids
UF	Ultra Filtration
US	United States (of America)
UK	United Kingdom
VS	Volatile Solids
WF	Wheat Feed
WRAP	Waste and Resources Action Programme
ww	Wet Weight
yr	Year

THESIS OUTPUTS

Journal Papers

Patterson, T., Esteves, S., Carr, S., Zhang, F., Maddy, J., Guwy, A. (2013). Life Cycle Assessment of the Electrolytic Production and Utilisation of Low Carbon Hydrogen Vehicle Fuel. International Journal of Hydrogen Energy, Submitted in May 2013.

Patterson, T., Esteves, S., Dinsdale, R., Guwy, A., Maddy, J. (2013). Life Cycle Assessment of Biomethane and Biohydrogen as a Transport Fuel. Bioresource Technology, 131, 235 – 245.

Patterson, T., Esteves, S., Dinsdale, R., Guwy, A. (2011). Life Cycle Assessment of Biogas Infrastructure Options on a Regional Scale. Bioresource Technology, 102, 7313 – 7323.

Patterson, T., Esteves, S., Dinsdale, R., Guwy, A. (2011). An evaluation of the policy and techno-economic factors affecting the potential for biogas upgrading for transport fuel use in the UK. Energy Policy, 39, 1806 – 1816.

Patterson, T., Dinsdale, R.M and Esteves, S. (2008). A review of energy balances and emissions associated with biomass derived transport fuels relevant to the UK context. Energy & Fuels, 22 (5) 3506–3512.

Conference Papers and Posters

Patterson T., Esteves, S.R., Dinsdale, R., Guwy, A.J. (2013). The use of LCA to optimise CO_2 savings in biogas production processes and biogas utilisation pathways. IWA 13th World Congress on Anaerobic Digestion: Recovering (bio) Resources for the World, 25th – 28th June, Santiago de Compostela, Spain. Poster presentation accepted in February 2013.

Patterson, T., Esteves, S., Dinsdale, R., Guwy, A. (2011). Life Cycle Assessment of Anaerobic Digestion of Source Segregated Food Waste with Various Biogas End Uses at a Regional Scale. Platform presentation at the IWA International Symposium on Anaerobic Digestion of Solid Waste and Energy Crops, August 2011, Vienna, Austria

Patterson, T., Esteves, S., Dinsdale, R., Guwy, A. (2011). Evaluation of the Policy and Economic Factors Affecting the Use of Biomethane as a Transport Fuel in the UK. Platform presentation at the IWA International Symposium on Anaerobic Digestion of Solid Waste and Energy Crops, August 2011, Vienna, Austria

EXECUTIVE SUMMARY

Transportation is the largest consumer of energy in the UK with the economy and lifestyle of the population underpinned by low cost, readily available mobility, largely based on the utilisation of fossil fuels. Reductions in greenhouse gas emissions, energy security and the potential economic consequences of diminishing fossil fuel availability are now concerns at national and international levels. The initial response to these concerns on a European and UK level was the introduction of liquid biofuels produced either from indigenous crops or imported from around the world. The subsequent realisation that much of the liquid biofuel being integrated into European and UK transport fuel delivered minor, if any, environmental benefits, has led to unambitious policies and subsidy mechanisms that are ineffective at delivering climate change goals. The technology for producing renewable biomethane fuel is well understood and offers some advantages over liquid biofuels in that a range of feedstocks including energy crops and organic wastes can be readily utilised. Where crops are utilised for fuel production, energy yields are shown to be higher for gaseous fuels than for liquid biofuels given current commercially available technology, with gaseous fuel leading to lower exhaust emissions. Anaerobic digestion of biodegradable wastes to produce biogas is undergoing rapid expansion in the UK, however, the upgrading of this biogas to biomethane is far less common. A review of upgrading technologies shows that there are many technology options available and that, where wastes are considered as feedstocks and an effective subsidy scheme is in place, it should be possible to produce a biomethane vehicle fuel at a competitive cost. A stable and functioning subsidy mechanism is required for this to be the case. Currently, however, Government support is focussed on the utilisation of biogas for the production of renewable electricity and, to a lesser extent, heat via combustion in a combined heat and power plant. Life cycle assessment of the production and primary end uses of biogas indicated that where heat utilisation was optimised this approach resulted in the lowest overall environmental burdens. However, to date the importance of the utilisation of excess heat has also been largely unrecognised in UK policy, leading to the majority of AD plants not effectively utilising excess heat. In this situation, other end uses for biogas, in particular utilisation as a vehicle fuel to displace the use of fossil fuels such as diesel, would be more preferable in terms of reducing environmental burdens. Modifications to the anaerobic digestion process provide the

opportunity to produce both biohydrogen and biomethane gases which could be combined into a single vehicle fuel gas, the inclusion of approximately 20% hydrogen providing benefits in terms of exhaust emissions reduction. Life cycle assessment of the process utilising food waste indicated that optimising the process for hydrogen production was likely to be detrimental to the overall energy yield of the process, as methane yields were reduced. Conversely, for wheat feed low hydrogen yields were achieved, however, the overall energy balance of the process was favourable due to an increase in methane production when compared to a single stage process. Further research is required to improve the process efficiency, particularly through reduction in water use, whilst aiming to increase hydrogen yield without lowering overall energy output. Hydrogen can also be produced through the electrolytic splitting of water, and, where this process is driven by renewable energy sources such as wind turbines or photovoltaic panels, a low carbon vehicle fuel could be produced. Life cycle assessment of this process along with other fossil and non fossil alternatives indicated that this would be a viable means of producing a vehicle fuel with low environmental burdens. However, for all biomethane, biohydrogen and hydrogen from electrolysis options there were limitations, primary among them was the availability of feedstock. Utilisation of waste materials for gaseous fuel production was shown to make a minor contribution to either our transport fuel or electricity needs. Utilisation of arable land at a level that does not affect food availability for the production of energy crops also contributed relatively little to overall energy needs. Consideration therefore needs to be given on how we utilise both arable and non arable land to maximise productivity whilst maintaining ecological diversity, regardless of what energy crop or biofuel is produced. Similarly, it is unlikely that the UK will be in a position where significant proportions of our renewable electricity infrastructure (i.e. wind turbines) can be dedicated to producing hydrogen – the priority will be to maintain availability of electricity. The integration of hydrogen production with large scale renewable electricity production, particularly during peak generation / low demand events should therefore be considered in detail. Greater integration of LCA into the research and development of all renewable, particularly biofuels, is required. More ambitious, long term UK policies would also assist with creating more sustainable renewable energy industries.

Chapter 1: Introduction

1.1 Background

The economy of the UK and the lifestyle of its population are underpinned by convenient and low cost mobility, as is the case for much of Western society. Much of this mobility is based on the consumption of fossil fuels. Some of the headline statistics describing the transportation sector in the UK reflect the scale on which we all rely on this mobility and of the resources that are consumed to maintain it.

- Figures for the UK in 2010 indicate that the transportation sector was the largest user of energy and was responsible for 35% of the UK's total energy consumption, some 55.7 million tonnes of oil equivalent out of a total of 159.1 million tonnes of oil equivalent (DECC, 2011a).

- In the UK in 2010 1,489 million tonnes of goods were transported by UK registered heavy goods vehicles which travelled a total of 18.8 billion vehicle kilometres (DfT, 2011a).

- Since 1950, the number of vehicles registered in the UK has increased by approximately 5 million per decade. In 2010 there were over 34 million licensed vehicles on the road (DfT, 2011b).

- In the UK in 2010/11 biofuels accounted for only 3.27% of total transport fuels compared to the government target of 3.5%. 78% of UK biofuel was imported or produced from imported feedstocks including soy from Argentina, corn from the USA and sugar cane from Brazil (DfT, 2011c).

- Despite European Directives stating a mandatory target of 10% of transport fuels being from renewable sources by 2020, the maximum deployment of biofuels planned by the UK Government at present is just 5% of total transportation fuels by 2013/14. This means that the sector will continue to be almost entirely based on non-renewable resources for the foreseeable future.

There are sound environmental reasons for encouraging a greater shift away from fossil fuels, however, serious and significant questions are being raised regarding the ability of existing biofuels to deliver real environmental benefits.

1.1.1 Global Warming / CO_2 Emissions

The Intergovernmental Panel on Climate Change (IPCC) has published detailed scientific evidence describing how anthropogenic activities such as agriculture, land use change and fossil fuel use has increased greenhouse gas levels over the past 250 years, and that it is extremely likely (i.e. over 90% certainty) that these activities have exerted a net warming influence on the global climate (IPCC, 2007). The main course of action available to governments to tackle climate change is to reduce greenhouse gas emissions. In 1991 thirty seven industrialised nations agreed to implement the Kyoto Protocol which committed the countries to an average 5% reduction in CO_2 emissions compared to 1990 levels (The United Nations, 1998). The details of how the Protocol would be implemented were agreed in 1997 with the first measured period of greenhouse gas reduction running between 2008 and 2012. Reductions could be achieved either by national measures to reduce physical atmospheric emissions, or by international market mechanisms such as carbon trading. Provisional figures for 2011 indicate that the UK had reduced its CO_2 emissions by 28% since 1990, although emissions from the transport sector remained largely static (DECC, 2012a).

On a global scale, it is questionable whether the Kyoto Protocol could ever achieve significant benefits. In 2001, the United States of America withdrew from the Kyoto Protocol and a number of countries currently seeing significant industrial development, including China, India and Brazil, are not subject to the agreement. On average global CO_2 emissions continue to increase at a consistent rate of 1.9% per year over the past 20 years (Olivier et al., 2011). The 17th Conference of the Parties (17th COP) international climate change talks held in December 2011 failed to reach an agreement on how to secure the future of the Kyoto Protocol post 2012, although the EU has committed to reduce overall emissions by 20% by 2020 (European Commission, 2008). The pressure on European authorities and member state governments to deliver significant greenhouse gas reductions is therefore set to continue for the foreseeable future.

1.1.2 Fossil Fuel Availability

UK data for 2010 suggests that current known oil reserves will be completely diminished in approximately 46 years, natural gas in approximately 59 years and coal in 118 years (BP, 2011). Given that our transportation infrastructure is so heavily dependent upon oil, it is clear that rapid development and implementation of alternative transport fuels and/or drive trains is required. Long before oil reserves are completely utilised it is likely that prices will increase significantly, meaning that low cost fossil fuelled transportation will no longer be available. Peak oil production is anticipated in around 2015 (Maggio et al., 2012) meaning that from around this date it is feasible that global supply will no longer meet global demand.

In the UK, crude oil and Natural Gas Liquids (NGL) production from the North Sea reduced by approximately one fifth between 2010 and 2011, the largest reduction since large scale production began, and for the first time UK primary oil imports exceeded indigenous production (DECC, 2011a). Whilst the UK's refining capacity currently exceeds internal demand for petroleum (for vehicle fuel use), gas oil and fuel oil (for electricity generation), the UK production of diesel is approximately 75% of demand and production of aviation fuel is approximately 50% of demand (DECC, 2011a). As stated above, this refining capacity is now predominantly met using imported primary oil. It is clear that whilst North Sea oil and gas production and refining will continue to be an important economic activity for the Country, we will increasingly be reliant upon imported fossil fuels.

1.1.3 Biofuel Supply & Sustainability

In order to combat the environmental problem of global warming and the resource / economic problem of future fossil fuel availability, European policies have increasingly sought to promote the production and utilisation of biofuels such as biodiesel and bioethanol in the transportation sector. However, there are growing concerns that the large scale production of many of the feedstocks required to produce these biofuels results in an overall increase in Greenhouse Gas (GHG) emissions, in particular through emissions associated with Indirect Land Use Change (ILUC) (Gallagher, 2008), increases general environmental damage such as habitat loss (Danielsen et al., 2009), and competes with food production (Sachs, 2008).

In order to attempt to address the primary issue of potential GHG impacts of biofuel production, the latest European legislation, EU Directive 2009/28/EC, stated that biofuels must result in 35% GHG saving compared to the fossil fuels which they replace, and in 2018 this target increases to 60% GHG savings for new production facilities (European Parliament, 2009a). Default GHG saving values within the legislation for a number of biofuel production pathways are:

- Wheat ethanol (production process not specified) 16%
- Wheat ethanol (natural gas as process fuel in boiler) 34%
- Wheat ethanol (straw as process fuel in CHP) 69%
- Rapeseed biodiesel 38%
- Soybean biodiesel 31%
- Corn ethanol (natural gas as process fuel in CHP) 49%
- Sugar cane ethanol 71%
- Palm oil biodiesel (production process not specified) 19%
- Palm oil biodiesel (methane capture at oil mill) 56%

Crops most suited to temperate European climates such as wheat and rapeseed may fail to meet either the existing 35% GHG saving target or the 2018 60% target unless the production process is configured in a highly efficient manner (i.e. utilisation of straw as a fuel for wheat ethanol production as shown above). Crop based fuels providing highest GHG savings such as sugar cane ethanol and palm oil may not automatically meet the 2018 targets, and, as they are grown in tropical or sub-tropical areas the concern and uncertainty over habitat destruction and losses of soil carbon will be subject to even closer scrutiny in the coming years. Liquid biofuels delivering the largest GHG savings are those derived from waste materials such as biodiesel production from cooking oil (83% GHG saving). Second generation biofuel production processes are under development and are likely to deliver increased environmental benefits compared to current, first generation biofuels (González-García et al., 2012) , however, to date these processes have not been deployed at industrial scales, largely due to economic reasons (Littlewood et al., 2013).

In recognition of the likely overall GHG *impact* of a number of biofuels, the European Commission has tabled a proposal (European Commission, 2012) for the amendment of both Directive 98/70/EC relating to the quality of petrol and diesel fuels and Directive 2009/28/EC on the promotion and use of energy from renewable sources (The Renewable Energy Directive). The aims of this proposal are to:

1. Limit the contribution that conventional biofuels (with potential indirect land use change emissions) make towards Renewable Energy Directive targets
2. Improve GHG performance of existing biofuel production process by raising the savings thresholds required for new installations
3. Encourage the uptake of advanced (low ILUC) biofuels by allowing them to make a greater contribution to Renewable Energy Directive targets
4. Improve reporting of GHG emissions including estimated ILUC emissions (European Commission, 2012).

In the UK, just 22% of the 1.57 million litres of biofuel supplied to the UK market during 2010/11 originated from UK sources, the majority of which was produced from waste cooking oil (DfT, 2011c). The remaining 78% was directly imported or produced from imported feedstocks. The verified Renewable Transport Fuel Obligation (RTFO) report for the period stated that a GHG saving of 57% was achieved (DfT, 2011c), although acknowledged that this did not include all emissions from direct land use change, and did not include any emissions from indirect land use change such as those highlighted in the Gallagher review. As such the actual GHG savings are likely to be significantly less than stated. Only 54% of biofuel supplied met the required environmental standard against the Government target of 80% (DfT, 2011c).

1.2 Focus of Research

1.2.1 Introduction to Gaseous Fuels

To date the implementation of alternative transport fuels in both the UK and on a European basis has focussed on the production of liquid biofuels such as bioethanol and biodiesel. However, gaseous transport fuels such as methane and hydrogen offer several advantages over liquid biofuels, in particular the flexibility of production options. Whilst the vast majority of methane and hydrogen used in the UK is manufactured or derived from fossil sources e.g. natural gas for methane, and Steam Methane Reforming (SMR) of natural gas for hydrogen production, there are alternative approaches to production that could deliver renewable fuel gases. Methane and hydrogen can be manufactured using a wide range of renewable or low carbon technologies including electrolysis of water using electricity derived from photovoltaic cells or wind turbines, thermal treatment of ligno-cellulosic materials such as wood, or biological treatment of complex biodegradable materials including municipal, commercial and industrial wastes (Figure 1).

The utilisation of methane as a vehicle fuel is not a new technology. Worldwide there are over 14.8 million vehicles fuelled by methane, predominantly as compressed natural gas (i.e. a fossil fuel). Highest Natural Gas Vehicle (NGV) deployment rates are seen in Iran (2.86 million vehicles), Pakistan (2.46 million vehicles) and Argentina (2.25 million vehicles) (NGV Communictions Group, 2012). In Europe, Italy is taking the lead with approximately 779,000 natural gas vehicles deployed, whereas, despite having one of the most advanced and widespread gas grids in the world, the UK is reported as having just 556 NGVs deployed (NGV Communictions Group, 2012).

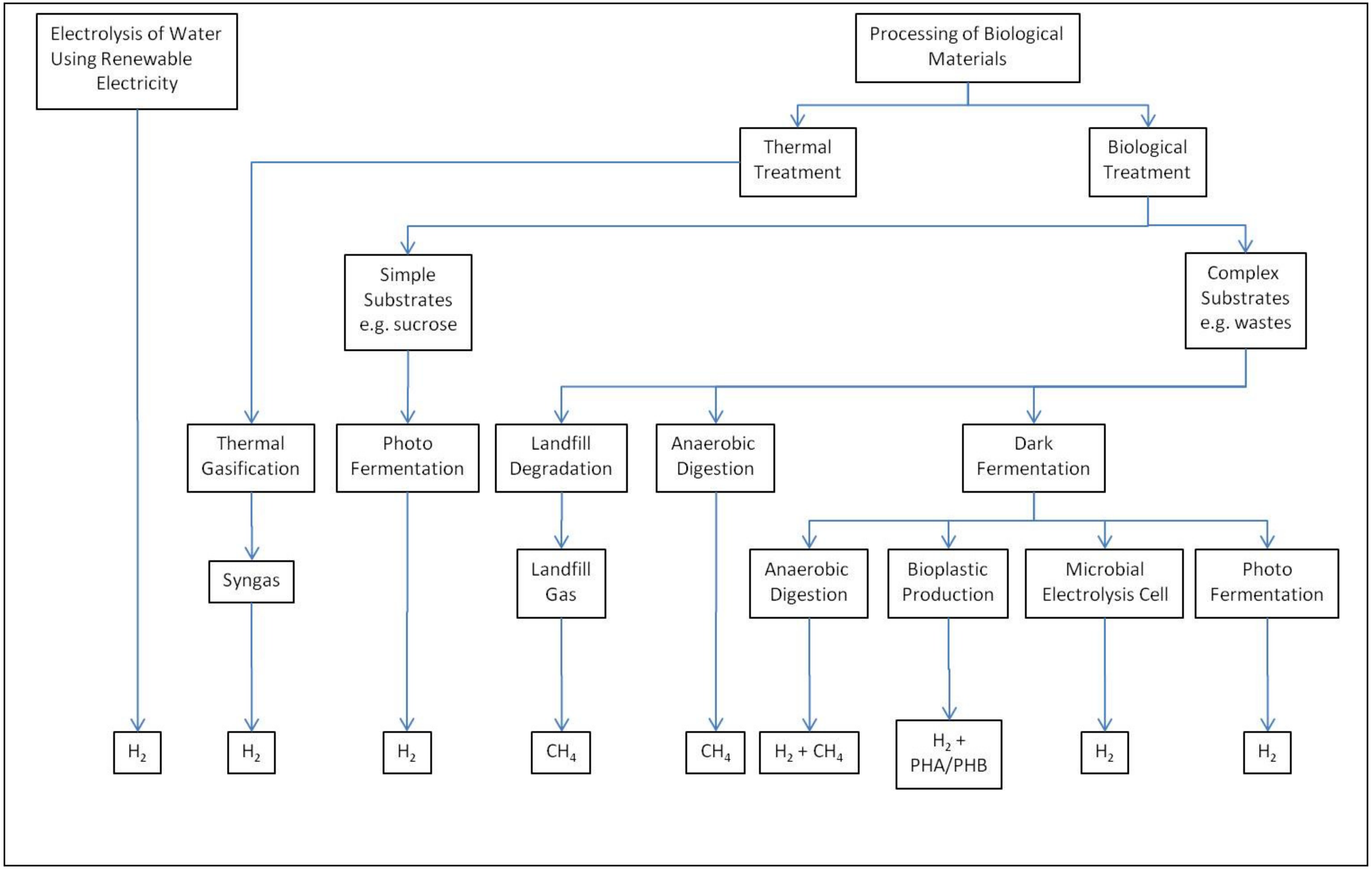


Figure 1 – Potential pathways for the production of renewable hydrogen and methane gas

The use of hydrogen as a vehicle fuel, as well as the development of a wider 'hydrogen economy' has been the subject of a considerable amount of research and development within academia (Barreto et al., 2003) and increasingly by governments (e.g. the US Department of Energy Hydrogen and Fuel Cells Program) and industry (Wallner et al., 2008). In the past decade there have been numerous hydrogen vehicle trials and demonstration activities undertaken throughout the world, including the UK (e.g. The London Hydrogen Partnership).

Gaseous fuels are efficient at the point of end use. The Volkswagen Passat 1.4 Ecofuel DSG which uses compressed natural gas (CNG) has a fuel consumption of 0.068 m^3 (approximately 0.045 kg) methane per km (Volkswagen AG, 2009), whilst the small SEAT Mii Ecofuel has an average fuel consumption of 0.029 kg methane per km (Green Car Congress, 2013). An assessment of the efficiency of (hydrogen) fuel cell vehicles undertaken by Ahluwalia *et al.* (2004) suggested a fuel consumption of 0.0134 kg hydrogen per km could be achieved in a standard Proton Exchange Membrane (PEM) fuel cell passenger car, which is 2.7 times the efficiency of a gasoline fuelled vehicle on an energetic basis. Suggested improvements to reduce vehicle mass, drag coefficients and rolling friction could improve performance to 3.3 times the efficiency of a gasoline vehicle (Ahluwalia et al., 2012). The specification for Honda's latest fuel cell production vehicle (Honda FCX Clarity) suggests a fuel efficiency of 0.0104 kg hydrogen per km (Honda, 2012), whilst for the larger ix35 Hyundai specify a fuel consumption of 0.0096 kg hydrogen per km (Hyundai, 2012). As summarised in Table 1 below, all of these gaseous fuelled vehicles consume less energy per km than a modern petrol fuelled vehicle.

Table 1 – Summary of energetic efficiency of petrol, CNG and Fuel Cell Vehicles (FCV)

Vehicle Type	Model	Engine	Av. Stated Fuel Consumption	litres (kg for gas) / km	LHV of Fuel (MJ/kg)	MJ / km
Medium Petrol Car	Ford Focus	2.0L Ti-VCT GDI I-4	30 mpg	0.078	43.448	3.39
Large CNG Car	VW Passat	1.4 TSI EcoFuel DSG	6.8 m^3/100 km	0.045	47.141	2.12
Small CNG Car	SEAT Mii	1.0 l CNG	2.9 kg/100 km	0.029	47.141	1.37
Medium FCV	Honda Clarity	PEM FC (100 kW)	60 miles / kg	0.0104	120.21	1.25
Large FCV	Hyundai ix35	PEM FC (100 kW)	0.96 kg / 100 km	0.0096	120.21	1.15

1.2.2 Biological Production of Methane and Hydrogen

One of the routes through which both methane and hydrogen could be renewably produced is the biological breakdown of readily biodegradable materials. Anaerobic Digestion (AD) is a process where organic matter is mineralised primarily to a mixture of methane and carbon dioxide (biogas) through a series of reactions mediated by several groups of microorganisms in the absence of oxygen. The process is widely used at an industrial scale for the treatment of organic wastes such as sewage sludge and is currently also being employed for the treatment of municipal, commercial and industrial organic wastes, or for energy generation using energy crops.

The majority of AD plants are configured such that the consortia of microbes converting the organic material to biogas are present within a single tank, meaning that conditions are necessarily sub-optimal for any groups of bacteria that require different environmental conditions. A variation to AD where trophic groups of microorganisms with differing optimal environmental condition requirements are separated into two different vessels has been developed, with potential advantages of this process being that (i) hydrogen can be liberated from the acid producing first phase (Dark Fermentation), and (ii) methane production in the second stage (Anaerobic Digestion) can be increased compared to the single stage process, giving an overall increase in energy output (Hawkes et al., 2007).

The UK has a good track record in utilising biogas. Since the 1970's AD facilities have been deployed throughout the UK water industry for the treatment of sewage sludge, and there are currently 146 AD plants on the ground for this purpose (National Non Food Crop Centre (NNFCC), 2012). The collection and utilisation of landfill gas also increased dramatically during the 1980s and 1990s to the extent that the UK is still the second largest producer of biogas (which includes landfill gas) in Europe behind Germany (L'Observitoire Des Energies Renouvelables, 2010). Whilst changes to waste management practices are resulting in a reduction in the landfill of wastes and therefore a reduction of landfill gas generation, the UK is currently seeing a rapid increase in the deployment of anaerobic digestion facilities, primarily for the treatment of municipal and agricultural wastes. Plant numbers for these purposes have increased from <10 in 2005 to over 106 in May 2013 (National Non Food Crop Centre (NNFCC), 2012). In almost all cases, the biogas collected is utilised for the production of electricity or the production of electricity and heat in a CHP plant.

Given the pressing need to deploy alternative vehicle fuels in the UK, the demonstrated availability of end use technology in the form of NGV's, and the concerns being raised regarding the sustainability of many current liquid biofuels, it is not immediately clear why gaseous fuels are not more prevalent in the UK and many other European countries.

1.2.3 Focus and Applicability of the Study

This thesis focuses on determining the environmental burdens associated with the 'low carbon' production and utilisation as a vehicle fuel of methane and hydrogen gases, in particular (i) the production of methane using anaerobic digestion, (ii) the production of a combination of hydrogen and methane using dark fermentation (for hydrogen production) followed by anaerobic digestion (for methane production), and (iii) the generation of hydrogen using renewable energy powered electrolysis. Consideration of the production and utilisation of biofuels is a country specific task as factors such as national policies, economics, demographics, climate and energy mixes are all variable, and therefore by necessity, this study will focus on current UK conditions.

It is very likely, however, that key findings of the study will be fully or partially relevant to other countries at a similar level of gaseous fuel deployment to the UK. Feedstocks considered for the biological production of methane and hydrogen were primarily waste materials, although an initial comparison of biogas with other liquid biofuels was based on agricultural feedstock production. The emphasis on waste as a feedstock for the detailed environmental evaluation was a reflection of the fact that, partially in recognition of the uncertainty over the long term sustainability of utilising domestic arable crops for energy production, and partially due the expectation that second or third generation biofuel production technologies will negate the need to utilise arable crops, current UK policy is focussed on optimising the capture and utilisation of waste materials (DECC, 2012b).

1.3 Aims and Objectives

The aims of the thesis were to evaluate the environmental burdens associated with the production and utilisation of fuel gases produced either by biological or electrolytic processes (i.e. methane, hydrogen / methane blend or hydrogen), to evaluate whether these fuels would deliver environmental benefits if they had a larger role in the UK, and to determine the factors that might limit this deployment in the future. Specific objectives that were completed to achieve these aims included:

- An investigation of the potential energetic and emission benefits that gaseous fuels may have when compared to current liquid biofuels (Chapter 3).
- Complete a techno / economic assessment of biomethane production to determine potential economic or policy driven barriers to implementation (Chapter 4).
- To quantify the environmental impacts of biomethane utilisation as a vehicle fuel in comparison to other end uses for biogas / biomethane in order to assess whether vehicle fuel is an environmentally advantageous end use (Chapter 5).
- To quantify the environmental impact of two stage biohydrogen / biomethane production in comparison to single stage biomethane production to investigate whether either process has a significant environmental benefit (Chapter 6).
- To quantify the environmental burdens associated with the electrolytic production of hydrogen using renewable energy sources and its utilisation as a vehicle fuel in comparison to other fuel options (Chapter 7).
- To critically analyse the potential for the technologies considered to contribute to the UK vehicle fuel infrastructure (Chapter 8).

1.4 Methodology

A description of the general methodologies employed in the completion of this thesis is provided in Chapter 2. The study is primarily desk based although a limited amount of laboratory work was necessary to determine some primary parameters for the assessment of two stage biohydrogen / biomethane production. Data sources for energetic, economic and emissions assessments were therefore primarily based on literature as well as experience and information gathered from liaising with various stakeholders within the AD industry in the UK and Europe. As the thesis covers a wide range of topics a traditional literature review chapter is not included. Given the diverse nature of the topics discussed, it was considered more appropriate to include a detailed review of relevant literature within each chapter. Methodologies utilised throughout the study have also been described in detail in each chapter and include simple economic assessment and Life Cycle Assessment (LCA). Results are used to inform a critical review of whether, and / or how, fuel gases considered can contribute to the future vehicle fuel infrastructure in the UK, with conclusions of the thesis summarised in the final chapter.

1.5 Thesis in Brief

The thesis comprises nine chapters. The main body of the thesis is summarised as:

Chapter 2 – The general methodologies used during the completion of the thesis are described. These include methods for the review of literature, simple economic analysis and a description of the rationale employed when undertaking life cycle assessment elements of the thesis.

Chapters 3, 4, 5 and 6 have largely been published as peer reviewed journal papers (Appendix A) in the scientific literature. Chapter 7 has been written in paper format and has been submitted for publication. All papers, data collection, modelling, analyses and interpretation within the papers, have been produced by the thesis author with contributions from the research supervision team in terms of direction and review of research. Additional direction relating to hydrogen production technologies was provided in Chapters 6 and 7. Sources of additional data have been fully referenced throughout. A summary of Chapters 3-7 is described below:

Chapter 3 – An energetic assessment of the production of biomethane has been undertaken and compared with traditional liquid biofuels. The energy balance was primarily used to quantify which option resulted in the most effective utilisation of agricultural land for the production of vehicle fuels. Tailpipe emissions at end use were also reviewed in order to assess any significant emissions benefit at end use.

Chapter 4 – An assessment of the various technologies available for the upgrading of biogas to biomethane was undertaken. Data was then utilised for an economic assessment of biomethane production in the UK given policy frameworks at the time in order to determine the extent to which these policies affect the deployment of biomethane as a vehicle fuel in the UK.

Chapter 5 – A Life Cycle Assessment (LCA) of biomethane production from source segregated food waste at a regional scale (Wales) was undertaken with various end use pathways considered including Combined Heat and Power (CHP) production with and without heat utilisation, grid injection followed by vehicle fuel utilisation, and grid injection followed by combustion in a domestic boiler. Each assessment was made based on either a centralised or more distributed treatment infrastructure within the region to determine the significance of variation in transport requirements between the two options.

Chapter 6 – A Life Cycle Assessment (LCA) of two stage biohydrogen / biomethane production and end use as a vehicle fuel was undertaken and compared with that of a single stage biomethane production system. Two feedstocks were considered, source segregated food waste and wheat feed, which is a commercial / industrial waste product. The assessment was based on two differing laboratory based processes, food waste being treated in a batch process whilst wheat feed was treated in a semi continuously fed apparatus. The assessment was aimed at identifying whether there were any clear environmental advantages of either single stage biomethane or two stage biohydrogen / biomethane production for the specific feedstocks considered.

Chapter 7 – A comparison of the environmental burdens associated with the production of hydrogen from the electrolysis of water using wind energy, PV generated electricity and UK grid electricity. For the UK grid electricity, an estimation of the energy generation mix in 2030 was also included in the assessment to evaluate the impact of an increased renewable energy component in this mix. Hydrogen produced was considered to be utilised in a Proton Exchange Membrane (PEM) fuel cell passenger vehicle. Comparisons were made with a range of reference fuels including hydrogen from steam methane reforming, petrol, and the utilisation of grid electricity to power electric (i.e. battery rather than fuel cell) vehicles.

Chapter 8 – Based on the findings of the previous chapters, a critical analysis of the potential for biomethane and / or biohydrogen / biomethane to contribute to the UK transport fuel infrastructure was performed, taking into consideration fuel production potential, associated costs and environmental burdens as well as required changes in policy, regulatory and fiscal incentives frameworks.

Chapter 9 – Conclusions were made with recommendations for future actions and research.

Chapter 2: Methodology

2.1 Introduction

Detailed methods are presented within each Chapter so that each represents a stand-alone study. The information below therefore represents an overview of the general rationale employed when completing the various elements of this thesis.

2.2 Review of Literature

This thesis covers a broad range of research approaches. Chapter 3 is largely based on a review of existing academic and technical literature in order to compare the energetic and emissions performance of biomethane with liquid biofuels. The first half of Chapter 4 is similarly a review of existing academic and technical literature in order to establish the status and relative performance of the most common biogas upgrading (i.e. biomethane production) technologies. The methodology employed to complete the literature review included the use of web based databases such as ‘Web of Knowledge’ and searches of the most relevant academic journals including Bioresource Technology, Biomass and Bioenergy, Biotechnology and Bioengineering, Waste Management, International Journal of Hydrogen Energy, Water Research, Energy Policy and Energy & Fuels. Other journals were consulted to source information on specific topics as and when required. Background information as well as data on UK and European energy, fuel and transport statistics and policies was sourced from a range of government and non-government organisations including the Department for Energy and Climate Change (DECC), the Department for Food and Rural Affairs (DEFRA), the Department for Transport (DfT), the European Commission and the European Parliament. Reports from a range of collaborative projects between industry, government and academic groups were consulted as these often provided basic, but useful, data and context relating to case studies describing the deployment of various technologies. These included the Swedish Gas Centre, CONCAWE, BiogasMax, and the International Energy Association (IEA). Some information and data was only available from companies active in the field of technology deployment, and where necessary (particularly in Chapter 4) this information has been included. Company data sources have included BP, Carbotech, Green Gas Energy, DGE GmbH and Questair. All data and information was fully referenced throughout the thesis, and a list of references can be found at the end of the document.

Selection of which references or data to utilise was, in the vast majority of cases, based on availability rather than comparative quality criteria. In many cases the availability of reliable technical or environmental data was limited (this indeed was one of the findings in Chapter 4). Varying methodologies and viewpoints were expressed in the literature relating to the life cycle analysis of biomethane or hydrogen fuels, and these have been summarised in Chapters 5, 6 and 7. Limitations to particular studies or potential biased viewpoints have been indicated as and when required.

It is worth noting that this thesis was written over a period of several years (2008 – 2013), and, to some extent, the research undertaken has evolved as the industry, economics and policies have altered. Data relating specifically to financial performance or viability is particularly vulnerable to external changes and pricing information described in Chapters 3 and 4 will be out of date at the time of the completion of the thesis. Similarly, the biofuels market is evolving so rapidly that financial information is out of date within a matter of months. Whilst the specific data presented in these chapters may therefore no longer be applicable, the conclusions drawn within each of the studies were still considered valid at the time of completion, and it was therefore decided not to fully update these chapters. A review of the current (e.g. immediately prior to submission) and foreseen factors affecting biomethane and biohydrogen deployment has been provided in Chapter 8.

Whilst Chapters 5, 6 and 7 were predominantly based on Life Cycle Analyses, it should be noted that these chapters also included a large amount of up front data sourcing or collection, either based on published literature following the methodology described above or, for Chapter 6, on laboratory work undertaken within the University of Glamorgan.

2.3 Simple Economic Analysis

Chapter 4 ends with a simple economic analysis of biomethane production compared with liquid biofuels, compressed natural gas and diesel. Production costs and any tax or subsidy incentives applicable for diesel, biodiesel, bioethanol and natural gas were sourced from literature. Simple biomethane production costs from the treatment of municipal wastes were calculated taking into account the following primary factors affecting financial performance of an AD plant:

- Feedstock biogas potential
- Parasitic energy demand of the plant
- Capital cost of the plant
- Interest on capital costs
- Operational costs of the plant
- Gate fee for waste treatment
- Financial incentives (Reduced tax and Renewable Transport Fuel Certificates as applicable)

Costs for upgrading and compression were taken to be the average of the literature values identified for a range of upgrading technologies assessed in Chapter 4.

A more detailed analysis of the payback times associated with the use of biogas in a CHP plant or the production of biomethane for vehicle fuel use, either within a captive fleet or for public sale, was also included in Chapter 4. This included additional parameters such as CHP cost and performance, grid connection costs, refuelling station costs, vehicle fleet capital and maintenance costs as described in Table 17 in Chapter 4.

2.4 Life Cycle Analysis

2.4.1 Overview of Approach

Chapters 5, 6 and 7 describe the detailed Life Cycle Analysis (LCA) research undertaken on biogas infrastructures with different end uses (Chapter 5); the biohydrogen production process (Chapter 6); and the electrolytic production of hydrogen (Chapter 7). Life Cycle Analysis, also known as Life Cycle Assessment, is a process by which a range of environmental burdens associated with the production, utilisation and final disposal of a product, process or service can be quantified. The concept of LCA developed in the 1960s and 1970s as companies recognised the need to evaluate the potential environmental impact of their activities, however, it was not until the 1990s that an organised approach to completing an LCA emerged and was finally formalised in an International Standard (EN ISO 14040:2006 and ISO 14044:2006, which have replaced the original Standards ISO 14041-43:2000).

ISO 14040 sets out the general methodological framework that should be applied when undertaking a life cycle assessment and defined the various stages and considerations that should be included when constructing a life cycle assessment. This primarily included the various potential inputs into the key stages that should be included in all LCAs:

- Goal and Scope Definition – A key stage at which the intended reasons for undertaking the study, intended application, audience as well as the impact categories of interest are defined (Goal) and the product system is defined in detail including function and functional unit, system boundary, allocation requirements, data requirements and assumptions made, all of which should be clearly stated. In the case of this thesis, the studies have largely been aimed at academics active within the relevant field of study, although where relevant, results have also been communicated to the Welsh Government and industry stakeholders. As such, whilst the impact categories of primary concern and with most relevance to current policies were considered to be climate change / CO_2 emissions and fossil fuel utilisation, the studies have been broadened to incorporate human health impacts such as carcinogen burdens and ecotoxicity impact categories as a means of evaluating the potential broader impacts associated with infrastructure changes.
- Inventory Analysis (IA) – This includes the procedures and processes associated with the collection or calculation of the data required to quantify the relevant inputs and outputs of the product system being investigated. Inputs to a product system

typically included raw materials, energy, ancillary inputs (e.g. transportation), or any other physical inputs. Outputs included products, co-products or wastes whilst emissions to air, water and soil were also quantified.

- Life Cycle Impact Assessment (LCIA) – The mathematical process by which the potential significance of the environmental impacts incurred as a result of the energetic and material flows identified in the Inventory analysis was evaluated. As a minimum this included classification of emissions and extractions to relevant impact categories and characterisation of the magnitude of their impacts. Normalisation and, in one instance, weighting of results, was also undertaken as described in Chapters 5 – 7.
- Interpretation – LCA is a highly iterative process, and throughout the completion of the above stages, a degree of interpretation was required. The above stages were re-visited to ensure that the goal and scope defined at the outset was achieved. The interpretation of the impact assessment results required a detailed understanding both the goal and scope and inventory analysis, and therefore great care was required when generating and applying LCA results.
- Application – Once it was clear that the goal and scope of the work had been achieved, the results generated were then applied for their intended purpose. In the case of this thesis, chapters comprising of LCA studies have been subject to peer review and published in the scientific literature. For Chapter 5, results were also communicated to the Welsh Government to assist in their development of waste management policy. Results for Chapter 6 were fed back to researchers active in biohydrogen production in order to direct their future research on how improvements in process efficiency can reduce environmental burdens.

The guidance was expanded upon in ISO 14044 which provided a more detailed description of the various components introduced above (e.g. system boundary within the goal and scope definition), and how various procedures (e.g. allocation of impacts in a multi output process) might be approached. The general principles of life cycle assessment were also embodied in the PAS 2050:2008 Specification for the assessment of life cycle greenhouse gas emissions of goods and services. ISO 14040 and 14044 also formed the basis of several other more specialised standards such as ISO 14025 (Environmental Product Declaration), ISO 14067 (Carbon Footprinting) and ISO 14045 (Eco-Efficiency), and so have been found to be largely fit for purpose, although further clarification on the requirements of critical peer review during the completion of a comparative LCA for public dissemination has been suggested (Klöpffer, 2012).

The guidance documents described only provided the general frameworks and key stages that were considered when the life cycle assessments were undertaken. They did not provide a detailed step by step guide on ‘how to’ undertake an LCA, partially because this would differ widely between studies. As such, it remained necessary to define the specific parameters associated with each study such as the various assumptions made, definition of system boundary, function and functional unit, data collection (or generation in some cases) and the impact assessment method employed.

The guidance documents describe the general requirements for the completion of an ‘attributional’ life cycle assessment. This is where the environmental impacts of the production, use and disposal of a well defined product system are quantified, but where indirect effects of the uptake of that product system are not considered. This is in contrast to a ‘consequential’ life cycle assessment where indirect changes outside of the immediate product system boundary are also taken into account. Consequential life cycle assessment is becoming increasingly relevant, particularly for the consideration of biofuel production and utilisation, however they are highly dependent on economic models and factors such as product and co-product supply, demand and value. These models are therefore even more transient in their accuracy and limited in their ability to be interpreted and applied than the already potentially complex attributional approach. For the purposes of this thesis therefore, the attributional approach was taken when developing the LCA models described in Chapters 5 - 7.

In addition to the ISO standards, the ILCD Handbook (European Commission Joint Research Centre (JRC) Institute for Environment and Sustainability, 2010) was referred to for more detailed guidance on the considerations required during the completion of life cycle analyses.

2.4.2 LCA Software

Quantifying the inventories (i.e. inputs, outputs and emissions) for anything but the simplest product system became a large and time consuming task. Similarly, quantifying the environmental impact of these inputs, outputs and emissions across a range of impact categories required a correspondingly large volume of computation. Since the early 1990's a range of software packages have been available in the market place to make the process of completing LCA's less time consuming, and to provide a relatively 'standard' framework in which an organisation could undertake successive LCAs over a period of time.

LCA software can be viewed as the users interface between a number of key components. Firstly, a database of inventories for various process, products or materials allowed the selection and the combination of these standard elements to build an inventory for the particular process system. Whilst the utilisation of this database allowed the building of relatively complex systems, the compromise was that by utilising 'standard' database values the modelled system would only be an approximation of the actual system of interest. In addition, many of the specific process of interest to this thesis (e.g. biohydrogen production) did not feature within the available databases, and even relevant processes that were available (e.g. anaerobic digestion) only included partially relevant inventories. As such, the core processes investigated in this thesis required the creation of new process entries or the significant modification of existing inventories.

The second primary function undertaken by LCA software was the impact assessment itself. This comprised a number of computational stages such as characterisation, normalisation or even weighting of results, to calculate burdens across a range of impact categories such as global warming, fossil fuel depletion or ecotoxicity. A number of specific methodologies to undertake these calculations were available and have been discussed in more detail below.

As well as the availability of a wide range of impact assessment methodologies, there was also a wide range of software packages available in the market. A review of the leading European LCA software providers in 1997 included twelve individual software packages, some of which were developed for specific industry applications, and all of which had varying characteristics such as provision of standard inventory data and the relevance of this data, impact assessment methodologies, functionality and cost (Rice, 1997). Although all of the software reviewed had advantages and disadvantages, four packages were highlighted as being likely market leaders including SimaPro (v3.1) (updated version still available), TEAM™ (now owned by Ecobilan, a division of Price Waterhouse Coopers), The Bousted Model (updated model is still available, but not widely used) and PEMS (3.0) (developed by the packaging company PIRA, no longer widely used). A survey of LCA software available in Sweden in 2000 identified twenty four commercially available packages, although no ranking of the performance of these was undertaken and the review reached the unremarkable conclusion that many of the packages were intended for a range of different purposes (Jönbrink et al., 2000).

To establish whether a particular software package was of good quality or appropriate for the specific applications for this study, factors were considered such as clarity of data display, process transparency, good quality database, sound calculation methodologies and provision of service support points that had already been referred by Unger et al. (2004).

At present the UK LCA software market is dominated by SimaPro (v7) (Pré Consultants bv) and GaBi (v5) (PE-International). Main alternatives included UMBERTO (IFU Hamburg GmbH) which was arguably more popular across mainland Europe than within the UK. The UK Environment Agency had also produced the Waste and Resources Assessment Tool for the Environment (WRATE), which was primarily aimed at assessing waste management options and therefore not necessarily appropriate for LCAs outside of this field. One of the major benefits of SimaPro was that it was provided with the Ecoinvent database (produced by the Swiss Centre of Life Cycle Inventories) which included over 4,000 inventories for major industrial and agricultural products and process. The key to the importance of the

Ecoinvent database was that the inventories were fully transparent so that all background data could be analysed, the assumptions identified as well as the methodologies used to calculate the inventories. Where necessary, adjustments were possible to accommodate specific user requirements. GaBi was provided with a larger database as standard (4,500 inventories), however, the data provided was far less transparent making it more difficult to determine how applicable a specific inventory might be to the process being modelled, or how the inventory could be adjusted to make it more accurate. The Ecoinvent database could have been used by GaBi, but it had to be purchased at additional cost. All of the most common impact assessment methodologies (discussed below) are provided in both packages. For the purposes of this thesis, although where possible process specific data was used, the ability to interrogate and potentially modify existing inventory data was considered important and as such SimaPro software including the Ecoinvent database was used.

2.4.3 LCIA Methodologies

As introduced above, the use of SimaPro also allowed the rapid calculation of a range of environmental burdens based on the materials, energy and emissions inventory assembled during the inventory analysis stage. This is known as the Life Cycle Impact Assessment (LCIA) stage. The assumptions (e.g. emission factors) and calculation methods employed during these calculations affect the final results generated (Dreyer et al., 2003). Unfortunately, there is a wide range of LCIA methods available, many derived using different assumptions, methods and for different purposes, whilst some share common elements and assumptions. To give an indication of the choices available, SimaPro v7.3.3 included sixteen European LCIA methods, twelve single issue methods (e.g. Global Warming Potential), two North American methods and a number of superseded or alternative methods. Clearly there is a wide scope for variability, and a need to understand the major differences between these methods so that an informed decision on which to use can be made.

Single issue LCIA methods (e.g. IPCC 2007 GWP) only calculated impacts associated with a particular impact category (e.g. Global Warming Potential (GWP)). In the case of this thesis, the LCA research was aimed at investigating a broader scope of impacts and as such single issue methods were not considered. Given that the thesis was focussed on UK conditions, and where possible data was gathered from UK or European sources, the European methodologies were considered the most appropriate to apply. The available methods (in SimaPro v7.3.3) are summarised in Table 2 below.

Table 2 – Summary of European LCIA Methods Available in SimaPro (v. 7.7.3)

Primary Method	**Midpoint / Endpoint**	**Weighting**
CML Baseline 2000	Midpoint	No
CML 2001	Midpoint	No
Eco-indicator 99	Endpoint	E, H, I (Optional)
Ecological Scarcity	Midpoint	Yes
EDIP 2003	Midpoint / Endpoint	Yes
EPD (2008)	Midpoint	No
EPS 2000	Endpoint	Yes
IMPACT 2002+	Midpoint	Yes
ReCiPe	Midpoint	E, H, I (Optional)
ReCiPe	Endpoint	E, H, I (Optional)

As indicated in Table 2, LCIA methods could broadly be divided into two approaches; (i) a midpoint (or problem orientated) approach, or (ii) an endpoint (or damage orientated) approach. Midpoint approaches calculated the relative importance of a particular emission or material extraction by characterising them at a point within the cause-effect chain. For example global warming impacts were calculated by aggregating the global warming potential of emissions expressed in terms of radiative forcing and half life differences. However, there was no consideration of cross-over of impacts between impact categories.

By contrast, end point approaches did include some trade off or aggregation across impact categories. For example an increase in climate change impact may ultimately (i.e. at the end point) also cause an impact to human health or ecosystem damage (Bare et al., 2000). There is, of course, a trade off between the two approaches. End point methods were accepted as being more relevant to establishing the true burdens of a product system as they attempted to include the interactions between impact categories and quantify the overall environmental burden. However, the interactions between macro-scale environmental systems are extremely complex, and therefore an additional element of uncertainty was inherent within the calculations. By omitting these interactions, and therefore uncertainty, midpoint methods benefitted from producing more certain results over a shorter timeframe, but with perhaps less overall relevance in terms of establishing the true environmental burdens associated with specific emissions or extractions (Bare et al., 2000).

As the LCA research within this thesis is concerned with a long term shift in the production and utilisation of transport fuel at a societal level, an endpoint approach was considered as the most relevant in establishing the long term environmental impacts or benefits of such a modal shift. However, in order to provide a balanced and objective approach, all LCA work also included undertaking a midpoint analysis at the sensitivity analysis stage.

Of the methods listed in Table 2, EPS 2000 was intended for internal use during a company's product design process whilst EPD (2008) was used for the production of Environmental Production Declarations in Sweden, and therefore given the specific applications that these were intended for, they were not considered for use in this thesis. The Ecological Scarcity method did not include impact categories that were considered of key importance when evaluating renewable gaseous fuel production, namely global warming potential and fossil fuel depletion (or similar), and as such this method was also discounted.

EDIP 2003 was an updated version of the EDIP 1997 method. The original methodology was a midpoint (problem orientated) approach, however the modifications made during the update rendered the method more in line with an endpoint (damage orientated) approach. The method did include a wide range of impact categories, however the update to EDIP 2003 did not include the impact category of resources (which would include fossil fuels), and the method was therefore not considered further.

Eco-indicator 99 (Goedkoop et al., 2001) was chosen as the end point method used for the primary analysis of life cycle impacts in this thesis. A broad range of impact categories were included in this method, as summarised in Table 3, and of key relevance to this thesis was the inclusion of climate change and fossil fuels as distinct impact categories. In order to provide an indication of burdens across damage categories other impact categories considered included carcinogens, respiratory inorganics and ecotoxicity. Following the classification of inventory emissions and extractions to the relevant impact categories and characterisation of the potential impacts, the method offered further damage assessment, normalisation and weighting stages. These all allowed results for individual impact categories to be aggregated into one of three damage (or endpoint) categories as in Table 3.

Table 3 – Damage and Impact Categories included in the Ecoindicator 99 method

Damage Category	Impact Category
Human Health	carcinogens, respiratory organics, respiratory inorganics, climate change, radiation, ozone layer
Ecosystem Quality	ecotoxicity, acidification / eutrophication, land use
Resources	Minerals, fossil fuels

As such, all stages within Ecoindicator 99 beyond data characterisation (i.e. damage assessment, normalisation and weighting) were based on a complex model including fate analysis, exposure, effect analysis and damage analysis. Any such model of complex environmental interactions over relatively long time periods (i.e. 10s – 1000s of years) inevitably involved the inclusion of assumptions and choices, which could have a dramatic influence on results. In order to accommodate for this potential variation, three versions of the Ecoinvent model were produced. These broadly reflected the outlooks expressed by a panel of 365 Swiss individuals with an interest in LCA that was used to establish 'archetypes' on which these models, as well as weighting factors, were based (Table 4).

For the purposes of this thesis, the Hierarchist (H) version of Ecoindicator 99 was employed as this was considered to reflect the most balanced approach, somewhere between that of the potentially optimistic Individualistic outlook and the potentially pessimistic Egalitarian outlook.

Table 4 – Three Ecoindicator 99 methods based on 'Archetypes'

Model Perspective	**Time Perspective**	**Manageability**	**Required Level of Evidence**
H (Hierarchist)	Balance between short and long term	Effective policy can avoid many problems	Inclusion based on consensus
I (Individualist)	Short term	Technology can avoid many problems	Only proven effects
E (Egalitarian)	Very long term	Problems can lead to catastrophe	All possible effects

Source: After (Goedkoop et al., 2001)

Normalisation of LCIA results is a useful procedure, particularly when results are to be presented to non-LCA experts. Normalisation allowed results across the different damage or impact categories to be expressed in the same unit, and therefore presented in a single figure. Ecoinvent 99 normalised results according to the damage caused by an average European over the course of one year, primarily based on data from 1993 with some updates of key emissions. As normalisation took place after damage assessment, the normalisation set depended on the model perspective chosen (i.e. H / I / E). Whilst it was generally acknowledged that normalisation made the communication of LCIA results a simpler task, a degree of understanding of the procedure and background assumptions, and how these could affect results, may be beneficial to those viewing results (Dahlbo et al., 2012).

Ecoindicator 99 allowed the weighting of results. This involved prioritising the importance of the various impacts (e.g. climate change might be considered to be more or less important than carcinogenic impacts) and as such is an entirely subjective process and falls outside of the ISO process. Developers of Ecoinvent used the views of the panel described above to determine the weighting factors, however, rather than prioritising between twelve impact categories it was considered a more manageable task to prioritise between the three damage categories (Human Health, Ecosystem Quality and Resources), with subsequent weightings then applied to the relevant damage and impact categories. Weighting sets were therefore either as per the model chosen (i.e. H / I / E) or an average of the three weighting sets was applied (A). Weighting was applied in the study described in Chapter 5, primarily because the outputs of the study needed to be presented to policy makers in a very simple format, and the ability to present results as a single score (which were weighted) fulfilled this criteria. The average (A) weighting set was used and the overall conclusions from the analysis were not different between weighted and non weighted results.

Midpoint analysis of results was also undertaken as part of the sensitivity analysis included in Chapters 5 - 7. The LCA work described in Chapter 5 used an earlier version of SimaPro (v. 7.2) and CML Baseline (2001) was utilised as the midpoint method. The extended impact categories included in the full CML method were not required as part of this thesis. In Chapter 6 and 7, where the latest version of SimaPro (v. 7.3) was used, ReCiPe (Midpoint) (H/A) was used as the comparative midpoint method. This was chosen in preference to CML Baseline as ReCiPe shared many operational procedures with Ecoinvent 99 (for example the incorporation of ‘archetype’ perspectives within the model) whilst maintaining a problem orientated midpoint approach.

Chapter 3: A Review of Energy Balances and Emissions Associated with Biomass Based Transport Fuels Relevant to the UK Context

3.1 Introduction

This Chapter focused on a numerical evaluation of the energy available from biomass derived transport fuels including biodiesel, bioethanol (both based on first generation conversion technologies) and biomethane that was undertaken based on available literature. This evaluation was completed in 2008 and focused on the energy balance, co-products and tailpipe emissions of the fuels.

Global warming and energy security have placed additional importance upon establishing viable alternatives to traditional petrochemical based transport fuels. In 2010, transport accounted for 35% of the total energy used in the UK and as such this sector represents the largest national consumer of energy (DfT, 2011a). Reducing the reliance of the transport sector on traditional petrochemicals will make a significant contribution to reducing greenhouse gas emissions and stimulating an environmentally and economically sustainable low carbon economy.

In 2003 the Biofuels Directive (2003/30/EC) set out a regime for incorporating a minimum proportion of biofuels and other renewables into petrol and diesel based transport fuels within EU Member States (European Parliament, 2003). By 2010, 5.75% of these traditional transport fuels were to comprise either biofuel or another renewable component. The Renewable Energy Directive (European Parliament, 2009a) further increased this target to at least 10% by 2020. Additives or fuels considered included, but were not necessarily limited to, any of the following.

a) bio ethanol	b) biodiesel
c) biogas	d) biomethanol
e) biodimethylether	f) bio-ETBE
g) bio-MTBE	h) synthetic biofuels
i) biohydrogen	j) pure vegetable oil

This study sought to compare a number of biofuels that could be produced from biomass crops capable of being grown within the UK, which could make a significant contribution to the transport sector either on a local, regional or national level, without relying on imported feedstocks. This evaluation related to energy balance, waste products and vehicle emissions. This area was of particular concern at the time of the study as the potentially negative ramifications of implementing the Biofuels Directive were becoming apparent (UK Parliament, 2006). Since the study was completed, in 2009 the EU formalized the 10% target for 2020 although the potential environmental damage associated with the production of some biofuels was acknowledged by introducing more stringent requirements to demonstrate the environmental benefits of the biofuels being utilized (European Parliament, 2009a). Subsequently, further acknowledgement of the limitations of some biofuels was provided when, in 2012, the EU issued a proposal to limit the implementation of food crop based biofuels to 5% by 2020 (European Commission, 2012)

3.1.1 Fuel vs Food

Studies have indicated that biomass could contribute between 33 – 1,130 Exajoules (EJ)/yr towards the future global energy needs by 2050, (Berndes et al., 2003; Hoogwijk et al., 2003; Holm-Nielson et al., 2006) however, large areas of land would be required. Figures for the European Union indicated that in 2001 some 5.7 million ha of land was under compulsory or voluntary set aside, of which 929,000 ha was dedicated to non-food crops, the majority of which were associated with the production of biofuels (Bauen et al., 2004). Land use statistics for the UK in 2005 indicated that over 70% of the UK land area was utilised for agricultural purposes and that 559,000 hectares of this agricultural land was designated as set aside (DEFRA, 2005). Although the set-aside regime has now been replaced, consideration of these areas in terms of non-impact upon food production would still be useful. Significant areas are therefore available within the UK upon which to establish a biofuels industry before food production is impacted. On a regional and global scale the conflict between using agricultural land for the production of food or energy is an extremely significant issue. Globally there are around 800 million car owners (this is expected to rise to 1 billion in the near future) requiring increasing proportions of biofuels, however, 2 billion of the world's poorest people need land to produce food (Brown, 2006). There are those that consider the production of food and fuel to be in direct conflict, with increasing production of biofuels resulting in decreased food supply particularly for the poorest (Brown, 2006; Ziegler, 2008). Others, such as José Graziano da Silva, the South American representative of the Food and Agricultural Organisation (FAO) of the United Nations, suggested that it was not necessarily a lack of food that leads to hunger, but a lack of purchasing power (Sachs, 2008). In a paper entitled 'The Biofuels Controversy' presented at the United Nations Conference on Trade and Development in 2008, Sachs (2008) described a number of the conflicts between food and fuel production and presented a number of policies that, if implemented, could greatly reduce the potential clashes identified. These policies were aimed at achieving the simultaneous consideration of food and energy security on a regional and local basis. Murphy et al. (2011) undertook a balanced assessment of how land can be utilized in the future and identified that whilst there is potential for conflict between food and energy production, better regulation, certification schemes and appropriate land management should limit detrimental economic or environmental impacts (Murphy et al., 2011).

The food versus fuel debate raises complex and potentially controversial social and economic issues, however, one point is clear. If as a global society large areas of land are to be utilized to produce fuel, the energetic yields from this land use should be as high as possible whilst not endangering the environment, and the fuels produced should be effective at reducing overall emissions of pollutants including CO_2.

3.1.2 Biofuels Considered

The majority of biofuel investment and development in the UK has been directed towards biodiesel and bioethanol. In 2006 UK biodiesel and bioethenol production was 169 million litres and 95 million litres respectively, totaling around 0.5% of transport fuel sales (DEFRA, 2007a). By the end of 2011, although UK biodiesel and bioethanol production capacity was estimated at 570 and 475 million litres, actual production was only around 201 million litres and 47.5 million litres respectively (DECC, 2012d). The UK is one of the largest producers of biogas (bio-methane) in Europe with 1,600 kilotonnes of oil equivalent (ktoe) of biogas produced in 2005 (European Commission, 2006) increasing to 1,750 ktoe by 2010 (L'Observitoire Des Energies Renouvelables, 2010). However, the majority of this biogas (>85%) was recovered from landfills with the remaining being produced from sewage sludge, and all of this biogas was converted directly into electricity (L'Observitoire Des Energies Renouvelables, 2010).

The biofuels considered most relevant to the UK, their production methods and the crops considered in this study are shown in Table 5.

Table 5 - Biofuels, production methods and source crops considered

Fuel	Production Method Considered	Crop Considered
Biodiesel	Extraction of plant oil followed by transesterification to biodiesel	(i) Rape seed
Bioethanol	Hydrolysis of sugars followed by fermentation and distillation	(i) Wheat grain (ii) Sugarbeet (roots only)
Bio-methane	Anaerobic digestion	(i) Rye grass (ii) Sugarbeet (whole crop) (iii) Forage Maize

The potential environmental benefits of these fuels were compared by considering the following characteristics:

- Energy Balance - the energy potential of the produced fuel off set against the energy required for crop growth, harvesting and processing.

- Co-Products – The nature of wastes / by-products generated and potential energy associated with them.

- Vehicle Emissions – Tailpipe emissions of pollutants from vehicles using the biofuels considered.

3.2 Energy Balances

An energy balance i.e. a comparison of the energy inputs associated with the production of a fuel and the overall energy output of the fuel produced, is an important factor when considering alternative fuels. Results are entirely dependent upon the extent of the analysis and the accuracy of input parameters and this has led to highly variable energy balance results for particular fuels (e.g. Shapouri reported a positive energy balance for bioethanol production from corn in the USA, where as Pimentel, with a greater range of energy inputs, reported a negative energy balance) (Shapouri et al., 2002; Pimentel et al., 2005). The ideal biofuel would have a large net energy balance, i.e. the potential energy associated with the fuel would significantly outweigh the energy required to grow, harvest and process the crop.

Gross energy output can be determined as a function of the average yield for a crop, the volume of crude fuel produced and the energy density of the fuel.

Gross Energy (MJ/ha) = Fuel Yield (l/ha) × Energy Content (MJ/l)

Table 6 below gives the gross energy associated with each of the biofuels considered as part of this study.

The primary energy inputs associated with the manufacture of biofuels from crops are (i) crop production, (ii) energy required to convert the crop to a fuel and (iii) gas upgrading and compression energy (for gases only). It is assumed that the energy required to transport the final fuel product to a distribution point would be equivalent for each biofuel and it has not therefore been included in this comparison. The energy losses associated with each of the fuels and feedstocks are shown in Table 7.

Table 6 - Gross Energy Output Associated with Biofuels produced from Energy Crops

Fuel	**Crop**	**Average Crop Yield for UK (t/ha FM)**	**Volume of Fuel Generated (l/ha)**	**Energy Density (MJ/l)**	**Gross Energy Output (MJ/ha)**
Biodiesel	Oilseed Rape	3.3 (DEFRA, 2007b)	1,455 (ESRU; The National Biodiesel Board, 2007)	34.45 (ESRU; Calais et al., 2000)	50,125
Bioethanol	Wheat Grain	7.96 (Grain) (DEFRA, 2007b)	3,184 (Kim et al., 2004)	21.2 (Rosenstrater, 2005)	67,501
	Sugar Beet (Root Only)	57.32 (DEFRA, 2007b)	6,191 (Rosenstrater, 2005)	21.2 (ESRU)	131,240
Bio-methane	Perennial Rye Grass	40.5 (DTi, 2004; Martínez-Pérez et al., 2007)	3,180,100 (Mähnert et al., 2005; Lehtomaki et al., 2006)	0.0359	114,164
	Sugar Beet	57.32 (DEFRA, 2007b)	4,808,900 (Lehtomaki et al., 2006; Plöchl et al., 2006)	0.0359	172,640
	Fodder Maize	40 (Living Countryside Ltd., 2007)	8,037,000 (Heiermann et al., 2004)	0.0359	288,544

The net energy balance of the biofuels considered is therefore expressed as:

Net Energy Balance = Gross Energy Output (
Table 6) – Total Energy Losses (Table 7)

The Net Energy Balance of each of the biofuels considered is presented in Table 8.

These figures indicate that, in general, biodiesel from oilseed rape has a relatively poor net energy balance (24,185 MJ/ha) when compared with other biofuels. In a techno-economic analysis of biodiesel by Enguídanos *et al* (2002) energy balances for biodiesel production from rape-seed (excluding by products) of 10,755 MJ/ha (ETSU), 14,374 MJ/ha (ARC) and 23,841 MJ/ha (Levington) were presented (Enguídanos et al., 2002).

Table 7 - Energy Losses Associated with the Production of Biofuels from Energy Crops

Fuel	Crop	Crop Production Energy (MJ/ha)	Crop Conversion Energy (MJ/ha)	Upgrade & Compression Energy (MJ/ha)	Total Energy Losses (MJ/ha)
Biodiesel	Oilseed Rape	17,440 (European Commission, 1994)	8,500 (European Commission, 1994)	N/A	25,940
Bioethanol	Wheat Grain	22,908 (Martínez-Pérez et al., 2007)	16,000 (Min) (European Commission, 1994)	N/A	38,908
	Sugar Beet (Root Only)	19,976 (Martínez-Pérez et al., 2007)	34,000 (Min) (European Commission, 1994)	N/A	53,976
Bio-methane	Perennial Rye Grass	4,710 (Martínez-Pérez et al., 2007)	4,338 (European Commission, 1994; Murphy et al., 2005)	11,949 (Bossel et al., 2003; NSCA, 2006)	20,997
	Sugar Beet	19,976 (Martínez-Pérez et al., 2007)	5,804 (European Commission, 1994; Murphy et al., 2005; Plöchl et al., 2006)	18,070 (Bossel et al., 2003; NSCA, 2006)	43,850
	Fodder Maize	17,630 (Martínez-Pérez et al., 2007)	3,702 (European Commission, 1994; Murphy et al., 2005; Plöchl et al., 2006)	30,201 (Bossel et al., 2003; NSCA, 2006)	51,533

Table 8 - Net Energy Associated with Biofuels from Energy Crops

Fuel	Crop	Gross Energy Produced (MJ/ha)	Total Energy Losses (MJ/ha)	Net Energy Balance (MJ/ha)
Biodiesel	Oilseed Rape	50,125	25,940	24,185
Bioethanol	Wheat Grain	67,501	38,908	28,593
	Sugar Beet (Root Only)	131,240	53,976	77,264
Bio-methane	Perennial Rye Grass	114,164	20,997	93,167
	Sugar Beet	172,640	43,850	128,790
	Fodder Maize	288,544	51,533	237,011

Based on the data used in this study bioethanol production using wheat grain provided an energy balance of 28,593 MJ/ha, comparable to that of biodiesel. Although not directly comparable to this study, or to each other, other investigations have reported energy output / input ratios (excluding by products) of 0.68 – 2.22 (Bernesson, 2004; Börjesson, 2004; Punter et al., 2004). The use of sugar beet as a feedstock dramatically increased the performance resulting in an energy balance of 77,264 MJ/ha. In an assessment of the energy and greenhouse gas balances of biofuels Armstrong et al. (2002) presented the results from previous studies which indicated positive energy balances for bioethanol production from sugar beet of between 4,000 MJ/ha (European Commission, 1994) and 62,760 MJ/ha (Levy, 1993).

The net energy balance of methane production from perennial rye grass was significantly higher than biodiesel or bioethanol at 93,167 MJ/ha. If this can be demonstrated at industrial scale, then there may be a place for the use of perennial rye grass as an energy crop within the UK agricultural industry. This would require little modification of existing agricultural practice, for example from dairy farming. The highest energy balance figures were obtained from the production of bio-methane from sugar beet (128,790 MJ/ha) and fodder maize (237,011 MJ/ha) crops. The additional energy required for gas compression for transport fuel use would be readily met by the higher bio-methane yield achievable from the anaerobic digestion of these crops.

3.3 Process Co-Products

Each of the biofuel production methods listed in Table 5 results in the generation of co-products, either in the solid, liquid or gaseous phase. Each co-product represented a proportion of the gross energy that could not be converted into the primary fuel product. However, in some cases the co-product could be utilised so that some or all of its energetic value could be realised.

The primary co-product generated through biodiesel production from rape seed is rape straw. In Europe this represented a mass of around 5.47 t/ha and had an energy value of approximately 78,190 MJ/ha (European Commission, 1994). Common practice has been to leave this straw in the field and although this may provide some benefits in terms of soil conditioning, there would be relatively little energetic gain. The other primary co-products from biodiesel production from rapeseed are:

(i) rapeseed cake (2,820 MJ/ha can be used as animal feed) (European Commission, 1994),
(ii) glycerol (2,200 MJ/ha) (European Commission, 1994), energetically, this is best used as a chemical feedstock, however the practical and economic experience is that it has generally been disposed either via anaerobic digestion or thermal means, both of which will enable recovery of some of this energy.
(iii) phosphatides (1,500 MJ/ha)(European Commission, 1994)

In cases where bioethanol is produced from grain only, giving a poor net energy balance, the waste straw would represent approximately 91,000 MJ/ha of energy, some of which could be recovered through combustion and heat utilization (DeWulf et al., 2000), or, in the future, could be utilized for bioethanol production using 2nd or 3rd generation processes (Littlewood et al., 2013). The primary waste product from the fermentation and distillation process was stillage. Up to 20 litres of stillage was generated for every litre of ethanol produced (i.e. up to 83,980 l/ha of wheat crop) (Wilkie et al., 2000). Inputs of large amounts of energy allowed for the recovery of Dried Distillers Grain and Solubles (DDGS), which could then be used as an animal feed. The large volumes of wastewater generated then required treatment. Where standard wastewater treatment plants were used, either on site or off site, a further input of energy was required for its operation. Where an on-site anaerobic digester was used to treat this wastewater, around 9,600 MJ/ha of energy could be recovered in the form of biomethane which could then be used within the production process (i.e. for heating / drying) (European Commission, 1994). Gluten, with an energetic value of around 44,000 MJ/ha, was also a significant by-product of bioethanol production (European Commission, 1994). This was a high value co-product which could be utilised in the food industry. Similar energy losses occur where bioethanol was produced from sugar beet. Around 38.2 t/ha of beet tops and leaves (representing around 78,500 MJ/ha) were either used as animal feed or returned to land as fertiliser which regained approximately 11,000 MJ/ha through reduced mineral fertiliser use (European Commission, 1994). Beet pulp mass of up to 2.8 t/ha (dry matter) representing 32,300 MJ/ha could be recovered for use as animal feed, although there was a significant energetic benefit in recovering heat during burning of this dried pulp (European Commission, 1994). Stillage volumes of up to 123,820 l/ha required separation, drying and treatment.

The primary by product of the anaerobic digestion of crops was digestate. The composition of the digestate varies according to the input characteristics and digester operating regimes, however, for crops with a relatively high moisture content (e.g. perennial rye grass), digestate may have been equivalent to approximately 85% of the input mass. Of this mass, the digestate comprised around 95% water. Digestate is a useful by product as it contains the majority of the nutrients from the source crop. As such it represented a fertiliser that could be spread to land, limiting the need for the use of mineral fertiliser. Using the digestate in this manner effectively recovers around 6,500 MJ/ha of energy through substituting of mineral fertilizer use (European Commission, 1994). Digestate also comprised a relatively small solid fraction, which generally included indigestible fibrous material. This could be separated and used as a soil conditioner or composted. Carbon dioxide was also produced as a waste gas during the anaerobic digestion processes. Assuming methane contents of around 55%, typical volumes of CO_2 generated would be around 1,285 m^3/ha for perennial rye grass, 3,934 m^3/ha for sugar beet and 6,575 m^3 / ha for maize. As this CO_2 would be absorbed from the atmosphere during plant growth and liberated on a short timescale, this did not represent an overall long term increase of atmospheric CO_2.

3.4 Combustion Emissions

The final environmental factor to be considered was the exhaust emissions from the biofuels following combustion in an internal combustion engine. As reduction in greenhouse gases has been one of the primary reasons for developing alternative energy carriers, it was important that atmospheric pollution following combustion was reduced. A review of literature indicated that research into exhaust emissions was variable in terms of the fuel used, the vehicle used, the parameters measured and the test conditions employed. Table 9 presents a summary of exhaust emission data from Beer (2000).

Table 9 - Exhaust Emission Data

GHG / Pollutant	Biodiesel	Bioethanol (E95)	CNG (Methane)
	g/km	g/km	g/km
CO_2	1948.35	2154	1343.51
CH_4	N/A	N/A	N/A
CO	11.41	20.62	5.33
NO_x	25.66	11.37	12.17
THC	1.21	7.02	6.9
Non Methane Hydrocarbons	N/A	N/A	N/A
Particulates	0.9	0.31	0.02

Average values of tailpipe emissions recorded for buses undergoing an urban (CBD) drive cycle on a dynamometer. (Beer et al., 2000)
N/A = Not analysed

Concentrations of CO_2, CO and particulates were significantly lower for CNG exhaust emissions than for biodiesel or bioethanol. NO_x concentrations for CNG and bioethanol were comparable, both being less than half that of biodiesel emissions. Concentrations of Total Hydrocarbons were again comparable between CNG and bioethanol, however biodiesel THC emissions were significantly lower.

Modern light vehicles are now generally fitted with catalytic converters to reduce exhaust emissions. These have been extremely successful in reducing concentrations of the principal emissions listed above (e.g. CO emissions for non catalytic gasoline fuelled vehicles was 25 g/km, whereas for a modern Category 3 vehicle with catalytic treatment it is just 0.1 g/km) (Nylund et al., 2000). As such for modern, light vehicles, a change in fuel type is unlikely to significantly reduce emissions any further.

Certain biofuels have specific emission characteristics that have been of some concern. In particular combustion of bioethanol / gasoline blends has shown to result in significantly increased acetaldehyde emissions (Niven, 2005). Acetaldehyde is a hazardous chemical, a suspected carcinogen and also a precursor to peroxylacetate nitrate (a respiratory irritant with acute toxicity and a known plant toxin).

The use of biomethane as a transport fuel raised a number of concerns over increased methane emissions, primarily due to un-combusted fuel. Emission tests undertaken on vehicles without catalytic exhaust treatment show that total hydrocarbon emissions for CNG vehicles (assumed to be equivalent to biomethane) were approximately half that of those fuelled by unleaded gasoline. However, the hydrocarbon emissions from the CNG vehicles comprised 92% methane where as the hydrocarbon emissions from gasoline fuelled vehicles comprised over 50% C1-C7 hydrocarbons and approximately 10% methane (Nylund et al., 2000). Further development of catalytic treatment of methane or complete methane combustion would therefore be required prior to the widespread utilisation of bio-methane vehicles in order to limit methane emissions from exhausts.

3.5 Discussion

Overall, biodiesel produced from rape seed oil performed the least well out of the fuels considered. Its energy balance was poor, primarily as it did not utilise the whole crop. In addition, a number of co-products were generated that, although they were potentially useful (e.g. glycerol), were proving difficult to utilise in an economic or sustainable way. However, biodiesel could be used in existing vehicles with minimum alteration, and, as a liquid fuel could also access existing distribution networks. These factors could make biodiesel more attractive to industry and end users in the short term. However, results of this study suggested that in the medium to long term, biodiesel produced from crops did not represent an efficient replacement of fossil transport fuels.

Bioethanol production using wheat grain also gave a poor energy balance, comparable to that of biodiesel, although the energy balance was improved significantly by utilising high carbohydrate crops such as sugar beet. A further major disadvantage of bioethanol was the large quantity of stillage produced. This took a large amount of energy to separate and dry in order to produce useful by-products, and the large volume of wastewater generated then required further treatment. Anaerobic digestion provided a means of recovering some of the energy from wastewater, however, where aerobic wastewater treatment systems were used further energy had to be invested. Emissions testing of E95 bioethanol fuel indicated an adequate performance, with emissions generally comparable to biodiesel and biomethane. Bioethanol also benefitted from being a liquid fuel that could be blended with existing fossil fuels and in this form could again be used with minimal or no vehicle alteration. As such, this would make bioethanol not only attractive to the consumer, but also to the petrochemical industry and national governments. Taking these factors into account, it was evident that crop based bioethanol provided a reasonable short term solution as an alternative transport fuel.

Biogas (methane) production using high carbohydrate crops such as sugar beet or maize appeared to be the most favourable option in terms of energy balance. Low carbohydrate crops such as perennial rye grass returned an energy balance broadly comparable to bioethanol from sugar beet, and could be of use in terms of providing diversity of crops, and in areas where grass production would be well understood (e.g. dairy farming). The primary co-product of anaerobic digestion was a liquid digestate. This could represent a fertiliser product and could be spread to land to provide nutrients for the next energy crop, providing a closed nutrient cycle. Solids within the digestate could also be returned to land as a soil conditioner. Exhaust emissions from biomethane were generally either lower than, or comparable to emissions from biodiesel and bioethanol. Although a cost comparison of the three fuels was not undertaken as part of this study, it was estimated that biogas could be produced from Rye Grass at a farm scale in the UK for between £0.27 - £0.55 per m^3. The additional infrastructure required to clean and compress the gas to vehicle fuel standard was only likely to be feasible at larger scale plants with an additional cost of approximately £0.1 per m^3. A study into the potential role of biogas as a vehicle fuel undertaken by the NSCA suggested a production cost from a centralized AD facility of £0.25 - £0.35 / m^3 (NSCA, 2006).

However, there are significant barriers against utilising biomethane as a transport fuel. Firstly, there are no recognised distribution and dispensing network for biomethane specifically as a vehicle fuel, and this will require a major investment programme to implement. However, there are examples within Europe such as Sweden where a nationwide distribution network has been developed and biogas is widely used as a transport fuel. The UK does have a very well developed natural gas network, and in the short term the feasibility of adding bio-methane to this network should be considered. The second major barrier is that vehicles would require conversion in order to use bio-methane, and new vehicles would have to be designed specifically to use bio-methane as a fuel. This is a well understood technology within Europe with bio-methane and dual fuel vehicles being widespread in a number of European countries such as Sweden, and major car manufacturers have the knowledge and technology to produce affordable biomethane powered vehicles. There are a number of large scale demonstrators across Europe where biogas is being upgraded to biomethane specifically for vehicle fuel use, such as Lille (France), Gothenburg (Sweden) and Stockholm (Sweden) (Pädam et al., 2010) However, these models are based on fairly large production facilities that co-digest food industry wastes (which generate additional income) with manures and municipal wastes. Bio-methane production using farmed crops alone may not be economically viable (CONCAWE, 2007) unless performed at large scale.

Flexibility of Production Technology

One key point to consider when trying to decide whether to invest at a national scale in a particular biofuel is the flexibility of the fuel and, perhaps more importantly, the flexibility of the production technology. In addition to adverse ecological impacts associated with fuel production based on limited crop types, it is unlikely that energy crops alone will be able to meet our growing needs for either transport fuels or other forms of energy including domestic electricity. As such, other feedstocks such as biodegradable wastes will become increasingly important in the future.

Production of biodiesel in the UK is limited to the use of grain crops and waste oils, a relatively small and specialised waste stream. Based on the energetic limitations of biodiesel production from crops shown above, and the limited availability of alternative feedstocks, the future large scale adoption of biodiesel as a transport fuel is considered unfavourable.

Current first generation bioethanol production could, in theory, utilise a wider range of high carbohydrate crops. In practice however, production is generally limited to the use of wheat grain and sugar beet as described above. Bioethanol production using ligno-cellulosic material such as waste wood or woody biomass is technically feasible, although still under development. This would open up a far larger portfolio of potential feedstocks, and has been shown to be energetically more favourable than current standard bioethanol production techniques in northern Europe (CONCAWE, 2007).

Fermentation technology has the potential to generate energy using a wide range of biodegradable materials, including biodegradable municipal waste, abattoir waste, manures and food production waste. When considering biofuels on a 'well to wheel' basis, biomethane production from manures or municipal waste was found to have a relatively high energy demand compared to gasoline production (CONCAWE, 2007). However, as the fuel was derived from low carbon feedstocks that constituted wastes, and the energy required during production was derived from the biomethane produced, the biomethane production had a very favourable fossil energy footprint. Additional investment in pasteurisation equipment would be necessary, as would the identification of a route for disposing of the digestate; however in principle this represented an extremely flexible technology capable of producing an energetically and environmentally beneficial transport fuel, as well as allowing conversion to electricity and heat. Converting biogas to electricity may provide a greater CO_2 replacement benefit than using it as a transport fuel, and the use of waste heat from electricity production would further enhance this benefit (NSCA, 2006). Biogas production also has the additional benefit of being compatible with state of the art and emerging technologies such as fuel cells utilising methane reformers. As such, biogas production technology could be considered to be the most versatile option for future fuel.

Land Use Pressures

Land that could be used for the growing of crops suitable for conversion to transport fuel is itself a limited resource. There are a number of growing concerns related to the use of large areas of agricultural land for the production of fuels that should be considered by policy makers.

1. As land is a finite resource that could otherwise be used for production of food, we have an obligation to use it efficiently and generate as much energy as possible from this land, not only through biomass but other renewables.
2. Ideally, energy generation should not be reliant on a single crop, but on a range of crop types. This allows energy to be produced across a greater area of the UK according to growing conditions, allows for production over a longer growing season, and reduces soil nutrient degradation.
3. There is also an obligation to protect biodiversity by avoiding large scale energy production based on single crop types.

Of the fuels considered above, biogas represented the largest energy yield per hectare, and was not reliant on a single crop but can be produced on a sustainable crop rotation basis (Amon et al., 2007) and even through the co-digestion of waste streams.

Total Contribution to Transport Fuel Use

Transport fuel use statistics for the UK in 2006 indicated that demand for petrol and diesel was 18.1 million tonnes and 20.1 million tonnes, respectively (BERR, 2007). This was equivalent to approximately 1.81×10^{12} MJ of energy. Taking the energetically best performing biofuel options (i.e. biogas), the theoretical contributions which crops could make to this total, and the land areas required can be seen in Table 10.

Using set aside land only, even the most energetically favourable biofuel would contribute only 2.87 – 7.18% of the total transport fuel required based on 2006 demand. In order to meet 100% of this demand, between 32 – 80% of the UK land area would have to be dedicated to biogas production. Clearly, this is not feasible and it demonstrates that significant social change, efficiency improvement and technology development is required before our reliance on fossil transport fuels is addressed. In particular other renewable resources such as wind, PV, wave or tidal energy generation will need to be utilised either through electric vehicles or electrolysis to hydrogen (Sherif et al., 2005; Conibeer et al., 2007; Mignard et al., 2007). This also demonstrates how utilising less energetically efficient biofuel options such as biodiesel and bioethanol is prolonging this reliance by limiting the amount of fossil fuels displaced.

Table 10 - Potential Contribution of Bio-methane to total UK transport Fuel Demand and Biofuels Directive Target

Crop	Energy / ha (MJ)	UK Set aside area (ha)	Biofuel Energy available (MJ)	Contribution to 2020 target of 10%*	% of total petrol & diesel energy demand*	Area required for 100% of petrol & diesel energy (ha)	% of UK land area required to meet 100% demand
Grass	93,167	559,000	5.2×10^{10}	28%	2.87	2.1×10^{7}	80%
Sugarbeet	128,790	559,000	7.2×10^{10}	40%	3.98	1.5×10^{7}	58%
Maize	237,011	559,000	1.3×10^{11}	72%	7.18	8.2×10^{6}	32%

* In relation UK statistics for 2006 (BERR, 2007).

Additional Biofuel Options

The production of hydrogen from biomass represented an area of considerable ongoing research. A theoretical assessment of energy output from hydrogen and methane production using a combination of dark fermentation followed by anaerobic digestion has been undertaken (Martínez-Pérez et al., 2007). Table 11 provides a summary of the results for crops relevant to this study.

These results indicated that energy yields from bio-hydrogen and methane production were greater than for biodiesel and bioethanol from wheat, and were comparable to those of single stage biomethane production. However, these results were theoretical and did not take into account the additional compression energy required to utilise the gas as a transport fuel. The most energy efficient way to achieve this would likely be to produce, clean and compress a hydrogen-methane blend as a single gas stream. Co-products from the production process would be similar to those from biogas production discussed previously, i.e. predominantly liquid digestate and undigested fibres. In terms of combustion emissions from hydrogen-methane blends, studies to date indicate a significant reduction in typical greenhouse gas emissions as exemplified in Table 12. Studies were undertaken under differing conditions to those of biodiesel, bioethanol and biogas and therefore direct comparison was not possible, however, a reduction in emissions was apparent.

Table 11 - Theoretical Energy Output from Bio-Hydrogen and Methane Production

Crop	Energy Output from H_2 (MJ/ha)	Energy Output from CH_4 (MJ/ha)	Total Gross Energy Output (MJ/ha)	Net Energy Output (MJ/ha)
Perennial Rye Grass	3,140	115,759	118,899	114,189
Sugar Beet	18,853	112,017	130,871	112,624
Forage Maize	13,429	125,723	139,152	121,522

Source: From Martínez-Pérez et al. (2007)

The production of hydrogen, methane, and hydrogen-methane blends from biomass crops and wastes has been proven at laboratory scale and work is ongoing to demonstrate the technology at pilot scale in the UK (Antonopoulou et al., 2007; Hawkes et al., 2007). Similar work is ongoing in other research groups within Europe (Lin et al., 2010; Cavinato et al., 2012). Small scale, two stage fermentation and anaerobic digestion has also been demonstrated at an industrial scale in Japan (Japanfs.org, 2007). Sapporo Breweries Ltd., Shimadzu Corp and Hiroshima University operated a small scale system generating hydrogen and methane from bread waste over a six month period. On a calorific basis the process generated around 10% more biogas than a conventional single stage digestion system. These studies also indicate that a two stage fermentation / anaerobic digestion system degrades the biomass substrate in significantly less time than a single stage. A study by Massanet-Nicolau et al (2013) also found an energetic advantage to the two stage fermentation process when treating high ligno-cellulosic wheat feed co-product from flour milling. Whilst the fundamental technology is therefore promising, the economic and energetic advantages of hydrogen /methane production are less clear at this stage. These advantages will become clearer as end use technology such as hydrogen fuel cells, (Oh et al., 2005) microbial fuel cells (Thomas et al., 2000) and hydrogen / methane transport fuels (Sierens et al., 2001) undergo further development.

Table 12 - Exhaust Emission Data for Hydrogen-Methane Blend

GHG / Pollutant	**$H_2 + CH_4$**		
	g/km[a]	g/km[b]	g/km[c]
CO_2	272.99	N/A	N/A
CH_4	0.05	10.08	7.63
CO	0.158	BDL	BDL
NO_x	0.048	2.86	2.24
THC	0.11	N/A	N/A
NMHC	0.014	0.34	0.18
Particulates	N/A	0.0034	0.0062

[a] Average figure for Ford F150 running on 28% hydrogen by volume (remaining fuel comprising CNG) completing 7 No. US FTP-75 road tests (Karner et al., 2003).
[b] Average figure from 2 No. buses running on 20/80 H_2/CNG blend subject to OCTA2X road tests (Del Toro et al., 2005).
[c] Average figure from 2 No. buses running on 20/80 H_2/CNG blend subject to CSHVR road tests (Del Toro et al., 2005).
N/A = Not analysed
BDL = Below Detection Limits

Second generation bioethanol production techniques will be able to utilise whole crops rather than just the grains, and as such energy balances should be significantly improved. Bentsen et al. (2006) provided a theoretical energy balance for the Integrated Biomass Utilisation System (IBUS) which combined second generation ethanol production from winter wheat with a power plant for combined heat and power generation. The simulation required a total (median) energy input of 65,927 MJ/ha to yield an output (median) of 133,962 MJ/ha giving a positive balance of 68,035 MJ/ha. However, it should be noted that of the energy yield of 133,962 MJ/ha, ethanol accounted for only 79,761 MJ/ha with the remaining energy gained from the utilisation of by products including DDGS (13,891 MJ/ha), biomass utilised for heat production (29,295 MJ/ha), and C-5 Molasses (3,133 MJ/ha). The report concluded that winter wheat should only be utilised in the short term, with alternative sources of biomass providing a greater energy input / output balance. An evaluation of the environmental burdens associated with utilisation of short rotation coppice willow for either bioethanol or for power generation (via gasification) found that whilst bioethanol delivered the best energy yield, the generation of power resulted in higher GHG savings (González-García et al., 2012).

During the preparation of this study, it became evident that there was a lack of comparable data, both between fuels and even when considering different studies of a single fuel. This was particularly true when comparing energy balances and combustion emissions. Processes and technologies are developing rapidly, difficult targets need to be met in the short to medium term, and the long term impact of climate change needs to be mitigated. In this context it would be beneficial to undertake detailed lifecycle analyses of these biofuels, and other emerging power sources, in consultation with industry and national government agencies on a national basis. When considering the full range of biodegradable feedstocks available for fuel and energy production (i.e. biodegradable waste products), again, anaerobic fermentation and biogas technologies appear to be the most flexible option.

3.6 Conclusions

A range of biofuels produced from crops grown in the UK were compared in terms of their energy balance, waste / co-products and exhaust emissions. Biomethane from the anaerobic digestion of crops was found to have a more favourable energy balance for the production of transport fuel than biodiesel or bioethanol (237,011 MJ/ha compared to 24,185 MJ/ha and 77,264 MJ/ha respectively). Tailpipe emissions were superior for methane with lower emission levels of CO, CO_2 and particulates, and lower NO_X levels than biodiesel which are comparable to bioethanol.

However, the lack of an established distribution network and the requirement to convert vehicles to use biogas provides significant, but not insurmountable, barriers that are primarily socio-economic rather than technical. Where land is to be used to produce energy or transport fuels rather than food, it is believed that we have an obligation to use this land in as efficient a way as possible by utilising the technology that yields the most energy.

Further feedstocks should also be considered, in particular the conversion of organic wastes to energy or transport fuels. Anaerobic digestion or second generation bioethanol production are both good candidates for utilising this valuable feedstock, but further research work needs to be undertaken on both options to determine which is technically and environmentally favorable.

Chapter 4: An Evaluation of the Policy and Techno-Economic Factors Affecting the Potential for Biogas Upgrading for Transport Fuel Use in the UK

4.1 Introduction

As discussed in the previous chapter, the Biofuels Directive (2003/30/EC) (European Parliament, 2003) set out a regime for promoting the use of biofuels or other renewable fuels to replace diesel or petrol transport fuels within EU Member States. By 2010, 5.75% (by energy) of these traditional transport fuels was to comprise either biofuel or another renewable component, and was set to increase to at least 10% by 2020. The UK's main strategy for developing biofuel use is the Renewable Transport Fuel Obligation (RTFO) (UK Government, 2007). Initial UK RTFO targets were a biofuel content of 3.75% in 2009/10 increasing to 5% by 2010/11. In response to the primary conclusions of the Gallagher Review of the indirect effects of Biofuel production (Gallagher, 2008) the UK Government has since revised these targets to 3.25% in 2009/10, 3.5 % 2010/11, 4 % in 2011/12 and 5% in 2013/14 (UK Government, 2009).

Previous studies, and the work described in the previous chapter, showed that biomethane delivered greater environmental benefits than either biodiesel or first generation bioethanol (Auer et al., 2006; Patterson et al., 2008; Murphy et al., 2009; Smyth et al., 2009). The ability to utilise a wide range of feedstocks such as biodegradable commercial, industrial and municipal wastes represents another potential advantage of gaseous fuels produced by anaerobic digestion. However, the requirement to upgrade the biogas to biomethane of adequate quality for transport fuel use, compression of the gas for storage and transport, and the lack of refuelling infrastructure were considered to be significant barriers to the deployment of biogas based vehicle fuels in the UK. There is currently no UK quality standard for biomethane fuels derived from biogas, however, national standards have been developed for grid injection of biogas or utilisation as a vehicle fuel in Sweden (Swedish Government, 1999), Switzerland (SVGW, 2008), Germany (DVGW, 2000a; DVGW, 2000b) and France (Gas de France, 2007).

This chapter aimed to assess the technical and economic performance of current biogas upgrading technologies. This was achieved through a review of available literature to establish, where possible, the energetic requirement of upgrade technologies, the cost per unit of upgraded biogas, and the loss of methane that can be expected for each technology type. The economic viability of upgrading biogas for direct use as a transport fuel, in particular the impact of the current and future RTFO regime, was then assessed.

4.2 Biogas Upgrading Technology Review

The technologies currently being utilised or developed for the upgrade of biogas include adsorption, absorption (physical and chemical), permeation and cryogenic. These technologies focus on the separation of methane (present at around 50-70% by volume in raw biogas) and carbon dioxide (25-45% in raw biogas). Whilst several of the technologies can also remove moderate concentrations of other contaminants, the majority require the reduction of high concentrations of contaminants such as water, H_2S and siloxanes (if present) in a pre-upgrade stage. It was noted that several of the case studies below incorporate significant over capacity within their upgrading capabilities. This was most likely to accommodate periods of down time associated with the maintenance of equipment and to allow for the developmental nature of some of the plants.

4.2.1 Pressure Swing Adsorption

Pressure Swing Adsorption (PSA) is a versatile technology for the separation and purification of gas mixtures (Sircar, 2002). Since the development of the process in the 1960s it has become one of the most widely used industrial gas separation technologies, primarily as a result of its flexibility, relatively low capital cost and efficiency.

PSA processes are based on the ability of various adsorbent materials to selectively retain one or more components of a gas mixture under varying pressure conditions. These adsorbent materials are highly porous and separate gas components under high pressure according to molecular size. In the case of CH_4 (molecular size of 3.8 angstroms) / CO_2 (molecular size 3.4 angstroms) separation is achieved by using an adsorbent with a pore size of 3.7 angstroms. Carbon dioxide is therefore allowed to enter into the matrix of the absorbent material and is retained, whilst methane is not allowed to enter the material but passes through interstitial spaces (Gladstone, 2007). The adsorbed component of the gas stream is then desorbed from the solid adsorbent by reducing the pressure, therefore allowing the regeneration and re-use of the adsorbent material (Sircar, 2002; Cruz et al., 2005). Adsorbents are packed into columns which are then arranged in sequence according to the gas components which require separation, or the level of output gas concentration required.

The reason that PSA technology is so flexible is the wide range of adsorbent materials available to separate the components of various gases and liquids. Adsorbent materials being utilised and developed include activated carbon (Sircar et al., 1996; Siriwardane et al., 2001), natural zeolites (alumina silicates) (Ackley et al., 2003; Siriwardane et al., 2003), synthetic zeolites (Inui et al., 1988; Sherman, 1999), activated aluminas (Alpay et al., 1996), silica gels (Lou et al., 1999) and polymeric sorbents (Kikkinides et al., 1993). The ability to combine various adsorbents within the overall PSA process provides added flexibility.

Where high concentrations of contaminants such as H_2S or siloxanes are present in the raw biogas, initial removal / reduction of these may be required prior to upgrading with PSA. This is because at high concentrations these contaminants cannot be desorbed from the adsorbent media.

The anaerobic digestion plant at Pliening (Germany) processes around 40,000 tpa of maize and other forage crop silage to generate around 920 Nm^3/hr of biogas (Schmack Biogas AG, 2007). This is upgraded using PSA incorporating carbon molecular sieve adsorbent (CarboTech AC GmbH) to >96% CH_4 content before being injected into the local gas grid.

Austria's first biogas injection project located in Pucking, Upper Austria generates raw biogas from the anaerobic digestion of chicken and pig manure. Approximately 10 m^3/hr of raw biogas is produced which is upgraded using PSA incorporating carbon molecular sieves (Linsbod, 2005). The resulting 6 m^3/hr of upgraded biogas (>97% methane), which is enough to supply biomethane to around 40 flats, is then injected into the local gas distribution grid. Unwanted sulphurous contaminants are removed in a preliminary stage using an activated carbon filter.

Sweden represents the most advanced nation in terms of the deployment of biogas upgrading technology with 32 biogas upgrading plants, 7 of which are based on pressure swing adsorption processes (Petersson, 2008).

In addition to biogas from AD, there is also ongoing research into the application of PSA for the upgrading of landfill gas (Cavenati et al., 2005) and field based demonstrations of this application (QuestAir Technologies Inc., 2006; QuestAir Technologies Inc., 2007).

4.2.2 Water Scrubbing

Water scrubbing or absorption in water is the most widely used gas upgrading technology in Sweden with a total of 15 plants in operation or under construction in 2007 (Persson, 2007). The method is used to remove CO_2 from the raw biogas (therefore increasing the methane content) and is also effective at removing H_2S. The method relies on the basic principle that CO_2 (and H_2S) are more soluble in water than CH_4. Any condensed moisture or particulates present within the raw gas stream are removed prior to water scrubbing. The raw gas is then pressurised (to around 9-12 bar) and introduced to the bottom of the scrubbing tower whilst water is flushed into the top of the tower. The scrubbing tower is packed with a high surface area media (e.g. pall rings) to provide a high contact area between gas and water. As the raw biogas moves up the column against the flow of water, CO_2 and H_2S become dissolved within the liquid stream (Persson et al., 2006a). Upgraded gas leaves the top of the column. Any methane dissolved within the water is usually captured by depressurising the water to 2-4 bar within a flash tank. Gases released are then returned to the bottom of the column (Håkansson, 2006). Upgraded gas is then available for drying and compression (to around 200 bar) for storage. Scrubbing water can be used once in a single pass system, or re-circulated following removal of dissolved gases.

The biogas plant in Linköping (Sweden) is one example where water scrubbing is used to upgrade biogas for vehicle fuel use. The plant digests around 45,000 tonnes per year of slaughterhouse waste (c. 55%) and food waste (c. 45%) in a mesophilic one stage process with a 30 day retention time (Swedish Gas Centre, 2008a). An upgrading plant based on water scrubbing with a capacity of 500 Nm^3/hr was opened in 1997, and a second water scrubbing plant was installed in 2002 with a capacity of 1,400 Nm^3/hr (Swedish Gas Centre, 2008a). With the addition of the biogas from an adjacent sewage treatment plant (upgraded using PSA with 150 Nm^3/hr capacity) a total of 65,000 MWh of upgraded biogas is produced annually which supplies the town's buses, refuse vehicles and a number of public filling stations (Swedish Gas Centre, 2008a).

In Lille (France) a pilot project (1994-1999) demonstrated the upgrading of surplus biogas from the digestion of sewage sludge and its use in local bus fleets. Following the success of the trial the decision was made to phase out diesel buses and replace these with biogas vehicles. As of the beginning of 2007, there were 200 gas powered buses in operation (fuelled by a mixture of upgraded biogas and CNG). Biogas is generated from the digestion of biodegradable municipal waste at a dedicated Organic Recovery Centre (ORC). Raw biogas is upgraded in two water scrubbing towers each with a capacity of 600 Nm^3/hr with an annual production of 4 million Nm^3/yr (Persson et al., 2006a). A new biogas bus depot with 100 buses was constructed adjacent to the ORC facility.

4.2.3 Physical Absorption

In physical absorption processes a non reactive fluid is used to physically absorb the unwanted component of the gas stream. Spent absorbents are then regenerated by depressurising and / or heating. The most widely used absorbent for biogas upgrading available on the market is Genosorb 1753 which is used in the Selexol™ process. The solvent, manufactured by Clariant, is a mixture of dimethyl ethers and polyethylene glycols and can remove H_2S, CO_2 and moisture from gas streams.

The biogas facility at Laholm on the western coast of Sweden produces around 2.4 million m^3 of methane from the anaerobic co-digestion of up to 70,000 tonnes per year of manure, abattoir, industrial and household waste (IEA, 2005). The raw biogas has a methane content of around 75% and this is upgraded to natural gas quality by Selexol™ scrubbing (500 Nm^3/hr capacity) following sulphur removal. The Wobbe index is adjusted to that of natural gas by adding 5-10% propane. The upgraded gas is then added to the local gas grid (including refuelling stations) and is used to power a local district heating scheme.

Selexol™ is also used for gas upgrading at the McCarty Road landfill in Texas, USA. The plant has a capacity to upgrade approximately 10,618 Nm^3/hr of raw landfill gas to produce 5309 Nm^3/hr of upgraded gas (Montauk Energy, 2008). The gas meets the demand of around 15,000 homes in the Houston area.

4.2.4 Chemical (Amine) Scrubbing

A further variation on scrubbing technology is to use amine based chemicals as the solvent. Organic amines such as monoethanolamine (MEA), diethanolamines (DEA) and diglycolamines (DGA) are used as they are not only highly selective at absorbing CO_2 but can dissolve significantly more CO_2 per unit volume when compared to water scrubbing, leading to smaller volumes and plant sizes. Amine scrubbing is also effective at lower pressures compared to water and Selexol™ scrubbing leading to reduced compression energy requirements, however, some heat input is required to regenerate the amine solution prior to recirculation. Due to the highly selective nature of the amine solvent less CH_4 absorption occurs and overall CH_4 losses are reduced.

The Gryaab biogas facility in Gothenburg, Sweden treats around 430,000 Nm^3/yr of thickened sludge from a local wastewater treatment works along with grease trap waste and food waste using single stage anaerobic digestion (Swedish Gas Centre, 2008b). This produces around 60,000 MWh / yr of raw biogas which is sold to Göteborg Energi for upgrading. Upgrading is largely undertaken at a facility in Arendal which uses Cooab (an amine based solvent) to remove carbon dioxide before it is regenerated for re-use. A small amount of propane is added to bring the energy content up to natural gas standard. The capacity of the plant is approximately 1,600 m^3/hr and upgraded gas is distributed to the city gas pipeline network and to a network of local vehicle filling stations.

More recently, upgrading plants utilising the Cooab process have been commissioned at Falkenberg in Sweden (800 Nm^3/hr) and Stavenger in Norway (500 Nm^3/hr) (Solomon et al., 2007).

4.2.5 Membrane Separation

Membrane separation relies on the preferential transfer of one gas from a mixture through a semi permeable membrane, whilst other components are retained. Membranes can be grouped into two types; high pressure membranes which have gases present on each side of the membrane, and low pressure systems which have a liquid adsorbent on one side of the membrane wall. High concentrations of contaminants such as H_2S and moisture are generally reduced prior to separation of CH_4 and CO_2 in a membrane system.

High pressure membrane separation is undertaken at around >20 bar, although some systems can operate at 8-10 bar (Persson et al., 2006b). Biogas is generally upgraded in a multiple stage process to yield a final CH_4 concentration of >96%. Waste gases from the first stages are recycled within the process to enhance CH_4 capture whilst waste gas from the final stage (which may contain 10-20% CH_4) is either flared, used for heat production (Wellinger et al., 1999) or captured catalytically. This technology has been applied for some time for the upgrading of natural gas.

Low pressure membrane systems work at close to atmospheric pressure. A micro porous hydrophobic membrane separates the raw gas stream from a liquid phase absorbent. Absorbents such as NaOH (e.g. for H_2S separation) or heat regenerative amine solutions (e.g. for CO_2 separation) are used. CH_4 concentrations of >97% are possible and the process can yield high purity CO_2 that can be sold as a product.

A novel membrane gas upgrading system has been demonstrated at a biogas plant in Bruck / Leitha in Lower Austria (Miltner et al., 2008). Hollow fibre membranes are used to separate methane from CO_2 with a pressure differential of around 8-9 bar across the membranes. Two stages of membrane separation are employed with permeate from the first stage being utilised in the biogas plant's CHP engine, and permeate from the second stage which contains a higher percentage of CH_4 being recycled back through the separation process. In this way, methane losses to atmosphere are limited. Upgraded biogas with methane concentration of 98% is fed to the local gas grid. Whilst the process is capable of removing small concentrations of H_2S, pre-treatment to remove the majority of H_2S prior to membrane separation has been employed at the demonstration facility.

4.2.6 Cryogenic Technique

The technology relies on the principle that different constituents of a mixed gas stream have different boiling points. For example methane has a boiling point of -160 °C at atmospheric pressure where as carbon dioxide has a boiling point of -78 °C. Therefore by progressively cooling the raw gas under pressure, each of the constituents will condense to a liquid at different temperatures and can be separated. Cooling is achieved by compression of the gas stream, cooling with heat exchangers followed by expansion, for example in an expansion turbine, to condense the target contaminant (e.g. CO_2) (Persson et al., 2006b). High purity CO_2 is produced which can be sold as a product. A pilot cryogenic upgrading plant has been operational in the Netherlands since 2009 and a commercial scale plant has been upgrading landfill gas in the USA since 2006 (Petersson et al., 2009).

4.3 Summary of Energetic Performance of Biogas Upgrading Techniques

The amount of energy required to upgrade raw biogas to biomethane is a key consideration when selecting a technology. The lower the energetic requirement for upgrading, the more net energy is available for end use. The energetic performance of each of the technologies is summarised in Table 13. Data has been gathered from a range of industrial and academic literature.

Table 13 – Summary of energetic requirements of biogas upgrading technologies

Technology	Energy Requirement (kWh / m^3 of upgraded biogas)					
	(Persson, 2007) [a]	(Beil, 2009)	CarboTech [b] (Berndt, 2006)	DGE GmbH [b] (Günther, 2007)	(Benjaminsson, 2006)	(Miltner et al., 2008)
PSA	0.5-0.6 (0.3-1.0)	0.24	0.335	0.285	-	-
Water Scrubbing	0.3 (0.45 – 0.9)	0.2	0.43	0.391	-	-
Chemical (Amine) Scrubbing	(0.15)	0.12 (elec.) 0.44 (therm.)	0.646	0.126[c]	-	-
Physical Absorption	0.4	-	0.49	0.511	-	-
Membrane Separation	-	0.19	0.769	-	0.27 (Low Pressure)	0.378
Cryogenic	-		-	-	0.42	-

[a] Figures reported are from operational plants. Figures in brackets () are from manufacturers
[b] CarboTech manufacture carbon molecular sieves used in PSA plants, and DGE GmbH design and build PSA plants
[c] This figure also accounts for methane losses and regeneration energy.

Data for the energetic requirements for relatively well established technologies such as water scrubbing appear relatively consistent with a range of 0.20 – 0.43 kWh/Nm^3. Data for PSA was less consistent with a range of 0.24 – 0.6 kWh/Nm^3. Similarly, there was a wide variation in the energetic requirement for amine scrubbing potentially due to the inclusion or omission of the thermal energy required to regenerate the amine absorbent, although data relating to the physical absorption process was more consistent. Membrane separation data displayed high variability although this most likely was a result of variations in membrane types and operating pressures. A report prepared by the German Energy Agency (DENA, 2009) presents a similar range of energy consumption to those shown in Table 13. The variation of energy use both across and within technology groups suggests that opportunities still exist for detailed independent assessment and the optimisation of energy use within upgrading plants.

4.4 Methane Losses

A further key parameter when considering the economic and environmental performance of upgrading technologies is the level of methane losses associated with the upgrading process. Any methane lost in the process not only represents lost revenue, but, as CH_4 has a Global Warming Potential (GWP) 25 times greater than CO_2 (Solomon et al., 2007), high methane losses also represent a significant contributor to climate change.

As shown in Table 14, available data relating to methane losses associated with upgrading processes was extremely limited. Whilst equipment manufacturers acknowledge that losses do occur with each of the above technologies it is clearly not in their interest to overstate their significance. The ideal way to quantify these losses would be through independent site monitoring. Some manufacturers allow a certain percentage of methane to remain in the waste gas in favour of high product gas quality. Where this is the case, exhaust gas which may contain 1-4% CH_4 can be blended with higher methane content gases for combustion in a CHP (Miltner et al., 2008) or boiler plant, or captured within a catalytic converter.

Table 14 – Summary of CH_4 losses associated with biogas upgrade technologies

	Methane losses (%)			
	CarboTech (Berndt, 2006)	**DGE (Günther, 2007)**	**(Persson, 2007)**	**(Benjaminsson, 2006)**
PSA	Medium	5.5	<2[a]	-
Water Scrubbing	Medium	4.7	<2[a]	-
Chemical (Amine) scrubbing	Low	0.03	<0.1	-
Physical Absorption	High	13.75	<2	-
Membrane (High Pres.)	High	-	-	-
Membrane (Low Pres.)	-	-	-	<1.5
Cryogenic	-	-	<2[b]	<2

[a] Manufacturers figures – higher losses have been noted at some plants (Persson, 2007)
[b] Manufacturers figures – losses of 10-18% have been noted at some plants (Persson, 2007)

In order to limit impact from excessive methane losses, the tariff system introduced in Germany in 2008 (EEG 2009) (Eyler, 2009) stipulated that to receive the biogas upgrading bonus methane losses within the upgrading process must be shown to be less than 0.5% which, given the above data, only amine scrubbing appears to consistently meet at the present time.

Whilst the minimisation of methane losses during upgrading are important, these should be considered alongside the potentially much more significant methane losses which may occur due to operational factors associated with biogas production such as incomplete stabilisation or unnecessary release of biogas during digestate storage. A laboratory investigation of the co-digestion of crops and manures at a loading rate of 2 kg VS m^{-3} day^{-1} and a HRT of 20 days indicated that digestates would have a further methane potential of 0.9-2.5 m^3/t (ww) (12-31% of total methane production) in northern climates (Lehtomaki et al., 2007), a proportion of which would be released to atmosphere when spread on land. A laboratory study investigating the co-digestion of source segregated food waste and manures found that potential methane emissions from digestates during storage at 5°C for 1 year were up to 10% of the total methane potential of substrates and up to 28% of the methane potential of the digestate itself (Paavola et al., 2008). A study of full scale biogas plants operating under varying conditions indicated that between 5 – 15% of methane was collected during digestate storage (Angelidaki et al., 2006). This illustrates that optimising and managing the whole bio-methane production process including effective digestate storage and use is required in order to generate maximum economic value and minimum environmental impact.

4.5 Economic Assessment of Upgrading Technologies

Beil (2009) undertook an analysis of the technical availability and maintenance costs associated with various upgrading technologies for a 1,000 m^3/hr (raw biogas) upgrading plant, the results of which are summarised in Table 15.

Data in Table 15 indicate that availability of membrane separation, water scrubbing and physical absorption systems were the highest. The lowest maintenance costs were associated with water scrubbing and membrane separation. A report by the German Energy Agency (DENA, 2009) suggested annual maintenance costs may be 42% lower for PSA, 200% higher for water scrubbing and 10% lower for amine scrubbing than the value presented below.

Table 15 – Technical Availability and Maintenance Costs of Upgrading Technologies (From (Beil, 2009))

	Technical Availability per year (%)	**Maintenance Cost (€/yr)**
PSA	94	56,000
Water Scrubbing	96	15,000
Chemical (Amine) Scrubbing	91	59,000
Physical Absorption	96	39,000
Membrane Separation	98	25,000

Data regarding the biomethane production costs using the various upgrading processes considered was limited. A review of the literature identified the following data related to plants with an output of between 200 – 300 m^3 / hr biomethane (Table 16). This range was chosen to limit the cost variations associated with economies of scale. This was also approximately the size of plant required to process the biogas generated from a 35,000 t/yr digestion facility primarily treating source segregated food waste (SSFW) which was the average size of AD plant treating SSFW in north western Europe in 2007 (Monson et al., 2007).

Table 16 demonstrates that costs for widely used technologies such as PSA and water scrubbing varied by up to 100% and that there was a lack of data relating to the costs associated with less widely used technologies. This data suggested that lowest costs were associated with water scrubbing, however, given the limited amount of data and varying sources it may be considered as indicative only. A report prepared by the German Energy Agency (DENA, 2009) presented data which was in broad agreement with Table 16.

Table 16 – Cost estimates of upgrading biogas to biomethane from studies undertaken 2007 - 2009

	Cost per m^3 of upgraded biogas (€/m^3)			
	(de Hullu et al., 2008)	**(Persson et al., 2007)**	**(Jönsson, 2009)**	**(Hammer et al., 2007)**
PSA	0.26			-
Water Scrubbing	0.15	0.11 – 0.16	0.11 – 0.22	0.11[a]
Chemical (Amine) Scrubbing	-			-
Physical Absorption	-	-	-	-
Membrane (Low Pres.)	0.22	-	-	-
Cryogenic	0.40	-	-	-

[a] Calculated from reported Capital and Operational costs of the Falköping upgrading plant assuming 5% interest rate and 10 year depreciation period using the method described in de Hullu et al., (2008).

In order to assess the economic potential of biogas upgrading in the UK it was necessary to consider these costs in conjunction with those of producing the biogas in the first instance. Given the ongoing expansion of AD in the UK for the treatment of source segregated municipal food wastes, primarily to meet 2012-13 landfill diversion targets, this feedstock was considered. A previous assessment of the use of biogas as a transport fuel (NSCA, 2006) used data from Sweden to arrive at a biogas production price of £0.11 – £0.18 / m^3 (€0.13 – €0.22 / m^3). Consideration of a 35,000 t SSFW / yr capacity facility under current UK market conditions suggested that this figure was likely to be an underestimate with a more realistic value being £0.20 - £0.25 / m^3 (€0.24 – €0.30 / m^3) given the assumptions listed in the following description of economic scenarios. Adding an upgrading cost of €0.18 / m^3 (based on Table 3) and a further compression cost of €0.08 / m^3 (NSCA, 2006), a total cost for the production of upgraded biomethane was estimated as €0.5 – €0.56 / m^3 (£0.41 – £0.46 / m^3). This represents a basic production cost and does not allow for addition distribution costs or profits.

At the time of writing (March 2010) the average cost of low sulphur diesel in the UK was £1.16 / litre (€1.39 / l). Excluding VAT (17.5%) and fuel duty (£0.56 / €0.67 / l) left a basic retail price for diesel of £0.45 / €0.54 / l). This included the costs required to transport and retail the fuel at the forecourt as well as the profit margins of the retail and oil companies.

As 1 litre of diesel is approximately energetically equivalent to 1 m^3 of upgraded biogas it was likely that upgraded biogas is not yet competitive with petroleum fuels in a market without fiscal incentives for biogas.

There are however a number of financial incentives acting in the UK to promote the uptake of biofuels. These are discussed below.

Fuel Duty Differential

Historically, the primary incentive for the production of biofuels in the UK has been a reduction in fuel duty (tax) compared with that levied on diesel or unleaded petrol, the so called ‘duty differential’. For liquid biofuels (biodiesel and bioethanol) a £0.2 / litre duty differential is due to be withdrawn in 2010, at which point the Renewable Transport Fuel Obligations (UK Government, 2007) (discussed below) will be the only mechanism incentivizing liquid biofuel production in the UK. A duty differential equivalent to £0.4 / l has been applied to the production of biomethane (and CNG) as a transport fuel and this will remain in place until April 2013 (Her Majesty's Revenue and Customs, 2007).[1]

Renewable Transport Fuel Obligation

The Renewable Transport Fuel Obligation (RTFO) was the UK’s primary approach to encouraging UK fuel suppliers to increase the proportion of bio-fuels supplied to the market with the overall aim of meeting the requirements of Directive 2003/30/EC (European Parliament, 2003) and subsequent amendments.

The RTFO was modelled on the existing Renewables Obligation Certificate (ROC) (Renewable Energy Agency, 2009) scheme that already existed in the UK for the production of green electricity. The Obligation requires transport fuel suppliers to increase the proportion of bio-fuels supplied to the market in line with the targets presented in the introduction of this study, and offers tradable Renewable Transport Fuel Certificates

[1] At the date of publication (May 2013) Duty for 1st Generation bioethanol and biodiesel was equal to that due on petrol and diesel (58 p/l), whilst for Biomethane the rate was 24.7 p/kg.

(RTFCs) to demonstrate compliance. RTFCs issued to a company can be 'banked' for future use or sale, although only 25% of a company's RTFO annual target can be met by using its own banked certificates. Companies which do not meet the targets for the proportion of biofuel supplied to the market can buy RTFCs at a market rate from companies that have exceeded their target, and therefore achieve compliance in this way. In the event that all fuel companies meet their RTFO targets, the RTFCs will in effect have no value. Conversely, if companies do not meet the required targets RTFC's will acquire value through trade on an open market, and the greater the shortfall the greater the demand and value.

The UK Government has included a third option in which companies can meet their RTFO obligations. A fixed 'buy-out' price for each unit of shortfall between the target and the actual volume supplied to the market has been set, and companies can opt to pay this 'buy out' fee rather than either supplying biofuels to the market or purchasing RTFCs in the open market. The funds collected through the Buy Out fees are then redistributed to those redeeming or surrendering RTFC's, therefore providing a financial incentive for producing biofuels. However, the Buy Out fee also has the effect of limiting the maximum value of an RTFC to the value of the 'buy-out' fee. For 2008/9 and 2009/10 the buy-out fee was set at £0.15 / litre (or 0.15p/kg for gaseous fuels) such that the total incentive (Buy Out + Duty Differential) was £0.35 / litre. Whilst the Chancellor guaranteed that the total incentive for 2010/11 will be at least £0.30 / litre (most or all of which will be based on the Buy Out value), it is worth noting that this is the maximum attainable value of RTFCs in an open market and their actual value, and therefore the financial incentive, could be much lower.
For example, in 2008/2009 the RTFO target of 2.5% biofuels reaching the market place was met, although this was partially due to an error in the drafting of the legislation which allowed the fossil fuel element of blended biofuels imported into the UK to be excluded from calculations. This had a significant market impact as it immediately created a situation where there was oversupply of biofuels in the UK market place and production therefore slowed down. In addition, sufficient RTFCs had been issued not only to meet the 2008/09 target, but also to meet the maximum 25% bankable allowance for the following financial year of 2009/10 (Renewable Energy Agency, 2009). There was therefore limited market demand to purchase excess RTFCs, and, as production targets had been exceeded, no

central Buy Out fund could accumulate. RTFCs issued were in effect worthless on an open market and any investment decisions or business plans associated with the biofuels industry as a whole that had attached a value to the RTFCs had to be re-considered.

Clearly there are significant implications for the whole of the biofuels industry. Although in theory the Duty Differential in place until April 2013 provides some protection for biomethane used as a vehicle fuel from variations in RTFC values, in practice very few upgrading plants for vehicle fuel generation will have been commissioned in the UK by this date. In order to determine the feasibility of upgrading biogas in the UK on a significant scale an assessment will have to be made of the future structure and requirements of the RTFO scheme.

The Future of the RTFO

Beyond 2011 the RTFO in the UK evolved further as the requirements of the Renewable Energy Directive (RED) (European Parliament, 2009a) and the Fuel Quality Directive (FQD) (European Parliament, 2009b) were incorporated directly into the RTFO. The RED set a binding target that 10% (energetically) of all transport fuels consumed in Member States should be from renewable sources. The FQD set a target of at least a 6% reduction in the life cycle greenhouse gas emissions per unit of energy from fuel and energy supplied compared to the EU average greenhouse gas emissions per unit of fossil fuel energy in 2010. One of the immediate impacts of these Directives was that Member States needed to determine the greenhouse gas intensity of the transport fuels used (and other energy sources).

In order to achieve these requirements it was necessary for the RTFO to reward biofuels according to the carbon savings that they bring. Two broad approaches to achieve this were considered in the UK:

1. A sliding scale approach where improvements in greenhouse gas emissions of biofuels were rewarded through increasing the number of RTFCs issued per litre.

This may be on a broadly linear scale where small increments are rewarded, or on a stepped scale where larger improvements are required in order to increase the possible financial reward.

2. A minimum threshold approach where by biofuels that do not meet a minimum greenhouse gas emission standard are not rewarded, whilst all biofuels that meet or exceed this threshold are rewarded equally.

There were many complex issues to consider before implementing either one of the options above, a combination of these options, or alternative approaches. For example, if a linear sliding scale was adopted this gives the potential to reward, and thus encourage, each small incremental improvement in the environmental performance of a transport fuel, however, there would be a large administrative and reporting burden to meet as each small scale change must be demonstrated and validated. If a stepped scale was introduced, only larger step changes in performance would be rewarded. If there was little prospect of a fuel improving to such an extent that it reaches the criteria for the next 'step' in the scale, there would be little incentive to improve its performance at all, therefore discouraging small improvements in fuel performance. Similarly, if a single cut off criterion was used, there is no ongoing incentive for continued improvement in fuel performance, and the best performing fuels would be treated in the same way as a fuel that just makes it past the cut off point with no recognition or reward for superior performance. If threshold points were set too low, environmental improvement will be slow and the market would be saturated with lesser performing fuels. However, if the threshold was set too high the industry would be faced with unattainable targets, short supply of high quality fuel imports and considerable increases in costs to companies and customers.

The Governments approach, as outlined in the amended RTFO (UK Government, 2011), was based on option 2 above where fuels that did not meet minimum sustainability criteria were considered as fossil fuels and not rewarded under the scheme. However, and of significance to this study, the facility to incentivise the production of biofuels with GHG intensity savings higher than the mandatory minima set out within the legislation

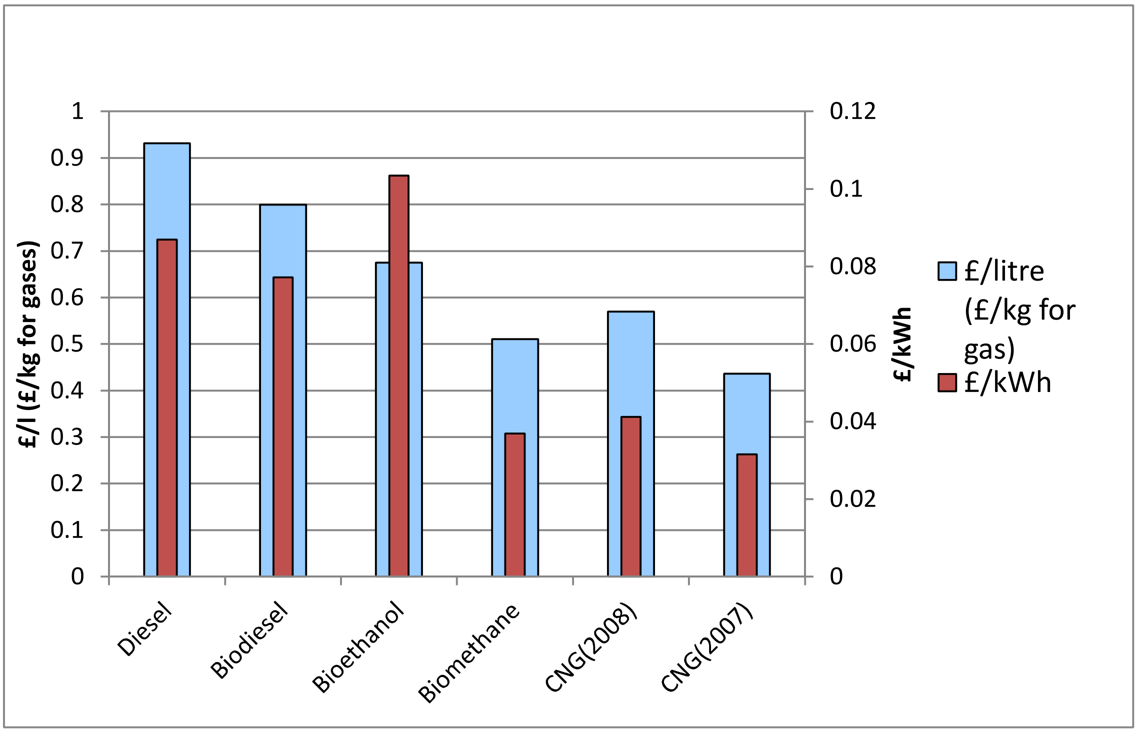
(European Parliament, 2009b) was also incorporated. This approach was consistent with Article 21 of the Renewable Energy Directive (European Parliament, 2009a) which states that biofuels produced from waste, residues, non-food cellulosic materials and ligno-cellulosic materials count twice towards GHG saving targets, which includes biomethane produced from waste feedstocks, which then qualifies for x2 RTFCs per kg of fuel.

4.6 Economic Viability of Upgrading Biogas for Direct Use as Vehicle Fuel in the UK

The viability of widespread adoption of biomethane for direct use as a transport fuel is dependent upon it being financially competitive with other non fossil fuels. This includes biodiesel and bioethanol, but potentially more relevant is the comparison with the most widely available gaseous fuel: natural gas.

An assessment of the current production costs (as previously described), duty rates (including the biomethane duty differential in force until 2012) and maximum RTFO buy out values suggested that when compared on an energetic basis (Figure 2) biomethane was more economically viable than diesel, biodiesel or bioethanol. Increases in natural gas costs (reflected in the large difference between CNG(2008) and CNG(2007)) also resulted in biomethane produced from waste (with waste attracting a gate fee of £40/t) being as competitive as CNG. The difference in production costs between liquid and gaseous fuels was more marked when considered on an energetic rather than mass or volumetric basis.

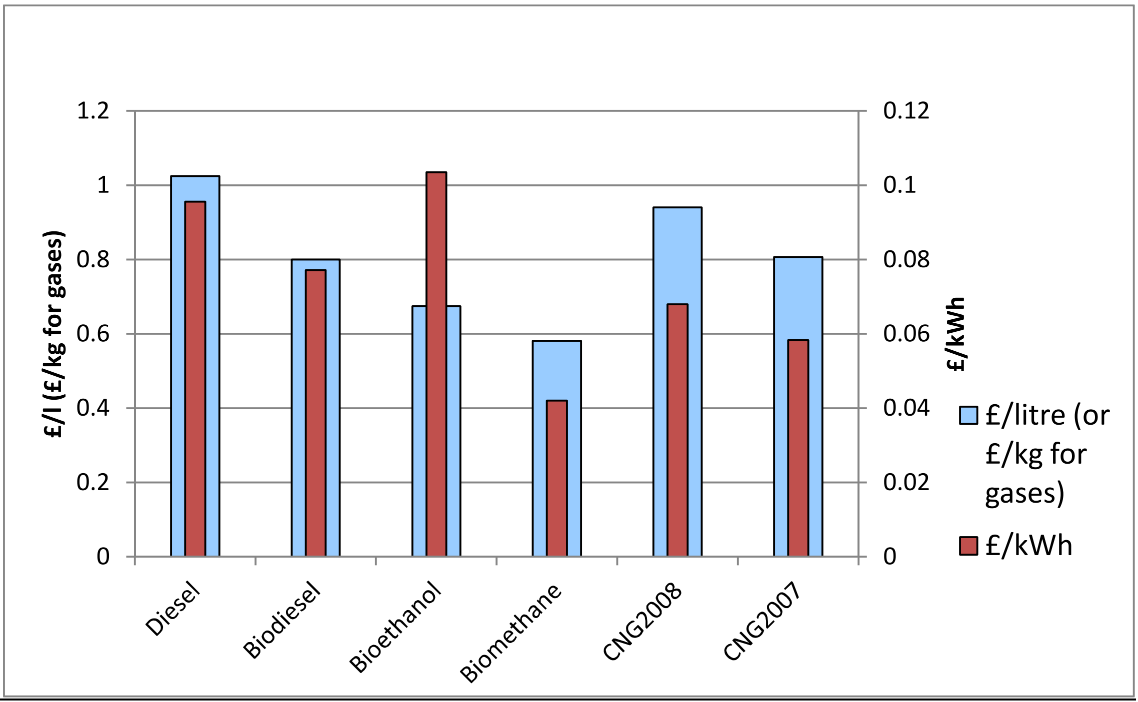
It should be noted that this analysis included the maximum attainable buyout value for RTFCs. Any reduction from this would significantly decrease the ability of biomethane to compete with CNG. If the aim of government policy is to encourage the use of biomethane as a transport fuel these results suggested that the RTFO should be applied on an energetic rather than volume or mass basis, and that further financial support for biomethane, or at least a mechanism to stabilise the potential subsidy, is required in order for it to compete with CNG.



Note: Pre-Tax Diesel price (Renewable Fuels Agency, 2008), Biodiesel and Bioethanol production costs (Kroes, 2007), Natural gas price based on Heren NBP Index value in (BP, 2009), UK Duty Rates (Her Majesty's Revenue and Customs, 2007)

Figure 2 – 2010/11 Fuel Production Cost Estimate (Aggregated Pre Tax Production Costs, Fuel Duty Rates, and Maximum RTFO Buyout Value

Post 2012/13 the duty differential for biomethane (and CNG) may no longer apply and the RTFO, incorporating RED and FQD requirements, will be the primary means of incentivising these fuels. As previously discussed fuels derived from waste (e.g. biomethane) are likely to attract double environmental credit and the financial comparisons shown in Figure 3 therefore assumed that these fuels (i.e. biomethane) would therefore attract double the financial incentive.



Note: Pre-Tax Diesel price (Renewable Fuels Agency, 2008), Biodiesel and Bioethanol production costs (Kroes, 2007), Natural gas price based on Heren NBP Index value in (BP, 2009)

Figure 3 – Post 2012 Production Cost comparison of vehicle fuels (No Duty Differential, Maximum RTFO Buyout Value, Double RTFCs for Biomethane)

These figures again show that biomethane (derived from waste) would be financially more competitive than diesel, biodiesel or bioethanol (both derived from crop) when produced from wastes which attract a gate fee. Biomethane would be less expensive than natural gas, primarily because natural gas does not receive any subsidy under the RTFO. Again, this analysis assumed that the full buyout value for RTFC was achieved. Any shortfall in the value of this subsidy would have the greatest impact on the ability of biomethane to compete with other biofuels and fossil fuels, in particular CNG.

4.7 Vehicle Fuel or Combined Heat and Power (CHP)?

The most common use for biogas in the UK is the generation of renewable electricity and heat through a CHP plant. An income can be gained from the sale of electricity to regional suppliers, and generators also receive a subsidy for each MWh of renewable electricity generated. Prior to 2010 this subsidy was delivered only through the Renewable Obligation scheme, however, post 2010 Feed In Tariffs (FITs) have also become available. The FIT for renewable electricity produced by anaerobic digestion at the scale being considered was initially set at 9 p/kWh plus a 3 p/kWh export bonus (DECC, 2009b).

In order to investigate whether upgrading biogas to transport fuel could compete with a CHP end use, two scenarios were developed based on potential options faced by the private sector when considering the upgrade of biogas derived from the anaerobic digestion of source segregated municipal food waste. These scenarios were (1) the provision of a waste collection service as well as treatment of the waste at an anaerobic digestion plant; and (2) the provision of anaerobic treatment capacity only. For each of these two scenarios, the economic performance of either (a) upgrading biogas for direct use as a vehicle fuel, or (b) utilising biogas in a CHP plant to produce renewable electricity and heat was considered. Where the service provider operated their own vehicle fleet, biomethane was assumed to be consumed by this fleet, and where no fleet was operated biomethane was assumed to be sold as a vehicle fuel at market rate. The scenarios considered are described in Table 17.

Whilst the production of biomethane from source segregated municipal food waste is particularly topical at present, it should be noted that the upgrading of biogas produced from sewage sludge and landfill gas is also relevant as the ROCs attracted by each of these is potentially significantly lower than the proposed RTFC value.

The analysis indicated that the capital costs required to produce a biomethane transport fuel were around 19% higher than that for CHP, however, operational costs were around 26% lower largely due to the elimination of diesel use within the captive fleet through the use of biomethane. All of the biomethane generated within Scenario 1a was utilised by the fleet of CNG waste collection vehicles with sufficient fuel for over 2.15 million km of urban work cycle produced.

Table 17 – Scenario parameters and assumptions included in economic model

	Scenario 1a	**Scenario 1b**	**Scenario 2a**	**Scenario 2b**
	Waste Collection (Biomethane Fuel) > AD > Biomethane	**Waste Collection (Diesel Fuel) > AD > CHP**	**AD > Biomethane**	**AD > CHP**
Capital and Operating Costs Assumptions				
35,000 t/yr Digester (Source segregated food waste) cost excluding energy conversion	✓	✓	✓	✓
Parasitic Electrical Use (10% of CHP electricity production)		✓		✓
Parasitic Heat Use (30% of CHP heat production)		✓		✓
1 MW CHP Plant and electricity grid connection cost??		✓		✓
56 No. 26t diesel fuelled refuse vehicles on an urban work cycle (López et al., 2009)		✓		✓
Imported Parasitic Electricity to match above	✓		✓	
Imported heat energy to match above	✓		✓	
250 m^3/hr Upgrading Plant and biomethane refuelling station	✓		✓	
56 No. 26t biomethane fuelled refuse vehicles (Fravolini et al., 2009; López et al., 2009) on an urban work cycle	✓		✓	
Borrowing and revenue related financial Assumptions				
Interest rate on capital of 5% per annum	✓	✓	✓	✓
Gate fee of £40 / t	✓	✓	✓	✓
Waste Collection fee of £15 / t	✓	✓		
RTFC of £0.35 / kg of biomethane	✓		✓	
Biomethane attracts x2 RTFCs	✓		✓	
Feed In Tariff (FIT) of 9p / kWh with 3p / kWh export bonus		✓		✓
Levy Exemption Certificate (LEC) Value of 0.441 p /kWh		✓		✓
Heat Value of £20 / MWh		✓		✓
No heat ROCs included		✓		✓

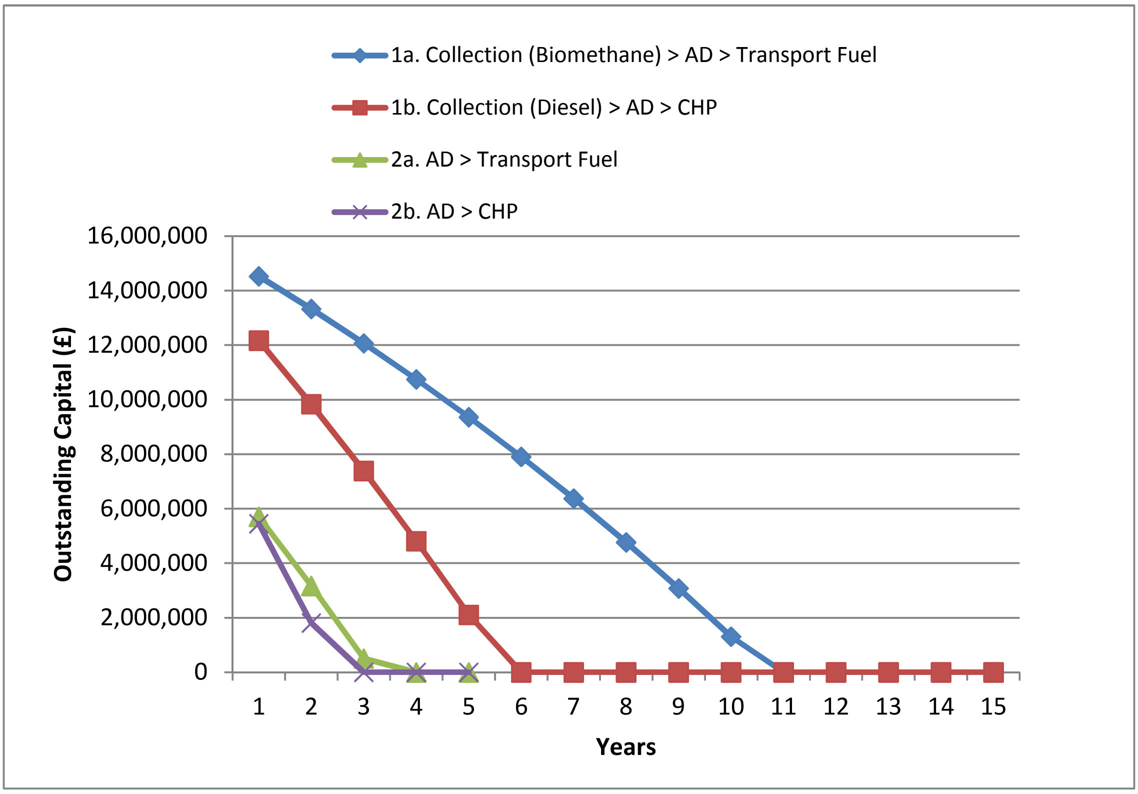


Figure 4 – Estimated Payback Times of CHP and Biogas to Transport Fuel Scenarios (Given Assumed Post 2012 Conditions)

Figure 4 indicates that for scenarios 1a and 1b where collection and fleet costs were included, the payback time of CHP (6 years with a 10 year IRR of 39%) is considerably more attractive than that for transport fuel (11 years with a 10 year IRR of 17%). The key difference between these two scenarios was the additional cost incurred to procure and maintain a fleet of biomethane fuelled refuse collection vehicles. This result is backed up by the performance of scenarios 2a and 2b where biomethane was sold directly to the consumer (or electricity to the grid) with no provision for vehicle purchase or maintenance. These indicated that the payback time for the manufacture and sale of biomethane as a transport fuel (4 years with a 10 year IRR of 63%) was comparable to that of the production and sale of renewable electricity and heat by CHP (3 years with a 10 year IRR of 81%). The results therefore suggest that the level of support assumed for the manufacture of biomethane (x1 RTFC, plus an equivalent RED/FQD bonus) is sufficient for biomethane production to compete with CHP. However, the additional costs associated with the purchase and maintenance of gas fuelled vehicles may prove prohibitive and as such consideration should be given to incentives to purchase or convert to biomethane fuelled vehicles.

4.8 Conclusions

A legislative framework has been put in place in the UK (and Europe) to increase the supply of non-fossil transport fuels to the market place. Biomethane produced from biogas provides greater environmental benefits than other biofuels currently available such as biodiesel and bioethanol, however the requirement to upgrade raw biogas was considered to be a potential barrier. The above assessment showed that upgrading of biogas is technically feasible and is being undertaken at city and regional scales across Europe. However, data associated with the operation of upgrading in terms of energy use and methane emissions is limited and further assessment of these parameters at operational plants would be desirable.

An assessment of the financial performance of biomethane production for transport fuel use suggested that it is at least as viable as liquid non-fossil fuels and CNG when it is produced from wastes with associated gate fees, given the current and anticipated regime of financial support. Compressed Natural Gas (CNG) was closely competitive. However, if financial subsidies that reflect the environmental benefits of biomethane were considered (i.e. double allocation of RTFCs for fuels derived from wastes) then biomethane was the more viable alternative post 2011-12, particularly if natural gas costs continue to increase as has been seen in recent years.

It should be noted that one of the key assumptions in the assessments above was that RTFC's achieved their maximum value, as limited by the 'buy out' option, currently estimated to be around £0.35 / litre (or kg for gases) after 2010. This is by no means guaranteed and 2008/09 saw RTFC values fall to zero, and, since this date the RTFO and its surrounding policy framework has, arguably, failed as a mechanism to encourage greater market penetration and improvements in quality of biofuels (Upham et al., 2009; Upham et al., 2011). Any shortfall from the maximum value would have significant implications for all biofuels. Liquid fuels such as biodiesel and bioethanol could remain less financially attractive than diesel, and biomethane could remain less financially viable than CNG. As well as applying the RTFO on an energetic rather than volumetric basis, the UK government should consider how best to ensure that an adequate and predictable subsidy is received across the biofuel industry.

The above analysis also indicated that the manufacture and sale of biomethane as a transport fuel could be financially competitive with the production of renewable electricity and heat using CHP, given the assumptions above. However, in the scenario presented the additional costs of purchasing and maintaining a fleet of biomethane fuelled vehicles meant that utilising biogas for CHP and using a diesel fuelled refuse fleet remained the more favourable option. If the UK Government is committed to the utilisation of biomethane as a vehicle fuel, directing financial support towards the purchase of biomethane fuelled vehicles or conversion of existing fleets to biomethane would result in a rapid expansion of infrastructure and significant long term environmental and economic benefits.

Chapter 5: Life Cycle Assessment of Biogas Infrastructure Options on a Regional Scale

5.1 Introduction

The number of Anaerobic Digestion (AD) plants treating biodegradable municipal, commercial and industrial wastes in the UK is set to increase rapidly in the next five years as central, regional and local governments implement strategies to meet the challenging targets for landfill diversion, CO_2 reduction and renewable energy generation. With such a large and rapid infrastructure development programme, decision makers must balance three key factors when deciding the nature and characteristics of the treatment infrastructure developed:

1. Economic – what solution provides the best economic value?
2. Technical – the solution must meet the technical requirements (i.e. effective waste treatment), and achieve high landfill diversion and recycling rates.
3. Environmental – ensure that the solution is environmentally sound and compares favourably with alternative options.

In the UK, anaerobic digestion is viewed as one of the most economic and technically appropriate methods for treating biodegradable municipal wastes such as source segregated food waste. The choice of whether to utilise biogas for electricity and heat generation, or upgrading the biogas to biomethane for transport fuel use or injection to the gas grid, is largely an economic decision or, in many cases, is influenced by specific site restrictions. Increasingly, however, stakeholders have requested guidance on the environmental costs and benefits of the various infrastructure options open to them.

The scale and distribution of the AD plants required to meet the treatment needs of the country or region may not be immediately clear. For example, population statistics for Wales (Welsh Assembly Government, 2009) indicate that the highest population densities occur in the south and south east of the principality e.g. Cardiff 2,397 p/km^2, with medium population densities in the north east e.g. Flintshire 342 p/km^2. Much of central, western and north western Wales is agricultural in nature with poorer transport and service networks and significantly lower population densities e.g. Powys 25 p/km^2. This high variability of population distribution, and hence municipal wastes, means that strategic decisions must be made regarding whether large centralised facilities with improved economies of scale should be commissioned in favour of smaller facilities treating wastes on a more localised basis with reduced transport requirements.

5.2 Review of Literature

Life Cycle Assessment (LCA) offers one way in which the environmental performance of various biogas infrastructures can be evaluated. Although some research has been undertaken to assess the environmental impacts of biogas production and utilisation, in many cases the approach has been limited to an assessment of energy balances or energy balances combined with emissions, which falls short of a full life cycle assessment. For example Berglund & Börjesson (2006) undertook an energy performance assessment of biogas production systems for several feedstocks including energy crops and wastes, however, end use of the biogas was not considered. The study concluded that the energy input to the system was approximately 20-40% of energy yield and highlighted the importance of assumptions made, quality of data and the allocation procedures followed for different feedstocks. It is notable and relevant to this study that transport distances could be relatively large, particularly for high organic content wastes, before energy balances approach negative. Whilst the assessment of energy is a useful indicator it does not necessarily reflect environmental performance. For example the study stated that methane emissions from upgrading of biogas were not considered as they would have a minor impact on energetic performance (Berglund et al., 2006), but the same is not necessarily true of environmental performance. Pöschl *et al.* (2010) also focussed on energetic system performance although the study included multiple feedstocks, multiple scales of biogas production plant, and multiple biogas end uses. Given the complexity and potential number of configurations of the system being investigated the limitation of the study to energetic performance is perhaps understandable. However, the conclusions reached are then also somewhat limited, for example an increased level of feedstock pre-treatment increased the overall energy demand (Pöschl et al., 2010), or in some cases confusing due to the number of potential scenarios modelled, and therefore, perhaps a more in depth assessment of fewer plant configurations may have been a more beneficial approach. Bohn *et al.* (2007) used the approach of calculating energy balances as a means of screening the viability of a novel anaerobic digestion plant designed to operate at low temperatures. This is a useful approach for screening specific technologies – the study found that biogas yields at ambient temperatures were 60% of those at 30 °C and therefore not likely to be viable at industrial scale (Bohn et al., 2007), however the approach does not meet the requirements of stakeholders wishing to establish the overall environmental impact of planned infrastructure developments.

Consideration of air emissions in combination with energetic assessment is somewhat more representative of a life cycle assessment, at least if global warming potential is the primary impact category under consideration, but does not necessarily address the full range of potential environmental impacts that could be of interest. Börjesson and Berglund (2006 & 2007) undertook such an approach. Again, the study was extremely comprehensive with a large number of feedstocks and biogas end uses considered, and by necessity therefore the studies were based on a large number of assumptions. Never the less, some important and relevant conclusions are reached, perhaps more so from the first of the studies which identified production of feedstocks from ley cropping as requiring high energy inputs and indicated that methane emissions throughout the system could have huge impacts on environmental performance (Börjesson et al., 2006). The second part of the study also highlighted the importance of methane emissions and indicated that whilst biogas production and utilisation often outperforms fossil fuel based reference systems in terms of global warming potential, there are other systems such as combustion of biomass for heat and methanol production and utilisation as a vehicle fuel that could provide additional benefits (Börjesson et al., 2007).

Whilst some studies have undertaken a limited assessment of a broad range of infrastructure configurations, others have chosen to complete an in depth study of a limited product system. In his thesis, Hartmann (2006) undertook a detailed life cycle assessment of a 1 MW biogas plant utilising a number of energy crops mixed with animal slurry. End use of the biogas was limited to production of electricity. Whilst the methodology taken is sound, the conclusions drawn from the study are somewhat limited, with the major conclusion being that crops with a higher Dry Matter (DM) yield result in a lower overall environmental impact (Hartmann, 2006). Ishikawa *et al.* (2006) used an LCA approach to determine the environmental impacts of two specific biogas plants in Japan, however the scope was limited to considering just global warming potential and an approximate energetic balance. Even though the study was based on existing plants, the level of detail included in the study was limited with very little description of the data used or assumptions made. As a result, the conclusion drawn from the study that utilisation of digesate is a key factor in establishing the environmental impact of biogas production (Ishikawa et al., 2006) is equally vague. A climate change evaluation of biogas production and utilisation using and LCA approach, focussed on distinguishing preferred biogas upgrading technologies, was undertaken by Pertl et al., (2010). The primary distinguishing features between the upgrading technologies are methane losses and energetic inputs, however, the study did not reflect the novel plant configurations being deployed on site to reduce methane losses such as combustion of off gases. This, coupled with the inclusion of carbon sequestration credits to the novel upgrading process that was the primary focus of the study, meant that this process option was potentially misleadingly identified as the best performing.

A number of studies have attempted to assess or compare the environmental impacts of entire waste management strategies using a life cycle assessment approach. For example Finnveden *et al.* (2005) compared the options of landfilling, incinerating or recycling (including AD of food waste) of municipal solid wastes in Sweden considering a range of impact categories. The study is well written and provides a good description of the systems and assumptions made. However, the study does suffer from a certain lack of clarity that is common to studies considering such a wide scope and diverse technologies, although this does not necessarily bring the conclusion that recycling of wastes is generally the best overall option into question (Finnveden et al., 2005). A similar study was undertaken by Cherubini *et al.* (2009) this time comparing the treatment of wastes in Rome either via landfill (with and without landfill gas capture), recycling (including biogas production from organic wastes) and incineration. The study arrived at a similar conclusion that re-cycling is the best environmental option, but also presented some interesting conclusions that this option does result in some point sources of potentially significant impacts and also that all options still result in a proportion of material going to landfill and that this is an area that could result in potential environmental improvements (Cherubini et al., 2009).

Other studies have been undertaken that concentrate on determining the environmental impact of limited biogas production pathways with a single end use. For example Jury *et al.* (2010) completed a detailed LCA of the production of biogas from energy crops with biogas then being upgraded and injected into the gas grid. The study concluded that whilst biogas generation from crops can generate lower GWP impacts than natural gas use, it definitively increases impacts associated with ecosystem quality and human health impacts primarily due to agricultural practices (Jury et al., 2010). Whilst the study itself is a good methodological example of how to complete an LCA, the lack of ability to directly compare possible production or utilisation options means that these conclusions must be considered in isolation, and may not provide stakeholders or policy makers with the information needed to make strategic infrastructure decisions.

As can be seen, there is a wide variety in both scope and approach of LCAs undertaken within the biogas and indeed the wider waste management fields, and the problems associated with this variety are thoroughly described by Cherubini *et al.* (2011). It is evident that there is an inevitable trade off between expanding the scope of a study and reducing the level of detail (or increasing the number of assumptions) included in the study, whilst potentially producing results that are of use to a wide audience, or undertaking a specialised study with limited scope, detailed data and fewer assumptions that will be directly relevant only to those considering the system modelled. However, this is true of all LCA's in all fields. It is perhaps only by building up a body of research and establishing a general consensus across studies that may not be directly comparable that conclusions to assist long term, strategic decisions can be drawn.

5.3 Study Aims

This chapter aimed to provide a life cycle assessment of biogas systems on a regional scale that can provide guidance on infrastructure development decisions. The study compared the environmental impacts, across a broad range of impact categories, associated with the transportation of wastes, construction and operation of anaerobic digestion plants, generation of biogas from source segregated municipal food waste and the utilisation of biogas for either CHP, or injection to the gas grid for end use as either transportation fuel or domestic heat. Whether there are significant environmental benefits from developing a centralised or more distributed infrastructure on a regional basis was also assessed. The objective was to determine whether any option presented clear environmental benefits when considered on a regional basis. The audience for the study was academics and stakeholders active within the anaerobic digestion sectors, and results were also communicated to representatives of the Welsh Government.

5.4 Methods

Environmental impacts were calculated using a life cycle assessment (LCA) approach undertaken in accordance with European guidance (BSi, 2006a; BSi, 2006b). Life Cycle Assessment modelling was undertaken using SimaPro v7.2 software (PRè Consultants b.v.) utilising the ecoinvent database v.2.1 (Swiss Centre for Life Cycle Inventories, 2009) and data as described below.

5.4.1 Function and Functional Unit

The product system assessed was the anaerobic digestion of source segregated municipal food waste in several plants on a regional scale. Co-digestion of other feedstocks was not considered. The function of both the centralised and distributed network of AD plants, regardless of the end use of the biogas generated, was to provide treatment capacity for the municipal source segregated food waste collected in Wales. The mass of municipal food waste generated in Wales was estimated as 16% of the total municipal solid waste stream

(Wasteworks Ltd. et al., 2010) which equates to 275,900 tonnes / yr. The functional unit for life cycle assessment purposes was therefore the treatment of 275,900 tonnes / yr of municipal source segregated food waste within the defined treatment infrastructure which had an assumed lifetime of 20 years.

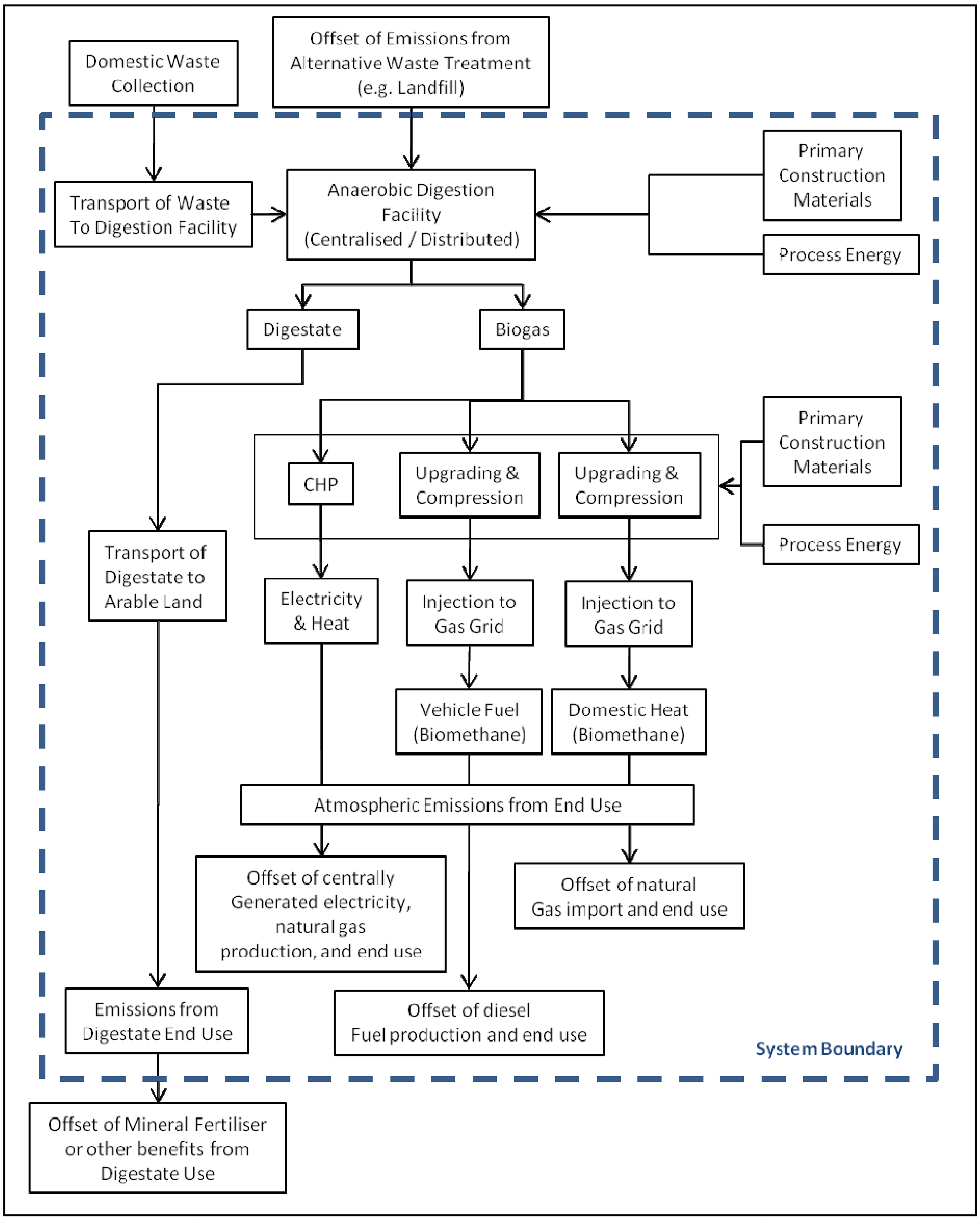


Figure 5 - System boundary of biogas production and end use included in the LCA

5.4.2 System Boundary

The system boundary defining both the centralised and distributed treatment infrastructures is shown in Figure 5. Collection of the source segregated food waste from individual households, displacement of mineral fertiliser and any alternative food waste treatment method was not included in the comparison as it was assumed that these would be equal for each scenario and therefore would not contribute towards a comparison. Studies have already identified the environmental benefits of treating biodegradable wastes by AD (Finnveden et al., 2005; Börjesson et al., 2007; Cherubini et al., 2009). Transportation of the collected waste to the nearest AD facility was included in the study. The centralised infrastructure was assumed to comprise of five anaerobic digestion plants located across Wales, whereas the distributed infrastructure comprised eleven plants. Further description is provided in Section 5.5.

5.4.3 Allocation Procedures

The production of biogas by AD was considered as a multi-output process with outputs being (i) ‘Biogas’, (ii) ‘Disposal’ of organic waste to anaerobic digestion plant, and (iii) ‘Digestate’ to agricultural land. As per the existing ecoinvent methodology for AD plants treating biodegradable wastes (Jungbluth *et al.*, 2007), the environmental burdens of the biogas production plant, co-generation of heat and electricity to power the plant (including air emissions) and the transportation of waste to the plant have been allocated to ‘Biogas’ and ‘Disposal’ on an economic basis. Economic allocation was made based on a ‘Biogas’ value comprising renewable electricity Feed In Tariff of £0.115 / kWh for plants <500 kW or £0.09 for plants >500 kW, and Export Tariff of £0.03 / kWh for electricity exported to the grid (DECC, 2010a), and an assumed Renewable Heat Incentive value of £0.035 / kWh. ‘Disposal’ income was based on an average gate fee of £65 / tonne. The allocation applied therefore varied according to plant scale between 37.1 – 40.6% towards ‘Biogas’ and 62.9 – 59.4% towards ‘Disposal’. The environmental burdens of the transportation of digestate, loading and spreading of digestate and emissions to land were allocated to ‘Disposal’ and ‘Digestate’. According to the ecoinvent process (Jungbluth *et al.*, 2007) 50% of the impact was allocated to each product. These allocation procedures allowed each of the products to be defined and investigated individually if required. In the case of this study, impacts associated with all of the products (i.e. Biogas, Disposal and Digestate) were brought together for each plant within the infrastructures modelled. The study therefore included environmental burdens associated with the transportation of waste and the production, processing and end use of biogas whilst only including the environmental benefits associated with the substitution of the equivalent fossil fuel production and end use, and not the additional potential benefits associated with use of digestate as a fertiliser or the diversion of organic material from alternative treatment methods (e.g. landfill or composting).

5.4.4 LCIA Methodology

Life cycle impact assessment was undertaken using the Eco-indicator 99 H/A methodology. The method was chosen as it provided a concise appraisal of end point damage on human health (including the human health impact of climate change), resources (including fossil fuel use) and ecosystem quality. Indicator values were calculated in three stages:

1. Damage factors for the pollutants or resources used were calculated for different impact categories
2. Normalisation of the damage factors on the level of damage categories
3. Weighting for the three damage categories and calculation of weighted Eco-indicator 99 damage factors.

Table 18 - Normalisation and Weighting factors used in the Ecoindicator 99 H/A LCIA methodology

Damage Category	Eco-indicator 99 Hierarchist/Average	
	Normalisation	**Weights**
Human Health	0.0154 DALYs (0,0)	40%
Ecosystem Quality	5130 PDF*m^2*yr	40%
Resources	8410 MJ	20%

DALYs – Disability Adjusted Life Years
(0,0) – Calculation does not include age weighting
PDF – Potentially Disappeared Fraction

Results for damage factors (non normalised or weighted) and normalised and weighted single scores were reported. It should be noted that weighting is a subjective process and falls outside of the scope of ISO14040 (BSi, 2006a) and ISO 14044 (BSi, 2006b). The Hierarchist approach was used to provide a balance between short term damage (e.g. 100s of years for Individualist approach) and long term damage (e.g. 1000s – 10,000s years for Egalitarian approach). The standard average normalisation and weighting factors were used as shown in Table 18. The model was also analysed using a non weighted methodology (CML 2, 2001 Baseline) within the sensitivity analysis.

5.5 Life Cycle Inventory (LCI) Analysis

Data gathered was first use to construct an Excel spreadsheet model that would calculate the necessary inputs for the LCA software (summarised in Appendix B) for a range of plant scales depending on the requirements of the centralised or distributed scenarios. This spreadsheet primarily calculated the parameters associated with the biogas production facilities required in each of the scenarios modelled. The key variables included in the spreadsheet model are shown in Figure 6. Output, primarily consisting of transport requirements, surplus biogas yields, parasitic biogas use and materials required for infrastructure construction was then used as input parameters in the LCA model constructed within SimaPro.

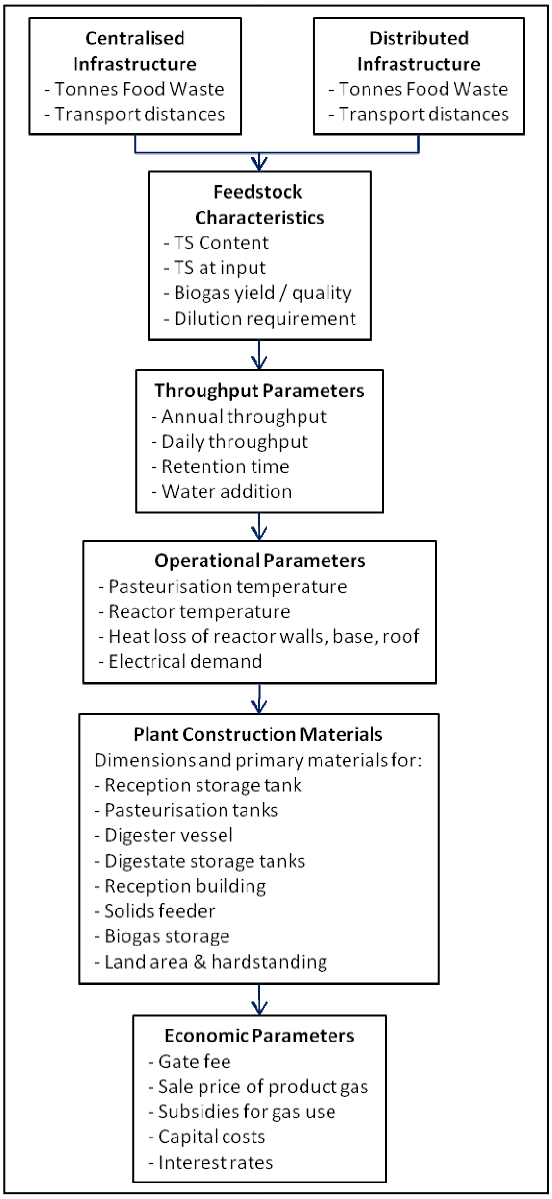


Figure 6 – Primary parameters considered in the Excel model

5.5.1 Waste Input & Defining the Centralised and Distributed AD Infrastructures

In order to construct the centralised and distributed models, Wales was divided into its constituent 22 local authority areas. Due to its large size, Powys was further divided into Powys north, central and south. The total food waste (275,900 tonnes) was divided by the total population (2,999,319) and re-assigned to each local authority according to population size. Local authorities were then clustered together according to geographic proximity such that their waste was treated in one of the five digesters in the centralised scenario, or one of the eleven digesters in the distributed scenario. The centralised model was based on the rationale that economics was the driving factor leading to large centralised plants. The distributed model was based on the rationale that minimisation of transport distance was the driving factor, whilst recognising that plants have to be of a certain scale to be economically viable, leading to a combination of small, medium and large plants. The smallest digester within the distributed scenario treated just 6,326 t/yr and was considered to be at the limit of what is currently financially viable in the UK.

5.5.2 Waste and Digestate Transportation

Digestion facilities were located at notional sites within each local authority cluster. Approximate road distances between the major population centre within each local authority area and the digestion facility within the cluster were calculated in order to determine approximate waste transport distances (by 20-28 t lorry). It was assumed that empty vehicles on return journeys had 80% of the environmental impact of full vehicles (Tung *et al.*, 2005). The centralised and distributed infrastructures are summarised in Table 19 and Table 20. On average each tonne of waste was transported a distance of 36.7 km and 22.9 km in the centralised and distributed infrastructures respectively, excluding the empty return journey.

It was assumed that digestates generated from the biogas plants were compliant with the Publicly Availably Specification (PAS) 110 (BSi, 2010) and the Anaerobic Digestate Quality Protocol (WRAP, 2009a), and could therefore be considered as a product and utilised as a fertiliser within agriculture, forestry or soil/field grown horticulture. No drying or treatment of digestate was included and therefore the whole digestate was transported to the point of end use. Application to arable land only was considered to represent a worst case scenario in terms of transportation requirement. Distances for transporting digestate to the nearest arable land were calculated based on the Land Cover Map 2000 (Centre for Ecology and Hydrology, 2000). Considerable savings in transportation could be achieved by utilising digestates on pasture land. Transportation of digestate was undertaken using a 20 m^3 road tanker modelled as 'Transport, lorry, 16-32 t, Euro3/RER U'. It was assumed that a mass destruction of 80% volatile solids (VS) content of food waste was achieved by the AD process. On average each tonne of digestate was transported a distance of 28.0 km and 20.9 km in the centralised and distributed infrastructures respectively, excluding the return journey. The difference in total transportation requirement (waste and digestate including return journeys) between the centralised and distributed infrastructures was 12,321,635 tkm per year.

Table 19 - Summary of Waste Input and Transport (including return journeys) for Centralised Infrastructure

Biogas Plant	Population*	Calculated FW (t / yr)	Calculated FW transport to digester (tkm** / yr)	Calculated digestate transport (tkm / yr)
1	601,198	55,303	1,414,693	6,271,292
2	754,026	69,361	3,850,839	5,243,661
3	845,033	77,732	4,893,382	4,642,641
4	239,079	21,992	2,678,855	1,385,506
5	603,895	55,551	5,376,011	1,959,821
Total	**2,999,319**	**275,900**	**18,213,780**	**19,502,921**

* (Welsh Assembly Government, 2010a)
** tkm = tonnes kilometre (i.e. 1 tonne transported 1 km = 1tkm)

Table 20 - Summary of Waste Input and Transport (including return journeys) for Distributed Infrastructure

Biogas Plant	Population*	Calculated FW (t / yr)	Calculated FW transport to digester (tkm / yr)	Calculated digestate transport (tkm / yr)
1	336,238	30,930	528,895	3,382,144
2	140,355	12,911	232,395	501,973
3	549,499	50,547	2,208,348	3,070,716
4	519,623	47,799	2,290,611	3,484,515
5	493,205	45,369	1,932,341	551,226
6	117,425	10,802	437,464	183,734
7	120,312	11,067	875,571	672,328
8	177,119	16,293	574,160	554,275
9	358,008	32,932	1,603,023	800,249
10	118,767	10,925	473,927	663,695
11	68,768	6,326	219,757	153,716
Total	**2,999,319**	**275,900**	**11,376,492**	**14,018,574**

* (Welsh Assembly Government, 2010a)

5.5.3 Feedstock and Biogas Characteristics

Parameters associated with the source segregated food waste treated within the AD plants were standardised for each plant. Based on (Hansen *et al.*, 2004; Monson *et al.*, 2007; Zhang *et al.*, 2007) a biogas yield of 130 m^3/t food wastes and a methane content of 60% was used. Based on (Nordberg, 2003; Esteves et al., 2010) a Total Solids (TS) content of 27.5% and a VS content of 80% of TS was used.

5.5.4 Inventory of Biogas Production Plant

Each AD plant within the centralised and distributed infrastructures was assumed to be a wet, single stage, continuously fed process operating at mesophilic temperatures and a hydraulic retention time of 30 days. An organic loading rate of 3.9 kg VS m^{-3} day^{-1} with a TS content at input of 14% was used. A water addition of 0.5 m^3/t waste was allowed for (Monson et al., 2007). It was assumed that water added to the feedstock was sourced from on site rainwater harvesting and therefore had no environmental burden associated with it prior to incorporation within the process. No materials for rainwater harvesting equipment were included as the impact over the lifetime of the plant was likely to be minimal. All feedstocks were pasteurised at 70 °C for 1 hour prior to water addition or digestion.

The conceptual model included the primary functional elements of a biogas plant with each functional element being assigned one or more of the main construction materials required. These included; tank bases and hardstanding / roadways (gravel sub-base, reinforced concrete); reception building slab (gravel sub-base, reinforced concrete), galvanised steel frame, dense concrete block walls and insulated galvanised steel cladding; shredder and screw pump (steel); reception tank (reinforced concrete); polyisocyanurate foam insulated steel pasteurisation tank; reinforced concrete digestion tank with polyisocyanurate foam insulation and aluminium cladding; and reinforced concrete digestate storage tanks with polyester and polyvinylchloride gas storage membranes.

The land required to accommodate each biogas plant was calculated using data gathered from a review of digestion plants across Europe (Monson et al., 2007). For all plants it was assumed that every tonne of construction materials was transported 100 km, the variation between plants is therefore dependent upon only the scale of plant.

Operation of the biogas plant required heat and electricity. In all scenarios this was generated from an on-site CHP plant using biogas as a fuel. Where biogas was upgraded (for injection to the gas grid) it was possible that a CHP plant may not be required and that heat may have been provided by a simple boiler system and electricity from importing from the grid. Given the increasing availability of smaller scale CHP plants and the current financial incentives for the generation of renewable electricity, it was considered that all heat and power for all biogas plants was generated from CHP.

In line with ecoinvent data, carbon dioxide fixed within the biowaste was included as an input of raw material (i.e. consumption of CO_2 from air) during the operation of the biogas plant. The model for each biogas plant is summarised in Figure 7.

The characterisation of food wastes (and therefore of digestates) within ecoinvent included relatively high concentrations of arsenic (2 mg/kg) and cadmium (0.1 mg/kg). Analysis of source segregated food waste in Wales indicated an arsenic concentration of 1.27 mg/kg DM and cadmium concentrations of 0.28 mg/kg DM (Unpublished internal data). These concentrations were therefore used as the input characteristics of the food waste, which had a subsequent impact on the concentrations within digestate.

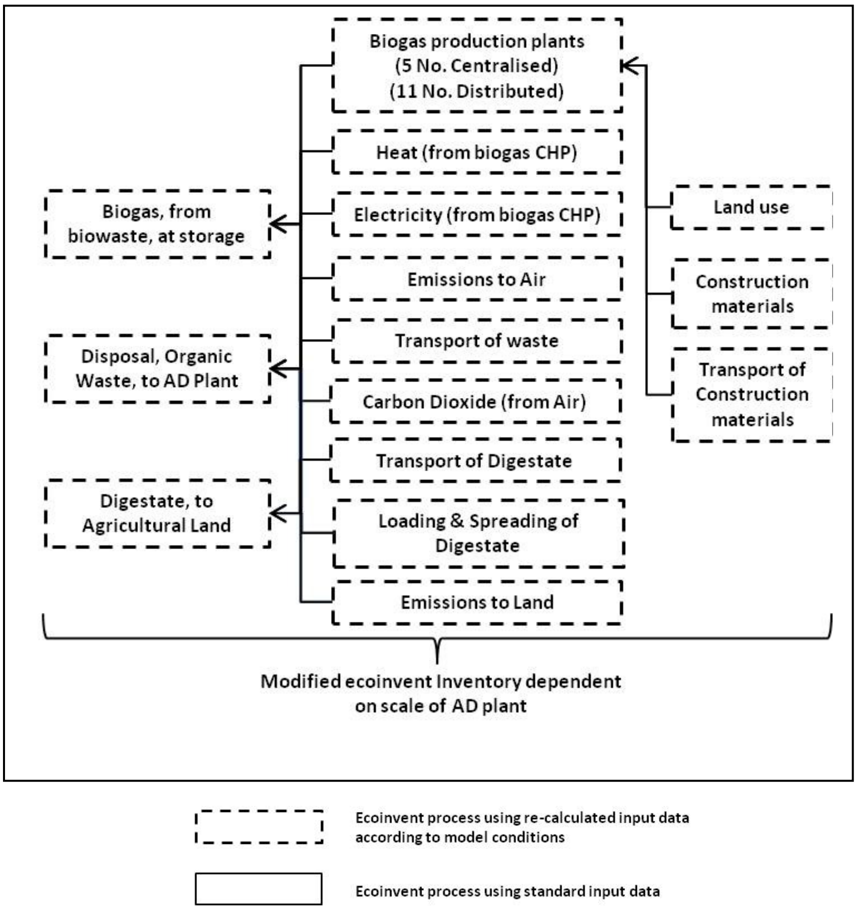


Figure 7 – Summarised inventory for each plant within centralised and distributed infrastructures; biogas plant

5.5.5 Inventory Analysis of CHP generation using biogas

The model for each CHP plant is summarised in Figure 8. Key parameters considered within the model for CHP generation using biogas were an electrical conversion efficiency of 32% and a thermal conversion efficiency of 50% (Jungbluth *et al.*, 2007; Monson *et al.*, 2007). The parasitic electricity demand of the biogas plant was assumed to be 20% of generated electricity (Monson et al., 2007) whilst the parasitic heat demand including pasteurisation energy was calculated on a plant specific basis. The mass of each CHP plant was included based on electrical output with an assumed composition of 85% cast iron and 15% aluminium.

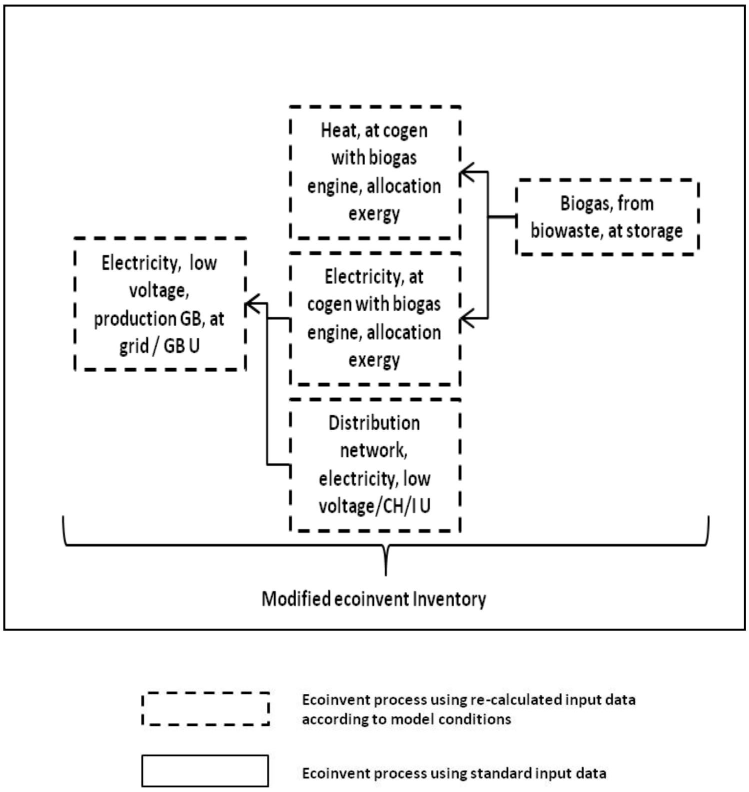


Figure 8 – Summarised inventory for each plant within centralised and distributed infrastructures; CHP end use

Two scenarios for CHP were included; firstly where excess heat was exported to an immediately adjacent end user (CHP 80%) where it was assumed that 80% of the heat exported was useful and substitutes the combustion of natural gas in an industrial boiler for heat production. In the second scenario, all excess heat generated in the CHP process was dissipated to atmosphere with no beneficial use i.e. 0% heat utilisation (CHP 0%). Electricity generated in the CHP process was exported to the grid as medium voltage electricity for end use as low voltage electricity. It was assumed that utilised electricity substituted the centralised generation of high voltage electricity according to the UK generation profile included within ecoinvent (33.4% coal, 1.1% oil, 40.9% natural gas, 19.5% nuclear, 1.4% hydro, 0.99% industrial gas, 0.68% pumped storage, 0.51% wind, 0.99% biomass). Grid losses and emissions associated with export to the grid and delivery of electricity to the end user were included.

5.5.6 Inventory analysis of biogas upgrading, grid injection and vehicle fuel use

Environmental burdens associated with the upgrading of biogas to 96% CH_4 content, calorific adjustment, transportation via the gas network, dispensing at a service station and

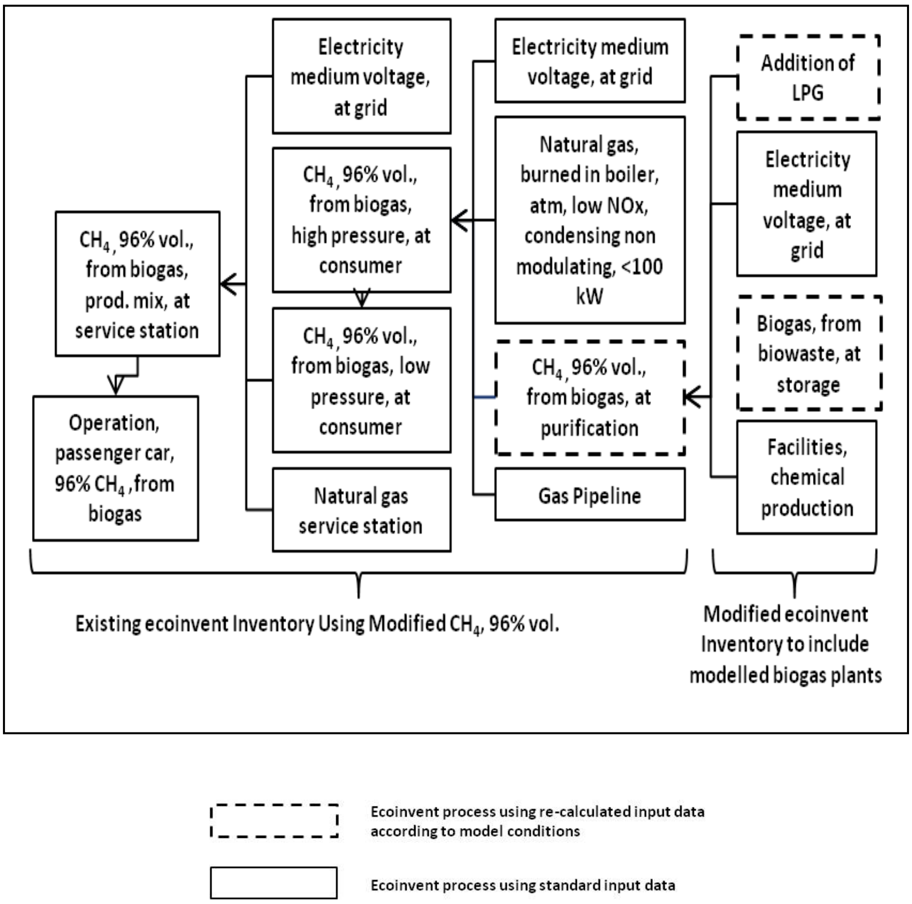


Figure 9 – Summarised inventory for each plant within centralised and distributed infrastructures; transport fuel end use.

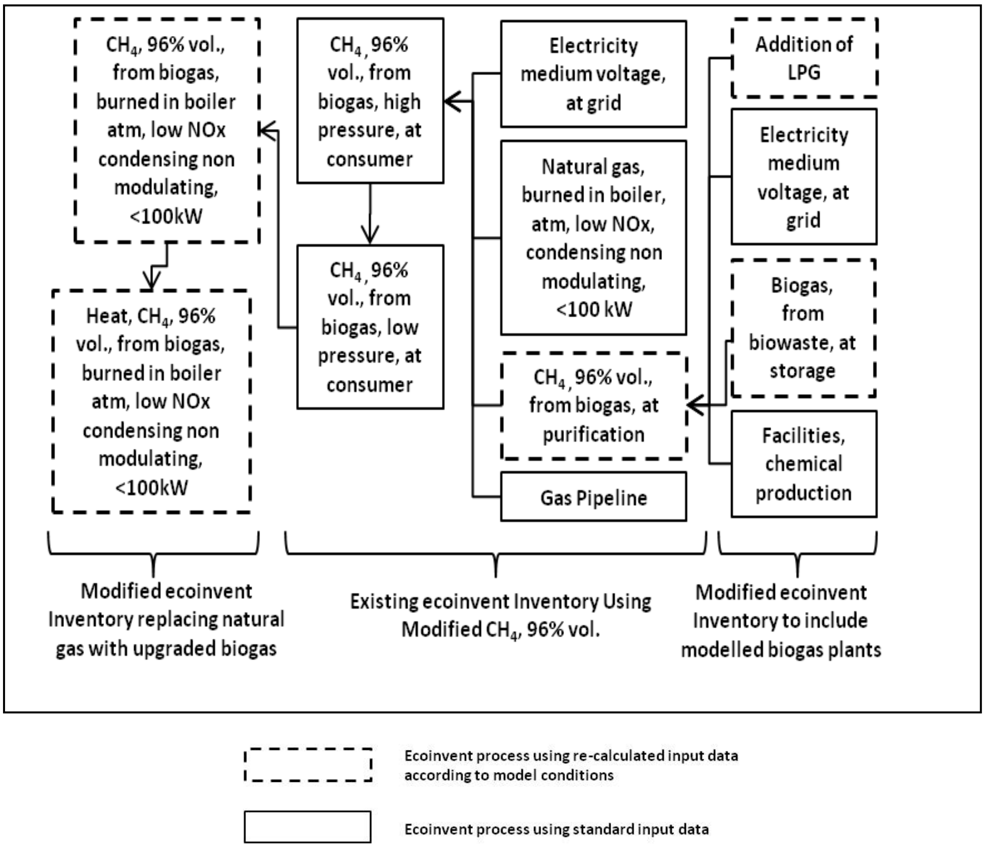


Figure 10 – Summarised inventory for each plant within centralised and distributed infrastructures; domestic heating end use

emissions associated with the end use of the fuel in a passenger car were modelled. The existing structure within the ecoinvent database was used with each element modified to reflect the mass flows appropriate for each plant. The methodology considered Pressure Swing Absorption (PSA) as the upgrading technology including a process methane loss of 3%. Calorific adjustment of the biomethane was considered by the addition of 0.03705 m^3 Liquefied Petroleum Gas (LPG) per m^3 of upgraded biomethane (Jury *et al.*, 2010). A simplified inventory for the end use of upgraded biogas in a passenger car is shown in Figure 9. It was assumed that the burdens of utilising biogas as a vehicle fuel substituted those of using diesel fuel. This substitution included the material flows and emissions associated with the production and transportation of diesel and the combustion of diesel at the end use.

5.5.7 Inventory analysis of biogas upgrading, grid injection and domestic heat use

The summarised inventory for determining the environmental burdens associated with the end use of the biogas for heating purposes following upgrading and transport via the gas grid is shown in Figure 10. The inventory for the production, upgrading and transportation of the biogas was as per the vehicle fuel scenario above. At the end use stage the low pressure gas was burned in a low NO_x condensing, non modulating boiler for heating purposes. Existing inventories within ecoinvent were modified to replace natural gas with upgraded biogas. It was assumed that the combustion of biomethane in a condensing boiler substituted the combustion of natural gas. This substitution therefore included the material flows and emissions associated with the production and transportation of natural gas and the combustion emissions of natural gas at end use.

5.6 Results

As noted in Section 4.2, results describe the impacts of biogas generation, processing and end use with no allowance for benefits gained from the diversion of organic materials from alternative treatment methods (e.g. landfill or composting). Figure 11 - Figure 13 shows the damage assessment results generated using Eco-indicator 99 H/A for all scenarios modelled per functional unit (i.e. 275,900 tonnes food waste treated). Figure 11 indicates that utilisation of biogas for domestic heating purposes via the gas grid displaced the most fossil fuel in both the centralised and distributed infrastructures (6.10×10^7 MJ and 6.60×10^7 MJ, respectively) closely followed by transport fuel use (5.83×10^7 MJ and 6.34×10^7 MJ) and CHP with 80% heat utilisation (5.48×10^7 MJ and 6.04×10^7 MJ). Not surprisingly, CHP with 0% heat utilisation was the worst performing in terms of fossil fuel displacement (1.46×10^7 MJ and 2.02×10^7 MJ). Mineral use ranged from a minimum of 1.2×10^5 MJ (Dist. CHP 80%) to a maximum of 3.98×10^5 MJ (Cent. Dom. Heat).

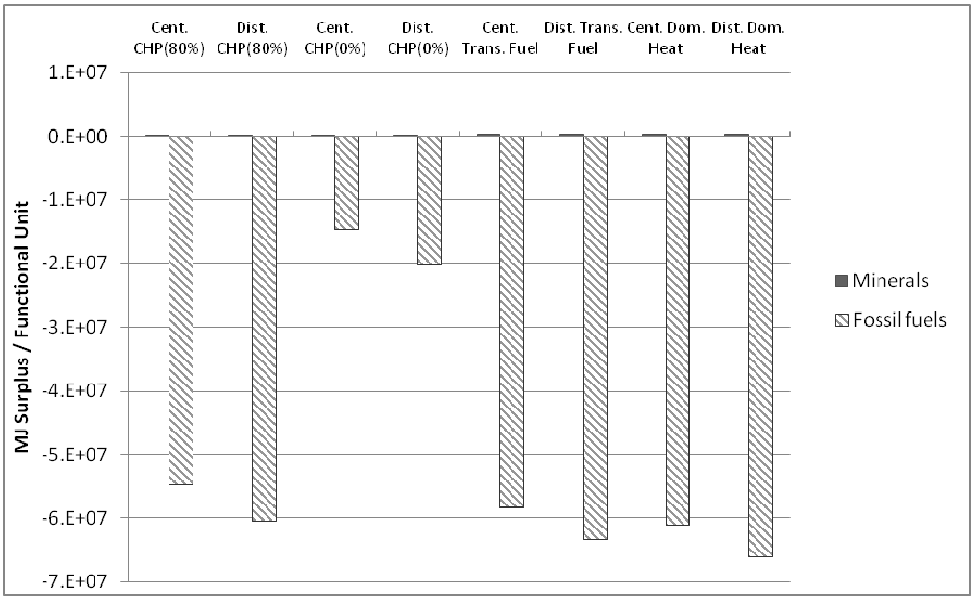


Figure 11 – Damage category level results using Eco-Indicator 99/H/A; for fossil fuels and minerals

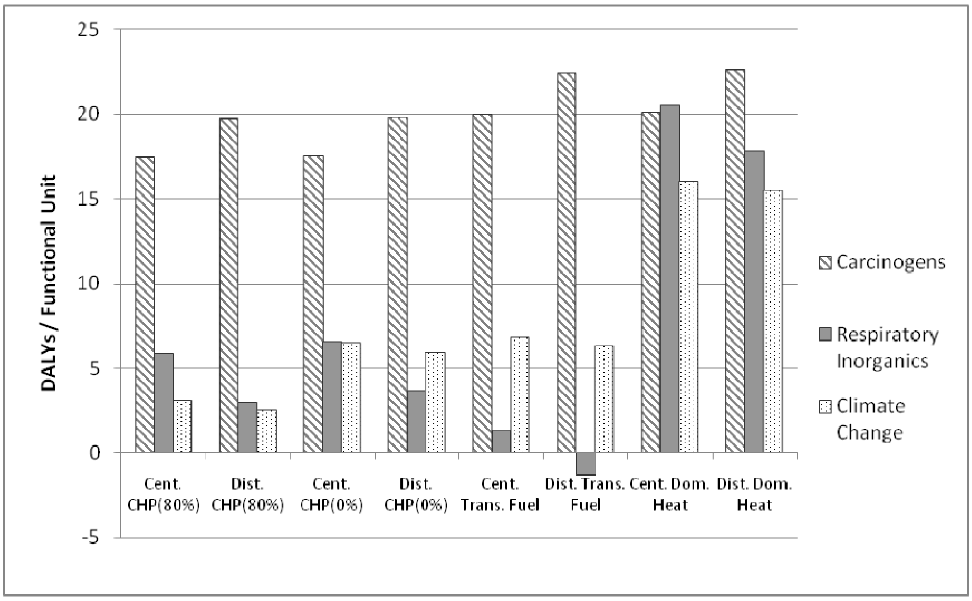


Figure 12 – Damage category level results using Eco-Indicator 99/H/A; for carcinogens, respiratory inorganics and climate change

Figure 12 indicates that impacts from emissions of carcinogens were fairly consistent across the infrastructures modelled with data values ranging from 17.46 DALYs (Cent. CHP 80%) to 22.62 (Dist. Dom. Heat). Impacts from respiratory inorganics showed a greater variability with transport fuel use providing the lowest impacts (-1.31 to 1.32 DALYs) and domestic heat use providing the greatest (17.86 – 20.52 DALYs). Climate change impacts were lowest for CHP with 80% heat utilisation (2.53 – 3.10 DALYs) and greatest for domestic heat use (15.53 – 16.03 DALYs).

Figure 13 indicates that transport fuel use performed significantly worse than other options in terms of ecotoxicity burdens, where as domestic heat use produced the greatest eutrophication / acidification and land use burdens. Ozone layer, radiation and respiratory organics have not been shown as calculated impacts were very small compared to those described above.

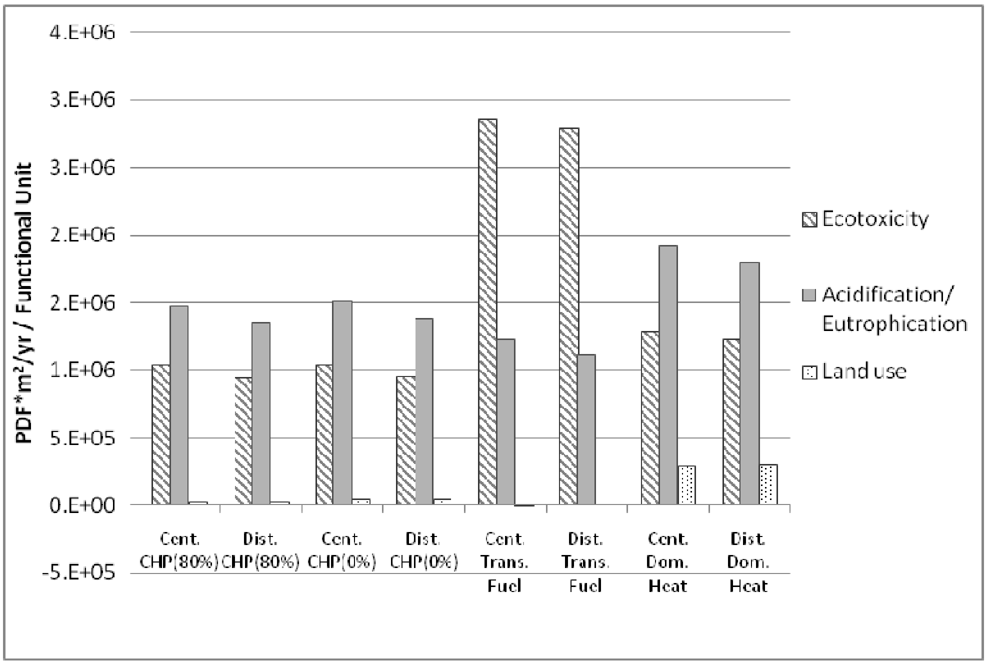


Figure 13 – Damage category level results using Eco-Indicator 99/H/A; for ecotoxicity, acidification/eutrophication and land use

Figure 14 shows the weighted life cycle impact assessment results (Eco-indicator 99 H/A) summarised as single scores at impact category level. Results are expressed in Points (Pt), which are dimensionless values generated following the normalisation and weighting procedures (Table 18). Results show that both CHP with 80% heat utilisation and transport fuel end uses had a comparable overall performance with a balance between positive and negative impacts for distributed infrastructures of -6.04E5 Pt and -4.75E5 Pt, respectively. Domestic heat use via the gas grid performed less favourably with a balance of +1.6E5 Pt with the worst performing option being CHP with 0% heat utilisation (+4.67E5 Pt). For all end uses, there was a small but consistent benefit associated with distributed infrastructures as opposed to centralised.

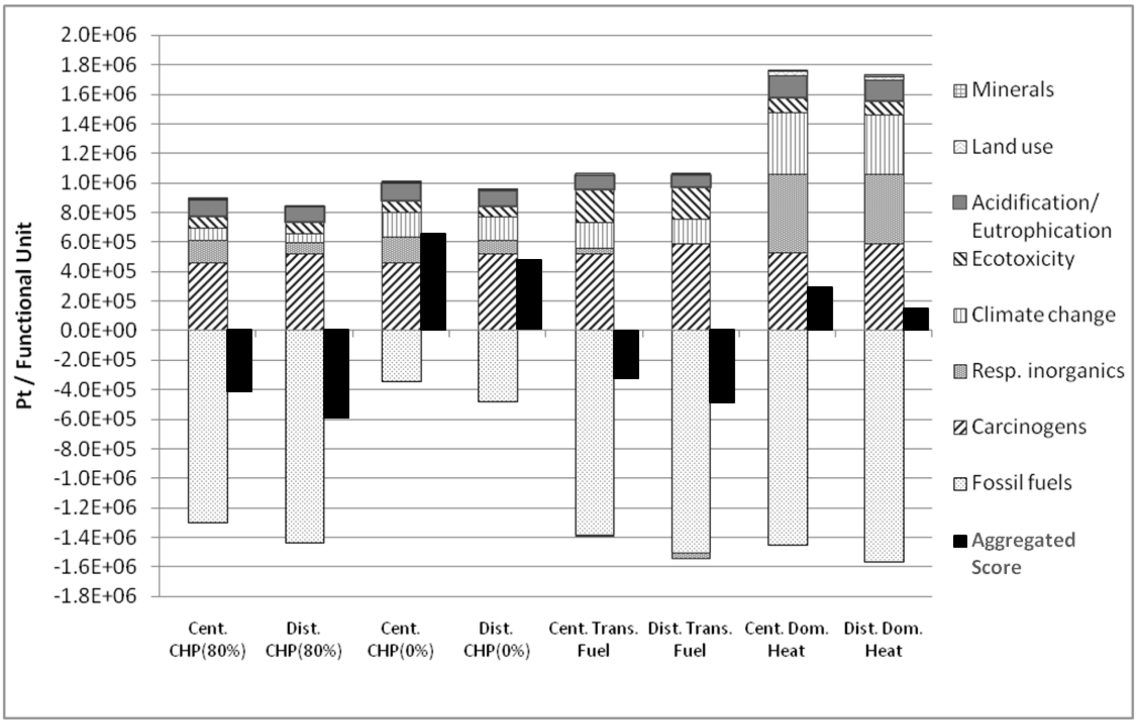


Figure 14 – Weighted, single score LCIA Results for all end uses and infrastructures using Eco-indicator 99 H/A

5.7 Discussion

Weighted results in Figure 14 suggest that the largest detrimental burden, which was consistent across all infrastructures, was associated with the emission of carcinogens. The majority of this burden was generated as a result of the presence of low concentrations of heavy metals such as cadmium and arsenic within digestates that are applied to agricultural land. Long term effects on human health associated with the accumulation of heavy metals in agricultural soils would be more significant in an end point, damage based method such as eco-indicator 99 H/A compared to mid-point, problem based approach such as CML, and the result was therefore likely to be a conservative one. The issue of heavy metals within MSW derived materials (Smith, 2009; Kumar *et al.*, 2010) and waste water sludges (Hospido *et al.*, 2010) applied to agricultural land is the subject of ongoing research and further quantitative analyses will give a more accurate indication of risks. In the UK the Anaerobic

Digestate Quality Protocol (ADQP) (WRAP, 2009a) and PAS 110:2009 (BSi, 2010) will ensure impact minimisation and that a consistent quality of digestate is supplied to end markets. Farmers and land owners should comply with the Code of Good Agricultural Practice (DEFRA, 2009), which describes best practice associated with the application of materials to land.

The CHP scenarios show the importance of utilising the surplus heat generated when converting biogas to electricity in combustion engines. The scenario with 0% heat utilisation stands out as performing worst compared to other options, whereas the scenario where 80% of the surplus heat is utilised in an adjacent process performed the best out of all the infrastructures modelled. The economic benefits associated with the use of excess heat, which have increased significantly since the introduction of the Renewable Heat Incentive (DECC, 2010b) in 2011, mean that AD schemes in the UK should be actively seeking to utilise excess heat. Where this high utilisation of excess heat at the end user cannot be achieved, CHP will result in higher impacts than alternative uses such as transport fuel. Heat losses within heat mains are a function of distance, thermal volume and flow rate and are estimated to be in the order of 1 – 3.5 % (0.5 km), 4 – 13.5% (2 km) and 6 – 20% (3 km) (Pöschl et al., 2010).

Upgrading biogas to biomethane and utilisation as a transportation fuel was shown to substitute a marginally higher amount of fossil fuel than CHP with 80% heat utilisation. The slightly larger environmental burden compared to CHP is largely due to the operation of the passenger vehicle during the end use including emissions from the combustion of fuel, brake dust and tyre wear as well as methane losses associated with the upgrading process. The results indicated that burdens associated with the upgrading of biogas are outweighed by the environmental benefits associated with reducing diesel fuel use.

The injection of biomethane to the gas grid and its end use for domestic heating was found to substitute marginally more fossil fuel than transportation end use, however, the impacts associated with the end use were considerably greater. This was because the end use of (biogenic) biomethane for domestic heating replaces the use of (fossil) natural gas which, in the context of this model, would have similar emission concentrations at end use i.e. at the domestic boiler. Therefore, whilst the other end use scenarios benefited through reduction in end use emissions by replacing centrally produced electricity or diesel fuel use, the use of biomethane for domestic heating did not incur such benefits. In essence, the emissions at end use were the same whether natural gas or biomethane is used as a fuel. The only savings are therefore those associated with the substitution of natural gas production and transportation, and the replacement of fossil CO_2 eq. with biogenic CO_2 eq. with no other emission benefits accrued at end use.

Given the assumption that the majority of natural gas within the UK grid is used for domestic heating, this result raises the interesting point that whilst the addition of biomethane to the gas grid provides (i) an efficient means of transporting the upgraded biogas, and (ii) corporate advantages associated with the reduction of the carbon footprint of the gas grid, it may not deliver the greatest environmental benefit at this stage. Results suggest that using biomethane to displace more polluting fuels such as liquid fossil fuels will have the greater overall environmental benefit.

The marginal but consistent difference between centralised and distributed infrastructure options indicate that, within the context of this study, the difference in the transportation of materials to and from the digestion plants has a relatively limited impact. This suggests that practical and economic factors should be allowed to be the important drivers in determining the physical distribution of AD plants on a regional basis rather than a specific requirement to reduce transport distances to a minimum. Further modelling was undertaken to assess the impact of increasing transport distances within the centralised scenario as described within the sensitivity analysis.

It is clear that the end use of the biogas has a major impact on the overall environmental performance. It is evident that centralised end uses such as CHP have less overall environmental impact than distributed end uses such as combustion in vehicle engines and boilers, partially due to the reduced material inputs required and partially due to tighter controls on emissions at point sources. Displacement of fossil fuels was however maximised in the decentralised end uses. It should also be noted that this study assumed that biogas will substitute predominantly fossil energy sources (diesel, natural gas, centralised electricity production). It will however become of increasing importance to compare how biogas performs in relation to other alternative or competitive fuels in the future such as biodiesel, bioethanol, hydrogen, imported LNG and wind or photovoltaic electricity generation as this will dictate the nature of future energy infrastructures.

5.8 Sensitivity Analysis

Many factors could justifiably form part of the sensitivity analysis of any LCA, however, in the context of this study, three key elements were assessed:

1. The effect of changing the methane losses associated with upgrading of biogas
2. The effect of increasing transportation distances in the centralised scenario
3. The effect of using an alternative LCIA methodology

5.8.1 Changing Methane Losses Associated with Upgrading

Upgrading biogas to biomethane using current systems involves a loss of a small percentage of the total methane content to atmosphere, with variations occurring due to upgrading technology and operating conditions. Given that methane is a significant greenhouse gas, and given that methane losses to atmosphere results in a loss of revenue for plant operators, there is clearly an incentive to minimise such losses. The models described above included a methane loss of 3% (i.e. 3% of the total methane within the raw biogas was lost to atmosphere). The sensitivity analysis included an assessment of the overall impact of increasing methane losses to 5% and reducing methane losses to 0.5% (Patterson et al., 2011a). These variations only effect end uses where upgrading of biogas was required (Transportation Fuel Use and Injection to Grid for Domestic Heat).

Results showed that increasing methane losses from 3% to 5% reduced the fossil fuel substituted for both transportation fuel and domestic heat via the gas grid by 3.5 – 3.9%

(due to less methane reaching the end use). Global warming impacts were increased by approximately 16% for transportation fuel end use and by approximately 6% for domestic heat use via the gas grid as a result of emissions of methane to atmosphere.

Decreasing methane losses to 0.5% had the effect of increasing the amount of energy delivered to the end use and therefore maximising the fossil fuel substituted. For transportation fuel the substitution was increased by 1.5%, where as for domestic heat use substitution was increased by around 2%. Global warming impacts were more significantly reduced by 30% for transportation fuel end use indicating that methane losses at upgrading are one of the primary contributors to global warming impacts for this end use. Global warming impacts were reduced by 8. 5% for domestic heat use via the gas grid again indicating that methane losses at upgrading do have significant impacts. Results therefore highlight the requirement to reduce technical and operational methane losses during the upgrading process as far as is practicable. This principle also applies to the biogas plant itself and the storage of digestate. The LCA model included a methane loss of 4.5% at the biogas plant, however, as with the upgrading stage, increasing methane losses would significantly alter the outcome of the assessment.

5.8.2 Increasing Transportation Distances

In order to determine the impact that transportation distances had on the result, distances for both the transportation of wastes to and digestates from the centralised anaerobic digestion plants were increased by 50%. Total transport requirements therefore increased from 37,716,701 tkm per year to 56,575,052 tkm per year and the difference in the total transportation requirements between the centralised and distributed infrastructures was therefore increased to 31,179,986 tkm per year.

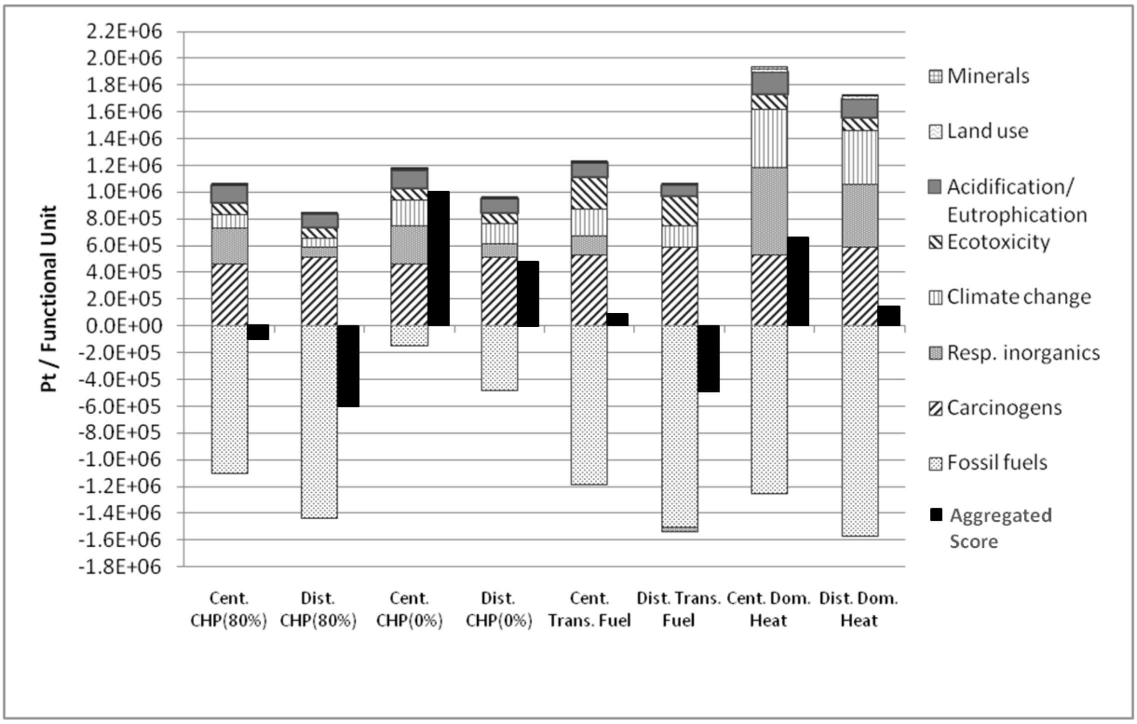


Figure 15 - LCIA results with transportation requirement for centralised infrastructure increased by 50%

Figure 15 indicates that this large increase in transportation requirement for the centralised infrastructure did produce a significant difference between the centralised and distributed infrastructures when Eco-indicator 99 H/A was used. Differences were particularly evident as a reduction in the amount of fossil fuels substituted within centralised infrastructures, and in an increase in emissions of respiratory inorganics. Where the difference in the total transportation requirements of a centralised or distributed infrastructure reaches the level of 20-30 million tkm per year, it would therefore be reasonable to encourage a more distributed infrastructure in order to reduce transportation requirements and maximise environmental benefits.

5.8.3 Alternative LCIA Methodology

The use of alternative LCIA methodology can have a significant impact on the final results generated. In this case, Eco-indicator 99 H/A was replaced by CML 2001 (Baseline) which is a problem orientated (midpoint) approach as opposed to a damage (end point) approach. Characterised output data is summarised in Figure 16 - Figure 19.

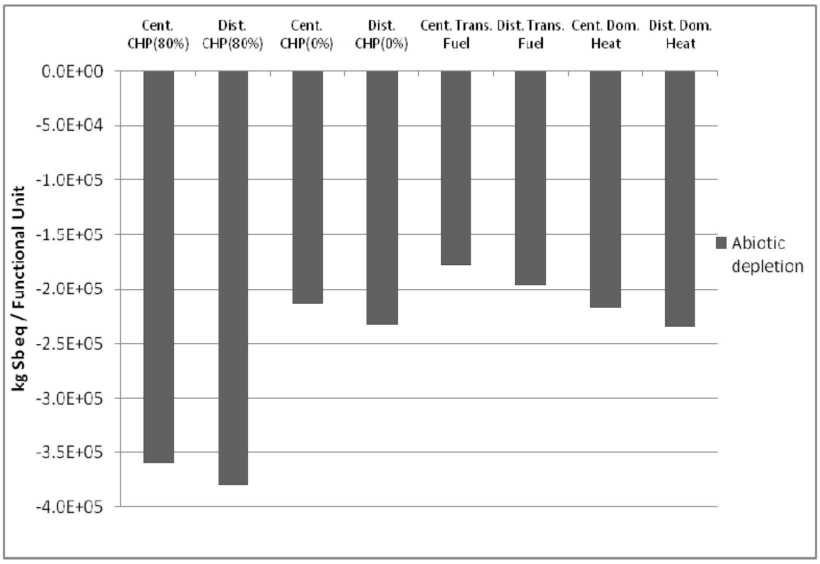


Figure 16 – CML 2001 (Baseline) characterised results; abiotic depletion (measured in kg antimony eq.)

Figure 16 suggests that burdens associated with abiotic resource depletion (depletion of fossil and mineral resources) were the most beneficial for CHP with 80% heat utilisation (-3.59 to -3.79 × 10^5 kg Sb eq.) and the least beneficial for transport fuel use (-1.78 to -1.96 × 10^5 kg Sb eq.).

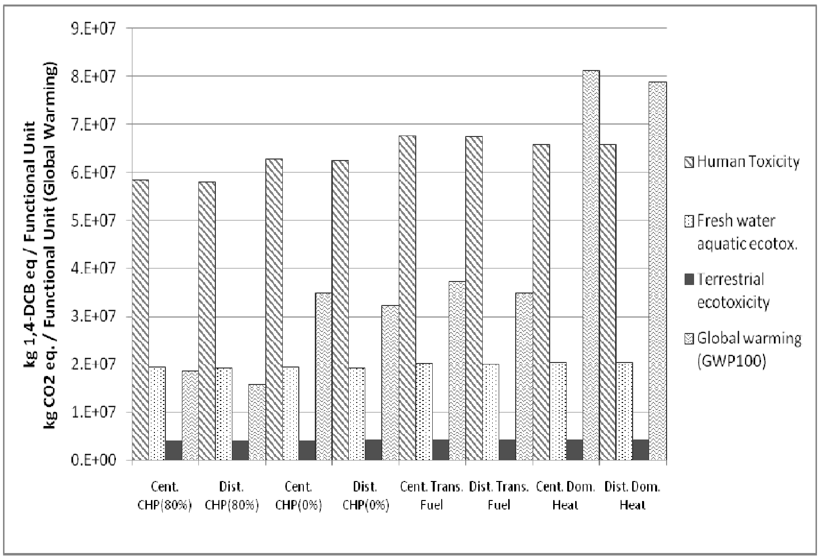


Figure 17 – CML 2001 (Baseline) characterised results; human toxicity, fresh water aquatic ecotoxicity, terrestrial ecotoxicity, global warming

Figure 17 shows that Global Warming Potential was greatest for domestic heat end uses (7.87 – 8.11 × 10^7 kg CO_2 eq.) and least for CHP with 80% heat utilisation (1.59 – 1.86 × 10^7

kg CO_2 eq.). Figure 17 also shows that impacts associated with fresh water aquatic ecotoxicity, terrestrial ecotoxicty and human toxicity were fairly consistent across the infrastructures. Transport fuel use was the worst performing in terms of marine aquatic ecotoxicty impacts (Figure 18) predominantly due to the additional materials and emissions associated with the construction and operation of passenger vehicles and refuelling stations.

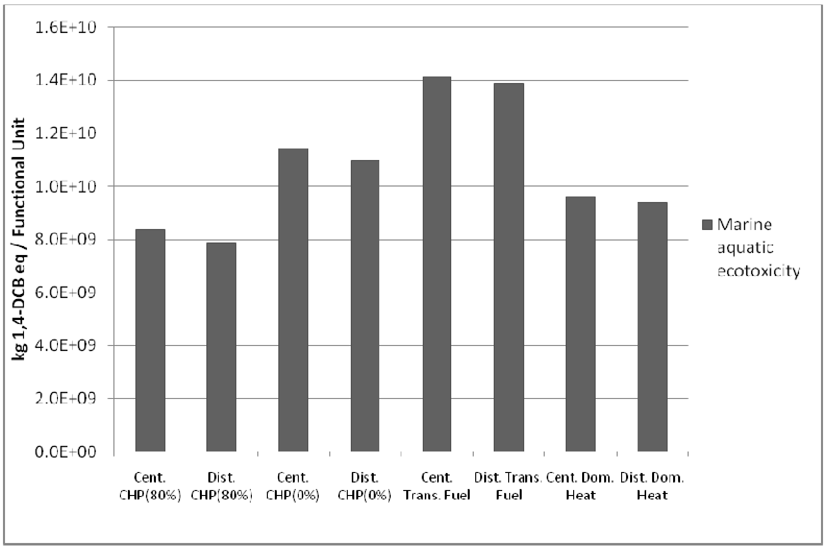


Figure 18 – CML 2001 (Baseline) characterised results; marine aquatic ecotoxicity (measured in 1,4, dichlorobenzene eq.)

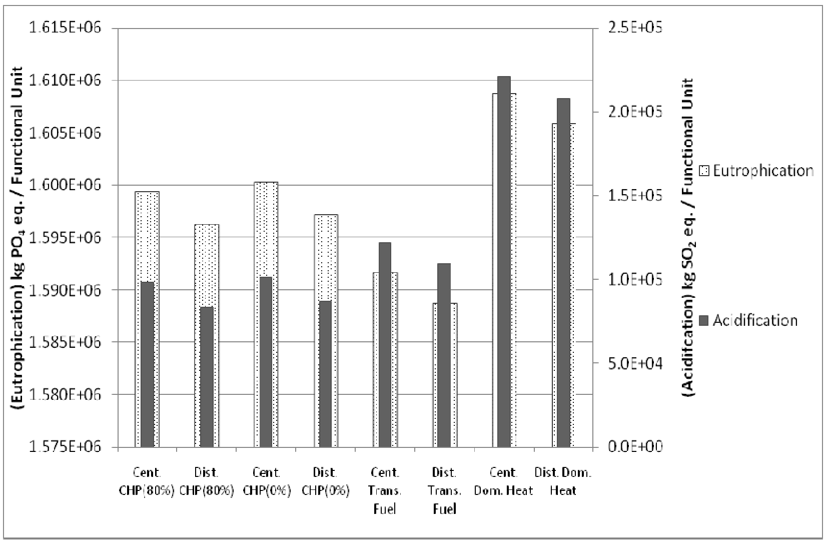


Figure 19 – CML 2001 (Baseline) characterised results; eutrophication / acidification

Figure 19 indicates that domestic heating use generated the greatest impacts associated with acidification (2.08 – 2.21 × 10^5 kg SO_2 eq.) with CHP with 80% heat utilisation generating the least (8.35 – 9.83 × 10^4 kg SO_2 eq.). All infrastructures performed similarly with regard to eutrophication burdens. Whilst both methods agree that CHP with 80% heat utilisation had the overall best environmental performance and domestic heat use the least favourable (other than for fossil fuel depletion), transportation fuel end use appeared to be significantly less favourable when CML methodology is used.

5.9 Conclusions

Damage orientated lifecycle impact assessment indicated that CHP with 80% utilisation of surplus heat resulted in the least overall environmental burden. However, where this high level of heat utilisation cannot be achieved, transportation fuel use would be the most favourable option. A 12.3 million tkm per year difference in transportation requirement between centralised and distributed infrastructures had a relatively small effect on the overall environmental impact. Human health impacts primarily from the deposition of heavy metals on agricultural land were highlighted. Methane emissions at the upgrading stage also had a significant impact highlighting the importance of minimising losses at upgrading.

Chapter 6: Life Cycle Assessment of Biohydrogen and Biomethane Production and Utilisation as a Vehicle Fuel

6.1 Introduction

The most common model for biogas utilisation in the UK and across much of Europe is the generation of electricity and heat using a Combined Heat and Power (CHP) plant (Monson et al., 2007). Where a high level of heat utilisation is achieved this is an efficient use of biogas which minimises environmental impacts. However, where only a low proportion of the heat is utilised, or none at all, other biogas end uses provide greater reductions in environmental burdens (Patterson et al., 2011b). One such use is as a vehicle fuel where biogas is cleaned and upgraded to biomethane and is combusted in an internal combustion engine. Both fuel production and engine technologies are based on well understood and readily available processes and products with the majority of major car manufacturers producing natural gas fuelled vehicles (e.g. Volkswagen, Volvo, Fiat, Ford). Barriers to large scale implementation are therefore largely economic.

The majority of AD plants are configured such that the consortia of microbes converting the organic material to biogas are present within a single tank, meaning that conditions are necessarily sub-optimal for any groups of bacteria that require different environmental conditions. A variation to AD where trophic groups of microorganisms with differing optimal environmental conditions are separated into two different vessels has been developed, with potential advantages of this process being (i) hydrogen can be liberated from the acid producing first phase, and (ii) methane production in the second stage can be increased compared to the single stage process, giving an overall increase in energy output (Hawkes et al., 2007). A blend of hydrogen and methane gas has been shown to reduce key exhaust emissions compared with methane alone when burned in an internal combustion engine (Wang et al., 2007; Dimopoulos et al., 2008; Graham et al., 2008). Therefore, the ability to biologically produce such blends has an obvious application as a means of producing non-fossil vehicle fuel, and could represent a further transition stage to a gaseous fuel economy.

However, as two stage biohydrogen / biomethane production requires a different process management strategy to single stage biomethane production (anaerobic digestion), it is not clear whether the potential advantages of biohydrogen / biomethane production will lead to real environmental benefits. One way to investigate this is via Life Cycle Assessment (LCA).

6.2 Review of Literature

As noted in the previous Chapter there have been several studies focussing on the energy balances and emissions of anaerobic digestion of various feedstocks, most notably Berglund et al., (2006), Börjesson et al., (2006), Börjesson et al., (2007), Patterson et al., (2008) and Pöschl et al., (2010) with relatively few studies undertaking full Life Cycle Assessments of AD infrastructure options, e.g. Patterson et al.,(2011b) Poeschl et al., (2012a) and Poeschl et al., (2012b).

Relatively little environmental assessment work has been completed for two stage biogas production processes. A study by Zielonka *et al.* (2010) undertook an energy balance for a two stage process for crop digestion, however this was not a hydrogen producing process and did not extend to an LCA. The leach bed and anaerobic filter process was found to be more suitable to crop substrates with a longer acid forming period in the leach bed reactor such as corn silage (Zielonka et al., 2010), however, as the process was not directly compared to other biogas producing technologies the advantages of the process were not clear.

An energy balance of the dark fermentation process has been undertaken by Ruggeri *et al.* (2010). Stirred batch reactors were used to treat a glucose substrate at various temperatures between 16 – 50 °C with pH controlled at a minimum of pH 5.2. The study found that due to reactor heat losses the hydrogen production process, when considered in isolation, is energy negative except at the lowest temperature investigated (Ruggeri et al., 2010). However, the study did not consider the production of hydrogen in combination with a second anaerobic treatment phase for the production of CH_4 / CO_2 biogas, and acknowledges that if this were the case the overall energy balance would be very different.

A comparative LCA of a two stage system producing hydrogen only was undertaken by Djomo and Blumberga (2011). Here, the second stage of treatment comprised of the photofermentation of the outputs from the first dark fermentation stage, thus producing additional hydrogen gas. The option of treating first stage outputs by anaerobic digestion to produce methane was not considered and, despite stating that the produced hydrogen was for vehicle fuel use the system boundary stopped at the compression and storage stage. The study concluded that energy ratios and greenhouse gas balances of the biological hydrogen production process compared favourably with conventional diesel and hydrogen production by steam methane reforming, and that consideration of the fate of process by-products (and therefore the allocation procedures used) could have a large impact on results (Djomo et al., 2011).

A similar energetic and CO_2 balance of a dark fermentation / photofermentation biohydrogen production process was undertaken by Ferreira *et al.* (2011), this time with the scope extended to include the utilisation of the fuel in one of five vehicle drive trains including internal combustion engines and fuel cells. The study found that whilst diesel and gasoline production had the lowest energy requirement and CO_2 emissions, the biological production of hydrogen was preferable when compared to steam methane reforming or hydrogen from electrolysis. However, due to lower emissions during the utilisation stage (i.e. the lower emissions of a hydrogen based fuel), the production of hydrogen from sugar cane and potato peelings was found to have lower overall CO_2 emissions than either hydrogen from electrolysis or fossil based alternatives (Ferreira et al., 2011).

No studies undertaking a life cycle assessment of the production of biohydrogen and biomethane gas using a combination of dark fermentation and anaerobic digestion could be located in the available literature.

A number of studies have undertaken life cycle approach investigations into the utilisation of hydrogen and / or methane as a vehicle fuel. Wang (2002) used the GREET model to assess various fuel options for fuel cell vehicles. Whilst the study does highlight the importance of considering fuel production pathways in life cycle assessments of fuel cell vehicles (Wang, 2002), the GREET model is highly focussed on conditions in the USA. Biological options for fuel production do not include biogas or biohydrogen production using anaerobic fermentation processes and the model only provides energy and emission outputs, not the expanded range of impact categories produced in other life cycle assessments. Neelis *et al.* (2004) undertook an exergetic life cycle assessment focusing on electrolytic production of hydrogen with utilisation in a fuel cell vehicle. The main focus of the study was to investigate various on board hydrogen storage options. Of the options considered, compressed hydrogen was the best performing storage option with metal hydrides performing less well due to additional vehicle weight and liquefaction also performing less well due to high energy inputs (Neelis et al., 2004).

Martínez et al. (2010) has completed an assessment of life cycle greenhouse gas emissions for CNG / H_2 mixtures for transportation use in Argentina. However, the study included a very large scope with fourteen fuel production pathways considered, and as such the system boundaries used to determine each pathway are very unclear, as are the life cycle assessment methodologies employed. The study concludes that of the fourteen pathways considered, only hydrogen production through wind electrolysis or nuclear electrolysis resulted in greenhouse gas savings when compared with natural gas alone (Martínez et al., 2010), although given the uncertainties associated with the study it is difficult to apply these conclusions elsewhere.

A number of studies have attempted to undertake a life cycle assessment of hydrogen production using electrolytic technologies, often in comparison with other production options (Koroneos et al., 2004; Ally et al., 2007; Granovskii et al., 2007(a); Lee et al., 2010; Cetinkaya et al., 2012; Dufour et al., 2012). None of these studies investigates the biological production option in specific detail.

6.3 Study Aims

This chapter aimed to compare the environmental burdens of a single stage biogas (methane) production system (i.e. anaerobic digestion) and a two stage (hydrogen /

methane) production system using two feedstocks with different characteristics, and to identify future research requirements for improving the environmental performance of the processes. For both production systems the raw biogas produced was assumed to be upgraded, compressed and utilised as fuel in a passenger vehicle. It is important to note that the assumptions made, the data used (summarised in Appendix B), the allocation procedures implemented and the LCIA methodology used, all of which have been described in as much detail as possible, have large effects on the final results generated.

6.4 Methods

Environmental burdens were calculated using a Life Cycle Assessment (LCA) approach undertaken in accordance with European guidance (BSi, 2006a; BSi, 2006b). LCA modelling was undertaken using SimaPro v7.3 software (PRè Consultants b.v.). Data relating to the single and two stage treatment of wheat feed was derived from experimental work described in Massanet-Nicolau (2012), whilst data for the treatment of food waste was obtained from separate laboratory work completed as part of this study. Both experiments were undertaken at the Sustainable Environment Research Centre (SERC) at the University of Glamorgan, UK. Where necessary, data has been supplemented with literature values and as a final option the Ecoinvent database v.2.1 (Swiss Centre for Life Cycle Inventories, 2009) has been utilised. The intended audience for the study was primarily considered to be researchers active in the field of biohydrogen and biomethane production and utilisation.

6.4.1 Function and Functional Unit

The product system assessed was the production of either (i) biomethane or (ii) biohydrogen / biomethane vehicle fuel with the primary biogas production process being either (i) single stage mesophilic anaerobic digestion or (ii) dark fermentation followed by mesophilic anaerobic digestion. Feedstocks considered were a laboratory prepared food waste, approximately representative of municipal food waste collected in Wales as described in Wasteworks Ltd. et al. (2010), and wheat feed, a by-product of the flour milling process that has been found to be appropriate for biohydrogen production (Hawkes et al., 2008). Raw biogas produced from both processes was assumed to be upgraded using Pressure Swing Adsorption (PSA), was compressed to 200 bar and was distributed at a refuelling facility for passenger vehicle fuel use. The functional unit of the study was the production of sufficient fuel to achieve 1 km of passenger vehicle transportation. Impacts were compared with a reference fuel of diesel derived from fossil sources.

6.4.2 System Boundary

The system boundary describing the processes modelled is shown in Figure 20. Energy requirements, emissions and primary materials for each sub process were included in the model. Burdens associated with the door to door collection of food waste or the production of wheat feed were not included. The primary purpose of the study was to compare the biogas production systems themselves. Energy requirements and emissions associated with the decommissioning of the service station, compression, upgrading and dewatering plant are also not included as these were anticipated to have a negligible burden compared with the energy and material flows associated with the operational phase of the plant (Berglund et al., 2006).

6.4.3 Allocation Procedures

The production of biogas by both the single and two stage digestion processes was considered as a multi-output process with outputs being (i) 'Biogas', (ii) the service of 'Disposal' of organic waste to a treatment plant (applicable to waste streams only), and (iii) 'Digestate' to agricultural land. Environmental burdens were allocated to each output on an economic basis. Upgraded biogas was assumed to have an economic value of an equivalent volume of diesel (on an energetic basis) which varied between 0.4452 £ / m^3 and 0.5196 £ / m^3 depending on hydrogen content, in addition to attracting Renewable Transport Fuel Certificates (RTFCs) with a value of 0.208 £ per RTFC (2011 average). Both feedstocks were considered as being derived from waste, residues or non food cellulosic material and therefore attract double RTFCs. A gate fee of £40 per tonne was applied to the disposal of food waste. No gate fee was applied to wheat feed as alternative disposal routes could be applicable in many circumstances (e.g. animal feed) and therefore the output of 'Disposal' did not apply. Digestate value was calculated according to measured nitrogen content in feedstocks and applying a value for purchasing an equivalent mass of mineral fertiliser. At the time of writing mineral fertilisers were commercially available at a cost of £330 per tonne (34.5% N). Allocation was applied in order to compare the environmental burdens of methane or hydrogen / methane fuel (derived from one of the outputs, "Biogas", from the multi output process described above) with the reference fuel of mineral diesel (which is itself produced from the multi output process of crude oil refining). However, results for the total process (without diesel comparison) including burdens allocated to all three process outputs (Biogas, Disposal and Digestate) were also considered and allowed direct comparison of all aggregated burdens between the single stage and two stage processes.

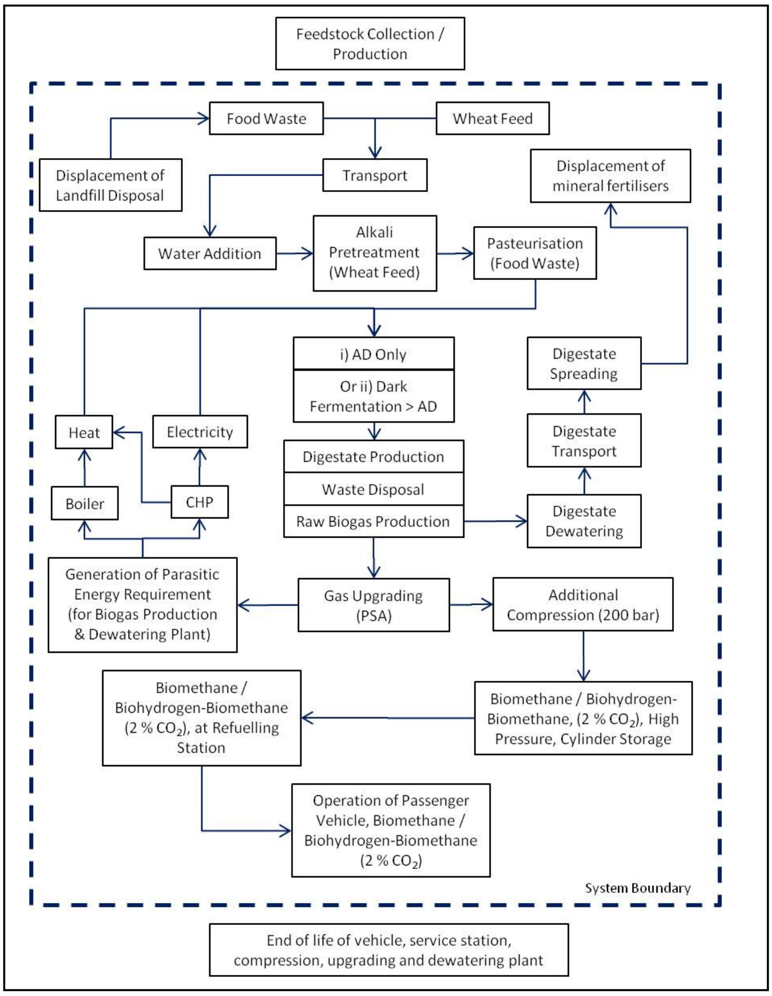


Figure 20 - System boundary of modelled process, all energy, emissions and primary materials are included for individual processes

6.4.4 LCIA Methodology

Life cycle impact assessment was undertaken using the Eco-indicator 99 H/A methodology. The method was chosen as it provided a concise appraisal of end point damage on human health (including the human health impact of climate change), resources (including fossil fuel use) and ecosystem quality. Indicator values were calculated in two stages:

1. Damage factors for the pollutants or resources used were calculated for different impact categories,
2. Normalisation of the damage factors on the level of damage categories.

As with the previous assessment the Hierarchist approach was used as this provided a balance between short term damage (e.g. 100s of years for Individualist approach where technology can be used to avoid many problems) and long term damage (e.g. 1,000s – 10,000s years for Egalitarian approach where problems lead to catastrophe). The normalisation factors as shown in Table 18 were used. The model was also analysed using a midpoint methodology (ReCiPe Midpoint (H)) within the sensitivity analysis. Impact categories of carcinogens / human toxicity, respiratory inorganics, climate change, ecotoxicity and fossil fuels were considered as being most relevant to UK energy and environmental policies.

6.5 Life Cycle Inventory (LCI) Analysis

As per the previous chapter, the treatment process was first modelled within an Excel spreadsheet in order to generate appropriate data for building a further model within SimaPro LCA software. In this case the Excel spreadsheet (summarised in
Figure 21) built upon that developed for the previous modelling exercise, but included additional and optional flows through an initial dark fermentation phase, digestate concentration, gas upgrading and additional options for meeting parasitic loads. The dark fermentation and anaerobic digestion models were informed by experimental work as described below.

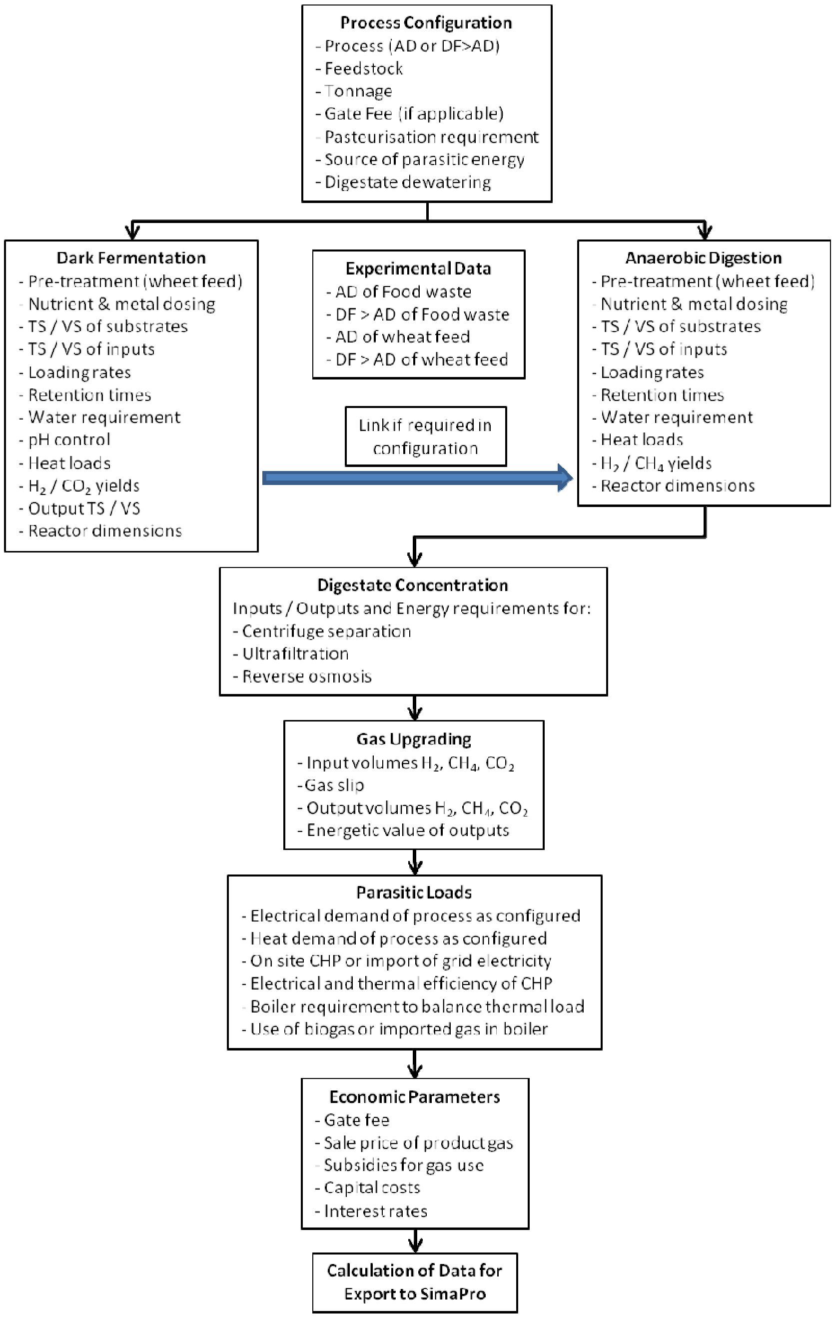


Figure 21 – Summary of primary parameters included in the Excel model of process

6.5.1 One and Two Stage Treatment of Food Waste

The major variations between the biogas infrastructures modelled are summarised in Table 21. Parameters associated with the production of biogas from food waste were determined through laboratory based work. Food waste with characteristics within the range anticipated within Wales as determined in (Esteves et al., 2010) was produced using common household foodstuffs.

Table 21 – Summary of the variations between the modelled biogas production processes

	Alkali Pre-treatment	Nutrient Addition	Trace Element Addition	Pasteur-isation	Dilution	Biohydrogen Production (Dark Fermentation)	Methanogenesis (Anaerobic Digestion)	Digestate Concentration (Centrifuge, UF, RO)
Food Waste_AD	No	No	Yes	Yes	Yes	No	Yes	No
Food Waste_DF_AD	No	No	Yes	Yes	Yes	Yes	Yes	No
Food Waste_DF_AD (w. RO)	No	No	Yes	Yes	Yes	Yes	Yes	Yes
Wheat Feed_AD (w. RO)	Yes	Yes	Yes	No	Yes	No	Yes	Yes
Wheat Feed_DF_AD (w. RO)	Yes	Yes	Yes	No	Yes	Yes	Yes	Yes

For biohydrogen production, food waste was added to a 9.5 litre working volume batch reactor at an organic loading rate of 20 g Volatile Solids (VS) l^{-1} with autoclaved digested sewage sludge added as inoculum at a loading rate of 3 g VS l^{-1}. The remaining volume was made up with clean water. The reactor was sealed and maintained at a temperature of 35°C (±2°C) with continuous mechanical mixing. A minimum pH of 5.5 was maintained using 1M sodium hydroxide. The volume, H_2, CH_4 and CO_2 content of the produced gas was measured. The batch experiment lasted 1.75 days at which point hydrogen production had ceased. Potential biomethane production from both the untreated food waste (i.e. single stage biomethane production) and the effluent from the hydrogen batch reactor (i.e. two stage biohydrogen / biomethane production) was determined at mesophilic temperatures over a 30 day period using an Automatic Methane Potential Test System “AMPTS” (BioProcess Control AB) using digested sewage sludge as an inoculant. The experimental parameters are summarised in Figure 22a.

6.5.2 One and Two Stage Treatment of Wheat Feed

The single and two stage treatment of wheat feed is described fully in Massanet-Nicolau et al., (2013). Wheat feed was sourced from the Premier Food flour mill in Barry, South Wales, UK. Wheat feed was diluted and subject to alkali pre-treatment before being treated in both a semi continuous single phase system (i.e. biomethane production) and a semi continuous two stage system (i.e. biohydrogen / biomethane production). Retention time in the hydrogen reactor and methane reactor was 0.75 days and 19.25 days respectively in the two stage system, and 20 days in the single stage system. The experimental parameters are summarised in Figure 22b.

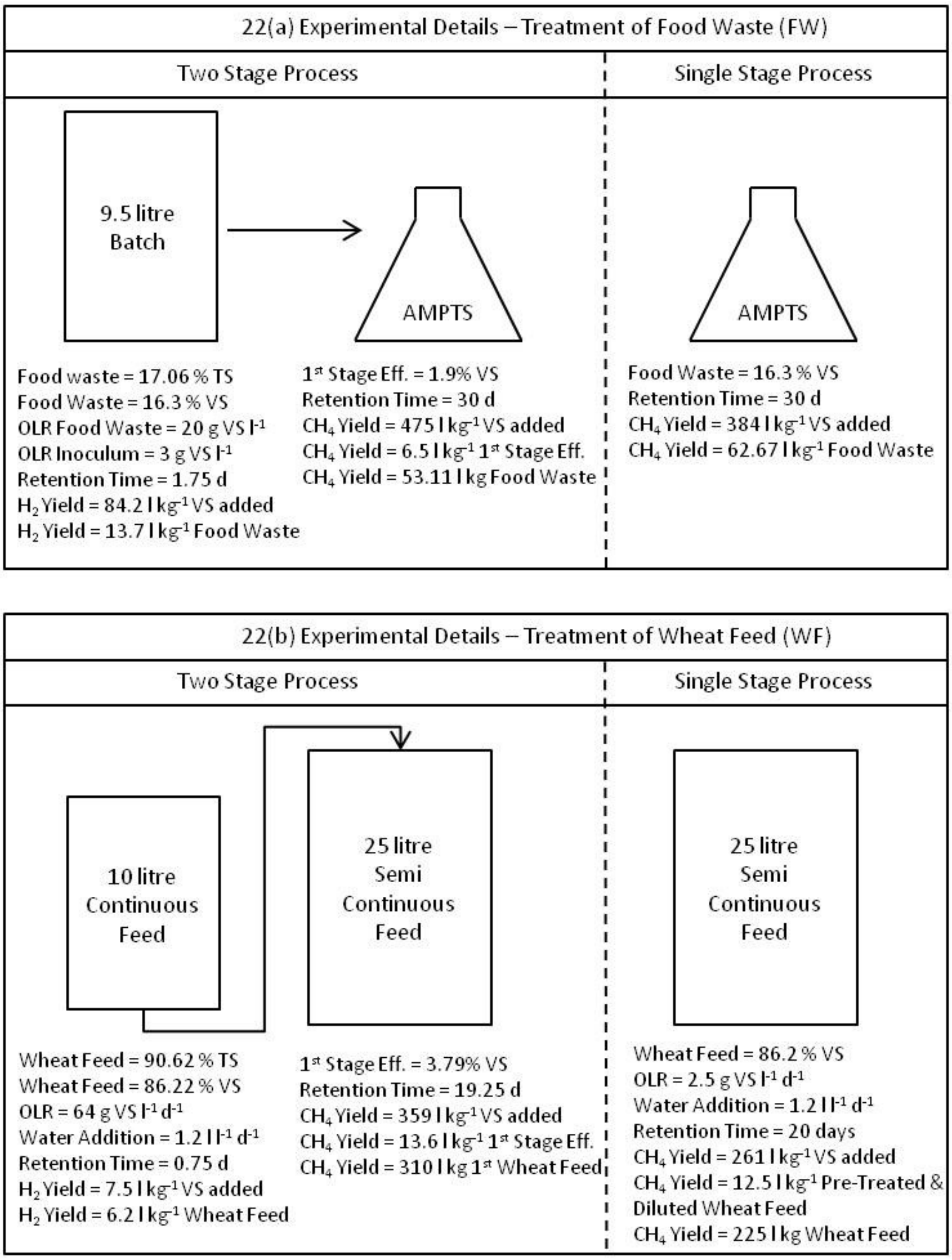


Figure 22 – Summary of experimental work used to derive gas yields for two stage and single stage treatment of, (a) food waste, and (b) wheat feed (from Massanet-Nicolau et al., 2013)

6.5.3 Transportation of Feedstocks and Digestate

Distances for transporting feedstocks to the treatment facility, and digestate to the point of end use, were assumed to be 20 km in all cases. Return journeys were considered to be empty and therefore accrued 80% of the impact of a full journey.

6.5.4 Inventory of Biogas Production Plant

Facilities treating wheat feed were considered to have the same operating conditions as those described above and in Massanet-Nicolau et al., (2013). Process conditions for the production of biohydrogen from food waste were also assumed to be as those described above. Single stage treatment of food waste (i.e. anaerobic digestion) was assumed to treat waste with a total solids input content of 12%. All water required for dilution of feedstocks was assumed to be from mixed potable sources. All food wastes required pasteurisation at a temperature of 70 °C for 1 hour prior to water addition or digestion. Pasteurisation was not required for wheat feed.

The conceptual model for both single and two stage plants included the primary functional components of a biogas plant with each component being assigned one or more of the main construction materials required. These included; tank bases and hardstanding / roadways (gravel sub-base, reinforced concrete); reception building slab (gravel sub-base, reinforced concrete), galvanised steel frame, dense concrete block walls and insulated galvanised steel cladding; shredder and screw pump (steel); reception tank (reinforced concrete); polyisocyanurate foam insulated steel pasteurisation tank (if required); reinforced concrete fermentation / digestion tank(s) with polyisocyanurate foam insulation and aluminium cladding; and reinforced concrete digestate storage tanks with polyester and polyvinylchloride gas storage membranes.

The land required to accommodate each biogas plant was calculated using data gathered from a review of digestion plants across Europe (Monson et al., 2007). For all plants it was assumed that every tonne of construction materials was transported 50 km, the variation between plants is therefore dependent upon only the scale of plant.

All dark fermentation to anaerobic digestion plants included the dewatering of digestates using centrifuge, ultrafiltration (UF) and reverse osmosis (RO) based on the mass balance presented in (Fuchs et al., 2010). Solids and concentrates produced by the dewatering process were exported to agricultural land with clean water being re-circulated to the process. The treatment of food waste via dark fermentation to anaerobic digestion was modelled both with and without dewatering / concentration in order to determine the environmental impacts of this option.

The parasitic electricity demand of both the raw biogas production plant and the digestate dewatering plant were met using an on-site CHP plant using upgraded biogas as a fuel with an electrical conversion efficiency of 32% and a thermal conversion efficiency of 50% (Jungbluth et al., 2007; Monson et al., 2007). Heat generated by the CHP plant was utilised for pasteurisation / reactor heating, and any additional heat required was generated using an on-site boiler using upgraded biogas as a fuel with a thermal efficiency of 98%. Components for an appropriately scaled CHP and boiler plant were included as per the Ecoinvent database. The parasitic electrical demand of biogas production processes was assumed to be 15% of total potential electrical output in all cases. Thermal demands were calculated for each plant.

In accordance with the existing Ecoinvent method for modelling the treatment of wastes via anaerobic digestion (Jungbluth et al., 2007) carbon dioxide fixed within the feedstock was also included as an input of raw material (i.e. consumption of CO_2 from air) during the operation of the biogas plant.

Concentrations of metals, nutrients and trace elements in food waste were as per Esteves et al. (2010) and associated feedstock characterisation work. Concentrations for wheat feed were from various literature sources including Dhuyvetter et al. (1999), Elliot et al. (2002), and Baxter et al. (2006). Where no data was available (Boron, Chloride, Bromine, Fluoride, Iodide, Tin, Vanadium, and Silicon), data included within the Ecoinvent database for biogas production from wastes was utilised.

6.5.5 Inventory analysis of biogas upgrading, compression and vehicle fuel use

Environmental burdens associated with the upgrading of biogas to 98% CH_4 content, compression to 200 bar, transporting gas cylinders a distance of 20 km, dispensing at a service station and emissions associated with the end use of the fuel in a passenger car were included. The existing structure within the Ecoinvent database was used with each element modified to reflect the mass flows appropriate for each plant. The methodology considered Pressure Swing Absorption (PSA) as the upgrading technology including a process methane loss of 1.5%.

6.6 Results

Figure 23 shows the normalised results generated using Eco-indicator 99 H/A for the operation of a passenger vehicle for 1km fuelled by biomethane or biomethane / hydrogen derived from food waste. Figure 23(a), which includes the burdens allocated to “Biogas” and therefore to the vehicle fuel product, indicates that the production and utilisation of either biomethane or hydrogen / biomethane from food waste had lower environmental burdens compared to diesel fuel. Both biomethane and biohydrogen / biomethane reported large negative results (i.e. environmental benefits) for carcinogens (-5.94 E-5 Pt to -7.70 E-5 Pt) and ecotoxicity (-2.76E-5 Pt to -3.33E-5 Pt). Both climate change burden (-2.48E-6 Pt to -3.99E-6 Pt) and fossil fuel use (8.96E-6 Pt to 2.25E-5 Pt) were significantly lower for both biogas production processes than for diesel fuel production. Figure 23(b), which includes the aggregated burdens of all three outputs of the multi output biogas production process (Biogas, Digestate and Disposal), indicated lower burdens for the production of biohydrogen / biomethane from food waste; however, as discussed below, with the exception of fossil fuel use which was significantly higher for the biohydrogen / biomethane production processes, the inefficiencies of the process compared with single stage biomethane production may not be immediately obvious.

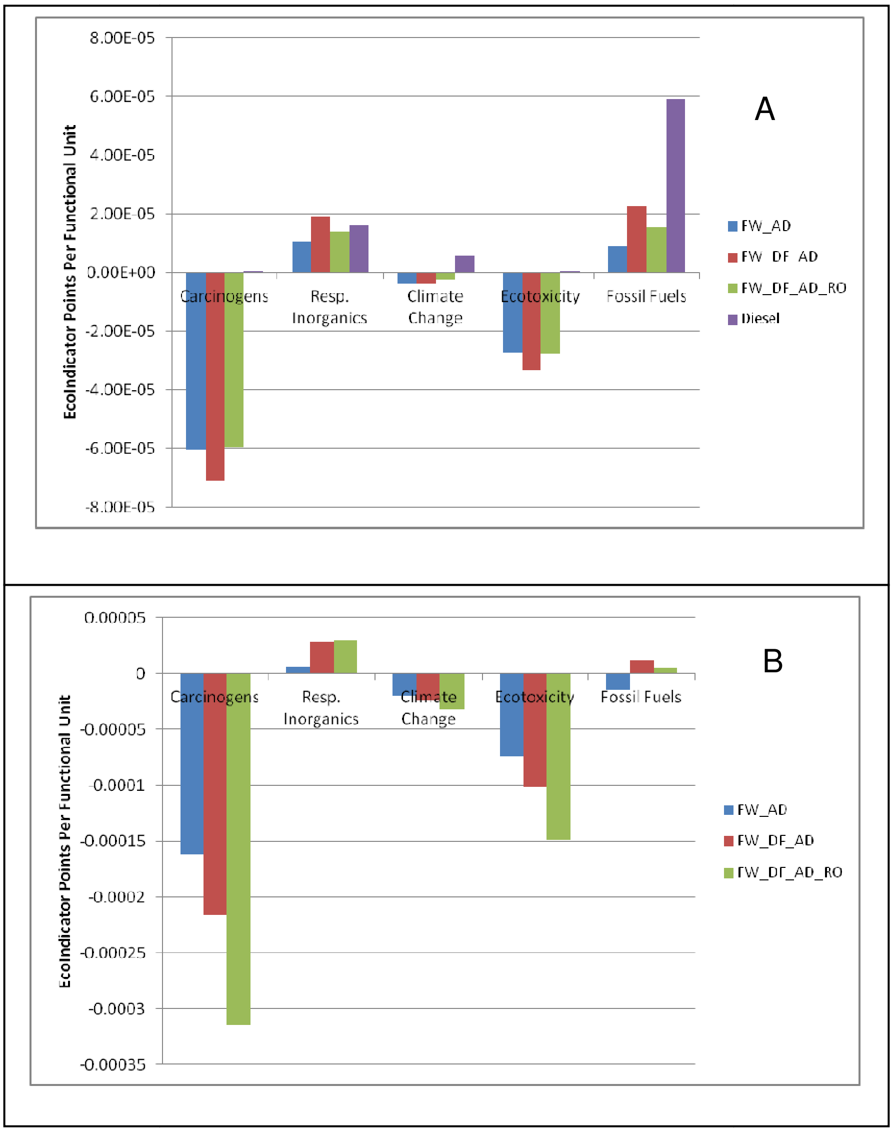


Figure 23 – Ecoindicator 99 (H/A) Normalised LCIA results for treatment of food waste via single stage and two stage systems; (a) impacts allocated to transport fuel utilisation, (b) impacts for whole biogas infrastructure (biogas production, disposal and application of digestate to agricultural land)

Figure 24 presents the normalised results for the operation of a passenger vehicle for 1 km fuelled by either biomethane or biomethane / hydrogen derived from wheat feed. Figure 24(a) indicates that the utilisation of biomethane and biohydrogen / biomethane had fossil fuel advantages over diesel (3.67E-5 Pt, 2.87E-5 Pt and 5.89E-5 Pt respectively), however, for all other impact categories diesel utilisation had lower burdens than either biomethane or biomethane / biohydrogen use. It was notable that climate change burdens of biomethane utilisation, biohydrogen / biomethane utilisation and diesel fuel were relatively similar (6.98E-6 Pt, 5.89E-6 Pt and 5.56E-6 Pt respectively), suggesting that modest and realistic improvements to process efficiencies could have a significant influence on this comparison. Figure 24(b), which quantifies environmental burdens for the whole biogas infrastructures (i.e. all impacts allocated to 'Biogas' and 'Digestate'), clearly demonstrates how the increased energy output of the two stage process when compared with the single stage process led to reductions in overall environmental burdens in all categories.

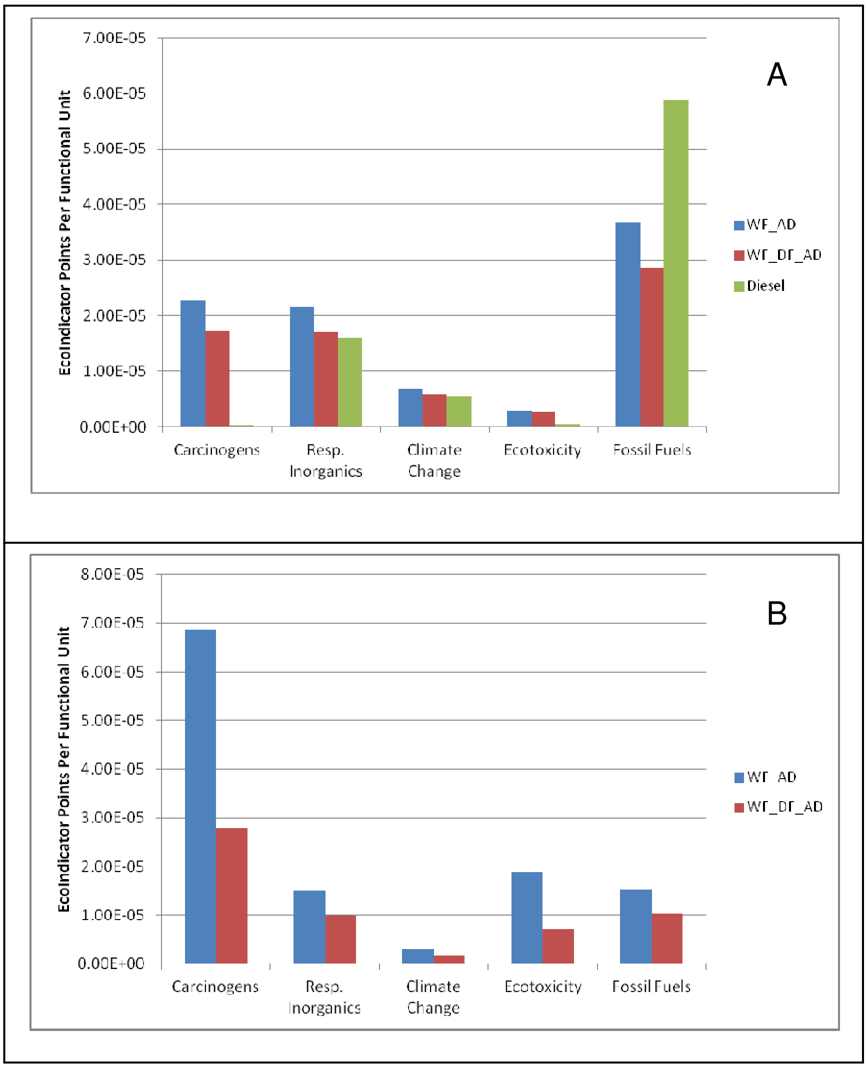


Figure 24 – Ecoindicator 99 (H/A) Normalised LCIA results for treatment of wheat feed via single stage and two stage systems; (a) impacts allocated to transport fuel utilisation, (b) impacts for whole biogas infrastructure (biogas production and application of digestate to agricultural land)

6.7 Discussion

As with all LCA results, an understanding of the process and modelling behind the results is key to reaching appropriate conclusions. Results for the production of biogas from food waste (Figure 23) indicate large reductions in terms of carcinogens and ecotoxicity burdens. These were almost entirely a reflection of the benefits accrued from the diversion of material from landfill which was considered as the alternative disposal route for food waste. The fact that the benefits increased for the biohydrogen / biomethane process compared with the single stage biomethane process was therefore a reflection of the greater inefficiency of the two stage process for this type of waste. Due to the lower overall energetic output of the two stage process when treating food waste under the conditions described, approximately 2.83 kg of waste was required to produce sufficient fuel for 1 km of transportation use compared with 1.97 kg in the single stage process. The waste required increased to 4.05 kg when the additional parasitic demands of digestate treatment by reverse osmosis were included. Whilst this displaced a greater volume of material from landfill and therefore accrued a subsequent environmental benefit, it is clearly neither environmentally or commercially viable to favour a more energetically inefficient process. Overall, the results indicated that the two stage biohydrogen / biomethane process for the treatment of food waste, as configured for this particular study, was not environmentally advantageous when compared with the single stage process.

In the batch experiment for biohydrogen / biomethane production from food waste conditions in the first phase (biohydrogen production) were optimised for hydrogen production with a relatively long residence time (1.75 days) to maximise hydrogen output. However, this resulted in an overall reduction in energy output as methane production in the second stage was detrimentally effected compared to the single stage. Even though hydrogen was present at approximately optimal concentrations in the final gas to maximise emissions benefits at end use (approx. 20% H_2 by volume), the benefits of reduced emissions at end use were not sufficient to counter act the overall lower energy output of the process. Reducing the retention time in the first stage would limit hydrogen production, however, this may have the effect of increasing overall process energy output. Process performance in semi-continuous or continuous configuration may also be significantly different from a batch process. Further research is required to provide additional data which directly compares single and two stage treatment of food waste in a continuous system, and this result highlights the importance of optimising treatment systems according to the specific feedstock or mix of feedstocks utilised.

As wheat feed was not considered as being diverted from landfill disposal, utilising it to produce a vehicle fuel did not attract the same environmental benefits as those afforded to food waste. This is the primary reason why it did not appear to perform well compared with the fossil fuel alternative of diesel. Processes utilising non waste feedstocks therefore need to pay particular attention when assessing their overall impacts, and it is important that any LCA work on these processes present clear and realistic assumptions for system boundaries, co-product end use and any allocation procedures utilised. As an example a review of LCAs undertaken on biodiesel production (Malça et al., 2011) indicated that there was wide variability in these assumptions and that they had a very significant impact on the results of the LCA, in particular for assessing GHG emissions. In many of the scenarios investigated, there was the possibility that GHG requirements of the EU Renewable Energy Directive, 2009/28/EC (European Parliament, 2009a), would not be met.

What is clear is that in the case of wheat feed case, the two stage biohydrogen / biomethane processes resulted in significantly lower environmental burdens than the single stage biomethane production process, primarily as a result of the higher overall energy output of the two stage process. However, whilst the overall energy yield for the two stage process was higher, the hydrogen yield from the first stage was very low. The overall percentage of hydrogen in the final gas was <2% by volume and therefore may not result in any significant emissions benefit at the point of end use. As such, in this particular case the two stage process should be viewed as such; a two stage anaerobic treatment process that increased overall energy yield, and not necessarily a biohydrogen production option. Additional research is required to increase hydrogen yields whilst maintaining energy outputs in the second stage.

A major factor contributing to the overall environmental burdens of both the single stage and two stage treatment of wheat feed was high dilution requirements with associated low organic loading rate in the laboratory scale process. For every kg of wheat feed treated, approximately 18 litres of water was added leading to large process energy requirements and to the production of large volumes of digestate which require treatment and disposal. The impact of increasing the loading rate of the two stage process was explored further in the sensitivity analysis.

6.8 Sensitivity and Uncertainty Analysis

The effect of three key variables was assessed for the two stage production of vehicle fuel from wheat feed.

1. The environmental effect of increasing the loading rate of the process,
2. The environmental effect of reducing digestate transport distances,
3. The effect of using an alternative LCIA methodology.

Another key process factor not included in the sensitivity analysis was the methane emissions from the process, which are currently modelled at 3% of output for the biogas production process and 1.5% of output for the upgrading process. Methane emissions have already been shown to be highly significant in influencing the overall environmental impact of biogas systems (Patterson et al., 2011b) and this was therefore not repeated in this study. Reduction of methane emissions at all stages of fuel and digestate production and utilisation remains as a key priority to the AD and biogas sectors.

6.8.1 Increasing the Loading Rate of Wheat Feed Treatment

The impact of increasing the organic loading rate in the single and two stage biogas production processes utilising wheat feed as a source material was investigated. Organic loading rates to the first stage were increased from 64 kg VS m^{-3} day^{-1} to 127 kg VS m^{-3} day^{-1} (increasing the total process loading rate from 2.4 to 4.8 kg VS m^{-3} day^{-1}) which was equivalent to increasing the first stage feeding rate from 74.23 kg m^{-3} day^{-1} of wheat feed to 147.3 kg m^{-3} day^{-1}. This gave a total solids loading rate to the first stage of approximately 10% which was in line with total solids addition to most single stage CSTR systems and was therefore considered a realistic and achievable goal, although this has not yet been reproduced in the laboratory. This theoretical increase in loading rate would have a number of system benefits including reduced reactor sizes, reduced fresh water use, reduced parasitic energy demand for heating and dewatering, increased volume of biogas available for export and decreased volumes of digestate and associated transport requirements.

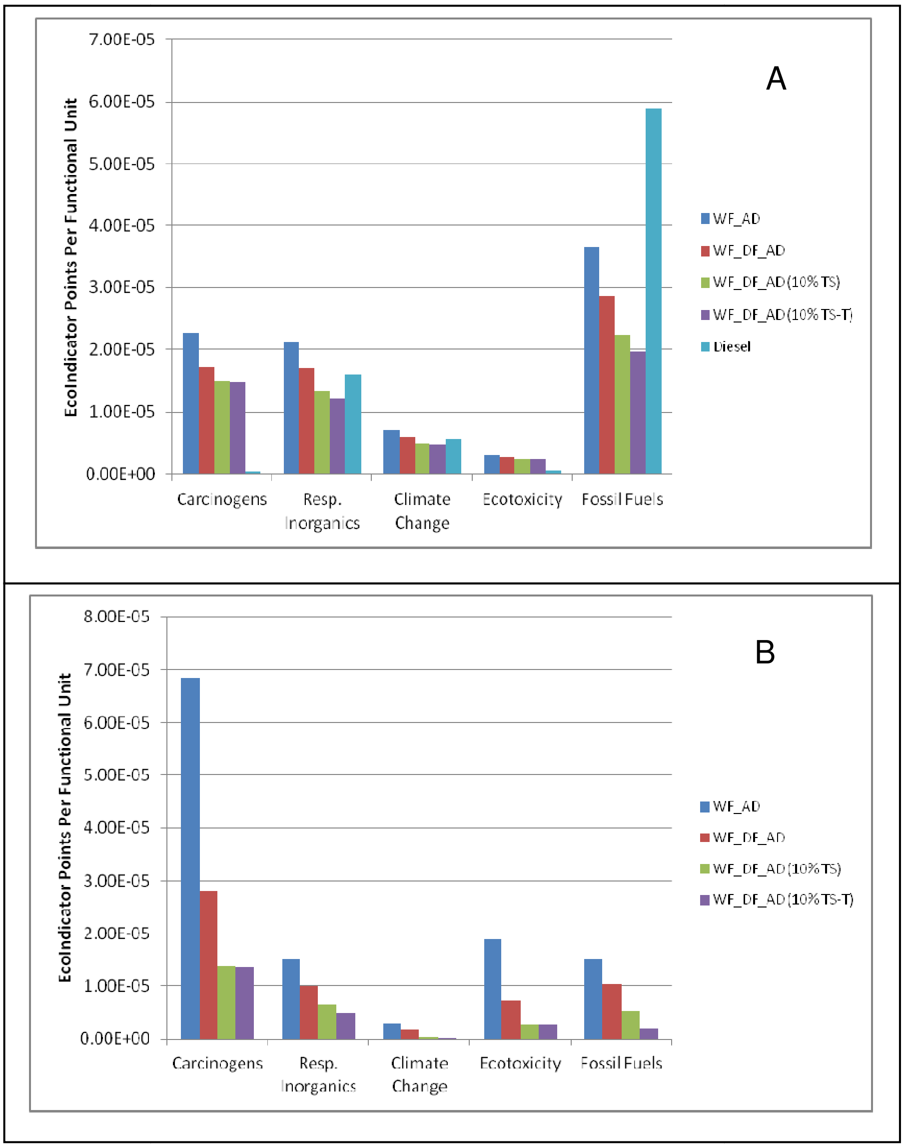


Figure 25 – Ecoindicator 99 (H/A) Normalised LCIA results for treatment of wheat feed via single stage and two stage process including sensitivity parameters; (a) impacts allocated to transport fuel utilisation, (b) impacts for whole biogas infrastructure (biogas production and application of digestate to agricultural land)

The effect of this increase in loading rate is shown in Figure 25 as WF_DF_AD (10%TS). The effect was to decrease overall burdens and as shown in Figure 25(a) the process now outperformed the diesel fuel reference case in terms of respiratory inorganics (1.35E-5 Pt compared with 1.59E-5 Pt) and climate change (4.89E-6 Pt compared with 5.56E-6 Pt) as well as fossil fuel use (2.25E-5 compared with 5.89E-5). Given that digestate residues were applied to land, impacts to categories such as carcinogens and ecotoxicity that are strongly influenced by soil emissions remained relatively high compared with diesel. Figure 25(b) highlights the significant step change to the reduction of impacts of the entire infrastructure (i.e. impacts allocated to both 'Biogas' and 'Digestate') that is possible if loading rates can be increased. Further research is therefore required to develop the process such that higher loading rates can be achieved without reducing energy outputs. Similar benefits associated with reduction in energy requirements for heating and increased availability of gas for end use could also feasibly be achieved using efficient thermal strategies such as solar heating, heat recovery, or thermal integration with other site processes (Ruggeri et al., 2010).

6.8.2 Decreasing Transportation Distances

Even with an increase in loading rates as described above, digestate volumes associated with wheat feed treatment were still large with every 1 tonne of wheat feed treated generating 6.28 tonnes of digestate which required transportation and offsite disposal. The effect of reducing the transportation requirements of digestate to zero was therefore considered. This could be feasible if all nutrients could be utilised on site, either through land application, or other intensive agriculture or aquaculture practices. For the purposes of this study, land application was assumed and emissions associated with land spreading were retained.

The effect of decreasing digestate transportation to zero is shown in Figure 25 as WF_DF_AD (10% TS-T). Impact categories strongly influenced by soil emissions (e.g. carcinogens and ecotoxicity) showed only a very slight reduction as these are dominated by the final destination of the digestate on land, not how it arrives there. Impact categories where air emissions have most influence such as respiratory inorganics and climate change showed significant impact reductions from 6.48E-6 Pt to 4.96E-6 Pt and 3.71E-7 Pt to 7.79E-8 Pt respectively. There was of course a significant fossil fuel saving associated with the reduction in transport requirements. It is perhaps interesting to note that overall, the savings associated with reducing transportation to zero may be considered to be not as significant as increasing the organic loading rate of the process. This highlights the fact that whilst reducing transport distances as far as is practicable undoubtedly makes environmental sense, improving the overall efficiency of the treatment process has a more significant overall effect.

6.8.3 Alternative LCIA Methodology

Alternative LCIA methodologies can affect the final results generated. Therefore, the damage orientated end point methodology of Eco-indicator 99 H/A was replaced with a problem orientated midpoint approach, in this case ReCiPe (Midpoint) H with European normalisation, to assess the effect of using an alternative impact assessment methodology. Normalised output for impact categories relevant to the previous assessment is shown in Figure 26.

Figure 26(a) and Figure 26(b) show the results for the production of biomethane or biohydrogen / biomethane from wheat feed, including sensitivity parameters described above, using ReCiPe (Midpoint) to quantify the burdens associated with the production of transport fuel only in comparison to the reference fuel of diesel. Results were largely in agreement with Eco-indicator 99 H/A in that the largest burdens were associated with human health and land ecotoxicity (Figure 26(a)) associated with the ultimate disposal of residues on agricultural land. Burdens to marine ecotoxicity and aquatic ecotoxicity were both comparatively minor. The greater energy output of the two stage process and the theoretical increase in solids loading both lead to significant reductions in burdens with the reduction in transport having relatively small effect. Figure 26(b) shows that for particulate matter formation and climate change, burdens of both biogas systems were greater than that for diesel fuel until the total solids loading rate was increased to approximately 10%. A small additional burden reduction could also be achieved through eliminating transportation of digestates from site.

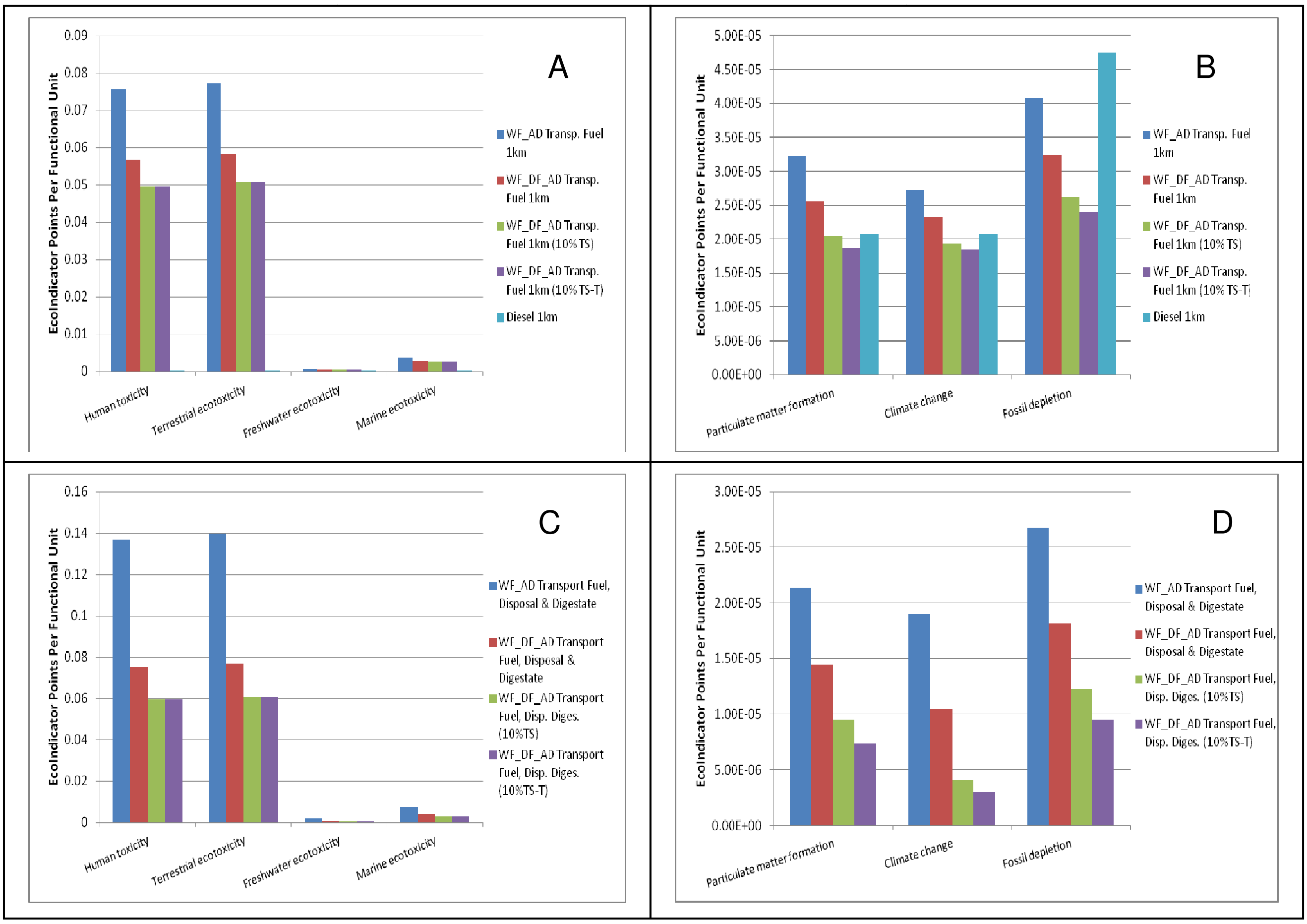


Figure 26 – ReCiPe (Midpoint) European (H) normalised results for wheat feed processes; (a) Transport fuel only, ecotoxicity, (b) Transport fuel only, particulate matter, climate change & fossil depletion, (c) whole process, ecotoxicity, (d) whole process, particulate matter, climate change & fossil depletion

Figure 26(c) and Figure 26(d) show the burdens quantified for the whole biogas infrastructure (i.e. negating allocation effects) which, as with Eco-indicator results, confirms the substantially reduced burdens of the two stage process compared to the single stage and the similarly significant burden reductions that could be achieved if loading rates could be increased to 10% TS. Whilst a dramatic reduction in transportation did lead to a substantial decrease in burdens, the reduction is not as significant as for process based factors that improve the overall efficiency of biogas production.

6.8.4 Uncertainty Analysis

Table 22 shows the results of a Monte Carlo analysis of the comparison between (i) the single stage treatment of wheat feed and the two stage treatment of wheat feed, (ii) the two stage treatment of wheat feed and diesel, and (iii) and (iv) the optimised two stage system for wheat feed and diesel. Results indicated that even accounting for inventory uncertainty the two stage process (WF_DF_AD) was likely to deliver climate change and fossil fuel reductions compared with the single stage process (WF_AD). The results back up the conclusion that diesel has slightly lower climate change burdens than fuel produced from the current experimental two stage treatment of wheat feed. The realistic proposed improvement in process efficiency (WF_DF_AD (10%TS)) was shown to be likely to reverse this conclusion such that climate change burdens for fuel derived from the two stage treatment of wheat feed would be lower than for diesel.

Table 22 – Monte Carlo simulation results of characterised LCIA comparisons between wheat feed fuel production options and diesel

	A	B	A>=B (%)				
			Carcinogens	**Respiratory Inorganics**	**Climate Change**	**Ecotoxicity**	**Fossil Fuels**
i	WF_AD	WF_DF_AD	99.4	79.0	66.4	99.5	63.7
ii	Diesel	WF_DF_AD	0	40.6	44.9	0	99.9
iii	Diesel	WF_DF_AD (10%TS)	0	71.6	72.8	0.3	100.0
iv	Diesel	WF_DF_AD (10%TS-T)	0	83.0	81.1	0.2	100.0

Note: Monte Carlo analysis comprised of 1000 iterations at the 95% confidence limit

6.9 Conclusions

Environmental burdens for the production and utilisation of biomethane vehicle fuel or a biohydrogen / biomethane blend produced from food waste or wheat feed, based on data from two different laboratory experiments, have been compared. For food waste treated by batch processes the two stage system gave high hydrogen yields (84.2 l H_2 kg^{-1} VS added) but a lower overall energy output than the single stage system. Reduction in environmental burdens compared with diesel was achieved, supported by diversion of waste from landfill.

For wheat feed, the semi continuously fed, two stage process gave low hydrogen yields (7.5 l H_2 kg^{-1} VS added) but higher overall energy output. The process delivered reduction in fossil fuel burdens, and improvements in process efficiencies could lead to reduction in CO_2 burdens compared with diesel. The study highlighted the importance of understanding and optimising biofuel production parameters according to the feedstock utilised.

Two stage treatment of wheat feed increased energy outputs and reduced overall environmental burdens compared to single stage treatment. Whilst further increases in process efficiencies are required prior to full scale deployment, this was the preferred treatment option for this ligno-cellulosic feedstock. Based on the limited experimental data available two stage biohydrogen / biomethane production using food waste resulted in increased environmental impacts compared with the single stage process due to lower energy yields. Utilisation of biomethane from food waste as vehicle fuel was beneficial compared with diesel fuel, largely due to savings in landfill emissions.

Chapter 7: Life Cycle Assessment of the Electrolytic Production of Hydrogen and its Utilisation as a Low Carbon Vehicle Fuel

7.1 Introduction

Hydrogen can be produced in an electrolyser by passing an electrical current through pure water, with hydrogen liberated at the negatively charged cathode and oxygen liberated at the positively charged anode. The ability to utilise primary electrical energy from renewable sources such as wind turbine or photovoltaic derived electricity to drive this process raises the potential to produce hydrogen (and oxygen) via low carbon, fully scalable distributed points. One potential future application of the process is to generate hydrogen vehicle fuel on a local or regional basis, either at the point of fuel distribution (i.e. the service station) or at a regional 'hub' for distribution to a number of local refuelling facilities.

State of the art industrial electrolysers including alkali cells and proton exchange membrane cells have a nominal hydrogen production efficiency of around 70-80% (Barbir, 2005; Mazloomi et al., 2012). Therefore there is still a strong argument at present to dedicate renewable technologies such as wind turbines to electricity production as only transmission losses of around 1.6% (National Grid, 2008) are incurred and overall energy output is maximised. However, as the deployment of renewable technologies becomes more widespread in the future, the ability to combine hydrogen generation with electricity generation becomes a more realistic prospect. For example wind turbines could be used to generate hydrogen during times of high wind speeds, or during times when electrical grid capacity is exceeded. UK Government plans to generate 50% of grid electricity using renewable technologies by 2050 (DECC, 2011b) and also raise the prospect of using this low carbon grid electricity for the production of hydrogen vehicle fuels.

An exergy based LCA of hydrogen production and storage technologies was undertaken by Neelis et al. (2004), which found that electrolysis driven by grid electricity incurred the most environmental burdens whilst wind driven electrolysis incurred the least. Gaseous hydrogen storage (340 bar) was found to minimise environmental burdens compared with liquefaction. A similar conclusion that wind driven electrolysis was a favourable option was reached by Koroneos et al. (2004) and Khan et al. (2005). Spath and Mann (2004) demonstrated that the production of the wind turbine itself incurred the most significant environmental burdens for this option. Granovskii (2007(a)) undertook a detailed exergetic LCA of hydrogen production using renewables which again found that wind and PV driven hydrogen production systems could deliver reductions in environmental burdens, but were not cost effective compared to fossil fuel alternatives. When considering GHG emissions it was found that utilising the primary renewable energy sources (e.g. wind turbines, PV) for electricity production rather than for fuel production was a more cost effective option unless fuel cell efficiency was double that of a fossil fuel internal combustion engine (Granovskii et al., 2007(b)). Lee et al. (2010) reached a similar conclusion that wind driven electrolytic hydrogen production could deliver reductions in environmental burdens compared to fossil alternatives, but that using grid electricity (in South Korea) as the primary energy source was more cost effective. The relatively well understood process of renewable energy driven electrolysis also performed comparably in terms of GHG emissions when compared to a number of novel, but as yet unproven, technologies such as two step thermochemical water splitting (Cetinkaya et al., 2012; Dufour et al., 2012).

This study aims to determine the environmental burdens associated with the electrolytic production of hydrogen using renewable technologies (wind and PV power) under UK conditions as compared with the electrolytic production of hydrogen using UK grid electricity. The impact of the future greening of the UK grid electricity mix will also be investigated by using a forecast UK grid mix for 2030, based on the projected pathways included in DECC (2010c). Hydrogen produced is utilised in a PEM fuel cell powered electric passenger vehicle. The fuel production and utilisation pathways were compared with the reference scenarios of the use of petrol as a vehicle fuel, the production of hydrogen using conventional steam methane reforming followed by utilisation in a PEM fuel cell vehicle, and the utilisation of grid electricity in a battery powered electric vehicle.

7.2 Methods

Environmental burdens were calculated using a Life Cycle Assessment (LCA) approach undertaken in accordance with European guidance (BSi, 2006a; BSi, 2006b). LCA modelling was undertaken using SimaPro v7.3 software (PRè Consultants b.v.).

7.2.1 Function and Functional Unit

The product system assessed was the production of a compressed hydrogen vehicle fuel and its utilisation in a fuel cell passenger vehicle. The function of the system was therefore to achieve the transportation of passengers and the functional unit was 100 passenger km (100 pkm). This functional unit also allowed comparison between differing fuel and vehicle options.

7.2.2 System Boundary

The system boundary describing the processes modelled is shown in Figure 27. Energy requirements, emissions and primary materials for each sub process were included in the model (summarised in Appendix B). The primary purpose of the study was to compare the hydrogen production systems themselves. Energy requirements and emissions associated with the decommissioning of the primary energy systems, service infrastructure or vehicles were not included as these were anticipated to have a negligible effect compared with the energy and material flows associated with the production and operational phase of the system (Berglund et al., 2006).

7.2.3 Allocation Procedures

It was assumed that hydrogen was the only product from the electrolytic conversion of water. Although oxygen is also liberated from the process, the volumes generated at the scales considered in this study were considered to be too small to be economic as an oxygen product. As such, no allocation was required within the system.

7.2.4 LCIA Methodology

Life cycle impact assessment was undertaken using the Eco-indicator 99 H/A methodology. The method was chosen as it provided a concise appraisal of end point damage to human health (including the human health impact of climate change), resources (including fossil fuel use) and ecosystem quality.

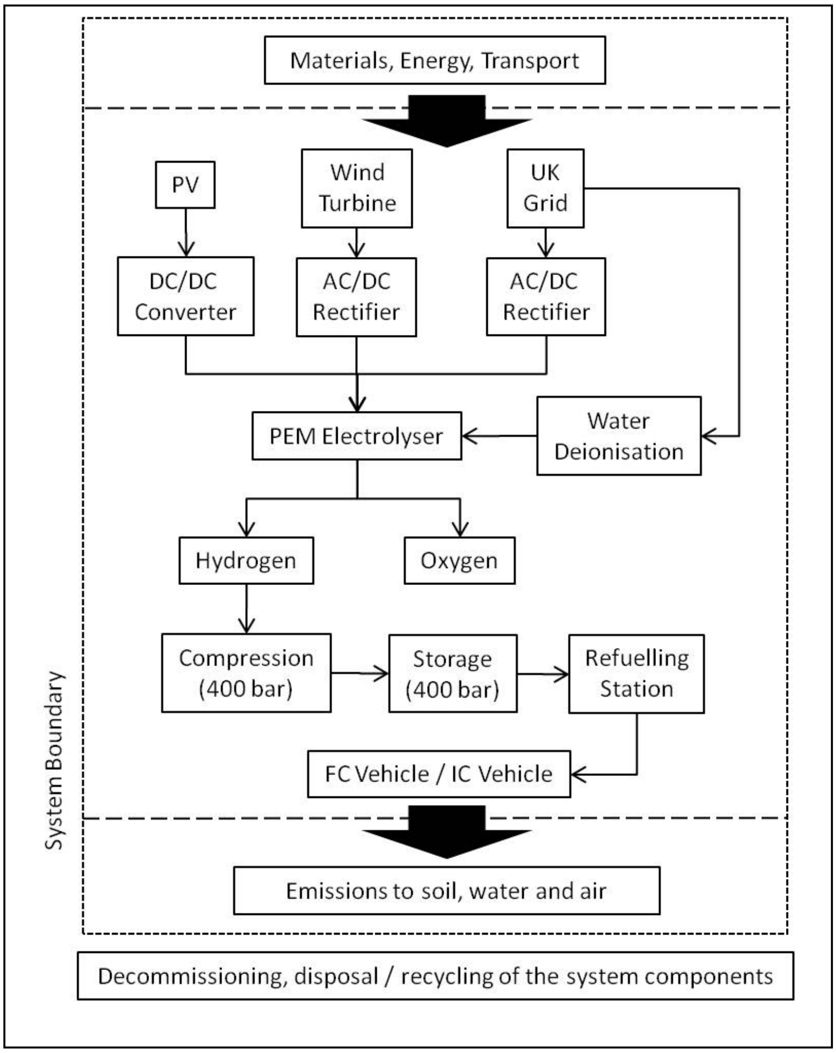


Figure 27 - System boundary of modelled process, all energy, emissions and primary materials are included for individual processes

Indicator values were calculated in two stages:

1. Damage factors for the pollutants or resources used were calculated for different impact categories,
2. Normalisation of the damage factors on the level of damage categories.

The Hierarchist approach was used as this provided a balance between short term damage (e.g. 100s of years for Individualist approach where technology can be used to avoid many problems) and long term damage (e.g. 1,000s – 10,000s years for Egalitarian approach where problems lead to catastrophe). The standard average normalisation factors have been used as shown in Table 18. The model was also analysed using a midpoint methodology (ReCiPe Midpoint (H)) within the sensitivity analysis. Impact categories of carcinogens / human toxicity, respiratory inorganics, climate change, ecotoxicity and fossil fuels are considered as being most relevant to UK energy and environmental policies.

7.3 Life Cycle Inventory (LCI) Analysis

7.3.1 Primary Energy Sources

The Ecoinvent inventory for a 3 kW (peak) silicon single crystal photovoltaic (PV) panel mounted on a sloping roof was utilised. Solar radiation at the location of the production system (Baglan, South Wales, UK) of 3.1 kWh / m^2 / day was used to calculate the total PV area required given an electrical conversion efficiency of 14%. Operational lifetime of the PV panels was 30 years. A total panel area of 278 m^2 was required to meet the demand of the electrolyser.

The inventory from a 30 kW wind turbine in the Ecoinvent database was included. Average wind speed at the location being considered was relatively low at just 5 m / s giving a relatively poor capacity factor of just 13% and an energy output of 34,231 kWh / yr (Wind Energy Resources Ltd., 2010).

Latest UK Grid electricity was based on the energy generation mix between April 2011 and March 2012 as published by DECC (2012c). This comprised of electricity generation by coal (29.2%), natural gas (40.7%), nuclear (19.1%), renewables (9.2%) and other unspecified technologies (1.8%). A further breakdown on the renewable energy technologies included in the grid mix is provided in Table 23. An estimate was also made of the potential grid electricity mix in the UK in 2030. This was based on the HM Government 2050 Pathways

Analysis report (DECC, 2010c) which sets out a range of potential pathways which the future UK energy strategy could take. Within this report a number of potential Levels of behavioural change and technology deployment were included in the analyses. These range from Level 1, which was a pessimistic outlook where either the current status quo was maintained or technology deployment was not favourable to meeting 2050 targets, through to Level 4 which was an optimistic outlook where energy demands are reduced through behavioural change and technology deployment is favourable for meeting 2050 targets.

Table 23 – Breakdown of renewable energy generation included in the UK 2011/12 grid mix

	April 2011 – March 2012*		**Estimate for 2030†**	
Technology	**GWh**	**% of total generation**	**GWh**	**% of total generation**
Total UK Electricity Generation	**374,022**	**100**	**607,000**	**100**
Non Renewable				
Natural Gas (with CCS for 2030)	152,227	40.7	75,000	12.4
Coal	109,214	29.2	ND	ND
Nuclear	71,438	19.1	180,000	29.6
Renewables (see below)	34,410	9.2	352,000	58.0
Other	6,732	1.8	ND	ND
Renewables				
Wind (Onshore)	10,372	2.77	70,000	11.5
Hydro	5,686	1.52	7,000	1.20
Wind (Offshore)	5,126	1.37	170,000	28.0
Landfill Gas	4,979	1.33	ND	ND
Co-firing	2,964	0.79	ND	ND
MSW Combustion	1,739	0.46	40,000	6.6
Plant Biomass combustion	1,683	0.45	30,000	4.9
Sewage sludge digestion	755	0.20	ND	ND
Animal biomass	614	0.16	ND	ND
Photovoltaic	252	0.07	10,000	1.6
Anaerobic digestion	239	0.06	8,000	1.3
Geothermal electricity generation	ND	ND	10,000	1.6
Tidal	ND	ND	4,000	0.7
Wave and tidal stream	ND	ND	3,000	0.5

Note: CCS = Carbon capture and storage ND = No data available
Source: * Digest of United Kingdom Energy Statistics (DUKES) (DECC, 2012)
† Estimate based on HM Government 2050 Pathways Analysis Report (DECC, 2010)

For the purposes of this study, 2030 figures midway between Levels 2 and 3 were selected as this was deemed to represent a balanced outlook towards future energy demand and generation. Figures for the 2030 energy mix used in this study are presented in Table 23. Inventories for the individual generation technologies existing within the Ecoinvent database were utilised.

7.3.2 Electrolyser and Hydrogen Production

Electrolyser performance characteristics were based on manufacturer's data provided for the HPac40 Proton Exchange Membrane electrolyser (ITM Power Plc). This included a hydrogen flow rate of 40 standard litres per minute, an output pressure of 15 bar, a power rating of 11 kW and a production efficiency of 4.8 kWh / Nm^3 (peak). As detailed inventory information relating to the electrolyser was not available, the Ecoinvent inventory for a 2 kW PEM fuel cell was used, given that a fuel cell and electrolyser would have very similar internal components and could be scaled according to output. This was based on a PEM fuel cell with a 32% electrical efficiency utilising natural gas (CH_4) as a fuel. All inventory items were scaled up by a factor of 3.0 according to the ratio between hydrogen utilisation efficiency in the Ecoinvent fuel cell inventory and the output efficiency of the HPac40 as shown in (1).

$$\frac{HPac_{eff}}{1 / \left(\left(\left(\frac{100}{FC_{eff} \times NG_{energy}} \right) \frac{\rho\, CH_4}{MM\, CH_4} \right) \times \left(\frac{MM\, H_4}{\rho\, H_2} \right) \right)} \qquad (1)$$

Where:

$HPac_{eff}$ = Efficiency of HPac electrolyser (4.8 kWh / m^3)

FC_{eff} = Inventory fuel cell electrical efficiency (32%)

NG_{energy} = Gross energy per m^3 of natural gas (9.92 kWh)

$\rho\ CH_4$ = Density of methane at STP (0.66 kg / m^3)

$MM\ CH_4$ = Molar mass of methane (16)

$MM\ H_4$ = Molar mass of H_4 (4)

$\rho\ H_2$ = Density of hydrogen at STP (0.082 kg / m^3)

Inventory items included land use, steel, chromium steel, aluminium, cast iron, titanium dioxide, charcoal, polyethylene, polypropylene, polystyrene foam, platinum, an inverter, PEM fuel cell stack, heating unit, sheet rolling (for steel, chromium steel and aluminium), injection moulding, transport, production energy, production water and housing within a building. The electrolyser was assumed to have an operational life of 15 years with 75% technical availability. Inventory items for the maintenance of the electrolyser and the generation and consumption of de-ionised water at a rate of 10 litres / m^3 hydrogen produced were also included. Emissions during the operation of the electrolyser were waste heat (1.29 kWh / m^3 H_2), oxygen (2.66 kg / m^3 H_2) both released to atmosphere, and unconverted de-ionised water (1.96 litres / m^3 H_2) discharged to surface water.

7.3.3 Hydrogen Compression and Storage

Hydrogen produced by the electrolyser was assumed to be compressed from the electrolyser output pressure of 15 bar to a pressure of 400 bar. As a worst case the energy required for adiabatic compression was calculated according to (2).

$$W=[\gamma/(\gamma-1)]p_oV_o[(p_1/p_o)^{(\gamma-1)/\gamma}-1] \qquad (2)$$

Where:

W = Specific compression work (J / kg)
p_o = Initial pressure (Pa)
p_1 = Final pressure (Pa)
V_o = Initial specific volume for hydrogen (11.11 m^3 / kg) (Bossel et al., 2003)
γ = adiabatic coefficient for hydrogen (1.41) (Bossel et al., 2003)

This gave a compression energy requirement of 13.42 MJ / kg (3.73 kWh / kg) hydrogen. A gas loss of 1.5% of the input volume was included with lost hydrogen included as an emission to atmosphere. Compressor lifetime was assumed to be 15 years. Inventory materials for a 4 kW screw compressor within the Ecoinvent database were used.

Compressed hydrogen was assumed to be stored in aluminium body, carbon fibre wrapped, high pressure storage cylinders. These had an unfilled cylinder mass of 170.5 kg (Dynetek Industries Ltd., 2006), of which 150 kg was assumed to be aluminium and 20.5 kg was assumed to be carbon fibre. 1 kg carbon fibre was assumed to be produced from 2 kg acrylonitrile and 0.1 kg epoxy resin with an input of 7.56 MJ of medium voltage electricity (De Vegt et al., 1997). Storage capacity of the cylinder was 8.64 kg H_2.

7.3.4 Fuel Transportation and Distribution

Cylinders of compressed hydrogen were assumed to be transported (16 – 32 t Euro4 lorry) a distance of 25 km to the point of fuel retail. Empty cylinders were transported back to the production facility by return journey. Refuelling facilities were included in the model as per a natural gas service station present within the Ecoinvent database.

7.3.5 Fuel Cell Vehicle and Fuel End Use

Hydrogen fuel was utilised within an 85 kW PEM fuel cell driven vehicle, which incorporated a $LiMn2O_4$ battery for storage of electrical energy. Total mass of the propulsion system was 600 kg comprising of the battery (40 kg), electric motors (104 kg) and the PEM fuel cell (456 kg) (Sørensen, 2004). The inventory for the 2 kW PEM fuel cell within the Ecoinvent database, which had a total mass of 110.6 kg including the stack, was scaled linearly using a scaling factor of 4.12 to determine the masses of inventory materials for the vehicle fuel cell. Annual maintenance of the fuel cell was also included with an anticipated operation lifetime of 15 years. Inventories for the appropriate mass of $LiMn2O_4$ battery and the electric motors from the Ecoinvent database were utilised. The vehicle had a chasis and body mass of 800 kg (Sørensen, 2004) and the inventory of a standard passenger vehicle within the Ecoinvent database (mass of 1,307 kg) was adapted using a scaling factor of 0.61.

A fuel utilisation efficiency of 96.5 km / kg H_2 was included (Honda, 2012) which gave a fuel utilisation per km of 0.0104 kg H_2. This included an allowance of 0.5% unconverted hydrogen which was emitted to atmosphere. The remaining 99.5% of hydrogen fuel was assumed to be converted to water vapour and emitted to atmosphere. Waste heat emitted from the vehicle was based on the electrical efficiencies of the major components including the fuel cell (55%), DC/AC control (97%), electric motor (92%) and transmission (98%) with the energetic balance being converted to heat (Campanari et al., 2009). Ecoinvent figures for particulate emissions to atmosphere, emissions to soil and emissions to water that are primarily associated with break and tyre wear were utilised.

7.3.6 Reference cases

A number of reference cases were included so as to provide a wide comparison of results. Crude oil extraction and the production and utilisation of petrol was included as the current fossil fuel based standard for passenger vehicle transportation fuel. 1 kg of hydrogen fuel was equivalent to 3.27 kg of petrol based on their respective higher heating values of 39 kWh / kg and 11.94 kWh / kg. The existing Ecoinvent inventory for the fleet average operation of a petrol passenger vehicle was used (total vehicle mass 1.3 tonnes).

Hydrogen production from large scale steam methane reforming was included as the current industrial standard method for hydrogen production. The existing inventory within Ecoinvent was used.

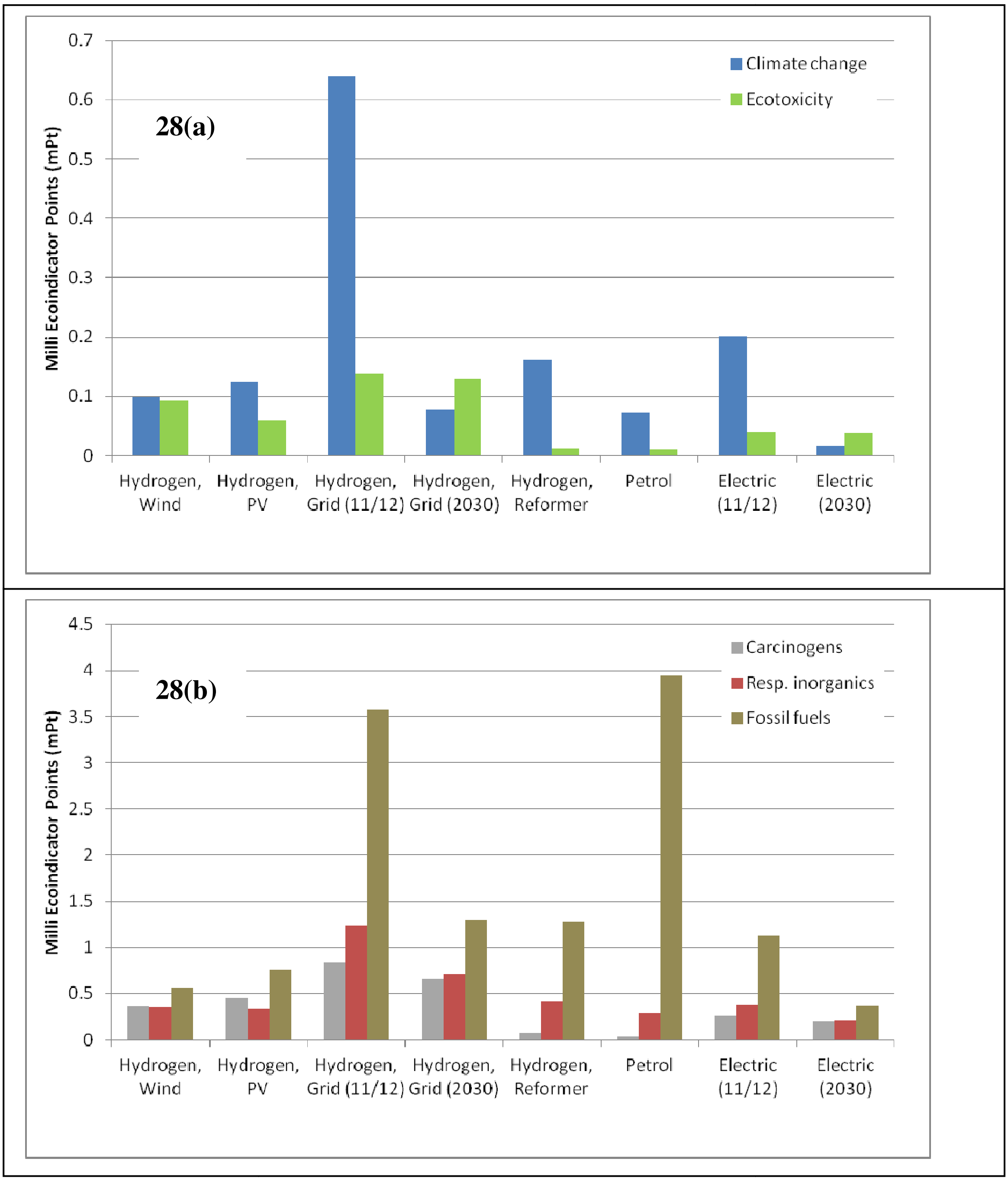
Finally the utilisation of an electric vehicle powered using grid electricity was also modelled. The existing Ecoinvent inventory for a $LiMn2O_4$ battery, electric passenger vehicle was used with the primary energy source being either the UK 2011/12 grid mix or the estimated UK 2030 grid mix as discussed above.

7.4 Results & Discussion

7.4.1 Results

Figure 28 shows the normalised results generated using Eco-indicator 99 H/A for the production and transportation to a service station of sufficient compressed hydrogen to achieve 100 pkm of transportation. Figure 28(a) indicates that climate change burdens for the production of hydrogen using wind powered electrolysis (0.09 mPt) and PV powered electrolysis (0.12 mPt) were lower than for the current standard hydrogen production method of steam methane reforming (0.16 mPt). Utilising current (2011/2012) UK grid electricity for hydrogen production realised the highest climate change burdens (0.64 mPt) although if the aspirational changes to UK electricity generation for 2030 can be achieved, climate change burdens for hydrogen generation using grid electricity could have low climate change impacts (0.08 mPt). This is reflected in the results for production of electricity sufficient to achieve 100 pkm of transportation in an electric (battery only) vehicle in 2011/12 (0.2 mPt) or using 2030 grid electricity (0.01 mPt). However, the existing fuel production option with the lowest climate change impact was the extraction of crude oil and production of petrol (0.07 mPt), although it should be noted that burdens associated with the initial carbonation of the fossil fuel are not accounted for until it is utilised and fossil CO_2 is emitted.

Fossil Fuel burdens (Figure 28b) were highest for the production of petrol (3.94 mPt) followed by hydrogen using current (2011/12) UK grid electricity (3.58 mPt) which reflected the UK's current reliance on fossil fuels for the majority of its electricity generation. Hydrogen production from steam methane reforming and hydrogen production from 2030 UK grid electricity had similar burdens (1.27 mPt and 1.30 mPt, respectively). Current (2011/12) grid electricity production and future (2030) grid electricity production had fossil fuel burdens of 1.13 mPt and 0.37 mPt respectively. Low fossil fuel burdens were also evident for hydrogen production using PV (0.76 mPt) and hydrogen production using wind power (0.57 mPt).



Note: Milli Ecoindicator Point (mPt) = 1/1000 Ecoindicator Points (Pt)

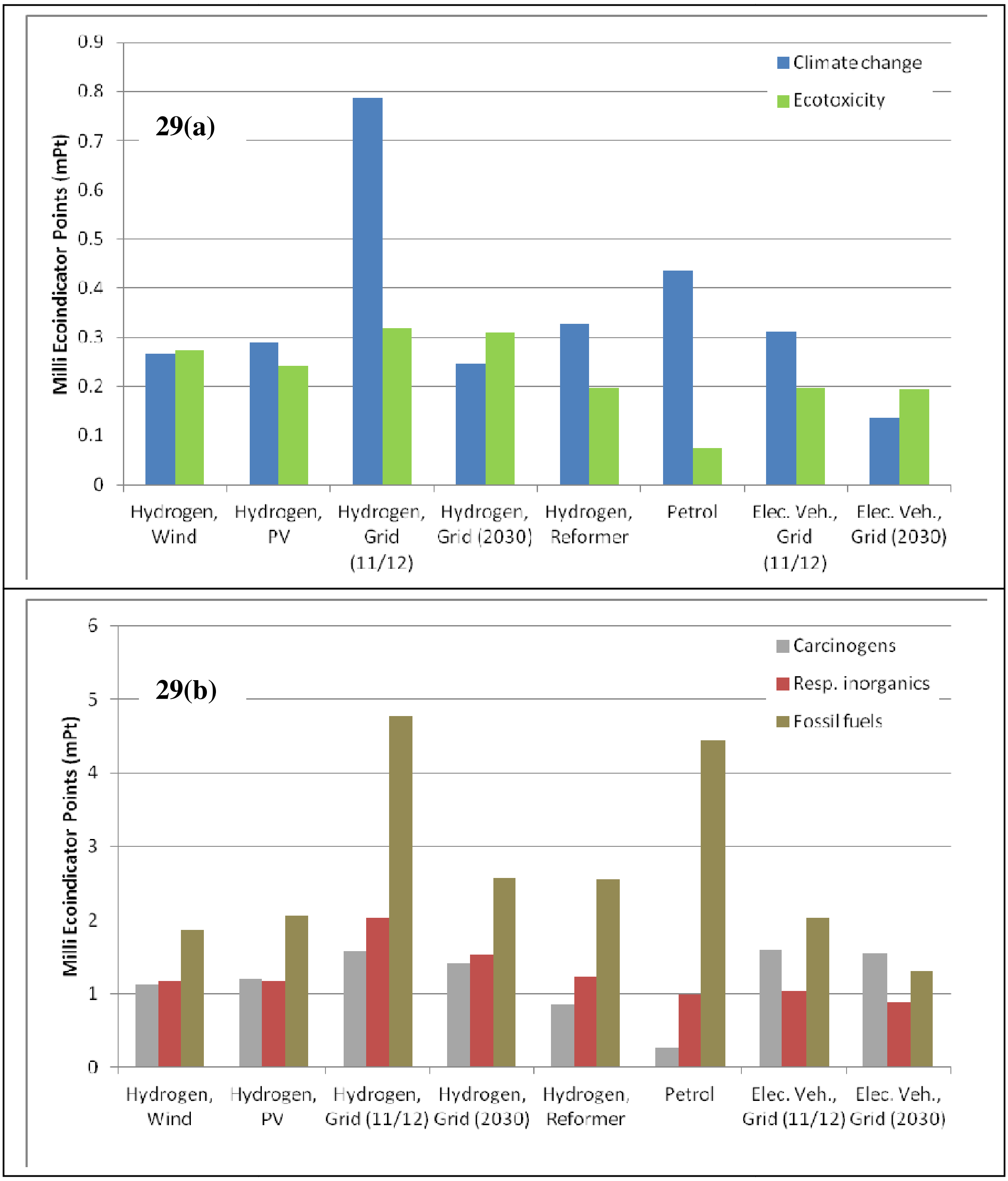
Figure 28 – Ecoindicator 99(H/A) normalised results for production of transport fuel and transport to retail point for impact categories (a) climate change and ecotoxicity; and (b) carcinogens, respiratory inorganics and fossil fuels.

Production of hydrogen using 2011/12 UK grid electricity had the highest burden for carcinogens (0.84 mPt), respiratory inorganics (1.24 mPt) and ecotoxicity (0.14 mPt), and although these burdens are reduced in the 2030 grid mix (to 0.66 mPt, 0.72 mPt and 0.13

mPt respectively), they were still higher than other fuel production options considered. The fossil fuel options of hydrogen from steam methane reforming and petrol had the lowest burdens for carcinogens (0.08 mPt and 0.03 mPt, respectively) and ecotoxicity (both 0.01 mPt) with petrol having the lowest burdens for respiratory inorganics (0.29 mPt) which again was a reflection of the relatively low inputs required to extract and process those fuels. Both hydrogen production using PV and wind energy had higher burdens for carcinogens (0.45 mPt and 0.36 mPt, respectively), respiratory inorganics (0.33 mPt and 0.35 mPt, respectively) and ecotoxicity burdens (0.06 mPt and 0.09 mPt).

Figure 29 shows the normalised results generated using Eco-indicator 99 H/A for the utilisation of a passenger vehicle to achieve 100 pkm of transportation which includes burdens for both fuel production and vehicle operation. Production of hydrogen using 2011/12 Grid electricity remained the option with the highest climate change (Figure 29a) burdens (0.79 mPt). Due to the relatively high emissions during vehicle use, petrol was the second worst option in terms of climate change burdens (0.44 mPt) followed by the production of hydrogen via steam methane reforming (0.33 mPt) and electric vehicles using the 2011/12 UK grid (0.31 mPt). Climate change impacts were comparable for hydrogen using PV (0.29 mPt) and hydrogen using wind energy (0.27 mPt), both being marginally higher than hydrogen from the UK 2030 grid (0.24 mPt). The option with the lowest climate change burden was the electric vehicle utilising the UK 2030 Grid electricity (0.13 mPt).

Fossil fuel burdens (Figure 29b) were highest for the utilisation hydrogen produced using UK 2011/12 grid electricity (4.77 mPt) and for petrol (4.44 mPt). Hydrogen from 2030 grid electricity had similar fossil fuel burden to hydrogen from steam methane reforming (2.58 mPt and 2.56 mPt, respectively). Utilisation of hydrogen from PV incurred fossil fuel burdens of 2.06 mPt, similar to that of electric vehicles powered by 2011 / 12 grid electricity at 2.03 mPt. Lowest fossil fuel burdens were achieved by utilisation of hydrogen from wind energy (1.88 mPt) and electric vehicles using 2030 grid electricity (1.31 mPt).



Note: Milli Ecoindicator Point (mPt) = 1/1000 Ecoindicator Points (Pt)

Figure 29 - Ecoindicator 99(H/A) normalised results for the production and utilisation of vehicle fuels for impact categories (a) climate change and ecotoxicity; and (b) carcinogens, respiratory inorganics and fossil fuels.

The utilisation of electric vehicles powered by 2011/12 grid electricity, hydrogen produced using UK grid electricity (2011/12) and electric vehicles utilising 2030 UK grid electricity resulted in the highest carcinogen burdens (1.6 mPt, 1.59 mPt and 1.54 mPt respectively), closely followed by the utilisation of hydrogen produced using 2030 grid electricity (1.41

mPt). These were followed by the production of hydrogen using PV generated electricity (1.21 mPt) and hydrogen production using wind energy (1.12 mPt). The fossil options of utilising hydrogen from steam methane reforming and the utilisation of petrol had the lowest carcinogen burdens (0.85 mPt and 0.27 mPt, respectively). Utilisation of grid electricity for the production of hydrogen utilising both the existing energy mix (Hydrogen Grid (11/12)) and the 2030 grid mix (Hydrogen Grid (2030)) also had the highest burdens for respiratory inorganics (2.04 mPt and 1.54 mPt, respectively), followed by the utilisation of hydrogen from steam methane reforming (1.24 mPt). Respiratory inorganic burdens for both the utilisation of hydrogen from wind energy and hydrogen from PV were comparable (1.18 mPt and 1.17 mPt, respectively) as were those for electric vehicles using the 2011/12 grid, petrol vehicles and electric vehicles using the 2030 grid (1.04 mPt, 1.0 mPt and 0.88 mPt, respectively). Ecotoxicity burden was also highest for production of hydrogen using both 2011/12 and 2030 grid mixes (0.32 mPt and 0.31 mPt, respectively) followed by the production of hydrogen using wind and PV generated electricity (0.27 mPt and 0.24 mPt, respectively). The lowest ecotoxicity burdens were associated with the utilisation of electric vehicles using either 2011/12 or 2030 grid mix, or hydrogen from steam methane reforming (all 0.19 mPt) and the utilisation fossil petrol (0.07 mPt).

7.5 Discussion

Results indicate that, with the exception of fossil fuel burdens, the extraction of crude oil and *production* of petrol resulted in lower environmental burdens than the fossil or renewable based hydrogen production methods analysed (Figure 28). This serves to demonstrate the efficiency of crude oil as an energy source and the efficiency of the refining process in producing multiple products and fuels. However, it should be noted that fossil fuels have the significant advantage in not requiring any inputs for the carbonisation of the original organic material as this was achieved by natural processes over geological timescales, whereas alternative fuels manufactured over short timescales require inputs of primary energy and materials either to produce this carbon (e.g. energy crops) or capture natural energy sources (e.g. wind powered electrolysis). When the inputs and emissions associated with the *extraction, production and utilisation* of the fuels were included in the analysis (Figure 29), petrol resulted in some of the highest burdens in terms of climate change and fossil fuel use, although carcinogen, ecotoxicity and respiratory inorganic burdens remained low. Given that the primary drivers for diversion from fossil fuels are reducing climate change impacts and future security of supply, it can be seen that alternative transport fuels can deliver substantial benefits in these areas.

The results indicated that the production and utilisation of hydrogen using electrolysis powered by the current (2011/2012) UK grid mix did not represent the best option in terms of limiting environmental burdens. This result reflected the UK's current reliance on fossil fuels for the generation of the majority of its electricity and the comparatively low contribution of renewable energy within this mix. Utilising the 2011/12 electricity grid for direct powering of electric vehicles did produce a reduction in climate change and fossil fuel burdens compared with petrol. The increased level of renewable energy included in the potential 2030 electricity grid mix used in this study was sufficient to reduce climate change impacts associated with grid electricity powered electrolysis to a level that was comparable to other options considered, although burdens associated with other impact categories remain significantly higher than these alternative approaches.

With the exception of the utilisation of the notional 2030 grid electricity mix for powering electric vehicles, the utilisation of hydrogen produced by electrolysis driven by renewable energy sources such as wind and PV electricity resulted in the lowest fossil fuel burdens with savings in climate change burdens also evident when compared to fossil or grid alternatives. This is largely due to the low emissions associated with the utilisation of hydrogen within a fuel cell powered vehicle. Carcinogen burdens remained higher than for other options, which was a reflection of the additional inputs and emissions associated with the manufacture and operation of the additional fuel production infrastructure required to produce hydrogen at the relatively small scales considered in this study. Never-the-less, the utilisation of hydrogen produced from wind or PV driven electrolysis was clearly an attractive option in terms of achieving UK policy ambitions.

However, in the event that the UK electricity grid can be decarbonised by 2030 to the extent considered in this study, the option with the lowest climate change and respiratory inorganic burdens, and almost the lowest fossil fuel burdens was the utilisation of grid electricity to power electric vehicles. This option also provided lower carcinogen and ecotoxicity burdens than the renewable hydrogen production methods. As such the utilisation of 2030 grid electricity to power electric vehicles represented the most favourable option in terms of limiting overall environmental burdens.

It is unlikely that a single transport fuel will be sufficient to reduce fossil fuel use to the extent required in the future, or that a single fuel will be able to meet the technical demands of our road transport requirements. For example this study only considered small passenger vehicles, where as heavy goods vehicles would have a different set of performance requirements for their fuels. As such it is likely that a number of primary energy sources and energy carriers will be required in the future (European Commission, 2011). For example when considering passenger vehicles, it is feasible that electric only vehicles could be used for short range intra-city applications, hydrogen fuel cell vehicles for short to medium range applications, and methane / synthetic fuelled vehicles for long distance / heavy goods transport applications (McKinsey & Company, 2010). Given the results described previously, the combination of electric vehicles powered by the grid with fuel cell vehicles utilising renewably derived hydrogen represents a favourable environmental option.

This assessment considered the environmental burdens associated with small scale renewable energy facilities that were dedicated to producing hydrogen fuels. Further assessment is required to assess the impacts of utilising large scale wind farms and PV arrays for part-time production of transport fuels at times of low electricity demand or low grid capacity.

7.6 Sensitivity and Uncertainty Analysis

The effect of two key variables was assessed for the production and utilisation of alternative vehicle fuels.

1. The environmental effect of increasing the capacity factors of wind turbine and PV panel for electrolytic hydrogen generation,
2. The effect of using an alternative LCIA methodology.

7.6.1 Increasing capacity factors for wind turbine and PV panels

The wind turbine capacity factor in the assessment above was just 13%, this being based on the sub-optimal average wind speed of 5 m/s anticipated at the test location of Baglan in South Wales. Wind speed was increased to 7 m/s to give a generation potential from the 30 kW turbine of 82,879 kWh / yr which gave a more realistic capacity factor of 32%. Similarly, the PV panel included in the above assessment had a conversion efficiency of 14%, which seemed typical of the average performance of currently deployed technology. However, panel efficiencies of 16 – 20% are being reported by product manufacturers (e.g. Sunpower E20 / 333, Sanyo Electronic HIT-N240SE10) and it was a reasonable assumption that improvements in manufacturing technologies would result in an overall increase in panel efficiencies in the future. For the purposes of the sensitivity analysis, a panel efficiency of 17% was therefore considered.

Figure 30 indicates that locating the wind turbine in a more optimal location reduced carcinogen burdens by approximately 12.5% and respiratory inorganics by 10%. Climate change burdens were reduced by 7.5%, ecotoxicity by 15% and fossil fuel burdens by 5.5%. Environmental burdens remain approximately between those of electric vehicles using 2011/12 electricity and those utilising 2030 electricity. The increase in PV panel efficiency saw a less marked improvement in environmental burdens with carcinogens reducing by 5%, respiratory inorganics by 2.5%, climate change, ecotoxicity and fossil fuel burdens all reducing by 3%.

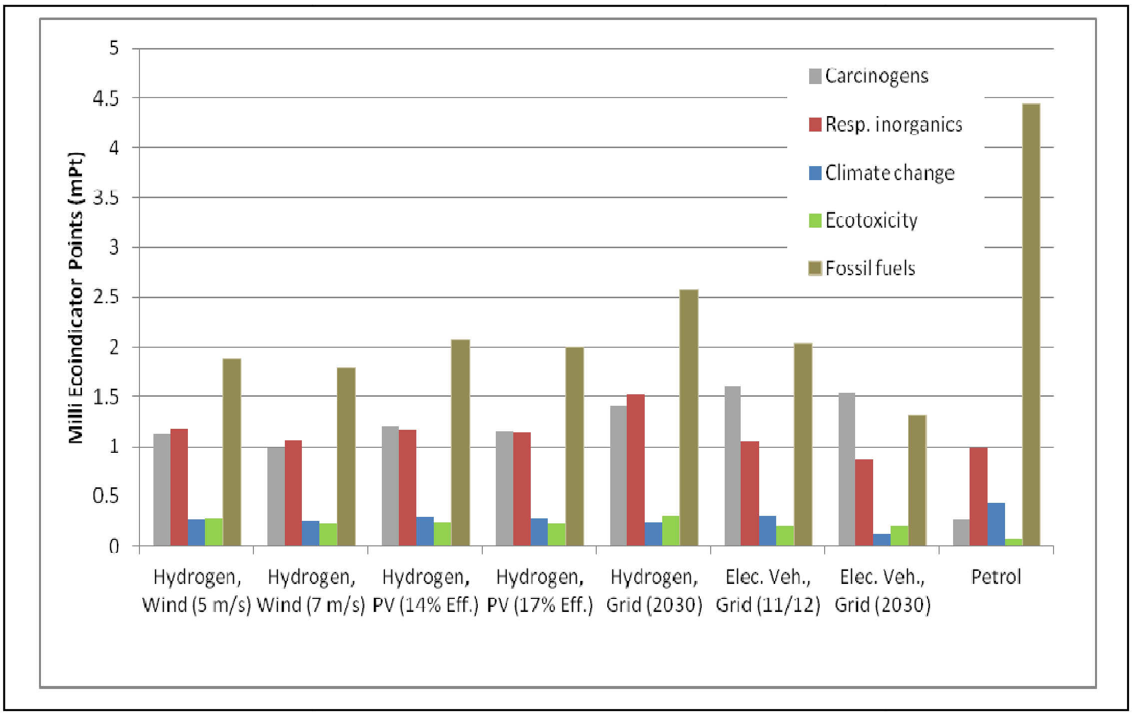


Figure 30 – Ecoindicator 99 (H/A) Normalised LCIA results including electrolytic hydrogen production using alternative primary energy efficiencies compared with 2030 electricity grid based options

7.6.2 Alternative LCIA Methodology

Alternative LCIA methodologies can affect the final results generated. Therefore, the damage orientated end point methodology of Eco-indicator 99 H/A was replaced with a problem orientated midpoint approach, in this case ReCiPe (Midpoint) H with European normalisation, to assess the effect of using an alternative impact assessment methodology. Normalised output for impact categories relevant to the previous assessment for the combined production and utilisation of fuels is shown in Figure 31 (a & b) with the increase in primary energy recovery also repeated using the alternative method in Figure 32 (a & b).

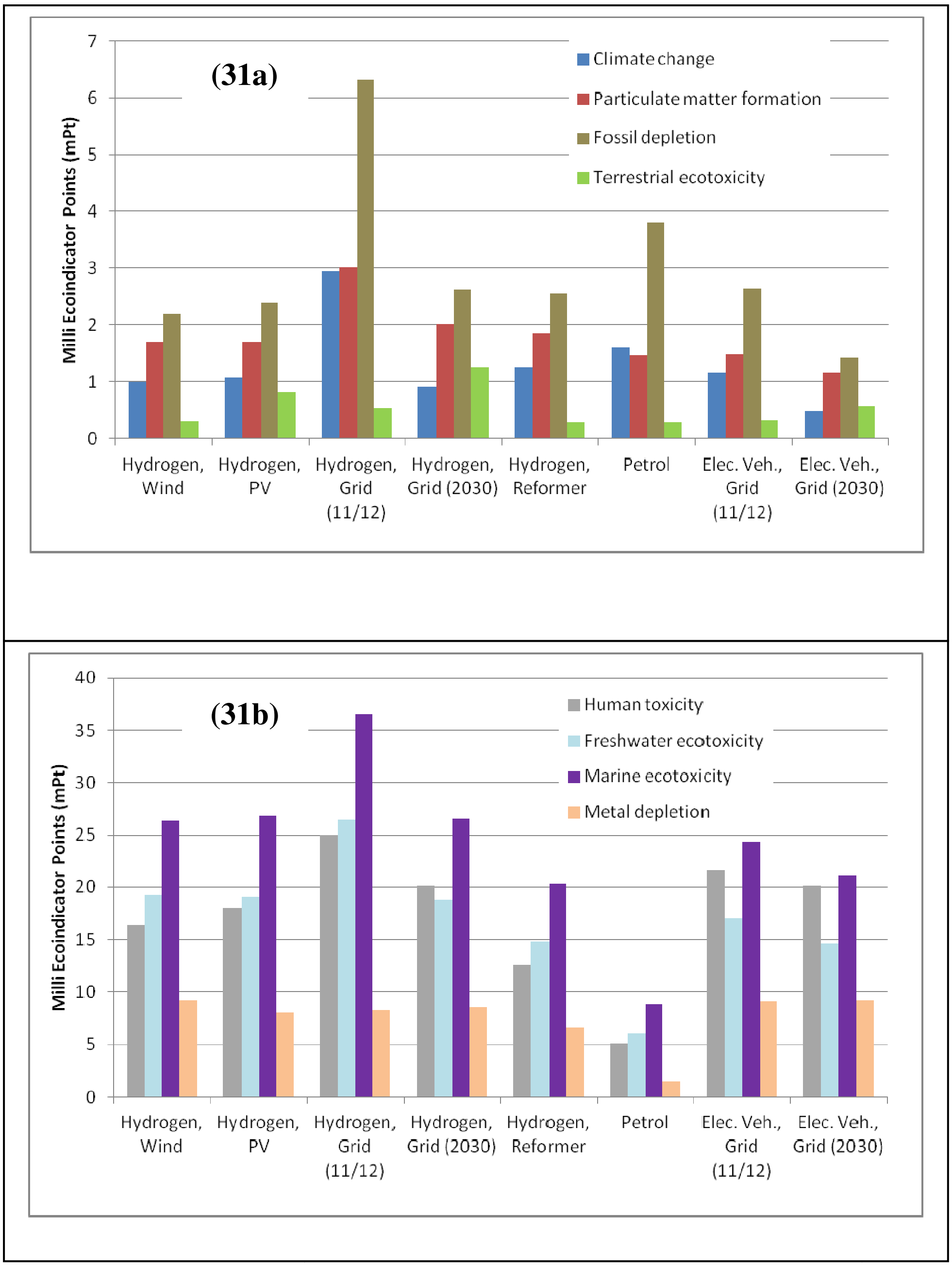


Figure 31 – ReCiPe (Midpoint) European (H) normalised results for production and utilisation of hydrogen fuel compared to fossil equivalent and electric vehicle

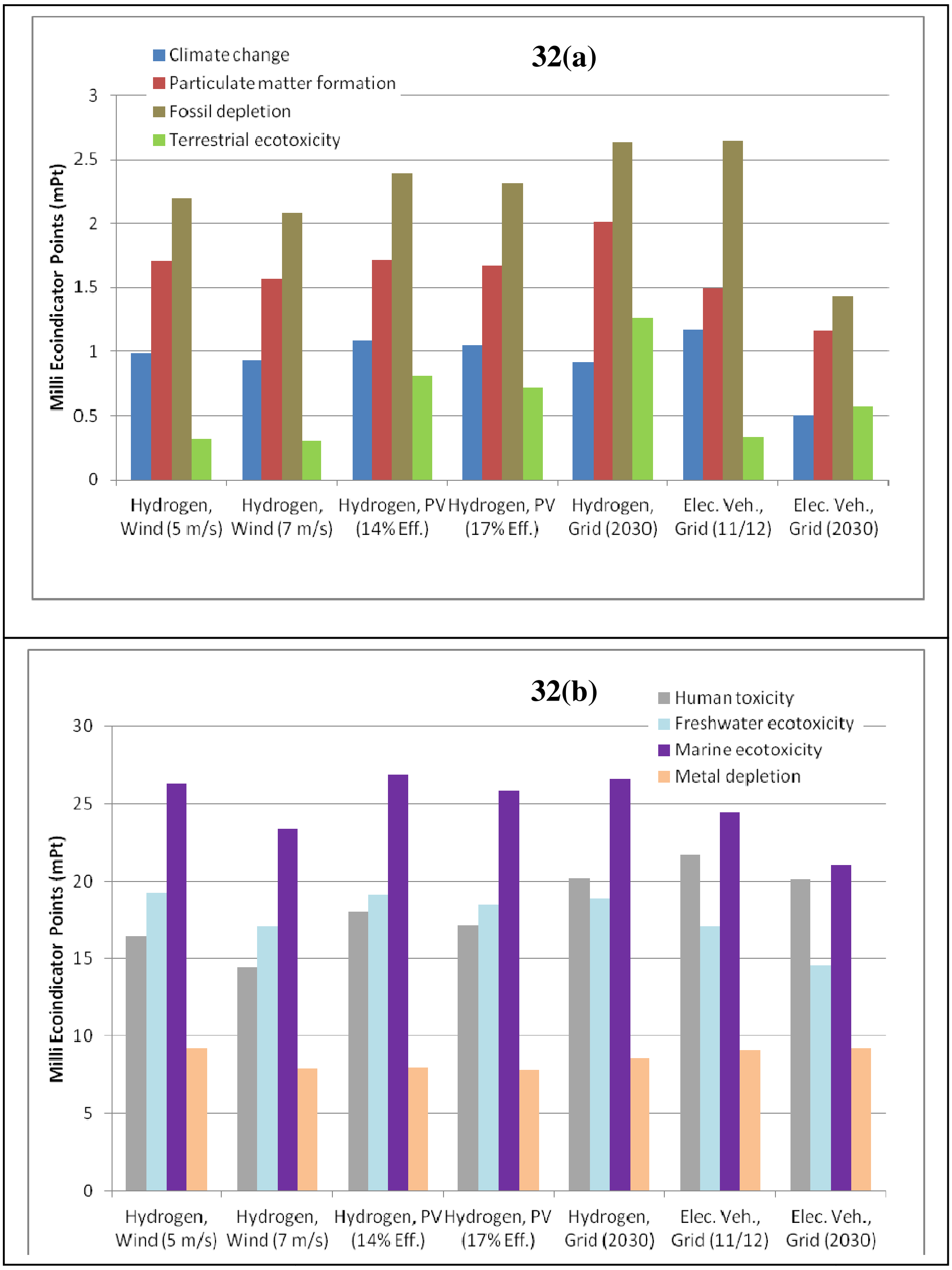


Figure 32 – ReCiPe (Midpoint) European (H) normalised results including electrolytic hydrogen production using alternative primary energy efficiencies compared with 2030 electricity grid based options

General conclusions remained consistent, with hydrogen from wind energy, hydrogen from PV and electric vehicles powered by the 2030 grid having amongst the lowest burdens for climate change and fossil fuel depletion (Figure 31a), which provided significant savings compared with petrol, although reductions compared with hydrogen generation using the

2030 grid and hydrogen from steam methane reforming were not as pronounced as the Eco-indicator results. Metal depletion was highest for the generation of hydrogen using wind energy due to the construction of wind turbines dedicated to transport fuel production. As a result of their large scale and centralised infrastructures the lowest human, freshwater and marine ecotoxicity burdens (Figure 31b) were associated with the fossil fuel options of utilising hydrogen from reforming and petrol. Including the increase in primary energy capture / utilisation for the two renewable hydrogen options (Figure 32 a & b) showed reductions in burdens of a similar magnitude to those realised in the initial assessment using Eco-indicator. As such the results and conclusions drawn from the study can be seen to be not significantly changed when different impact assessment methodologies were applied.

7.6.3 Uncertainty Analysis

Table 24 shows the results of a Monte Carlo analysis of the comparison between the production and utilisation of hydrogen fuel produced by electrolysis powered by wind energy, as the electrolysis option with the lowest climate change burden considered, with (i & ii) petrol, (iii & iv) hydrogen produced by electrolysis using the 2030 electricity grid, (v & vi) hydrogen from steam methane reforming, (vii & viii) an electric vehicle powered using the 2011/12 electricity grid and (ix & x) an electric vehicle powered using the 2030 electricity grid.

Results indicated that even accounting for inventory uncertainty the production of hydrogen using wind energy was likely to deliver fossil fuel burden reductions compared to all options, with the exception of notional 2030 electricity. Climate change burdens were likely to be lower compared to petrol, hydrogen from steam methane reforming and electric vehicles using current grid electricity, but were likely to be higher than for utilisation of hydrogen derived from, or electric vehicles using, 2030 grid electricity. Carcinogen burdens were likely to be lower than all options utilising current or future grid electricity, but higher than for fossil fuel options. Respiratory inorganic burdens associated with utilising hydrogen from wind energy were likely to be lower than hydrogen from the

2030 grid or from reformed natural gas, but higher than for all other options. The comparison of ecotoxicity burden were at approximately 50% for all options, suggesting that there was no clear advantage of any option in terms of ecotoxicity burden reduction.

Table 24 – Monte Carlo simulation results of characterised LCIA comparisons between production and utilisation of hydrogen produced by wind powered electrolysis with alternative fuels

	A	B	A>=B (%)				
			Carcinogens	Respiratory Inorganics	Climate Change	Ecotoxicity	Fossil Fuels
i	Wind, Hydrogen (5m/s)	Petrol	100.0	3.2	0	51.3	0
ii	Wind, Hydrogen (7m/s)	Petrol	100.0	0	0	49.8	0
iii	Wind, Hydrogen (5m/s)	Hydrogen, Grid (2030)	9.1	0	96.0	49.7	0
iv	Wind, Hydrogen (7m/s)	Hydrogen, Grid (2030)	1.4	0	64.2	52.6	0
v	Wind, Hydrogen (5m/s)	Hydrogen, Reformer	100.00	11.5	0	52.8	0
vi	Wind, Hydrogen (7m/s)	Hydrogen, Reformer	100.00	0.3	0	51.0	0
vii	Wind, Hydrogen (5m/s)	Elec. Veh., Grid (11/12)	67.6	37.7	0	47.7	0.1
viii	Wind, Hydrogen (7m/s)	Elec. Veh., Grid (2030)	31.7	1.8	0	51.9	0
ix	Wind, Hydrogen (5m/s)	Elec. Veh., Grid (2030)	93.0	92.2	100.0	52.1	40.6
x	Wind, Hydrogen (7m/s)	Elec. Veh., Grid (2030)	77.1	49.0	100.0	47.7	18.5

Note: Monte Carlo analysis comprised of 1000 iterations at the 95% confidence limit

7.7 Conclusions

Utilisation of hydrogen produced using renewable primary energy sources was shown to deliver reductions in fossil fuel and climate change burdens compared with petrol, even at the small scale considered here (640 kg H_2 / yr). Additional burdens associated with increased infrastructure development, such as carcinogens and respiratory inorganics, are incurred by small scale hydrogen production using renewable energy. The combination of renewably derived hydrogen fuelled vehicles with grid powered electric vehicles to contribute towards short and medium range transport requirements is a realistic means of achieving UK policy objectives in terms of energy security and climate change. A greatly increased renewable energy component within the UK electricity generation mix is required to maximise these benefits. Further research is required to determine the benefits of integrating hydrogen production with large scale renewable generation assets such as on and off shore wind farms.

Chapter 8: Critical Analysis of the Future Growth Of Renewable Hydrogen and Methane in the UK

8.1 Introduction

Research presented in Chapters 3 – 7 indicated a number of positive results for the potential future deployment of biologically derived renewable gaseous vehicle fuels such as hydrogen and methane, namely;

1. The production of methane using crops grown on agricultural land in a UK setting was more energy efficient than the production of biodiesel or bioethanol from first generation processes utilising arable crops.
2. Methane or a hydrogen / methane blend could potentially reduce key exhaust emissions when compared with biodiesel or bioethanol, although a solution for unburnt methane emissions would be required.
3. Several technology options for the production of renewable methane from biodegradable materials are widely available.
4. Biomethane production using waste materials that attract a gate fee for disposal was cost competitive with biodiesel, bioethanol and CNG providing that planned financial incentive mechanisms work effectively.
5. Utilisation of biomethane as a vehicle fuel delivered overall environmental benefits and compared favourably with other potential biogas end uses.
6. Hydrogen produced from electrolysis powered by renewable energy sources delivered savings in climate change burdens compared with fossil fuel alternatives.

Given these points, there appears to be little reason why the use of biomethane as a vehicle fuel should not be widespread in the UK, or at least expanding in line with the growth of the anaerobic digestion and biogas industries in the UK. However, of the 78 anaerobic digestion plants established in the UK at present (excluding water industry plants), none of these is focussed on the production of vehicle fuels. Clearly there is still a significant barrier to the deployment of biomethane as a vehicle fuel in the UK. In this Chapter, the previous conclusions have been challenged through a wider critical analysis of the industry and financial and legislative framework to identify these barriers and to suggest potential means of addressing them.

It is notable from the list above that biohydrogen does not feature strongly. It has been shown that the incorporation of a percentage (10-20%) of hydrogen in a predominantly methane (80-90%) fuel gas does provide emissions benefits and does not necessarily and detrimentally reduce engine power. A process to produce a fuel mixture at approximately the correct ratio via a single biological process is therefore a potentially attractive proposition. However, the research presented in Chapter 5 indicated potential problems with biohydrogen production which will be expanded upon in the coming analysis. This does not mean that the future portfolio of vehicle fuels will not include hydrogen – indeed it still remains as one of the most promising fuel vectors of the future, and the results of Chapter 7 demonstrated that the fuel is capable of delivering reductions in climate change burdens. The discussion regarding the future deployment of electrolytic hydrogen production was therefore also expanded upon within this chapter.

8.2 Critical Analysis of Raw Material Availability and Production Technology

8.2.1 Municipal Wastes

Using waste materials for fuel (or energy) production that would otherwise be disposed to landfill has been shown to be the best option in terms of both environmental and economic performance. Municipal wastes that are suitable for treatment by anaerobic digestion include source segregated food waste, non ligno-cellulosic green wastes, or residual organic materials recovered mechanically from the waste stream where source segregation does not occur. Accurately measuring the volumes of these wastes generated in the UK, or predicting how these volumes might change in the future is a difficult task. Many local or regional authorities have attempted to measure waste production rates and waste composition (e.g. Wasteworks Ltd & AEA, 2010 and Esteves and Devlin, 2010) however there are large temporal and geographic variations in both waste production rates, the characteristics of the waste produced and participation in recycling or waste segregation schemes. This, coupled with the impact of evolving waste management policy and practice, means that widespread surveys which take some time to complete will almost immediately be out of date.

Estimates from the Waste and Resources Action Programme (WRAP) indicated that in 2010/11 approximately 7.2 million tonnes of household food waste was disposed of, with around 4.6 million tonnes collected by local authorities, 1.9 million tonnes disposed to sewer and 0.7 million tonnes either home composted or fed to animals (WRAP, 2011).

Figures for 2009/10 stated that 3.71 million tonnes of ‘green waste’ was collected in England (DEFRA, 2011a). However, in the majority of cases, this ‘green waste’ would have comprised a combination of domestic garden waste and household food waste, and it was not possible to determine the proportion of each within the waste stream. Figures for the same year in Wales indicated that approximately 119,000 tonnes of green waste was recycled along with a further 63,000 tonnes of co-mingled green and food wastes (Welsh Government, 2011). Again, it was not possible to accurately determine the proportion of food waste present in the co-mingled ‘green waste’ stream.

A survey of municipal solid wastes generated in Wales in 2009 indicated that food waste and garden waste represented 16% and 14% of the total solid waste stream, respectively (Wasteworks Ltd. et al., 2010). For the year 2010/11 a total of 1,620,911 t of municipal waste was collected in Wales (Welsh Assembly Government, 2011), giving an approximate mass of food waste and garden waste production of 259,345 t and 226,927 t, respectively.

In 2010/11 in England approximately 26,200,000 t of municipal waste was collected by the local authorities (DEFRA, 2011b). Assuming that the proportion of food waste and garden waste was approximately similar to that in Wales, this would give approximate masses of 4.19 million tonnes of food waste and 3.66 million tonnes of garden waste.

In Scotland, approximately 3,141,202 t of municipal waste was collected by local authorities (SEPA, 2012). Applying the same proportion of food and garden waste as above suggested approximate masses of 502,592 t of food waste and 439,768 t of garden waste. Figures for Northern Ireland indicated a total municipal waste production in 2010/11 of 985,176 t (DEONI, 2011), suggesting in the order of 157,628 t food waste and 137,924 t garden waste.

This crude estimate suggested in the order of 5.1 million tonnes of food waste being collected in the UK, slightly higher than the WRAP estimate of 4.6 million tonnes. Not all of this waste will however be available for treatment via anaerobic digestion. A trial of kerbside food waste collections in 21 local authority areas in England and Scotland undertaken by WRAP between 2004 and 2009 achieved capture rates of 43 – 77% (WRAP, 2009b). Assuming that, over time, participation rates approach the upper level of this range, a capture rate of 70% would provide in the order of 3.29 – 3.57 million tonnes of source segregated food waste available for anaerobic processing (based on 2010 / 11 food waste production estimates). It is acknowledged that over time households will most likely produce less food waste, however, these figures are considered as indicative of the current situation.

Garden waste is not currently processed via anaerobic digestion. The potentially high ligno-cellulosic content of the feedstock coupled with low moisture content makes this waste stream more suitable for aerobic composting. However, certain elements of this waste stream are readily treatable by anaerobic digestion (e.g. grass clippings). It is not impossible that, given a scenario where biogas production was to be maximised, material suitable for anaerobic digestion could be segregated from the garden waste stream. As an estimate and a likely best case scenario, it is assumed that 25% of the garden waste stream (which totals approx. 4.46 million t) could be suitable and available for anaerobic digestion giving a further 1.11 million t of potential feedstock in the form of green garden waste.

Table 25 – Estimates of the theoretical availability of municipal solid waste stream components suitable for AD

Municipal Feedstock	Min. (t/yr)	Max. (t/yr)
Source Segregated Food Waste	3,290,000	3,570,000
Garden Waste suitable for AD	0	1,110,000

The fine fraction of municipal wastes is also likely to contain a high proportion of biodegradable material, however, for the purposes of this study the major municipal solid waste feedstocks that are potentially suitable for AD are summarised as Table 25.

8.2.2 Commercial and Industrial Wastes

Many commercial and industrial (C&I) wastes, in particular those generated through the manufacture or processing of food and drink, are suitable for treatment via anaerobic digestion for the production of biogas. C&I waste comprises of a huge variety of materials and is collected by both municipal authorities and private / commercial waste companies, and therefore gaining accurate data or statistics relating to the nature and masses of C&I waste produced in the UK is even more taxing than for the municipal waste stream.

The industry sector likely to contribute the largest mass of material suitable for anaerobic digestion is food and drink manufacturers. In addition, food waste from the catering, hotel and restaurant sector could also be significant. It is likely that other sectors will contribute smaller masses of waste (e.g. public services) however, for the purposes of this investigation these will not be considered.

The majority of data relating to C&I waste comes from periodic surveys undertaken by central or regional government in the UK. In 2007, the Welsh Assembly Government commissioned a study of commercial and industrial wastes which was managed by the Environment Agency Wales and completed by Urban Mines Ltd. The survey included collecting data from 1,500 businesses in Wales with results being published in 2009. Results indicated a total C&I waste production in Wales of 5,359,980 t (2007/08) of which 478,690 t was generated by the food and drink industry and a further 315,300 t was generated by the accommodation and food service industry (Environment Agency Wales, 2009).

A survey undertaken by DEFRA in 2002/3 indicated that a total of 67,907,000 tonnes of C&I waste was produced in England for that year of which 7,230,000 was generated by the food and drink industry (DEFRA, 2011c). A later 2009 survey indicated that the total had reduced to 47,928,000 tonnes with 4,667,000 t associated with the food, drink and tobacco industries (DEFRA, 2010). The 2009 survey indicated that the total waste production from the hotel and catering industry sector was 2,707,000 t, a decrease from 3,352,000 t indicated in the 2002/03 survey (DEFRA, 2010; DEFRA, 2011c).

Similarly in Scotland, SEPA undertook a C&I waste survey in 2002/03 with figures updated in 2010 to provide data with a 90% confidence limit. These figures indicated a total of 6,500,826 tonnes of C&I waste in total, with 567,377 t arising from the food and drink industries and 685,024 t from hotels and restaurants (SEPA, 2012). An estimate of commercial and industrial waste production in Northern Ireland in 2009 completed by WRAP indicated a total C&I waste stream of 1,288,996 t of which 243,856 was derived from the food, drink and tobacco sectors with 78,402 t derived from the hotel and catering industries (WRAP (Northern Ireland), 2011).

Whilst it was possible to gain a reasonable understanding of the total masses of C&I wastes being produced, it was not possible to quantify what proportion of these wastes would be suitable or available for treatment via anaerobic digestion. As an example Category 1 Animal By-Products cannot be treated by AD, and the majority of Category 2 material can only be treated by AD if subjected to an up-front rendering process. Some materials such as egg shells and bones are not degraded by the anaerobic digestion process and are therefore not suitable for treatment via this route.

As such, it is necessary to consider a range of proportions of these materials that could be processed by anaerobic digestion, as shown in Table 26. As a minimum, 50% the mass of material currently disposed via ‘land disposal’ (i.e. landfill), and 100% of the material currently disposed via ‘land recovery’ (i.e. disposed to agricultural land) and composting will be used. As a maximum threshold, it was assumed that 75% of waste masses could be treated via anaerobic digestion.

Table 26 – Potential availability of Commercial and Industrial Waste for biogas production in the UK

C&I Waste Feedstock	Total UK Production (t/yr)	Available for AD	
		Min. (t/yr)	Max. (t/yr)
Food, Drink & Tobacco Sector	5,956,923	1,999,180	4,467,792
Catering, Hotel & Restaurant Sector	3,785,726	685,930	2,839,294

8.2.3 Agricultural Wastes (Slurries)

Figures for the national or regional production of agricultural slurries are not currently collected in the UK. However, it is estimated that between 90 and 100 million tonnes of animal slurry is collected from housed animals and spread to land each year (Bywater, 2011). At present, the stabilisation of this material using anaerobic digestion prior to spreading to land is limited in the UK. However, there are many drivers working towards improved management of slurries and their nutrients. These factors include compliance with the EU Nitrogen Directive (1991), increasing costs of mineral fertilisers and the desire (which is not yet formalised into legislation) to reduce GHG emissions from agriculture.

There are, however, some significant barriers to the widespread adoption of on farm digesters to treat slurries including economies of scale for small digester plants, lack of feedstock where cattle are housed for only six months, and lack of capacity to produce or utilise imported feedstocks. Never the less, these barriers could all be addressed by the creation of farming co-operatives for the treatment of slurries, or the co-digestion of slurries with crops or other feedstocks. Such models have been shown to be effective in other countries including Germany and Austria.

Table 27 – Potential availability of agricultural slurries for biogas production in the UK

Agricultural Waste	Total UK Production (t/yr)	Available for AD	
		Min. (t/yr)	Max. (t/yr)
Slurries and manure	90,000,000	45,000,000	67,500,000

For the purposes of this study, it is assumed that a minimum of 50% and a maximum of 75% of collected agricultural slurries could be available for treatment via anaerobic digestion (Table 27).

8.2.4 Energy Crop Cultivation

The production of energy crops specifically for the production of biogas is not currently explicitly encouraged by UK policies and subsidies, and therefore the large scale cultivation of mono crops, predominantly maize, for biogas production as seen in countries such as Germany is unlikely to take place in the UK in the near future. This lack of government support is largely as a result of the debate over the utilisation of food producing agricultural land for the production of low carbon fuels, the so called food vs. fuel debate. However, limited growth of energy crops is occurring (e.g. Swancote Energy Ltd., UK) either for mono or co-digestion, and as government subsidies change, this is likely to continue to slowly increase on farms large enough to make an economically viable operation. It is therefore valid to consider the potential amount of energy crops that could be produced in the UK for biogas production. In reality crops for biogas production would have to compete with agricultural land being used for crops being grown for liquid biofuel production (e.g. rape seed), heat production (e.g. short rotation coppice) or, potentially, second generation biofuel production (e.g. straw, *miscanthus*). For the purposes of this study it is assumed that the findings of Chapter 3 remain relevant for the foreseeable future and that biogas production remains the most efficient means of producing energy using agricultural land.

The total cropable land (i.e. arable & ley grass) in England in 2011 was 4.7 million hectares (DEFRA, 2011d). In Wales the arable land use (including ley grass) in 2010 was recorded as 189,777 hectares (Welsh Assembly Government, 2010b), for Scotland 994,147 hectares (The Scottish Government, 2011) and in Northern Ireland 54,000 hectares (Department of Agriculture and Rural Development, 2012). This gives a total of 5.93 million hectares of arable and ley grass land in the UK.

Clearly the majority of this land is required for the continued production of food. However, a small percentage of land could be utilised for energy crop production without a detrimental impact on future food production, potentially up to approximately 350,000 ha (Rural Economy and Land Use, 2007). 5% of the total cropable area in the UK would equal 296,000 hectares, so as a minimum future scenario, if required this land could be made available for energy crop cultivation, specifically for biogas production. This is in line with the UK Governments Bioenergy Strategy which suggested the utilisation of 200,000 – 300,000 ha of land for cereals production specifically for bioenergy production (DECC, 2012). A study to assess the potential for utilising all arable land in Austria for both food and energy crop production using sustainable crop rotations was completed (Amon, 2007) which suggested that this would be a more productive route than dedicating a small percentage of land specifically for energy crop production in place of food. The study concluded that producing a smaller amount of energy crop over a larger area increased the yield on a national level. The practicalities and economics of delivering this option on a farm by farm basis in the UK would require significantly more investigation.

The UK Governments '2050 Pathways Analysis' (HM Government, 2010) investigated a range of potential pathways which might enable future renewable energy and CO_2 emissions targets to be met. This analysis included an assessment of future biomass production for energy production. With reference to the utilisation of agricultural land for biomass production, four potential pathways are described:

- Trajectory A – 550,000 ha arable land used for energy crop production
- Trajectory B – 350,000 ha arable land used for energy crop production
- Trajectory C – 1.5 million hectares of arable land used for energy crop production
- Trajectory D – 1.2 million hectares of arable land used for energy crop production

It should be noted that in the Pathways Analysis these areas are primarily considered for second generation biomass feedstocks, predominantly short rotation coppice and miscanthus grass, with a smaller reliance on cereal crops such as rape seed oil, wheat and maize. The figures do however give an indication of the order of magnitude of production conversion which might be acceptable on a political level, regardless of the end technology utilised to generate the low carbon energy or fuel. For the purposes of this investigation, and in order to gain an indication of the potential contribution that biogas / biomethane could make to future energy requirements if government policies favoured this technology, a maximum area of 550,000 hectares of land utilisation for biogas production was considered.

The largely maize based mono-crop approach taken in Germany may not necessarily be followed in the UK, although the economic requirement to maximise biogas output would most likely be the dominant factor determining which crops might be grown. Regional and local growing conditions along with potential ecological and soil benefits associated with crop rotation might result in less dominance of maize, despite its higher gas yields per hectare. To account for this, it was assumed that 60% of the land will be utilised for maize production, 20% for sugar beet production and 20% for grass production. A summary of the potential contribution of energy crops to biogas production on this basis is provided in Table 28.

Table 28 – Potential energy crop cultivation for biogas production in the UK

	Land Area (ha)	**Crop Yield (t FM / ha)**	**Total Feedstock Production (t / yr)**
Min. Maize	177,600	40	7,104,000
Min. Sugar Beet	59,200	57.32	3,393,344
Min. Grass	59,200	40.5	2,397,600
Min. Total	**296,000**		**12,894,944**
Max. Maize	330,000	40	13,200,000
Max. Sugar Beet	110,000	57.32	6,305,200
Max. Grass	110,000	40.5	4,445,000
Max. Total	**550,000**		**23,950,200**
Permanent Grassland (5%)	**324,500**	**20**	**6,490,000**

Permanent grassland (inc. rough grazing) covers a further 3.78 million ha in England, 1.0 million ha in Wales, 946,372 ha in Scotland and 770,000 ha in Northern Ireland. However, the environmental benefit, political will or economic advantages of managing part of this land to produce grass for energy production, potentially at lower yields due to sub optimal growing conditions, is not clear, although Corton et al. (2013) have investigated the utilisation of such land for transport fuel production. For illustrative purposes, a scenario where 5% of this area was used for grass production at half of the yield achieved in intensive agriculture is also shown in Table 28.

8.2.5 Potential Biomethane and Biohydrogen Yields

Given the above assumptions, it was possible to estimate the potential future levels of biogas and biomethane production in the UK, and gain an understanding of the potential scale of the contribution that biogas could make to future energy requirement (Table 29). This assumed that all biogas produced is utilised for a single end use (e.g. if all biogas produced were used for electricity production by CHP, etc). It can be seen that even if all biogas generated was pooled to one specific end use, the overall contribution to national demand was small - electricity (2.1 – 4.4% of total demand), heat (1.4 – 3.0%), natural gas (2.4 – 4.9%). The potential contribution to vehicle fuel use was of more significance (7.4 – 15.4% of total demand). This assumed current operational practices and yields. The potential for a feedstock specific two stage process to increase overall energetic output by up to 30% as suggested in Chapter 6 would be of strategic importance to UK energy requirements and should be investigated further.

Another important factor shown in Table 29 is the importance of agricultural feedstock production (i.e. energy crops). Even at the relatively conservative level of agricultural production included in this calculation, the contribution that agriculture makes to overall biogas production is almost twice that from municipal wastes, C&I wastes and agricultural wastes combined. Given that all waste management policies are directed at reducing waste production volumes (i.e. reducing feedstock availability for AD plants) it is clear that, given current available technologies, greater utilisation of agricultural land for energy crop cultivation will be required in the future.

Table 29 – Potential future level of biogas production and contribution to energy demand

Feedstock	Availability	Biogas Yield	Biogas Total	CH_4 Content	AD Parasitic Elec & Heat	CH_4 Available	Potential Elec. (CHP)	Potenial Heat (CHP)	Upgrading Parasitic	CH_4 Available	Potential NG Displaced	Potential Diesel Displaced
	t/yr	m^3/t	m^3/yr	m^3/yr	% of output	m^3	GWh	GWh	% of CH_4	m^3	m^3	m^3
Municipal Waste												
SSFW (Min.)	3,290,000	130	4.28E+08	2.57E+08	15	2.18E+08	778.98	1272.84	5	2.07E+08	2.07E+08	1.86E+05
SSFW (Max.)	3,570,000	130	4.64E+08	2.78E+08	15	2.37E+08	845.27	1381.16	5	2.25E+08	2.25E+08	2.02E+05
GW (Min.)	0	100	0.00E+00	0.00E+00	15	0.00E+00	0.00	0.00	5	0.00E+00	0.00E+00	0.00E+00
GW (Max.)	1,110,000	100	1.11E+08	6.66E+07	15	5.66E+07	202.17	330.34	5	5.38E+07	5.38E+07	4.84E+04
C&I Waste												
Food & Drink (Min.)	1,999,180	110	2.20E+08	1.32E+08	15	1.12E+08	400.52	654.45	5	1.07E+08	1.07E+08	9.59E+04
Food & Drink (Max.)	4,467,792	110	4.91E+08	2.95E+08	15	2.51E+08	895.10	1462.58	5	2.38E+08	2.38E+08	2.14E+05
Catering (Min.)	685,930	130	8.92E+07	5.35E+07	15	4.55E+07	162.41	265.37	5	4.32E+07	4.32E+07	3.89E+04
Catering (Max.)	2,839,294	130	3.69E+08	2.21E+08	15	1.88E+08	672.26	1098.47	5	1.79E+08	1.79E+08	1.61E+05
Agricultural Waste												
Slurry (Min.)	45,000,000	20	9.00E+08	5.40E+08	12	4.75E+08	1697.03	2678.40	5	4.51E+08	4.51E+08	4.06E+05
Slurry (Max.)	67,500,000	20	1.35E+09	8.10E+08	12	7.13E+08	2545.55	4017.60	5	6.77E+08	6.77E+08	6.09E+05
Energy Crops												
Maize (Min.)	6,624,000	300	1.99E+09	1.19E+09	12	1.05E+09	3747.05	5913.91	5	9.97E+08	9.97E+08	8.97E+05
Maize (Max.)	13,200,000	300	3.96E+09	2.38E+09	12	2.09E+09	7466.95	11784.96	5	1.99E+09	1.99E+09	1.79E+06
Sugar Beet (Min.)	3,164,064	113.3	3.58E+08	2.15E+08	12	1.89E+08	675.96	1066.86	5	1.80E+08	1.80E+08	1.62E+05
Sugar Beet (Max.)	6,305,200	113.3	7.14E+08	4.29E+08	12	3.77E+08	1347.03	2125.99	5	3.58E+08	3.58E+08	3.22E+05
Grass (Min.)	2,235,600	130	2.91E+08	1.74E+08	12	1.53E+08	548.01	864.91	5	1.46E+08	1.46E+08	1.31E+05
Grass (Max.)	4,445,000	130	5.78E+08	3.47E+08	12	3.05E+08	1089.59	1719.68	5	2.90E+08	2.90E+08	2.61E+05
Perm. Grass (Min.)	0		0.00E+00	0.00E+00	12	0.00E+00	0.00	0.00	5	0.00E+00	0.00E+00	0.00E+00
Perm. Grass (Max.)	6,490,000	130	8.44E+08	5.06E+08	12	4.45E+08	1590.88	2510.85	5	4.23E+08	4.23E+08	3.81E+05
TOTAL (MIN.)	62,998,774		4.27E+09	2.56E+09			8009.96	12,717			2.E+09	1.92E+06
TOTAL (MAX.)	103,437,286		8.88E+09	5.33E+09			16654.79	26,432			4.E+09	3.99E+06
2011 UK Generation / Consumption							**374,343**	**891874**			**8.99E+10**	**2.58E+07**
% Contribution By Biogas / Biomethane (MIN.)							**2.1**	**1.4**			**2.4**	**7.4**
% Contribution By Biogas / Biomethane (MAX.)							**4.4**	**3.0**			**4.9**	**15.4**

And what of biohydrogen? The research in Chapter 6 indicated that the combined dark fermentation > anaerobic digestion approach is unlikely to be viable as a hydrogen production route. Where hydrogen yields were maximised, this was at the expense of second stage methane yields leading to lower process wide energy yields than a single stage digestion process. Where the process was optimised for energy production (i.e. higher second stage methane outputs) first stage hydrogen production decreased to <2% of gas by volume – a level that is not likely to be viable as a hydrogen production technology. However, only two feedstocks were studied within a limited range of operational conditions with non-optimised performance, so further research would be required to reach a definite conclusion.

However, as mentioned in Chapter 1, one of the major advantages of hydrogen as a future energy vector is that it can be produced by a number of different processes. In addition to Dark Fermentation > Anaerobic Digestion as investigated in this study, Dark fermentation > photo fermentation (Huibo et al., 2009; Eroglu et al., 2011) and bio-photolysis (Huesemann et al., 2010) are both potential future pathways to produce hydrogen from organic matter. However, yields are still below those required for a practical or economic deployment of these technologies at a significant scale. Extensive further research is required in process design, improvement of metabolic performance of hydrogen production pathways and also in the pre-treatment of feedstocks to improve hydrogen yields, and this is a long term prospect (Hallenback et al., 2012). As with the production of biogas (methane), the availability of suitable feedstocks and competition with other biofuels for agricultural land is likely to be a limiting factor in the future.

The potential to produce a hydrogen rich 'syngas' through the gasification of high lignin (i.e. woody) feedstocks could contribute greatly to increasing the availability of hydrogen produced from sustainably sourced biomass. This approach would allow the utilisation of forestry wastes, waste wood and non fermentable municipal and commercial and industrial wastes, and grown crops such as short rotation coppice and *miscanthus* as feedstocks. Although the gasification process is relatively well understood, research is ongoing to enhance yields (Franco et al., 2003) address technical challenges associated with processing biomass feedstocks with varying characteristics, and upgrading outputs to produce fuels or chemicals with consistent quality (Kumar et al., 2009). Once again, however, competition for feedstocks from conventional thermal energy plants means that a market for the hydrogen produced will be needed before gasification plants become widespread.

The potential for producing hydrogen from the electrolysis of de-ionised water, with the electricity derived from renewable source such as wind turbines or photovoltaic cells is perhaps one of the most captivating possibilities for large scale hydrogen production. This approach offers the benefits of not being geographically limited, is scalable and ties in with the management of power grids with high input from intermittent renewable energy sources (Ursua et al., 2012). Provisional life cycle assessments undertaken on these technologies are also promising (Chapter 7). Whilst the production of the fuel and fuel cell vehicles generally incurs relatively high environmental impacts, these are more than off-set by emissions reduction through the vehicle's use phase resulting in overall environmental benefits compared to most technologies being developed (Granovskii et al., 2007(a); Cetinkaya et al., 2012; Dufour et al., 2012).

This is indeed a promising prospect, however, there is a potential bottleneck in the system, and, in a similar way to the production of vehicle fuel from biomass, the bottleneck is the availability of feedstock, in this case renewable electricity. Taking a closer look at some UK figures, the approximate number of passenger vehicle km travelled in 2010 is around 6.74×10^{11} km. Given current hydrogen consumption in fuel cell vehicles of in the order of 100 km kg^{-1} H_2, this gives a total hydrogen requirement of approximately 6.74×10^{09} kg in order to meet all passenger vehicle requirements (not necessarily a realistic requirement, but it serves these illustrative purposes). Electrical input to produce 1 kg of hydrogen via electrolysis is approximately 50 kWh, therefore an electrical demand of 3.37E11 kWh (337,000 GWh) would be required to produce sufficient hydrogen to fuel all UK passenger vehicles which is 90% of the UK's current total electricity generation capacity, and over 10 times the UKs total renewable electricity generation capacity. Given that only a proportion of renewable energy will be available for hydrogen generation, the majority being directly used as electricity, there clearly needs to be a step change in renewable energy generation and drive train efficiency before renewable electrolytic hydrogen will make large contributions to national mobility.

Another potentially significant source of hydrogen is in the form of industrial off gases, which, on the whole, are currently combusted as waste products in processes such as chlorine manufacture (Pastowski et al., 2010) and steel making (Chen et al., 2011). The UK has a number of major steel manufacturing plants (e.g. Tata Steel, Port Talbot, South Wales) and chlorine manufacturing plants (e.g. Ineos ChlorVinyls, Runcorn, Cheshire) which could produce regionally significant volumes of usable hydrogen. A study has shown that chlorine manufacturing sites in one region of Germany could satisfy a significant proportion of local demand for fuel cell vehicles with the advantage that the hydrogen produced is of reasonable purity with relatively little clean up required (Pastowski et al., 2010). Further research and development of the processing of such industrial off gases is required, and economic benefits to the gas producers must also be clear before deployment of this approach is possible.

Finally, the use of non renewable sources of hydrogen such as reforming of natural gas, and the production of syngas from coal are likely to be significant sources in the short to medium term (Cormos et al., 2008). In the long term, the electrolytic production of hydrogen utilising nuclear power is a potential means of producing large volumes of hydrogen, although the long term benefits of current nuclear power technology are less than clear (Dincer et al., 2012).

This sounds like a bleak forecast for both methane and hydrogen adoption in the future, and indeed there are very significant problems with supply of the raw materials (biomass or renewable electricity) needed to produce sufficient masses of gas to make significant in-roads to national demand. However, methane and hydrogen are still candidates for future energy carriers. At this stage it is worth noting that all options for future vehicle fuels suffer from exactly the same supply issues. Whilst utilising indigenous waste materials to produce fuels makes absolute environmental sense, there is insufficient material available to make significant contributions to fuel demand using this material alone, whether the fuel is methane, hydrogen, bioethanol or biodiesel. Even with large scale agricultural production, our ability to meet indigenous demand is limited. Given this, a future where a range of transportation drive trains and fuels are utilised depending on the application, and some re-evaluation of society's expectation of low cost mobility, is likely.

8.3 Critical Review of Infrastructure Requirements

In conjunction with any future growth in supply and demand of gaseous vehicle fuels, there will have to be an equivalent roll out of infrastructure to be able to meet this demand. This infrastructure can broadly be divided into:

1. Fuel Production Infrastructure
2. Fuel Supply / Refuelling Infrastructure
3. End Use / Vehicle Infrastructure

Fuel production requirements and its limitation due to feedstock or primary energy availability have been discussed in the previous section. End use technology in the form of vehicles will be briefly discussed in the next section. This section considered some of the key issues associated with the development of a suitable refuelling infrastructure for gas fuelled vehicles in the UK, some perceived barriers to this deployment and how they might be addressed.

It is of some value to note that there is a global and national track record in developing, deploying and drastically altering fuel and energy infrastructure. The rise of the motor car in the early 20th Century in the USA was facilitated by the widespread, but small scale, availability of gasoline which was built around the pre-existing kerosene refining industry and was distributed in a wide variety of methods before the service station became the dominant mode during the 1930s (Melaina, 2007). This wholesale change in both the refining of crude oil to provide cheap gasoline, the widespread availability of affordable motor cars, and the development of an infrastructure to fuel them developed over a period of 30 years.

Since the onset of the industrial revolution, UK industry was largely driven by the combustion of coal. Up until the late 1960s coal was still the dominant means of heating domestic homes and generating electricity. The utilisation of coal to produce coal (town) gas (a mixture predominantly of hydrogen, methane and carbon monoxide) in urban areas facilitated the development of local and regional gas networks that formed the basis of today's gas grid. The discovery of high quality North Sea oil and gas in the 1960's and the onset of its exploitation during the 1970s led to a rapid decline in coal gas production, and the 'dash for gas' was completed in the 1980s and 1990s as gas fired power stations were developed. Electricity generation using natural gas increased from 0% in 1980 to its peak level of 47% of total electricity generation in 2010 (DECC, 2011a) largely at the expense of coal fired generation.

These historical examples serve to show that large scale transitions in fuel and energy infrastructures are indeed possible. It is often argued that introducing a gaseous vehicle fuel infrastructure will be too expensive. However, there is again some precedent in making huge investments in infrastructure developments, this time in the communications industry. The upgrading of the UK mobile phone network to 3G standard in the late 1990s and early 2000s is estimated to have cost between £20 billion - £40 billion, which illustrates the point that when the economic and political conditions demand such large investments, the money can be found.

Indeed the engineering required to physically build gaseous refuelling stations, whether methane or hydrogen based, is not a barrier to deployment. Natural gas filling stations are prolific across the world, particularly in areas where surplus gas from oil production makes it the most cost effective vehicle fuel (e.g. Iran, Russia, Ukraine). The substitution of natural gas with the methane component of biogas would make no material difference to the nature of the refuelling infrastructure. There are approximately 56 No. hydrogen filling stations of varying scale in Europe (10 of which are in the UK) with approximately another

40 No. stations in the development state (TÜV SÜD Industrie Service GmbH, 2012), so clearly for hydrogen also, the technical ability to build and operate appropriate refuelling stations is not a significant barrier to wider deployment.

The nature of future refuelling and distribution infrastructures has been the subject of a reasonable amount of research, predominantly in the mathematical modelling of the distribution and management of primary energy sources, hydrogen production and delivery nodes on a regional basis. Studies indicated that matching infrastructure requirements with demand and production is feasible and can be optimised if required (Pastowski et al., 2010; Dagdougui et al., 2012).

A number of studies have also looked at the economic environment associated with the development of a hydrogen infrastructure. The ability of hydrogen to be produced by a wide range of technologies, whilst potentially beneficial in terms of manufacture, has been identified as a factor that significantly complicates investment decisions – in effect, all possible infrastructure configurations must be understood and evaluated before accurate pricing systems can be arrived at (Michalski et al., 2011). Whilst market forces will undoubtedly determine the eventual location of infrastructure such as refuelling stations, the optimal location of filling stations early in the transition process could result in significantly faster uptake of the technology (Stephens-Romero et al., 2010), and therefore the modelling approaches to determining such optimal locations should be seriously considered by politicians, planners and industry stakeholders. Building on the practical demonstration of hydrogen fuel cell buses in London, a study investigating the changes in the economics of hydrogen production and distribution throughout the transition between diesel and hydrogen reached a number of interesting conclusions including; 1) that widely differing production costs between technologies converged as transition took place, 2) that demand for hydrogen was the dominant factor in determining the supply price for hydrogen, and 3) that at a fleet size of 100-200 vehicles, the cost of hydrogen became comparable with current diesel prices (Shayegan et al., 2009).

The strategic storage of large volumes of hydrogen does not seem to present an insurmountable barrier. The natural gas grid in the UK aims to operate by matching supply with demand with a latent amount of storage within the network in the form of high pressure mains. The amount of storage external to the gas mains is generally limited to a number of strategic sites and a dwindling number of low pressure gasometers to meet local demand during peak periods. For hydrogen, at least in the initial phases of deployment, the availability of compression and liquefaction technologies will meet local demands before localised and eventually regional and national grids could manage the balancing of supply with demand. In the future, as with all energy distribution grids, this balancing of supply and demand will be of greater importance and will require more intelligent and flexible ways of moving energy in multiple directions through our grids.

Given the above, it is clear that the development of a gaseous refuelling infrastructure, whilst not without its challenges, is possible from a technical and even an economic basis given the appropriate market and political conditions. This, perhaps, is a potential stumbling block at present. Market forces are not yet strong enough to warrant a significant shift away from fossil fuels, and as such businesses are unlikely to invest in infrastructure and there is little political will to push for major changes in transport infrastructure.

8.4 Critical Analysis of End Use Technologies

As described in Chapter 1, the utilisation of methane as a vehicle fuel in an internal combustion engine is widespread throughout the world, particularly in areas where affordable gas is available largely as a by product of other industrial activities. In the UK, many manufacturers now offer methane fuelled passenger vehicles (e.g. VW Passat, Volvo Bi-Fuel) and light duty vehicles (e.g. Fiat, Ford, Mercedes) although production numbers are still low. Companies specialising in the conversion of heavy goods vehicle diesel engines to gas fuel or dual diesel / gas systems (e.g. Hardstaff Group) are expanding rapidly as the reduced diesel consumption begins to bring economic benefits to transport fleet operators. It is therefore clear that development of methane fuelled vehicles is not a barrier to their wider adoption. Whilst adoption of methane as a vehicle fuel can deliver environmental benefits, it is of critical importance that methane emissions including unburned methane from vehicle exhausts are minimised. Whilst these emissions are subject to regulation in European countries, this is not necessarily the case in all countries where methane utilisation is currently, or may in the future be, widespread. Failure to minimise methane emissions throughout the fuel production, transport, delivery and end use pathway could result in a net increase in global warming impact (See Chapter 5). Further research should be conducted to support the reduction of methane emissions from the fuel supply and utilisation chain.

The research and development of hydrogen fuelled power trains for road vehicles has been an active and productive field over the past 20 years within both academia and industrial organisations. There are a broad range of potential drive train configurations subject to research and development, some of which will play a larger part in a possible future transition to gaseous transport fuels than others. Broadly speaking, these options can be classed as:

- Internal Combustion (Spark Ignition)
  Based on standard gasoline internal combustion engines where the fuel or fuel mixture is injected into the cylinder and ignited with a spark e.g. (Karim, 2003; Mohammadi et al., 2007). This configuration can operate using a range of fuels including methane only, hydrogen only, methane / hydrogen blend, or a blend of liquid and gaseous fuels (bi-fuels). Manufacturers that have developed the technology to demonstration level include BMW, Mazda and Ford.

- Internal Combustion (Compression)
  Based on diesel engine technology where the fuel mixture is compressed to initiate ignition. Hydrogen compression engines often use small volumes of diesel as a pilot (dual fuel) before compressed hydrogen is added to the combustion chamber (Boretti, 2011). This configuration is generally more fuel efficient than spark ignition and can deliver similar engine performance to standard diesel engines. The

technology is appropriate for heavy goods vehicle type applications. Manufacturers that have developed the technology to demonstration level include MAN, Iveco (Fiat) and Ford.

- Fuel Cell – The electro-chemical conversion of hydrogen and oxygen (air) within a fuel cell, which generally includes a cathode, an anode and a conducting electrolyte, results in a difference in electrical potential between the anode and cathode resulting in the generation of a direct current through an external circuit. This external current can be utilised to drive electric motors as part of a vehicle drive train (or perform other electrical work, i.e. stationary power applications). There are several types of fuel cells, however, the Proton Exchange Membrane (PEM) design is currently seen as the most viable for transport applications. Fuel is generally directly delivered as compressed hydrogen gas, although this can be sourced from other hydrocarbons such as methane or methanol using on board reformers. Fuel cell vehicles are considered to be most suitable to mid range passenger vehicle duty cycles. Fuel cell and vehicle configurations are still the subject of considerable research and development (Veziroglu et al., 2011). The majority of major vehicle manufacturers have developed fuel cell demonstration vehicles including Honda, Toyota, Daimler-Mercedes, GM, Nissan, and Ford.

- Fuel Cell Hybrid – Incorporates fuel cell technology as described above with secondary power storage in an on board battery. The fuel cell then provides power for the vehicle drive train (motors) as well as charging the battery. Features such as regenerative braking also provide additional charge to the battery. In this way, the range of the vehicle can be increased (or the hydrogen storage requirement decreased) and issues associated with fuel cell only vehicles (e.g. slow cold weather start up) are reduced, and this configuration is likely to represent the most viable in terms of early market deployment (Offer et al., 2010). Many major manufacturers have developed fuel cell hybrid demonstration vehicles including Toyota, GM, Kia, Mercedes and Honda.

Research and development of hydrogen vehicles is ongoing, and some significant challenges remain including on board hydrogen storage (Ahluwalia et al., 2012), long term fuel cell stack durability (Burlatsky et al., 2012; Hidai et al., 2012) and optimising fuel cell efficiency (Ratlamwala et al., 2012). However, progress in all fields associated with development of hydrogen vehicles has been rapid in the last decade and the majority of major vehicle manufacturers have and are continuing to invest heavily in development of the technology. Whilst commercially available hydrogen vehicles have so far failed to become a reality, and, despite many predictions to the contrary are unlikely to be widely deployed in the immediate future, the prospect of the deployment of a commercially viable hydrogen vehicle in the medium term future remains the goal of the majority of major car manufacturers (Frenette et al., 2009).

8.5 Critical Analysis of Environmental Performance

As demonstrated in the research presented in Chapters 3 - 7, determining the environmental impacts or benefits of a future fuel infrastructure is not a simple process. The multiplicity of primary energy sources, conversion technologies and end use technologies that are relevant for both methane and hydrogen infrastructures means that there is a huge variety of configurations that could be considered. In addition, society rightly wants to know how possible future technologies compare with each other, therefore increasing the complexity of the task by an order of magnitude.

Life cycle assessment is, for now, the best tool that we have for undertaking these types of assessments. However, the approach is limited by the availability of data (especially when modelling processes and products that are still subject to research and development), data accuracy, the calculation methods of individual impact assessment methodologies, and, perhaps most significantly of all, by the assumptions made during the modelling as well as choice of system boundary.

Research completed as part of this thesis indicated that:

1. The production of methane as a vehicle fuel was an energetically more efficient use of arable land than currently available crop based biodiesel or bioethanol routes.
2. Once produced, the utilisation of biogas for electricity production combined with high heat utilisation was the most environmentally beneficial means of utilisation. However, where heat utilisation was not possible or sub-optimal, conversion to vehicle fuel was a preferable option.
3. The utilisation of methane derived from waste products provided significant environmental benefits compared with fossil diesel.
4. The utilisation of non waste feedstocks for biomethane production may not necessarily result in environmental benefits compared with fossil fuel use where the production process was not optimised according to feedstock characteristics.
5. Methane emissions at all stages in the fuel chain including feedstock production, biogas production and upgrading, digestate storage and use, and combustion and tailpipe emissions at end use are critical in determining whether the use of methane as a vehicle fuel was beneficial from a GHG perspective.
6. Given current energetic yields, biohydrogen production based on dark fermentation was unlikely to contribute significantly to future hydrogen requirements, although the two stage dark fermentation > anaerobic digestion process could be valuable in terms of optimising methane production.

For those with an interest in the development and deployment of biomethane technologies, there are perhaps some worrying aspects to the above conclusions. The issue of methane emissions throughout the supply chain has for some time been something of an ‘elephant in the room’ amongst those involved in the industry and it is an issue that is seeing renewed research effort in order to quantify (Hrad et al., 2011; Liebetrau et al., 2011; Menardo et al., 2011; Daelman et al., 2012). The adoption of improved engineering and management practices for processes operated under Environmental Permits or their European equivalent will result in a reduction in methane emissions over time, and it is important to allow the industry this time in order to make appropriate changes as required. It is also important to view the UK and European biogas industry in a global context. Perhaps of a far greater concern should be the growth of methane production and utilisation in populous but less well regulated countries where there will be little incentive to quantify or reduce emissions. Similarly, the emission of unburned methane from exhaust emissions in Europe is highly regulated, and a significant part of the engineering expertise and expense of a biomethane fuelled vehicle is the treatment of exhaust gases. Again, it is unlikely that this level of environmental diligence would be followed in all countries either with substantial existing methane vehicle fleets or with plans for future fleet expansion. As such, there is no guarantee that, on a global basis, the adoption of methane as a vehicle fuel in place of fossil diesel or gasoline will result in significant GHG savings.

Reductions in exhaust emissions are a clear benefit of utilising hydrogen as a vehicle fuel, particularly where fuel cells are utilised and exhaust emissions are limited to water. However, even exhaust emissions from hydrogen fuel utilised in internal combustion engines are lower than current fossil and bio fuels with potential increases in NOx due to high temperature combustion with air (mainly nitrogen) managed through a combination of lean burn and engine gas recirculation (EGR) to boost engine power (Dimopoulos et al., 2008). The vast majority of the environmental impact of hydrogen as a vehicle fuel therefore is associated with the production and distribution (well to tank) and it is not clear which production pathway will represent the best environmental option. This situation is perhaps exacerbated by the plethora of production options available, and, of course, any future deployment will be based primarily around financial viability and availability of the primary energy resource (e.g. wind energy for electrolysis, biomass for biological or thermal production) rather than a clear focus on environmental benefit. For example, many studies suggest that hydrogen production using electrolysis powered by renewable energy such as wind turbines is an environmentally favourable option, however, this assumes that sufficient wind generation capacity is available on a national or regional basis to provide sufficient electricity peaks (i.e. high generation + low demand) to generate useful quantities of hydrogen. Given the plans for deployment of wind energy in the UK over the coming decade, it seems unlikely that this excess of renewable energy will be available, and, a point rather forcibly made by Kreith et al., (2004), it is unlikely to make environmental sense to divert primary renewable electricity to hydrogen production by electrolysis if there is a ready grid demand for the primary energy. Despite this, it seems that in Scotland wind energy is routinely providing energy in excess of grid capacity to the extent that the National Grid is forced to pay wind farm operators a fee to shut off renewable energy production rather than take the more technically and economically challenging route of limiting production from fossil or nuclear power plants (The Telegraph Online, 2012). As deployment of wind energy generation infrastructure increases, these issues will become more widespread across the National Grid and the integration of hydrogen generation technology could provide a technically feasible, environmentally sound and economically viable means of managing these issues.

8.6 Critical Analysis of Financial Viability

Research in Chapter 4 indicated that biomethane production for vehicle fuel utilisation was cost competitive with other, more widely used, biofuels. However, this was based on two key assumptions:

1. That the biomethane was produced from waste, residues, non-food cellulosic materials or ligno-cellulosic materials, and therefore attracted double Renewable Transport Fuel Certificates (RTFC's) i.e. double the financial incentive compared to non-waste derived fuels.
2. That the value of the RTFC's approached the maximum allowable level of £0.35 / kg of biomethane.

As discussed above, whilst there are clear environmental and economic advantages of utilising waste materials for vehicle fuel production, in reality this end use would be competing with a range of other processes such as biogas combustion (CHP), thermal processes (CHP and syngas production), feed production (e.g. wheat feed to animal feed) as well as a range of new and novel processes currently under development (e.g. bioplastic production), all competing for waste products and each bringing their own environmental advantages and adding to the complexity of determining which pathway is the most beneficial. It seems unlikely, therefore, that sufficient volumes of waste materials will be available specifically for biomethane production to make a strategic contribution to the vehicle fuel portfolio in the UK.

Even if sufficient volumes of wastes / residues were available, the assumption that the RTFC value will approach £0.35 / kg (or the future equivalent) has so far proved flawed. As discussed in Chapter 4, the RTFC, like the ROC, is a market driven scheme based on the volume of renewable fuels produced or imported to the UK, and the targets set for the volume of renewable fuel supplied to the market. As the targets increase, the likelihood of the RTFC achieving a higher value also increases as fuel suppliers begin to struggle to produce or source sufficient volumes of compliant fuels. To date, however, the targets in the UK in terms of volumes of fuel brought to the market are modest at best, and the majority of pressure on fuel suppliers has come from the requirement to source fuels with a demonstrable carbon saving over their fossil equivalents. One of the UKs major trading platforms achieved peak RTFC values of 23.5 p/l (or per kg) in August 2011 although more recently in March and April 2012 values were just 8.0 p/l and 7.5 p/l respectively, and 20 million RTFCs offered for sale in June 2012 at a price of 12.5 p/l received no bidders (Non Fossil Purchasing Agency Ltd., 2012).

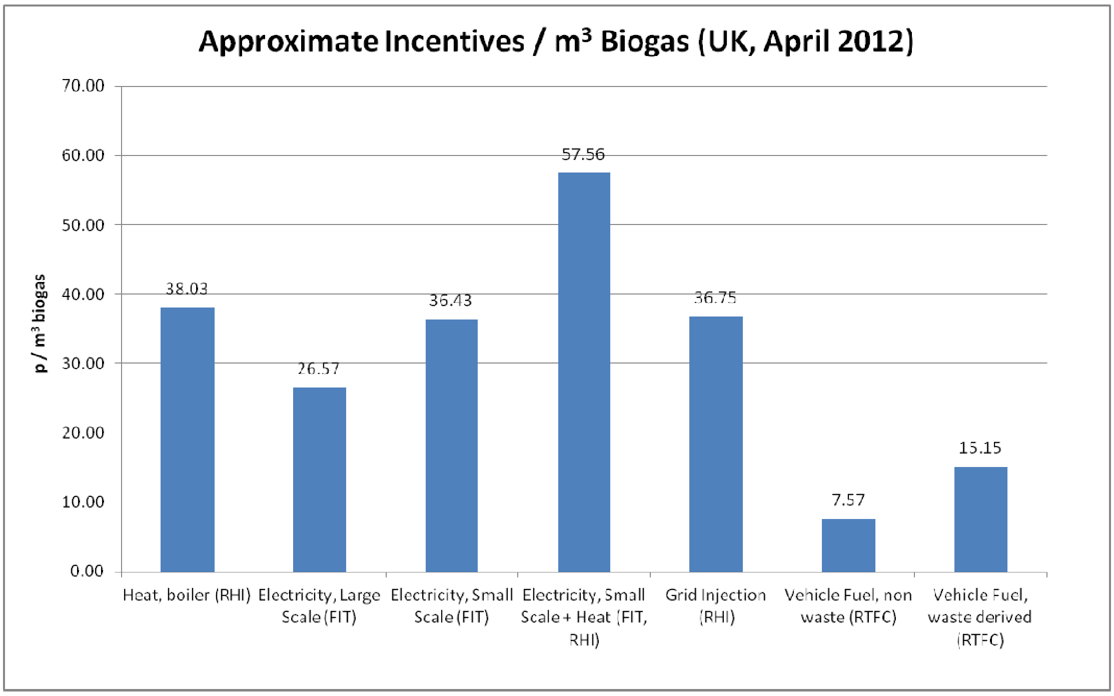


Figure 33 – Subsidies (as of April 2012) that could be achieved for biogas production for each end use.

Clearly this poor financial incentive has an impact on the viability of the production of biomethane as a vehicle fuel. In particular, the utilisation of biogas to produce vehicle fuel would no longer compare favourably to the alternative end uses of biogas, such as renewable heat and electricity production. This can be seen in Figure 33, which indicates just the level of subsidies that could be attracted through the production and utilisation of biogas.

Figure 33 has assigned the average 2011 RTFC price of 20.11 p/kg to biomethane vehicle fuel use and, given the above, it is far from clear that this will be achieved in 2012/13. This raises another limitation of the funding mechanism which is its unpredictability. Variations in value of RTFCs on a year in year out basis make vehicle fuel utilisation a risky prospect for potential investors, where as the guaranteed income from the Feed In Tariff and Renewable Heat Incentive make equivalent schemes using biogas to produce renewable heat and electricity far more bankable. One option that may well be considered by the industry in the UK is to inject to the gas grid and accrue the RHI and then use the biomethane within the gas grid (together with natural gas) as vehicle fuel.

The comparatively low subsidies available for production of biomethane for transport fuel use are an issue which the UK and European authorities are aware of. In September 2012 it was announced that the European Union was considering allocating x4 credits for biomethane production from waste, residues, non-food cellulosic materials or ligno-cellulosic materials under the Renewable Energy Directive, doubling from the current position of x2 credits. This would clear the way for the UK to award x4 RTFCs per kg of biomethane vehicle fuel produced, effectively doubling the current subsidy and bringing the subsidy value approximately in line with Feed In Tariffs. This move would go some way to providing a more level playing field on which biomethane vehicle fuel utilisation can compete with other end uses. However, the limitation to waste and residue type feedstocks brings us back to the apparent shortage of these materials in terms of making significant contribution to future energy needs, and the strong competition for these materials from other processes. At some point, the issue of utilising land for the mass production of energy crops will have to be assessed and debated in detail.

The economics associated with the development of a future hydrogen refuelling infrastructure including fuel production and distribution is currently based more on theory than practice. Unsurprisingly in this situation there is great uncertainty as to the economic feasibility of specific hydrogen production pathways. The production of hydrogen using fossil fuels is still generally accepted to be cheaper than utilising renewable energy alternatives. A review by Bartels et al. (2010) showed that the renewable alternatives increased costs by between 1.5 - 7 times compared to fossil fuel alternatives (e.g. SMR) but did conclude that increasing fossil fuel costs and decreasing costs of alternative production pathways could rapidly change this situation. Financial modelling of the interaction between hydrogen production (initially including by-product hydrogen liquefaction, followed by SMR) and refuelling station operators showed that hydrogen production could be financially viable in the short term, where as the retail of hydrogen fuel requires a strong demand before viability can be achieved (Michalski et al., 2011). An analysis of hydrogen infrastructures in London echoed the conclusion that demand has the strongest overall influence on final hydrogen cost and that SMR remains as the most economic production route, however, the study also found that given sufficient demand, the (renewable) electrolysis pathway could compete with SMR (Shayegan et al., 2009).

Clearly, the economics of hydrogen utilisation is a dynamic system dominated by fossil fuel prices, production technology costs, and demand for the end product. What is also clear is that fossil fuel costs have increased rapidly in recent years, and are highly unlikely to decrease by any significant amount, and that hydrogen production and utilisation technologies are increasing in efficiency and decreasing in costs as research and development continues (Schoots et al., 2010). However, the tipping point at which non fossil hydrogen production and utilisation becomes cost competitive with fossil alternatives is far from clear.

Chapter 9: Conclusions and Recommendations

9.1 Conclusions

The aim of the study was to evaluate whether gases produced either by biological or electrolytic processes (i.e. methane, hydrogen / methane blend or hydrogen) would deliver environmental benefits of they had a larger role as a vehicle fuel in the UK than at present, and to determine the factors that might limit this deployment in the future.

From the studies performed, it can be concluded that:

- Production of methane fuel utilising waste as a feedstock, which has already been shown to deliver environmental benefits compared to current fossil fuels, can also be cost competitive with alternative fuels currently being deployed in the UK.

- Utilising methane from AD plants in CHP plants where a high degree of excess heat is utilised represents the most effective option in terms of limiting environmental burdens. However, where this high heat utilisation cannot be achieved, as is the case in many existing and planned AD facilities in the UK, alternative uses for the methane, including transport fuel, will deliver greater reductions in environmental burdens.

- However, the mass of organic wastes available in the UK are such that biofuels produced from these materials are only likely to make a small contribution to the overall future vehicle fuel mix. Significant deployment of biomass based biofuels beyond this level will require the re-assessment of the utilisation of agricultural land for biomass / fuel production, or the import of feedstocks / fuels.

- In the event that raw materials for biofuel production are to be produced on agricultural land in the UK, the processing of this organic material by anaerobic digestion represents the existing and economically viable option which delivers the most energy per hectare of land utilised, and should therefore be encouraged in favour of current options such as bioethanol or biodiesel production from crops. Environmental impacts associated with land use change and the import of biomass from tropical and sub tropical areas are yet to be resolved and require further research.

- The treatment of substrates high in ligno-cellulose which may otherwise be difficult to digest is enhanced when a two stage acidogenic and methanogenic digestion process was used. However, these benefits may not be realised for substrates that are relatively easy to digest (e.g. food wastes), but further research is required to fully determine this fact. This highlights the need for biological processes to be tailored to suite the feedstocks being treated. The two stage process was not considered as viable for hydrogen production under the conditions studied in this thesis.

- Uncontrolled methane emissions should be limited at all points of the process including at the digestion stage, storage and utilisation of digestates, and at the point of end use of the biogas / biomethane. Failure to manage methane emissions could negate any potential climate change benefits that the process could deliver.

- The utilisation of electrolytic hydrogen fuel produced using renewable energy sources (e.g. wind turbine generated electricity) is a good option in terms of minimising burdens to climate change, fossil fuel utilisation and respiratory inorganics. However, additional burdens to impact categories such as ecotoxicity and carcinogens are likely, at least given the conditions (i.e. small scale, dedicated hydrogen production facilities) included in this thesis. Even so, the combination of electrolytic hydrogen with grid powered electric vehicle represents a favourable option for mass deployment in the future, providing that sufficient renewable generation technology can be deployed across the UK.

9.2 Recommendations

The research work described in Chapters 3 – 7 indicated that there are many benefits associated with the production and utilisation of renewable gaseous vehicle fuels. However, Chapter 8 outlines some of the key limitations and barriers that go some way towards explaining why their deployment in the UK is so limited. A number of recommendations can be made aimed at improving the viability and availability of these fuels.

- Research undertaken as part of this thesis suggests that whilst utilisation of waste materials for energy generation undoubtedly makes environmental sense, the masses available are only likely to contribute a small percentage of total UK energy demand, no matter what conversion technology is deployed. Further research needs to be carried out in order to accurately quantify the masses of municipal, commercial and industrial wastes. This work needs to be far more detailed than exercises undertaken in the past. In particular, future work needs to quantify how these masses may change over the coming 10 – 20 years, and determine which wastes are suitable for specific reprocessing technologies. This work would allow a more strategic approach to be taken as to how these waste materials are best managed in order to maximise the environmental and economic benefits associated with their utilisation.

- It is clear that if any degree of UK self sufficiency in terms of indigenous biofuel production is to be achieved, whether in gaseous or liquid form, a greatly increased use of agricultural land for feedstock production will be required. To date, the UK Governments position has been not to overtly encourage the large scale production of energy crops in the UK, and, perhaps, given the important debate surrounding the food vs fuel issue this standpoint has so far been justified. However, ongoing research seems to indicate that importation of some feedstocks and fuels produced in sub-tropical regions will not meet future sustainability criteria. Further research is required into establishing the feasibility of utilising UK agricultural land for energy crop production, and the associated environmental and economic impacts of doing so. Of particular interest is the approach of utilising crop rotations so that the same land can be used to produce both food and fuel over 1 – 2 years. This approach may require a larger area of land to be utilised for energy crop production, however, with lower intensity and interspersed with food production, the environmental and social costs could be minimised.

- The subsidy system for renewable transport fuels is severely flawed and is not currently providing a reliable income for fuel producers and does not encourage investment into the sector. These problems with a market value based reward system have been shown to exist with the Renewable Obligation Certificates for electricity generation, and the introduction of Feed In Tariffs which provide a guaranteed income has seen a rapid increase in investment and deployment of renewable energy technologies. A similar system is required for sustainable transport fuel production. The production of biomethane and utilisation as a vehicle fuel is particularly hampered by the RTFO system. Issuing certificates on a mass rather than energetic basis is shown to disadvantage gaseous fuels significantly. An open, industry wide discussion or consultation regarding the legality, practicality, economic costs and likely outcomes of changing the subsidy system should be undertaken as a matter of urgency.

- Anaerobic digestion plants are being deployed now. However, all too often the utilisation of excess heat is not considered, or is considered only as an afterthought. The Renewable Heat Incentive (RHI) currently provides a reliable income to small plants for using this heat, and planned changes to the RHI suggest that subsidies will be extended to larger plants which meet CHP Quality Assurance standards. Therefore, continuation of the practice of under utilising excess heat should no longer be acceptable. All plants should be required to assess of their environmental benefits, indicate whether and how these could be improved, and either implement these improvements or provide justification as to why these improvements cannot be made. In many cases excess heat could be utilised locally with relatively little additional investment.

- Further research work into the whole field of biological hydrogen production is required, whether utilising dark or photo fermentative processes. Research in this thesis has highlighted the requirement to reduce water utilisation in the dark fermentation > anaerobic digestion process and to increase hydrogen yields before the process can be either economically or environmentally viable. This crosses over into the issue of the utilisation and pre-treatment of high ligno-cellulosic feedtocks, the assessment of the environmental and economic costs of these pre-treatment methods and comparison with process that may require less pre-processing of feedstocks. Chapter 6 highlighted the need to fully understand the origin of feedstocks and the necessity to tailor processes to treat specific materials in order to maximise environmental (and economic) performance.

- A strategic approach is also required when considering the demand placed on our future renewable energy and transmission networks (both electricity and gas). To date there seems to be an acceptance that the production of renewable electricity for direct industrial and domestic consumption will require infrastructure modification over the coming decades. However, there seems to have been relatively little assessment relating to the potential demands that a more sustainable transportation sector could also place on this infrastructure. A combination of centralised and distributed renewable energy production, electric vehicle use, centralised and distributed hydrogen production (both fossil, electrolytic, and potentially nuclear) and hydrogen transmission would have major impacts on infrastructure capacity and performance requirements. Research and planning as to what these requirements might be, and how they can be implemented should be ongoing. A large increase in the deployment of renewable energy generation is required in order to maximise the environmental benefits of hydrogen or electric vehicles.

- Demand for the end product has been shown to be the key driving force in lowering production and distribution costs. So far, the majority of policies and financial incentives have focussed on the upstream side of the fuel chain, however, in the very near future, the UK should be seriously considering how to best stimulate downstream demand. For biomethane, vehicle technology at affordable prices is available now. The UK should be seeking to encourage the development of small networks of gaseous refuelling facilities, potentially through the encouragement of biomethane captive vehicle fleets including local authority vehicles, utility companies or other regionally significant businesses, particularly those that operate significant HGV fleets. A similar approach will be needed for hydrogen fuel, although this may not necessarily be limited to niche captive fleets but will be equally applicable to any inter-city travel mode including passenger cars. Hydrogen vehicle technology is perhaps in the late development / early commercialisation phase, and it seems that stimulation of demand is perhaps one of the biggest barriers to rapid deployment.

9.3 Future Development

Further detailed research is required on the economics of gaseous fuel production and utilisation, particularly for hydrogen fuels. The economic viability of the utilisation of wind farms for combined electrical and hydrogen production, specifically in areas where grid capacity will be reached the soonest is potentially of strategic importance. If possible, this research needs to be integrated with developing renewable energy policy and subsidy mechanisms.

This thesis included detailed models of biogas production and end use systems. However, some of the longer term costs or benefits of the system have been neglected through excluding effects such as land use change and the fate of carbon and fertilisers within agricultural soils. Further LCA research, backed up with monitoring data from field application of digestates, is required to fully quantify these impacts.

Discharge of digestate liquors to sewer as opposed to transporting digestates by road to agricultural land appears to be a preferred option to some within the AD industry. Further research to compare the LCA costs and benefits of these options is required, however, this will be highly dependent on the treatment processes undertaken at the destination waste water treatment plant.

On completion of the research the background numerical models and associated LCA models for biogas are in place which could be readily utilised to assess the effect of changes within the system in the form of an expanded sensitivity analysis. Changes in construction materials, operating temperatures, pre-treatments, upgrading technologies and biogas end use efficiencies can all be accommodated to determine optimal plant configurations. With additional effort, models could also be adapted to investigate the burdens associated with novel fermentation processes such as biopolymer production.

Life cycle assessment is currently the best tool available for assessing the potential environmental burdens of products, processes or services. As a modelling process, it can only ever represent an approximation of the product system under investigation, and quantification of the complex environmental interactions within and between impact categories can at best only be considered as an estimate. LCA will always involve uncertainty, whether in datasets, interpretation of system boundaries, requirements for allocation, it seems an impossible task to standardise an approach to the extent that uncertainty can be eliminated. An understanding of the model and modelling processes will always be required to interpret results correctly. This need not be considered as a negative outlook towards LCA – that is not intended to be the case. It is a tool, and it will be improved over time. The way in which LCAs are undertaken and applied can, and should change.

When modelling existing or well established and well understood processes, uncertainty can be minimised and even quantified in a meaningful way. System boundaries should be clear, process data will be available, inputs and outputs will be readily quantifiable. However, modelling of bioenergy systems is especially challenging; they involve interactions between waste generation and processing, land use, multi output industrial processes, material and nutrient recycling, various end use technologies and are heavily influenced by social, political and economic conditions. The system being modelled is inherently complex. Many processes are at the research and development stage and may not be optimised, and yet this is the stage at which life cycle assessment can be critical –

directing research to improve processes, identifying potential future problems or conflicts, assessing burdens across a range of impact categories to better understand the potential future implications of large scale process implementation. Life cycle assessment should not be undertaken in isolation, or remotely from the researchers developing bioenergy processes. LCA needs to be integrated into all aspects of bioenergy research and process development. By participating, or even driving the LCA of a process, bioenergy researchers who are often focussed on a specific technical aspect can gain an overview of the whole bioenergy chain, and this should be encouraged. Whilst the development of the LCA tools and methodologies could be viewed as an independent science, the use and application of those tools should not, but should be integrated into other research disciplines as a matter of routine.

LCA clearly has a role to play in better informing policy developers and this is an area that has improved significantly in recent years as a greater understanding of the global implications of biofuels is sought. However, in the UK policy makers appear to be satisfied with meeting minimum targets set by Europe with the consequence that growth of the bioenergy sector itself is stifled. By contrast countries such as Denmark take a far more ambitious approach of maximising indigenous renewable energy production by promoting a diverse range of technologies, integrating these technologies into the national energy infrastructure, as well as making improvements in efficiency at the point of end use. Whether Denmark manages to meet its ambitious energy targets (100% renewable energy by 2050), is arguably irrelevant as they will not only exceed any European targets but will, most likely, develop a vibrant indigenous renewable energy industrial sector. For this to be replicated even in part in the UK, clearer, more ambitious, and more long term renewable energy policy is required.

REFERENCES

Ackley, M., W., S. Rege, U. and H. Saxena (2003). "Application of natural zeolites in the purification and separation of gases." Microporous and Mesoporous Materials **61**: 25-42.

Ahluwalia, R. K., T. Q. Hua and J. K. Peng (2012). "On-board and Off-board performance of hydrogen storage options for light-duty vehicles." International Journal of Hydrogen Energy **37**(3): 2891-2910.

Ally, J. and T. Pryor (2007). "Life-cycle assessment of diesel, natural gas and hydrogen fuel cell bus transportation systems." Journal of Power Sciences **170**: 401-411.

Alpay, E., N. Haq, L. S. Kershenbaum and N. F. Kirkby (1996). "Adsoprtion parameters for strongly adsorbed hydrocarbon vapours on some commercial adsorbents." Gas Separation and Purification **10**(1): 23-33.

Amon, T., B. Amon, V. Kryvoruchko, A. Machmüller, K. Hopfner-Sixt, V. Bodiroza, R. Hrbek, J. Friedel, E. Pötsch, H. Wagentristl, M. Schreiner and W. Zollitsch (2007). "Methane production through anaerobic digestion of various energy crops grown in sustainable crop rotations." Bioresource Technology **98**: 3204-3212.

Angelidaki, I., A. Heinfelt and L. Ellegaard (2006). "Enhanced biogas recovery by applying post-digestion in large-scale centralized biogas plants." Water Science Technology **54**(2): 237 - 244.

Antonopoulou, G., H. N. Gavala, I. V. Skiadas, K. Angelopoulos and G. Lyberatos (2007). "Biofuels generation from sweet sorghum: fermentative hydrogen production and anaerobic digestion of the remaining biomass." Bioresource Technology **99**(1): 110-119.

Armstrong, A. P., J. Baro, J. Dartoy, A. P. Groves, J. Nikkonen and D. Rickeard (2002). Energy and greenhouse gas balance of biofuels for Europe - An update, CONCAWE Report No. 2/02.

Auer, S., M. Haulio, L. Lekawska and M. Sonnleitner (2006). Life Cycle Assessment Study, Ethanol vs. Biogas used as car fuels, Institute of Environmental Strategies Research, Sweden. Available at http://www.infra.kth.se/fms/utbildning/lca/projects%202006/Group%2010%20(Biofuels%20in%20cars).pdf. Accessed on 07th October 2009.

Barbir, F. (2005). "PEM electrolysis for production of hydrogen from renewable energy resources." Solar Energy **78**: 661 - 669.

Bare, J. C., P. Hofsetter, D. W. Pennington and H. A. Udo de Hacs (2000). "Midpoints versus endpoints: The scarifices and benefits." International Journal of Life Cycle Assessment **5**(6): 319 - 326.

Barreto, L., A. Makihira and K. Riahi (2003). "The hydrogen economy in the 21st century: a sustainable development scenario." International Journal of Hydrogen Energy **28**: 267 - 284.

Bartels, J. R., M. B. Pate and N. K. Olson (2010). "An economic survey of hydrogen production from conventional and alternative energy sources." International Journal of Hydrogen Energy **35**(16): 8371-8384.

Bauen, A., J. Woods and R. Hailes (2004). Biopowerswitch - A biomass blueprint to meet 15% of OECD electricity demand by 2020, Prepared for WWF International and Aebiom.

Baxter, E. D. and S. Salmon (2006). Monitoring the quality and safety of grain and grain derived co-products destined for animal feed, Home Grown Cereals Authority. Project Report No. 387. Available at http://www.hgca.com/cms_publications.output/2/2/Publications. Accessed on 02/05/12.

Beer, T., T. Grant, R. Brown, J. Edwards, P. Nelson, H. Watson and D. Williams (2000). Life-cycle emissions analysis of alternative fuels for heavy vehicles, CSIRO Atmospheric Research, Australia.

Beil, M. (2009). Overview on Biogas Upgrading Technologies. European Biomethane Fuel Conference, Goteborg, Sweden, Biogasmax. Available at http://www.biogasmax.eu/media/3t3_overview_on_upgrading_iset__062510600_0654_30092009.pdf. Accessed on 06th October 2009.

Benjaminsson, J. (2006). NYA Renings Och Uppgraderingstekniker for Biogas, Swedish Gas Centre, Report 163. Available from http://www.sgc.se/rapporter/resources/SGC163.pdf. Accessed on 24 July 2008.

Bentsen, N. S., C. Felby and K. H. Ipsen (2006). Energy balance of 2nd generation bioethanol production in Denmark, The Royal Vetinary and Agricultural University. Available from www.bioethanol.info.

Berglund, M. and P. Börjesson (2006). "Assessment of energy performance in the life-cycle of biogas production." Biomass and Bioenergy **30**: 254-266.

Berndes, G., M. Hoogwijk and v. d. R. Broek (2003). "The contribution of biomass in the future global energy supply: a review of 17 studies." Biomass and Bioenergy **25**: 1-28.

Berndt, A. (2006). CarboTech Engineering GmbH, Intelligent Utilisation of Biogas - Upgrading and Adding to the Grid. Jonkoping, May 2006. Available from www.german-renewable-energy.com/Renewables/Redaktion/PDF/es/es-World-Bioenergy-2006-Carbotech-schulte-biogas,property=pdf,bereich=renewables,sprache=es,rwb=true.pdf. Accessed on 18 July 2008.

Bernesson, S. (2004). Life cycle assessment of rape seed oil, rape methyl ester and ethanol as fuels – a comparison between large- and smallscale production. Statens energimyndighet, Sverige, 2004.

BERR (2007). UK Energy in Brief. Available from http://www.berr.gov.uk/files/file39881.pdf Accessed on 30/08/07.

Bohn, I., L. Björnsson and B. Mattiasson (2007). "The energy balance in farm scale anaerobic digestion of crop residues at 11-37 °C." Process Biochemistry **42**: 57-64.

Boretti, A. (2011). "Advances in hydrogen compression ignition internal combustion engines." International Journal of Hydrogen Energy **36**: 12601 - 12606.

Börjesson, P. (2004). Energianalys av drivmedel från spannmål og vall. Rapport nr. 54, Lunds Tekniska Högskola.

Börjesson, P. and M. Berglund (2006). "Environmental systems analysis of biogas systems - Part I: Fuel-cycle emissions." Biomass and Bioenergy **30**: 469-485.

Börjesson, P. and M. Berglund (2007). "Environmental systems analysis of biogas systems - Part II: The environmental impact of replacing various reference systems." Biomass and Bioenergy **31**: 326-344.

Bossel, U. and B. Eliasson (2003). Energy and the hydrogen economy. Report available on line at www.eere.energy.gov/afdc/pdfs/hyd_economy_bossel_eliasson.pdf.

BP (2009). BP statistical review of world energy., BP. Available at http://www.bp.com/liveassets/bp_internet/globalbp/globalbp_uk_english/reports_and_publications/statistical_energy_review_2008/STAGING/local_assets/2009_downloads/statistical_review_of_world_energy_full_report_2009.pdf. Accessed on 20th October 2009.

BP (2011). BP Statistical Review of World Energy, BP. Available at www.bp.com/.../bp.../statistical...review_2011/.../pdf/statistical_review. Accessed on 13th July 2012.

Brown, L. (2006). Ethanol could leave the world hungry. Fortune.

BSi (2006a). BS EN ISO 14040:2006 Environmental management - Life cycle assessment - Principles and framework. ISO 14040:2006(E).

BSi (2006b). BS EN ISO 14044:2006 Environmental management - Life cycle assessment - Requirements and guidelines. ISO 14044:2006(E).

BSi (2010). PAS110:2010 Specification for whole digestate, separated liquor and separates fibre derived from the anaerobic digestion of source segregated biodegradable materials. ISBN 978 0 580 61730 0. ICS 65.080. Issued by WRAP. Available at http://www.wrap.org.uk/farming_growing_and_landscaping/landscape_and_design/how_to_use_and_specify_compost_and_digestate/bsi_pas_110.html. Downloaded on 19th May 2010.

Burlatsky, S. F., M. Gummalla, J. O'Neill, V. V. Atrazhev, A. N. Varyukhin, D. V. Dmitriev and N. S. Erikhman (2012). "A mathematical model for predicting the life of polymer electrolyte fuel cell membranes subjected to hydration cycling." Journal of Power Sources **215**(0): 135-144.

Bywater, A. (2011). A review of anaerobic digestion plants on UK farms, Royal Agricultural Society of England. Available at http://www.rase.org.uk/what-we-do/core-purpose-agricultural-work/index.asp#publications. Accessed on 13th September 2012.

Calais, P. and R. Simms (2000). A comparison of life-cycle emissions of liquid biofuels and liquid and gaseous fossil fuels in the transport sector Solar 2000. Brisbane, Australia.

Campanari, S., G. Manzolini and F. Garcia de la Iglesia (2009). "Energy analysis of electric vehicles using batteries or fuel cells through well-to-wheel driving cycle simulations." Journal of Power Sources **186**: 464 - 477.

Cavenati, S., C. Grande, A. and A. Rodrigues, E. (2005). "Upgrade of methane from landfill gas by pressure swing adsorption." Energy and Fuels **19**: 2545-2555.

Cavinato, C., A. Giuliano, D. Bolzonella, P. Pavan and F. Cecchi (2012). "Bio-hythane production from food waste by dark fermentation coupled with anaerobic digestion process: A long-term pilot scale experience." International Journal of Hydrogen Energy **37**(15): 11549-11555.

Centre for Ecology and Hydrology (2000). Land Cover Map 2000, Natural Environment Research Centre.

Cetinkaya, E., I. Dincer and G. F. Naterer (2012). "Life cycle assessment of various hydrogen production methhods." International Journal of Hydrogen Energy **37**: 2071 - 2080.

Chen, W. H., M. R. Lin, T. S. Leu and S. W. Du (2011). "An evaluation of hydrogen production from the perspective of using blast furnace gas and coke oven gas as feedstocks." International Journal of Hydrogen Energy **36**(18): 11727-11737.

Cherubini, F., S. Bargigli and S. Ulgiati (2009). "Life cycle assessment (LCA) of waste management strategies: landfilling, sorting plant and incineration." Energy **34**: 2116-2123.

CONCAWE (2007). Well to wheels analysis of future automotive fuels and powertrains in the european context - Well to wheels report, Version 2c. CONCAWE/EUCAR/JRC. Report available to download from http://ies.jrc.ec.europa.eu/WTW.

Conibeer, G. J. and B. S. Richards (2007). "A comparison of PV / electrolyser and photoelectric technologies for use in solar energy storage systems." International Journal of Hydrogen Energy **32**: 2703 - 2711.

Cormos, C.-C., F. Starr, E. Tzimas and S. Peteves (2008). "Innovative concepts for hydrogen production processes based on coal gasification with capture." International Journal of Hydrogen Energy **33**(4): 1286-1294.

Corton, J., L. Bühle, M. Wachendorf, I. S. Donnison and M. D. Fraser (2013). "Bioenergy as a biodiversity management tool and the potential of a mixed species feedstock for bioenergy production in Wales." Bioresource Technology **129**(0): 142-149.

Cruz, P., F. D. Magalhães and A. Mendes (2005). "On the optimization of cyclic adsorption separation processes." American Institute of Chemical Engineers Journal **51**(5): 1377-1395.

Daelman, M. R. J., E. M. van Voorthuizen, U. G. J. M. van Dongen, E. I. P. Volcke and M. C. M. van Loosdrecht (2012). "Methane emission during municipal wastewater treatment." Water Research **46**(11): 3657-3670.

Dagdougui, H., A. Ouammi and R. Sacile (2012). "Modelling and control of hydrogen and energy flows in a network of green hydrogen refuelling stations powered by mixed renewable energy systems." International Journal of Hydrogen Energy **37**: 5360 - 5371.

Dahlbo, H., S. Koskela, H. Pihkola, M. Nors, M. Federley and J. Seppälä (2012). "Comparison of different normalised LCIA results and their feasibility in communication." International Journal of Life Cycle Assessment **Published on line on 18th September 2012. DOI 10.1007/s11367-012-0498-4**.

Danielsen, F., H. Beukema, N. D. Burgess, F. Parish, C. A. Bruhl, P. F. Donald, D. Murdiyarso, B. Phalan, L. Reijnders, M. Struebig and E. B. Fitzherbert (2009). "Biofuel Plantations on Forested Lands: Double Jeopardy for Biodiversity and Climate." Conservation Biology **23**(2): 348-358.

de Hullu, J., J. Maassen, P. van Meel, S. Shazad, J. Vaessen, L. Bini and J. Reijenga (2008). Biogas Upgrading: Comparing Different Techniques, Eindhoven University of Technology. Available at students.chem.tue.nl/ifp24/Final%20presenation.pps. Accessed on 19th September 2009.

De Vegt, O. M. and W. G. Haije (1997). Comparative environmental life cycle assessment of composite materials, ECN (Netherlands). Available at http://www.ecn.nl/publications/. Accessed on 19th October 2012.

DECC (2009b). Consultation on Renewable Electricity Financial Incentives 2009, Department of Energy and Climate Change (DECC). Available at http://www.decc.gov.uk/en/content/cms/consultations/elec_financial/elec_financial.aspx. Accessed on 16th December 2009.

DECC (2010a). Feed-in Tariffs: Government's response to the summer 2009 consultation. D. o. E. a. C. Change, Available to download at http://www.decc.gov.uk/assets/decc/Consultations/Renewable%20Electricity%20Financial%20Incentives/1_20100204120204_e_@@_FITsconsultationresponseandGovdecisions.pdf. Accessed on 11th May 2010.

DECC (2010b). Renewable Heat Incentive: Consultation on the proposed RHI financial support scheme. D. o. E. a. C. Change, Available to download at http://decc.gov.uk/en/content/cms/consultations/rhi/rhi.aspx. Accessed on 04th October 2010.

DECC (2010c). 2050 Pathways Analysis, Department of Energy and Climate Change (DECC). Available at http://www.decc.gov.uk/en/content/cms/tackling/2050/2050.aspx. Accessed 25th September 2012.

DECC (2011a). Digest of United Kingdom Energy Statistics, Department of Energy and Climate Change. Available at http://www.decc.gov.uk/en/content/cms/statistics/publications/dukes/dukes.aspx. Accessed on 10th July 2012.

DECC (2011b). UK Renewable Energy Road Map, Department of Energy and Climate Change (DECC). Available at http://www.decc.gov.uk/en/content/cms/meeting_energy/renewable_ener/re_roadmap/re_roadmap.aspx. Accessed n 25th September 2012.

DECC (2012a). 2011 Provisional UK Figures (of CO_2 Emissions). Department of Energy and Climate Change, Available at http://www.decc.gov.uk/en/content/cms/statistics/climate_stats/data/data.aspx. Accessed on 12th July 2012.

DECC (2012b). UK Bioenergy Strategy, Department of Energy and Climate Change (DECC), Department for Transport (DoT), Department for Food and Rural Affairs (DEFRA). Available at http://www.decc.gov.uk/en/content/cms/meeting_energy/bioenergy/strategy/strategy.aspx. Accessed on 01st August 2012.

DECC (2012c). Digest of United Kingdom Energy Statistics 2012, Department of Energy and Climate Change (DECC). Available at http://www.decc.gov.uk/en/content/cms/statistics/publications/dukes/dukes.aspx. Accessed on 09th September 2012.

DECC. (2012d). "National Renewable Statistics, Summary of 2011 survey." Available at https://restats.decc.gov.uk/cms/national-renewables-statistics/. March 2013.

DEFRA (2005). Key Facts About: Land Use and Land Cover, Data Table 01, Land by agricultural and other uses, 2005 Department of the Environment, Food and Rural Affairs (DEFRA). Available from www.defra.gov.uk/environment/statistics/land/download/xls/ldtb01.xls.

DEFRA (2007a). UK Biomass Strategy, Available from http://www.defra.gov.uk/environment/climatechange/uk/energy/renewablefuel/pdf/ukbiomassstrategy-0507.pdf.

DEFRA (2007b). SFFS Indicators Workbook, Section 1.04 Commodity Yields, Available from http://statistics.defra.gov.uk/esg/indicators/d104_data.htm.

DEFRA (2009). Protecting our Water, Soil and Air: A code of good agricultural practice for farmers, growers and land managers, Published by the Stationary Office. Available to download from http://www.defra.gov.uk/foodfarm/landmanage/cogap/documents/cogap090202.pdf. Accessed on 30th November 2010.

DEFRA (2010). DEFRA survey of commercial and industrial waste arisings [2009] - data tables. DEFRA, Available at http://www.defra.gov.uk/statistics/environment/waste/wrfg03-indcom/. Accessed on 12th September 2012.

DEFRA (2011a). Local authority collected waste statistics - tables for November 2011 statistical release. DEFRA. Available at http://www.defra.gov.uk/statistics/environment/waste/wrfg23-wrmsannual/. Accessed on 06th September 2012.

DEFRA (2011b). Local authority municipal and household waste statistics 2009/10. DEFRA. Available at http://www.defra.gov.uk/statistics/environment/waste/wrfg23-wrmsannual/. Accessed on 07th September 2012.

DEFRA (2011c). Industrial and commercial waste arisings by business sector and waste management method (2002/03). Environment Agency & DEFRA, Available at http://www.data.gov.uk/dataset/industrial-and-commercial-waste-arisings-by-business-sector-and-waste-management-method. Accessed on 10th September 2012.

DEFRA (2011d). Crop areas, livestock populations and agricultural workforce 2011 England - Final results, DEFRA. Available at http://www.defra.gov.uk/statistics/foodfarm/landuselivestock/junesurvey/. Accessed on 13th September 2012.

Del Toro, A., M. Frailey, F. Lynch, S. Munshi and S. Wayne (2005). Development and demonstration of hydrogen and compressed natural gas (H/CNG) blend transit buses, National Renewable Energy Laboratory, Technical Report NREL/TP-540-38707.

DENA (2009). Executive Report: The German Market for Biomethane, Deutsche Energie-Agentur GmbH (DENA), German Energy Agency.

DEONI (2011). Northern Ireland Muncipal Waste Management Statistics, Annual report 2010/11, Department for the Environment in Northern Ireland. Available at http://www.doeni.gov.uk/niea/waste-home/municipal_data_reporting.htm. Accessed on 07th September 2012.

Department of Agriculture and Rural Development, N. I. (2012). The agricultural census in Northern Ireland, Results for June 2011, Available at http://www.dardni.gov.uk/index/publications/pubs-dard-statistics/ni-agri-census-2011.htm. Accessed on 13th September 2012.

DeWulf, J., H. v. Langenhove, J. Mulder, M. M. D. v. d. Berg, H. J. v. d. Kooi and J. d. Swaan-Arons (2000). "Illustrations towards quantifying the sustainability of technology." Green Chemistry **2**: 108-114.

DfT (2011a). Domestic Activity of GB Registered Heavy Goods Vehicles, Department for Transport. Available at http://www.dft.gov.uk/statistics/releases/road-freight-statistics-2010/ . Accessed on 10th July 2012.

DfT (2011b). Transport Statistics for Great Britain: Vehicles Summary, Department for Transport. Available at http://www.dft.gov.uk/statistics/releases/tsgb-2011-vehicles/. Accessed on 10th July 2012.

DfT (2011c). Renewable Transport Fuel Obligation (RTFO) Statistical Releases: Verified Data 2010/11, Department for Transport. Available at http://www.dft.gov.uk/statistics/series/biofuels/. Accessed on 10th July 2012.

Dhuyvetter, J., K. Hoppe and V. Anderson (1999). Wheat Middlings: A Useful Feed for Cattle, North Dakota State University.Available at www.ag.ndsu.edu/pubs/ansci/livestoc/as1175w.htm. Accessed on 02/05/12.

Dimopoulos, P., C. Bach, P. Soltic and K. Boulouchos (2008). "Hydrogen - natural gas blends fuelling passenger car engines: combustion, emissions and well-to-wheels assessment." International Journal of Hydrogen Energy **33**: 7224 - 7236.

Dincer, I. and C. Zamfirescu (2012). "Sustainable hydrogen production options and the role of IAHE." International Journal of Hydrogen Energy **37**(21): 16266-16286.

Djomo, S., Njakou and D. Blumberga (2011). "Comparitive life cycle assessment of three biohydrogen pathways." Bioresource Technology **102**(3): 2684-2694.

Dreyer, L. C., A. L. Niemann and M. Z. Hauschild (2003). "Comparison of three different LCIA methods: EDIP97, CML2001 and Eco-indicator 99." International Journal of Life Cycle Assessment **8**(4): 191 - 200.

DTi (2004). Rye grass as an energy crop uding biogas technology, Contract Number B/CR/00801/00/00, URN Number 05/1804.

Dufour, J., D. P. Serrano, J. L. Gálvez, A. González, E. Soria and J. L. G. Fierro (2012). "Life cycle assessment of alternatives for hydrogen production from renewable and fossil sources." International Journal of Hydrogen Energy **37**: 1173 - 1183.

DVGW (2000a). Deutsche Vereinigung des Gas und Wasserfaches e. V. (DVGW) Standard G260 "Gas Quality".

DVGW (2000b). Deutsche Vereinigung des Gas und Wasserfaches e. V. (DVGW) Standard G262 "Usage of gases from renewable sources in the public gas distribution".

Dynetek Industries Ltd. (2006). Specifications: Dynecell 450 bar / 6527 psi Hydrogen Dynecell cylinders. Available at http://www.dynetek.com/cylinders.php. Accessed on 19th October 2012.

Elliot, D., C, R. Orth, J., G. Johnway, T. Werpy, A., D. Eakin, E., A. Schmidt, A., G. Neuenschwander, G., A. Flagg, J., J. Murry, L. Lahman, C. J. Lin, D. Mennel, L. and R. Landucci (2002). Biorefinery concept development based on wheat flour milling, US Department of Energy, Pacific Northwest Laboratory (PNL). Available at www.pnl.gov/biobased/docs/wheat.pdf. Accessed on 02/05/12.

Enguídanos, M., A. Soria, B. Kavalov and P. Jensen (2002). Techno-economic analysis of Bio-diesel production in the EU: A short summary for decision makers, European Commission Joint Research Centre, Report Ref: EUR 20279 EN. May 2002.

Environment Agency Wales (2009). Survey of commercial and industrial waste arisings in Wales data. Urban Mines Ltd., Available at http://www.environment-agency.gov.uk/research/library/publications/107692.aspx. Accessed on 12th September 2012.

Eroglu, E. and A. Melis (2011). "Photobiological hydrogen production: Recent advances and state of the art." Bioresource Technology **102**(18): 8403-8413.

ESRU. "Biodiesel production including oil and biodiesel yields. Available from www.esru.strath.ac.uk/EandE/Web_sites/02-03/biofuels/quant_biodiesel.htm." 2007.

ESRU. "Bioethanol production from wheat and sugarbeet, downloaded from www.esru.strath.ac.uk/EandE/Web_sites/02-03/biofuels/quant_bioethanol.htm." 2007.

Esteves, S. R. and D. Devlin (2010). Food Waste Chemical Analysis. Published by WRAP. Available to download at http://www.wrapcymru.org.uk/about_wrap_cymru/compositional_report.html. Accessed on 04th August 2010.

European Commission (1994). Biofuels: Application of biologically derived products as fuels or additives in combustion engines, Agro-Industrial Research Division, European Commission, Directorate General XII.

European Commission (2008). 20 20 by 2020: Europe's Climate Change Opportunity. Brussels, Commission of the European Communities.

European Commission (2011). Report of the European expert group on future transport fuels, European Commission. Available at http://ec.europa.eu/transport/themes/urban/events/2011_04_13_future_transport_fuels_en.htm. Accessed on 07th January 2013.

European Commission (2012). Proposal for a Directive of the European Parliament and of the Council amending Directive 98/70/EC relating to the quality of petrol and diesel fuels, and amending Directive 2009/28/EC on the promotion of the use of energy from renewable sources, European Commission.

European Commission Joint Research Centre (JRC) Institute for Environment and Sustainability (2010). General Guide for Life Cycle Assessment - Detailed Guide. ILCD Handbook, International Reference Life Cycle Data System.

European Parliament (2003). The promotion of the use of biofuels or other renewable fuels for transport. Directive 2003/30/EC of the European Parliament and of the Council, May 2003. Official Journal of the European Union.

European Parliament (2009a). Directive 2009/28/EC On the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. Official Journal of the European Union. Available at http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2009:140:0016:0062:EN:PDF. Accessed on 23rd September 2009, The European Parliament and Council of the European Union.

European Parliament (2009b). Directive 2009/30/EC amending Directive 98/70/EC as regards the specification of petrol, diesel and gas-oil and introducing a mechanism to monitor and reduce greenhouse gas emissions and amending Council Directive 1999/32/EC as regards the specification of fuel used by inland waterway vessels and repealing Directive 93/12/EEC. Official Journal of the European Union, The European Parliament and the Council of the European Union.

Eyler, D. (2009). Biomethane in Germany, Injection into the NG grid and Vehicle Fuel, Biogasmax. Available at http://www.biogasmax.eu/media/eifer__biomethane_in_germany__023022200_1136_24062009.pdf. Accessed on 06th October 2009.: EIFER - Europäisches Institut für Energieforschung / European Institute for Energy Research.

Ferreira, A., F., J. Ribau, P. and C. Silva, M. (2011). "Energy consumption and CO_2 emissions of potato peel and sugar cane biohydrogen production pathways, applied to

Portuguese road transportation." International Journal of Hydrogen Energy **36**: 13547 - 13558.

Finnveden, G., J. Johansson, P. Lind and Å. Moberg (2005). "Life cycle assessment of energy from solid waste - part 1: general methodology and results." Journal of Cleaner Production **13**: 213-229.

Franco, C., F. Pinto, I. Gulyurtlu and I. Cabrita (2003). "The study of reactions influencing the biomass steam gasification process." Fuel **82**(7): 835-842.

Fravolini, D. and S. Proietti (2009). Biogas use as fuel in waste collector captive fleet: the experience in the city of Rome, future plans. European Biomethane Fuel Conference, Goteburg, September 2009, BiogasMax.

Frenette, G. and D. Forthoffer (2009). "Economic & commercial viability of hydrogen fuel cell vehicles from an automotive manufacturer perspective." International Journal of Hydrogen Energy **34**(9): 3578-3588.

Fuchs, W. and B. Drosg (2010). Technologiebewertung von Gärrestbehandlungs - und Verwertungskonzepten, Universität für Bodenkultur Wien. ISBN 978-3-900962-86-9.

Gallagher, E. (2008). The Galagher Review of the indirect effect of biofuels production., The Renewable Fuels Agency. Available at www.renewablefuelsagency.org/_db/_documents/Report_of_the_Gallagher_review.pdf. Accessed on 17th November 2009.

Gas de France (2007). Prescriptions techniques du distributeur Gaz de France., Gas de France. Available at www.biogasmax.co.uk/media/prescriptions_gaz_de_france__055742400_1047_05112008.pdf. Accessed on 07th January 2010.

Gladstone, R. W. (2007). High-BTU Projects Using Pressure Swing Adsorption (PSA) Technology, Green Gas Energy LLC.

Goedkoop, M. and R. Spriensma (2001). The Eco-indicator 99 A damage orientated method for life cycle impact assessment methodology report (Third Edition), PRÉ Consultants B.V. NL Amersfoort.

González-García, S., D. Iribarren, A. Susmozas, J. Dufour and R. J. Murphy (2012). "Life cycle assessment of two alternative bioenergy systems involving Salix spp. biomass: Bioethanol production and power generation." Applied Energy **95**(0): 111-122.

Graham, L., A., G. Rideout, D. Rosenblatt and J. Hendren (2008). "Greenhouse gas emissions from heavy duty vehicles." Atmospheric Environment **42**: 4665 - 4681.

Granovskii, M., I. Dincer and M. Rosen, A. (2007(a)). "Exergetic life cycle assessment of hydrogen production from renewables." Journal of Power Sources **167**: 461 - 471.

Granovskii, M., I. Dincer and M. Rosen, A. (2007(b)). "Greenhouse gas emissions reduction by use of wind and solar energies for hydrogen and electricity production: Economic factors." International Journal of Hydrogen Energy **32**: 927 - 931.

Green Car Congress. (2013). "SEAT introducing natural gas Mii Ecofuel with 79 g/km CO2." Available at http://www.greencarcongress.com/2013/03/seat-20130303.html. Accessed on 04th March 2013.

Günther, L. (2007). DGE GmbH Presentation: Purification of biomethane using pressureless purification for the production of biomethane and carbon dioxide. INNOGAS, Available from www.dge-wittenberg.com/english/vortraege/DGE%20Fachtagung%20WB%202006%20teil1-EN.pdf. Accessed on 22 July 2008.

Håkansson, A. (2006). Preventing microbial growth on pall rings when upgrading biogas using absorption with water wash, Swedish Biogas Centre, Report SGC166. Report available from http://www.sgc.se/rapporter/resources/SGC166.pdf. Accessed on 18th July 2008.

Hallenback, P. C., M. Abo-Hashesh and D. Ghosh (2012). "Strategies for improving biological hydrogen production." Bioresource Technology **110**: 1 - 9.

Hammer, M., K. Pettersson and B. Svensen (2007). Guide to developing a local biogas strategy, for sustainable large-scale consumption and production in collaboration between town and countryside. BiogasMax Report Biogas as vehicle fuel - Market Expansion to 2020 Air Quality, Available at www.medcoast.org/download/18.726a005111ed95650fa80002283/Guide+to+developing+a+local+biogas+strategy.pdf. Accessed on 14th September 2009.

Hansen, T., L., J. E. Schmidt, I. Angelidaki, E. Marca, J. l. C. Jansen, H. Mosbæk and T. Christensen, H. (2004). "Method for determination of methane potentials of solid organic waste." Waste Management **24**: 393-400.

Hartmann, J. K. (2006). Life cycle assessment of industrial scale biogas plants. Department for Agricultural Science. Göttingen, George-August-Universität. **PhD:** 205.

Hawkes, F., I. Hussey, G. Kyazze, R. Dinsdale and D. L. Hawkes (2007). "Continuous dark fermentative hydrogen production by mesophilic microflora: Principles and progress." International Journal of Hydrogen Energy **32**: 172-184.

Hawkes, F., R., H. Forsey, G. Premier, C., R. Dinsdale, M., D. Hawkes, L., A. Guwy, J., J. Maddy, S. Cherryman, J. Shine and D. Auty (2008). "Fermentative production of hydrogen from wheat flour industry co-product." Bioresource Technology **99**: 5020 - 5029.

Heiermann, M. and M. Plöchl (2004). A big potential for biogas production. World Renewable Energy Congress VIII, Denver, Colorado.

Her Majesty's Revenue and Customs (2007). BN53: Hydrocarbon Oil Duty: Rates, Her Majesty's Revenue and Customs. Available at http://www.hmrc.gov.uk/budget2007/bn53.htm. Accessed on 22nd September 2009.

Hidai, S., M. Kobayashi, H. Niwa, Y. Harada, M. Oshima, Y. Nakamori and T. Aoki (2012). "Platinum oxidation responsible for degradation of platinum–cobalt alloy cathode catalysts for polymer electrolyte fuel cells." Journal of Power Sources **215**(0): 233-239.

Holm-Nielson, J. B., M. Madsen and P. O. Popiel (2006). Predicted Energy Crop Potentials for Bioenergy Worldwide and for EU-25. World Bioenergy 2006 Conference on Biomass for Energy. Jönköping, Sweden.

Honda. (2012). "Honda FCX Clarity specifications."

Hoogwijk, M., A. Faaij, v. d. R. Broek, G. Berndes, D. Gielen and W. Turkenburg (2003). "Exploration of the ranges of the global potential of biomass for energy." Biomass and Bioenergy **25**: 119-133.

Hospido, A., M. Carballa, M. Moreira, F. Omil, J. Lema, M. and G. Feijoo (2010). "Environmental assessment of anaerobically digested sludge reuse in agriculture: Potential impacts of emerging micropollutants." Water Research **44**: 3225 - 3233.

Hrad, M., M. Huber-Humer and M. Piringer (2011). Greenhouse gas monitoring for optimization of energy balance and process efficiency of biogas. International IWA

Symposium on Anaerobic Digestion of Solid Waste and Energy Crops, Vienna, Austria.

Huesemann, M. H., T. S. Hausmann, B. M. Carter, J. J. Gerschler and J. R. Benemann (2010). "Hydrogen Generation Through Indirect Biophotolysis in Batch Cultures of the Nonheterocystous Nitrogen-Fixing Cyanobacterium Plectonema boryanum." Applied Biochemistry and Biotechnology **162**(1): 208-220.

Huibo, S., J. Cheng, J. Zhou, W. Song and K. Cen (2009). "Combination of dark- and photo-fermentation to enhance hydrogen production and energy conversion efficiency." International Journal of Hydrogen Energy **34**: 8846 - 8853.

Hyundai. (2012). "Hyundai is first to launch series production of zero-emissions hydrogen fuel cell vehicle." Available at http://www.hyundai.com.au/about-hyundai/news/Articles/Hyundai-is-First-to-Launch-Series-Production-of-Zero-emissions-Hydrogen-Fuel-Cell-Vehicle. Accessed on 04th March 2013.

IEA (2005). IEA Bioenergy Task 37 Summary: Injection of biogas into the natural gas grid in Laholm, Sweden, Document available from http://www.iea-biogas.net/Dokumente/casestudies/sucess_laholm.pdf Accessed on 21st July 2008.

Inui, T., Y. Okugawa and M. Yasuda (1988). "Relationship between properties of various zeolites and their CO_2 adsoption behaviours in pressure swing adsorption operation " Industrial and Engineering Chemical Research **27**: 1103-1109.

IPCC (2007). Climate Change 2007: The Physical Science Basis. Contribution of Working Group 1 to the Fourth Assessment Report of the Intergorvernmental Panel on Climate Change. S. Solomon, D. Qin, M. Manninget al**:** 996.

Ishikawa, S., S. Hoshiba, T. Hinata, H. T and S. Morita (2006). "Evaluation of a biogas plant from life cycle assessment (LCA)." International Congess Series **1293**: 230-233.

Japanfs.org. (2007). "Japanese research group generates hydrogen from bread waste. Japan for Sustainability, downloaded from www.japanfs.org/db/database.cgi?cmd=dp&num=858&dp=data_e.html on 04th May 2007."

Jönbrink, A. K., C. Wolf-Wats, M. Erixon, P. Olsson and E. Wallén (2000). LCA Software Survey, IVL Swedish Environmental Research Institute Ltd. Available at http://www3.ivl.se/rapporter/pdf/B1390.pdf.

Jönsson, O. (2009). Biogas upgrading – Technologies, framework and experience. Microphilox Project Workshop, Barcelona, Spain 26th March 2009. Available at www.microphilox.com/pdf/OJonsson%20Natural%20gas%20grid%20injection.pdf. Accessed on 14th September 2009.

Jungbluth, N., M. Chudacoff, A. Dauriat, F. Dinkel, G. Doka, M. Emmenegger, E. Gnansounou, M. Keller, N. Kljun, K. Schleiss, M. Spielmann, C. Stettler and J. Sutter (2007). Life Cycle Inventories of Bioenergy (Data v2.0). Ecoinvent Report No. 17, Swiss Centre for Life Cycle Inventories.

Jury, C., E. Benetto, D. Koster, B. Schmitt and W. Joëlle (2010). "Life Cycle Assessment of biogas production by monofermentation of energy crops and injection into the national gas grid." Biomass and Bioenergy **34**: 54-66.

Karim, G. A. (2003). "Hydrogen as a spark ignition engine fuel." International Journal of Hydrogen Energy **28**(5): 569-577.

Karner, D. and J. Francfort (2003). Low percentage hydrogen / CNG blend Ford F-150 operating summary, US Department of Energy FreedomCAR and Vehicle Technologies, Advanced Vehicle Testing Activity. Report Ref. INEEL/EXT-03-00008.

Khan, F. I., K. Hawboldt and M. T. Iqbal (2005). "Life cycle analysis of wind-fuel cell integrated system." Renewable Energy **30**: 157-177.

Kikkinides, E. S. and R. T. Yang (1993). "Gas separation and purification by polymeric adsorbents: Flue gas desulfurization and SO_2 recovery using styrenic polymers." Industrial and Engineering Chemical Research **32**: 2365-2372.

Kim, S. and B. E. Dale (2004). "Global potential bioethanol production from wasted crops and crop residues." Biomass and Bioenergy **26**: 361-375.

Klöpffer, W. (2012). "The critical review of life cycle assessment studies according to ISO 14040 and 14044." International Journal of Life Cycle Assessment **17**: 1087 - 1093.

Koroneos, C., A. Dompros, G. Roumbas and N. Moussiopoulos (2004). "Life cycle assessment of hydrogen fuel production processes." International Journal of Hydrogen Energy **29**: 1443-1450.

Kreith, F. and R. West (2004). "Fallacies of a hydrogen economy: A critical analysis of hydrogen production and utilisation." Journal of Energy Resources Technology **126**: 249 - 257.

Kroes, N. (2007). 04:I 2007 C(2007)8 State aid n° N 418/2006 - United Kingdom, Renewable Transport Fuel Obligation (RTFO). Brussels, European Commission. Available at http://ec.europa.eu/community_law/state_aids/comp-2006/n418-06.pdf. Accessed on 01st October 2009.

Kumar, A., D. Jones, D. and M. Hanna, A. (2009). "Thermochemical biomas gasification: A review of the current status of technology." Energies **2**: 556 - 581.

Kumar, S., Javanta, N. R. Panwar and M. V. Singh (2010). "Determination of lead and cadmium concentration limits in agricultural soil and municipal solid waste compost through an approach of zero tolerance to food contamination." Environ Monit Assess **168**(1-4): 397 - 406.

L'Observitoire Des Energies Renouvelables (2010). Biogas barometer, EurObserv'ER. Available at http://www.eurobserv-er.org/downloads.asp. Accessed on 02nd August 2012.

Lee, J.-Y., S. An, K. Cha and T. Hur (2010). "Life cycle environmental and economic analyses of a hydrogen station with wind energy." International Journal of Hydrogen Energy **35**: 2213 - 2225.

Lehtomaki, A. and L. Björnsson (2006). "Two stage anaerobic digestion of energy crops: methane production, nitrogen mineralisation and heavy metal mobilisation." Environmental Technology **37**: 209-218.

Lehtomaki, A., S. Huttunen and J. Rintala (2007). "Laboratory investigations on co-digestion of energy crops and crop residues with cow manure for methane production: Effect of crop to manure ratio." Resources, Conservation and Recycling **51**: 591 - 609.

Levy, R. H. (1993). Les Biocarburants. A report to the French Government based on figures from the Commission Consultative Pour la Production des Carburants de Substitution, 1991.

Liebetrau, J., C. Reuschel, J. Clemens, C. Hafermann, J. Friehe and P. Weiland (2011). Analysis of greenhouse gas emissions from 10 biogas plants within the agricultural sector. International IWA Symposium on Anaerobic Digestion of Solid Waste and Energy Crops, Vienna, Austria.

Lin, C. Y., S. Y. Wu, P. J. Lin, J. S. Chang, C. H. Hung, K. S. Lee, F. Y. Chang, C. Y. Chu, C. H. Cheng, C. H. Lay and A. C. Chang (2010). "Pilot-scale hydrogen fermentation system start-up performance." International Journal of Hydrogen Energy **35**(24): 13452-13457.

Linsbod (2005). "fuchs'nhof" - Austrias first feeding of upgraded biogas to the natural gas grid. Available on the IEA Task 37 website at www.iea-biogas.net/casestudiesoperators.htm. Accessed on 17th July 2008.

Littlewood, J., R. J. Murphy and L. Wang (2013). "Importance of policy support and feedstock prices on economic feasibility of bioethanol production from wheat straw in the UK." Renewable and Sustainable Energy Reviews **17**(0): 291-300.

Living Countryside Ltd. (2007). "UK fodder maize yields from www.ukagriculture.com/crops/forage_maize.cfm." 2007.

López, J. M., A. Gomez, F. Aparicio and J. Sanchez (2009). "Comparison of GHG emissions from diesel, biodiesel and natural gas refuse trucks of the City of Madrid." Applied Energy **86**: 610 - 615.

Lou, H., H. Miyajima, F. Dong, A. Kodama, M. Goto and T. Hirose (1999). "Experimental study of thermal phenomenon in PSA air dehumidification." Separation and Purification Technology **17**: 65-75.

Maggio, G. and G. Cacciola (2012). "When will oil, natural gas and coal peak?" Fuel **98**: 111 - 123.

Mähnert, P., M. Heiermann and B. Linke (2005). "Batch and semi continuous biogas production from different grass species." Agricultural Engineering International: the CIGR e-journal **VII**: Manuscript EE 05 010.

Malça, J. and F. Freire (2011). "Life cycle studies of biodiesel in Europe: A review addressing the variability of results and modeling issues." Renewable and Sustainable Energy Reviews **15**: 338 - 351.

Martínez-Pérez, N., S. J. Cherryman, G. C. Premier, R. M. Dinsdale, D. L. Hawkes, F. R. Hawkes, G. Kyazze and A. J. Guwy (2007). "The potential for hydrogen enriched biogas production from crops: Scenarios in the UK." Biomass and Bioenergy **31**: 95-104.

Martínez, P., L. Dawidowski, D. Gómez and D. Pasquevich (2010). "Life cycle greenhouse emissions of compressed natural gas-hydrogen mixtures for transportation in Argentina." International Journal of Hydrogen Energy **35**: 5793 - 5798.

Massanet-Nicolau, J., R. Dinsdale, M., A. Guwy, J. and G. Shipley (2013). "Use of real time gas production data for more accurate comparison of continuous single-stage and two-stage fermentation." Bioresource Technology **129**: 561 - 567.

Mazloomi, K., N. B. Sulaiman and H. Moayedi (2012). "Electrical Efficiency of Electrolytic Hydrogen Production." International Journal of Electrochemical Science **7**(4): 3314-3326.

McKinsey & Company (2010). A Portfolio of Power-Trains for Europe: A Fact-Based Analysis, Available at http://www.europeanclimate.org/index.php?option=com_content&task=view&id=35&Itemid=53. Accessed on 12th February 2013.

Melaina, M. (2007). "Turn of the century refueling: A review of innovations in early gasoline refueling methods and analogies for hydrogen." Energy Policy **35**: 4919 - 4934.

Menardo, S., F. Gioelli and P. Balsari (2011). "The methane yield of digestate: Effect of organic loading rate, hydraulic retention time, and plant feeding." Bioresource Technology **102**(3): 2348-2351.

Michalski, J., U. Bünger and C. Stiller (2011). "Business analysis of the hydrogen refuelling station infrastructure and the role of the pricing system." International Journal of Hydrogen Energy **36**: 8152 - 8157.

Mignard, D. and C. Pritchard (2007). "A review of the sponge iron process for the storage and transmission of remotely generated marine energy." International Journal of Hydrogen Energy **32**: 5039 - 5049.

Miltner, M., A. Makaruk and M. Harasek (2008). Application of gas permeation for biogas upgrade - Operational experiences of feeding biomethane into the Austrian gas grid. 16th European Biomass Conference and Exhibition. Valencia, Spain.

Mohammadi, A., M. Shioji, Y. Nakai, W. Ishikura and E. Tabo (2007). "Performance and combusiton characteristics of a direct injection SI hydrogen engine." International Journal of Hydrogen Energy **32**: 296 - 304.

Monson, K. D., S. R. Esteves, A. J. Guwy and R. M. Dinsdale (2007). Anaerobic Digestion of Biodegradable Municipal Wastes: A Review, The Sustainable Environment Research Centre, University of Glamorgan. ISBN 978-1-84054-156-5.

Montauk Energy (2008). Montauk Energy, Case Study: LFG to high Btu pipeline quality gas, McCarty Road Landfill, Houston, Texas, Available from http://www.montaukenergy.com/mccarty.asp. Accessed on 23rd July 2008.

Murphy, J. D. and K. McCarthy (2005). "The optimal production of biogas for use as a tranport fuel in Ireland." Renewable Energy **30**: 2111-2127.

Murphy, J. D. and N. Power (2009). "Technical and economic analysis of biogas production in Ireland utilising three different crop rotations." Applied Energy **86**: 25-36.

Murphy, R., J. Woods, M. Black and M. McManus (2011). "Global developments in the competition for land from biofuels." Food Policy **36, Supplement 1**(0): S52-S61.

National Grid (2008). Investigation into transmission losses on UK electricity transmission system, National Grid.

National Non Food Crop Centre (NNFCC) (2012). Official Biogas Plant Map Tool. Available at http://www.biogas-info.co.uk/index.php/ad-map.html

Neelis, M. L., H. J. van der Kooi and J. J. C. Geerlings (2004). "Exergetic life cycle analysis of hydrogen production and storage systems for automotive applications." International Journal of Hydrogen Energy **29**: 537 - 545.

NGV Communictions Group. (2012). "NGV Journal, Worldwide NGV Statistics." Retrieved 01st August 2012. http://www.ngvjournal.dreamhosters.com/en, 2012.

Niven, R. K. (2005). "Ethanol in gasoline: environmental impacts and sustainability review article." Renewable and Sustainable Energy Reviews **9**: 535-555.

Non Fossil Purchasing Agency Ltd. (2012). "e-TOC Online Auctions." http://www.nfpas-auctions.co.uk/etoc/trackrecord.html., Accessed on 11th October 2012.

Nordberg, A. (2003). Treatment of animal waste in co-digestion biogas plants in Sweden. The Future of Biogas In Europe II. Esbjerg, Denmark, 2-4 October 2003.

NSCA (2006). Biogas as a road transport fuel. ISBN 978 0 903 47461 1.

Nylund, N. O. and A. Lawson (2000). Exhaust emissions from natural gas vehicles. IANGV Technical Committee. Report available from www.apvgn.pt.

Offer, G. J., D. Howey, M. Contestabile, R. Clague and N. P. Brandon (2010). "Comparative analysis of battery electric, hydrogen fuel cell and hybrid vehicles in a future sustainable road transport system." Energy Policy **38**(1): 24-29.

Oh, S. E. and B. E. Logan (2005). "Hydrogen and electricity production from a food processing wastewater using fermentation and microbial fuel cell technologies." Water Research **39**: 4673 - 4682.

Olivier, J., G.J., G. Janssens-Maenhout, J. Peters, A.H.W. and J. Wilson (2011). Long Term Trends in Global CO_2 Emissions: 2011 Report, European Union Joint Research Council (JRC) / Netherlands Environmental Assessment Agency. Available at www.pbl.nl/en. Accessed on 12th July 2012.

Paavola, T. and J. Rintala (2008). "Effects of storage on characteristics and hygienic quality of digestates from four co-digestion concepts of manure and biowaste." Bioresource Technology **99**: 7041 - 7050.

Pädam, S., D. Waluszewski, B. Rehnlund and B. Hugosson (2010). Biogasmax: Biomethane vehicles in five European cities, Biogasmax.

Pastowski, A. and T. Grube (2010). "Scope and perspectives of renewable hydrogen production and infrastructure for fuel cell vehicles in North Rhine - Westphalia." Energy Policy **38**: 5382 - 5387.

Patterson, T., R. M. Dinsdale and S. R. Esteves (2008). "Review of Energy Balances and Emissions Associated with Biomass Based Transport Fuels Relevant to the United Kingdom Context." Energy and Fuels **22**: 3506-3512.

Patterson, T., S. R. Esteves, R. M. Dinsdale and A. J. Guwy (2011a). "An evaluation of the policy and techno-economic factors affecting the potential for biogas upgrading for transport fuel use in the UK." Energy Policy **39**: 1806 - 1816.

Patterson, T., S. R. Esteves, R. M. Dinsdale and A. J. Guwy (2011b). "Life Cycle Assessment of biogas infrastructure options on a regional scale." Bioresource Technology **102**: 7313 - 7323.

Persson, M. (2007). Biogas upgrading and utilisation as a vehicle fuel. Proceedings of The Future of Biogas in Europe III, University of Southern Denmark. Available from http://websrv5.sdu.dk/bio/Probiogas/down/work07/Proceedings.pdf. Accessed on 17th July 2008.

Persson, M., O. Jönsson and A. Wellinger (2006a). Biogas upgrading to vehicle fuel standards and grid injection, IEA Bioenergy, Task 24.

Persson, M. and A. Wellinger (2006b). Biogas upgrading and utilisation: Biogas upgrading to vehicle fuel standard and grid introduction, IEA Bioenergy.

Persson, M., A. Wellinger, B. Rehnlund and L. Rahm (2007). Report on technological applicability of existing biogas upgrading processes. Biogasmax report del_3.1_STO_v3_final, Available from http://www.biogasmax.co.uk/media/report_on_technological_2007__041639600_1025_22052007.pdf Accessed on 21st July 2008.

Pertl, A., P. Mostbauer and G. Obersteiner (2010). "Climate balance of biogas upgrading systems." Waste Management **30**: 92-99.

Petersson, A. (2008). Biogas as a Transport Fuel - Upgrading Technique and Application. . Central European Biomass Conference 2008. Graz, Austria. Presentation available from http://www.biomasseverband.at/static/mediendatenbank/root01/3.%20Veranstaltungen/3.2%. Accessed on 17th July 2008.

Petersson, A. and A. Wellinger (2009). Biogas upgrading technologies - developments and innovations, IEA Bioenergy. Available at http://www.iea-biogas.net/Dokumente/upgrading_rz_low_final.pdf. Accessed on 14th April 2010.

Pimentel, D. and T. W. Patzek (2005). "Ethanol production using corn, switchgrass, and wood: Biodiesel production using soybean and sunflower." Natural Resources Research **14**(1): 65-76.

Plöchl, M. and M. Heiermann (2006). "Biogas farming in Central and Northern Europe: A strategy for developing countries?" Agricultural Engineering International: the CIGR e-journal **VIII**: Invited overview No. 8.

Poeschl, M., S. Ward and P. Owende (2012a). "Environmental impacts of biogas deployment - Part I: Life cycle inventory for evaluation of production processes and emissions to air." Journal of Cleaner Production **24**: 168 - 183.

Poeschl, M., S. Ward and P. Owende (2012b). "Environmental impacts of biogas deployment - Part II: Life Cycle Assessment of multiple production and utilisation pathways." Journal of Cleaner Production **24**: 184 - 201.

Pöschl, M., S. Ward and P. Owende (2010). "Evaluation of energy efficiency of various biogas production and utilization pathways." Applied Energy **87**: 3305-3321.

Punter, G., D. Rickeard, J.-F. Larive, R. Edwards, N. Mortimer, R. Horne, A. Bauen and J. Woods (2004). Well to wheel evaluation for production of ethanol from wheat, A report by the Low Carbon Vehicle Partnership, UK. Ref. FWG-P-04-024.

QuestAir Technologies Inc. (2006). QuestAir Technologies Inc. Press Release - QuestAir demonstration plant at City of Vancouver landfill recovers methane from landfill gas. Available at http://www.questairinc.com/investor_relations/press_releases/archived_releases/2006/03-13.htm. Accessed on 18th July 2008.

QuestAir Technologies Inc. (2007). QuestAir Technologies Inc. press release - QuestAir to supply methane recovery system for New Hampshire landfill gas-to-energy project. Available at http://www.questairinc.com/investor_relations/press_releases/archived_releases/2007/pdf/070904.pdf. Accessed on 18th July 2008.

Ratlamwala, T. A. H., A. H. El-Sinawi, M. A. Gadalla and A. Aidan (2012). "Performance analysis of a new designed PEM fuel cell." International Journal of Energy Research **36**(11): 1121-1132.

Renewable Energy Agency (2009). UK biofuels sector unsettled by "error" in legislation. Bioenergy Business. Environmental Finance Publications, Available at the Renewable Energy Association website www.r-e-a.net/info/rea-news/REAinthePress/090107BioenergyBusiness/?searchterm=RTFO.

Renewable Fuels Agency (2008). Carbon Calculator Version 1.0 User Manual, Produced by E4Tech and the Renewable Fuels Agency, UK: Available at http://www.renewablefuelsagency.org/_db/_documents/User_manual.pdf. Accessed on 23rd September 2009.

Rice, G. (1997). "LCA software review: A comparison of currently available European LCA software." International Journal of Life Cycle Assessment **2**(1): 53 - 59.

Rosenstrater, K. A. (2005). "Expanding the role of systems modelingL considering byproduct generation from biofuel production." Ecology and Society **11**(1: r2 [online]. Available from http://www.ecologyandsociety.org/vol11/iss1/resp2/).

Ruggeri, B., T. Tommasi and G. Sassi (2010). "Energy balance of dark anaerobic fermentation as a tool for sustainability analysis." International Journal of Hydrogen Energy **35**: 10202-10211.

Sachs, I. (2008). The Biofuels Controversy. The United Nations Conference on Trade and Development. **UNCTAD/DITC/TED/2007/12**.

Schmack Biogas AG. (2007). "Case Study, Feeding biogas into the natural gas grid - using the Pliening biomethane plant as an example. Available from www.schmack-biogas.com. Accessed on 16th July 2008."

Schoots, K., G. J. Kramer and B. C. C. van der Zwaan (2010). "Technology learning for fuel cells: An assessment of past and potential cost reductions." Energy Policy **38**(6): 2887-2897.

SEPA (2012). Waste data digest 12: data tables. Scottish Environment Protection Agency. Available at http://www.sepa.org.uk/waste/waste_data/waste_data_digest.aspx. Accessed on 07th September 2012.

Shapouri, H., J. Duffield, A. and M. Wang (2002). The energy balance of corn ethanol: An update, US Department of Agriculture, Office of the Chief Economist, Office of Energy Policy and New Uses.

Shayegan, S., P. J. C. Pearson and D. Hart (2009). "Hydrogen for buses in London: A scenario analysis of changes over time in refuelling infrastructure costs." International Journal of Hydrogen Energy **34**: 8415 - 8427.

Sherif, S. A., F. Barbir and T. N. Veziroglu (2005). "Wind energy and the hydrogen economy - A review of tehcnologies." Solar Energy **78**: 647-660.

Sherman, J., D. (1999). "Syntheti zeolites and other microporous oxide molecular sieves." Proceedings of the National Academy of Sciences **96**: 3471-3478.

Sierens, R. and S. Verhelst (2001). "Experimental study of a hydrogen fuelled engine." Journal of Engineering for Gas Turbines and Power **123**(1): 211 - 216.

Sircar, S. (2002). "Pressure Swing Adsorption." Industrial and Engineering Chemical Research **41**: 1389-1392.

Sircar, S., T. C. Golden and M. B. Rao (1996). "Activated carbon for gas separation and storage." Carbon **34**(1): 1-12.

Siriwardane, R., V., M.-S. Shen and E. Fisher, P. (2003). "Adsorption of CO_2, N_2, and O_2 on natural zeolites." Energy and Fuels **17**: 571-576.

Siriwardane, R., V., M.-S. Shen, E. Fisher, P. and J. Poston, A. (2001). "Adsorption of CO_2 on molecular sieves and activated carbon." Energy and Fuels **15**: 279-284.

Smith, S., R. (2009). "A critical review of the bioavailability and impacts of heavy metals in municipal solid waste composts compared to sewage sludge." Environment International **35**: 142-156.

Smyth, B. M., J. D. Murphy and C. M. O'brien (2009). "What is the energy balance of grass biomethane in Ireland and other temperate northern European climates?" Renewable and Sustainable Energy Reviews **13**: 2349-2360.

Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor and H. L. Miller (2007). Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change., Intergovernmental Panel on Climate Change (IPCC). Available at http://www.ipcc.ch/publications_and_data/publications_ipcc_fourth_assessment_report_wg1_report_the_physical_science_basis.htm.

Sørensen, B. (2004). Total life-cycle assessment of PEM fuel cell car, Roskilde University, Denmark.

Spath, P. L. and M. K. Mann (2004). Life cycle assessment of renewable hydrogen production via wind/electrolysis, US Department of Energy, National Renewable Energy Laboratory (NREL).

Stephens-Romero, S. D., T. M. Brown, J. E. Kang, W. W. Recker and G. S. Samuelson (2010). "Systematic planning to optimise investments in hydrogen infrastructure deployment." International Journal of Hydrogen Energy **35**: 4652 - 4667.

SVGW (2008). Schweizerischer Verein des Gas und Wasserfaches, Richtlinie G13, Entwurf vom 20.5.2008, SVGW.

Swedish Gas Centre (2008a). Case Study: The biogas plant in Linköping; in Biogas from Manure and Waste Products - Swedish Case Studies, , Available from http://www.sbgf.info/_filer/Biogasinfo_english.pdf. Accessed on 21st July 2008.

Swedish Gas Centre (2008b). Case Study of the biogas plant in Gothenburg; in Biogas from Manure and Waste Products - Swedish Case Studies, Available from http://www.sbgf.info/_filer/Biogasinfo_english.pdf. Accessed on 22nd July 2008.

Swedish Government (1999). Swedish Standard, Motorbränslen – Biogas som bränsle till snabbgående ottomotorer, SS 15 54 38.

Swiss Centre for Life Cycle Inventories (2009). Ecoinvent data version 2.1, the life cycle inventory data version, ISBN 3-905594-38-2.

The National Biodiesel Board. (2007). "Density of Biodiesel, downloaded from www.biodiesel.org." 2007.

The Scottish Government (2011). Final results from 2011 June agricultural census, Part 4, Available at http://www.scotland.gov.uk/Publications/2011/09/27083355/4. Accessed on 13th September 2012.

The Telegraph Online (2012). Wind farm operators 'Paid millions more than previously thought' by Grid to turn of turbines. The Telegraph. London, Available at http://www.telegraph.co.uk/earth/energy/windpower/9578104/Wind-farm-operators-paid-millions-more-than-previously-thought-by-Grid-to-turn-off-turbines.html. Accessed on 04th March 2013.

The United Nations (1998). Kyoto Protocol to the United Nations Framework Convention on Climate Change, Available onlne at http://unfccc.int/kyoto_protocol/items/2830.php/. Accessed on 12th July 2012.

Thomas, C. E., B. D. James, F. D. Lomax Jr and I. F. Kuhn Jr (2000). "Fuel options for the fuel cell vehicle: hydrogen, methanol or gasoline?" International Journal of Hydrogen Energy **25**: 551 - 567.

Tung, H., D. Cédric, A. Ventura, A. Jullien and G. Laurant (2005). A global tool for environmental assessment of roads: Application to transport for road building. European Conference of Transport Research Institutes. The Hague, Available to download at http://www.ectri.org/YRS05/Papiers/Session-3bis/ventura.pdf. Accessed on 11th October 2010.

TÜV SÜD Industrie Service GmbH. (2012). "H2Stations.org." Available at http://www.netinform.net/h2/H2Stations/Default.aspx. Accessed on 27th September 2012.

UK Government (2007). Statutory Instrument No. 3072 The Renewable Transport Fuel Obligations Order 2007. UK, Office of Public Sector Information. Available at www.opsi.gov.uk/si/si2007/uksi_20073072_en_1. Accessed on 09th September 2009.

UK Government (2009). Written Ministerial Statements, Wednesday 28th January 2009, RTFO Levels (Response to Consultation). UK Parliament. Available at http://www.publications.parliament.uk/pa/cm200809/cmhansrd/cm090128/wmstext/90128m0001.htm#column_20WS. Accessed on 09th September 2009.

UK Government (2011). The Renewable Transport Fuel Obligations (Amendment) Order 2011, No. 2937. UK Parliament. Available at http://www.legislation.gov.uk/ukdsi/2011/9780111516966/contents. Accessed on 07th March 2013.

UK Parliament (2006). Eighth Report of the Select Comittee on Environment, Food and Rural Affairs, Published by The Stationary Office, available from www.publications.parliament.uk/pa/cm200506/cmselect/cmenvfru/965/96501.htm .

Unger, N., P. Beigl and G. Wassermann (2004). General requirements for LCA software tools. International Environmental Modelling and Software Society, iEMSs 2004 International Conference University of Osnabrück, Germany.

Upham, P., P. Thornley, J. Tomei and P. Boucher (2009). "Substitutable biodiesel feedstocks for the UK: a review of sustainability issues with reference to the UK RTFO." Journal of Cleaner Production **17, Supplement 1**(0): S37-S45.

Upham, P., J. Tomei and L. Dendler (2011). "Governance and legitimacy aspects of the UK biofuel carbon and sustainability reporting system." Energy Policy **39**(5): 2669-2678.

Ursua, A., L. M. Gandia and P. Sanchis (2012). "Hydrogen Production From Water Electrolysis: Current Status and Future Trends." Proceedings of the IEEE **100**(2): 410-426.

Veziroglu, A. and R. Macario (2011). "Fuel cell vehicles: State of the art with economic and environmental concerns." International Journal of Hydrogen Energy **36**(1): 25-43.

Volkswagen AG (2009). The Passat - Environmental Commendation (Detailed Version), Validated by TUV Nord.

Wallner, T., H. Lohse-Busch, S. Gurski, M. Duoba, W. Thiel, D. Mertin and T. Korn (2008). "Fuel economy and emissions evaluation of BMW hydrogen 7 mono-fuel demonstration vehicles." International Journal of Hydrogen Energy **33**: 7607 - 7618.

Wang, J., Z. Huang, Y. Fang, B. Liu, K. Zeng, H. Miao and D. Jiang (2007). "Combustion behaviours of a direct injection engine operating on various fractions of natural gas - hydrogen blends." International Journal of Hydrogen Energy **32**: 3555 - 3564.

Wang, M. (2002). "Fuel choices for fuel cell vehicles: well-to-wheel energy and emission impacts." Journal of Power Sources **112**: 307 - 321.

Wasteworks Ltd. and AEA (2010). The composition of municipal solid waste in Wales. Published by WRAP. Available to download at http://www.wrapcymru.org.uk/about_wrap_cymru/compositional_report.html. Accessed on 04th August 2010.

Wellinger, A. and A. Lindberg (1999). Upgrading and utilisation of biogas, IEA Bioenergy, Task 24. Available from http://www.iea-biogas.net/Dokumente/Biogas%20upgrading.pdf Accessed on 22 July 2008.

Welsh Assembly Government (2009). Population by place of residence, area size and population density, Welsh Assembly Government. Downloaded from www.statswales.wales.gov.uk/ReportFolders/reportFolders.aspx on 02nd August 2010.

Welsh Assembly Government (2010a). Population by place of residence, area size and population density (003647), STATSWALES. Available at http://www.statswales.wales.gov.uk/ReportFolders/reportFolders.aspx. Accessed on 04th August 2010.

Welsh Assembly Government (2010b). Welsh agricultural statistics 2010: Chapter 1 - Agricultural area, Available at http://wales.gov.uk/topics/statistics/publications/welshagriculturalstatistics2010/?lang=en. Accessed on 13th September 2012.

Welsh Assembly Government (2011). Municipal waste management report 2009/10. Produced by StatsWales. Available at http://wales.gov.uk/topics/statistics/headlines/environment2010/101116/?lang=en . Accessed on 07th September 2012.

Welsh Government (2011). Local Authority municipal waste recycling / composting by material, tonnes, 2009-10 onwards. StatsWales. Available at http://www.statswales.wales.gov.uk/TableViewer/tableView.aspx?ReportId=26737. Accessed on 06th September 2012.

Wilkie, A. C., K. J. Riedesel and J. M. Owens (2000). "Stillage characterisation and anaerobic treatment of ethanol stillage from conventional cellulosic feedstocks." Biomass and Bioenergy **19**: 63-102.

Wind Energy Resources Ltd. (2010). Energy yield from 30kW wind turbine. Available at http://www.wind-energy-resources.com/wer_30kw_wind_turbine.html. Accessed on 09th September 2012.

WRAP (2009a). Anaerobic Digestate Quality Protocol (ADQP). The Environment Agency. Available at http://www.environment-agency.gov.uk/business/topics/waste/114395.aspx. Downloaded on 20th May 2010.

WRAP (2009b). Food waste collection guidance, Waste and Resource Action Programme. Project No. ROT-020-000. Available at http://www.wrap.org.uk/content/local-authority-separate-food-waste-collection-trials. Accessed on 07th September 2012. ISBN 1-84405-419-5.

WRAP (2011). New estimates for household food and drink waste in the UK, Waste and Resource Action Programme. Available at http://www.wrap.org.uk/content/new-estimates-household-food-and-drink-waste-uk. Accessed on 06th September 2012.

WRAP (Northern Ireland) (2011). Northern Ireland Commercial and Industrial (C&I) Waste Estimates, Available at http://www.doeni.gov.uk/niea/waste-home/waste-publications-2.htm. Accessed on 12th September 2012. ISBN 1-84405-456-X.

Zhang, R., H. M. El-Mashad, K. Hartmann, F. Wang, G. Liu, C. Choate and P. Gamble (2007). "Characterization of food waste as a feedstock for anaerobic digestion." Bioresource Technology **98**: 929-935.

Ziegler, J. (2008). Report of the Special Rapporteur on the Right to Food. Promotion and protection of all human rights, civil, political, economic, social and cultural righs, including the right to development.

Zielonka, S., A. Lemmer, H. Oechsner and T. Jungbluth (2010). "Energy balance of a two-phase anaerobic digestion process for energy crops." Engineering in Life Sciences **10**(6): 515 - 519.

APPENDIX A

COPIES OF PUBLISHED MATERIAL

(REMOVED FROM FINAL VERSION TO COMPLY WITH COPYRIGHT CONDITIONS OF ACADEMIC JOURNALS – SEE pg. xv FOR REFERNENCES)

APPENDIX B

SUMMARY OF ASSUMPTIONS & INVENTORIES

Inventory Assumptions for LCA Models (Chapters 5, 6 & 7)

Chapter 5 – LCA of Biogas Infrastructure Options on a Regional Scale		
Functional Unit	275,900	t/yr (of waste treated)
Infrastructure Lifetime	20	Years
Biogas Yield	130	m^3/t
Waste Characteristics		
Total Solids of Waste	27.5	%
Volatile Solids of Waste	80	% (of Total Solids)
Arsenic Concentration in Waste	1.27	mg/kg Dry Matter
Cadmium Concentration in Waste	0.28	mg/kg Dry Matter
Other metals concentrations in Waste	As per Ecoinvent database	
Biogas Plants		
Hydraulic Retention Time	30	Days
Organic Loading Rate	3.9	kg vs^{-1} d^{-1}
Pasteurisation Temp	70	°C (for 1 hour)
Typical Material Inventory (35,000 t/yr plant)		
Concrete	2681.49	m^3
Reinforcing Steel	132.92	t
Construction Steel	139.12	t
Aggregate	3554.78	t
Concrete Blocks	111.47	t
Zinc	10.28	t
Aluminium	5.09	t
Polyisocynurate	18.92	t
Polyester	10.07	t
Polyvinylchloride	9.89	t
CHP Plants		
Electrical Conversion Efficiency	32	%
Thermal Conversion Efficiency	50	%
Mass of CHP Plant	As per Jenbacher data according to output	
Gas Upgrading Plants		
Final methane concentration in gas	96	%
Methane slip	3	%
LPG Addition	0.03705	m^3 LPG / m^3 biomethane
Chapter 6 – LCA of Biohydrogen and Biomethane Production and Utilisation as a Vehicle Fuel		
Functional Unit	1	km passenger vehicle transportation
Pasteurisation Temp (Food waste only)	70	°C (for 1 hour)
Feedstock transport distance	20	km

Digestate transport distance	20	km
Burden of return journey	80	% of outward journey
Food Waste (Two Stage Batch)		
Total Solids of Waste	17.6	%
Volatile Solids of Waste	16.3	%
Organic Loading Rate	20	g VS l^{-1}
Inoculum	3	g VS l^{-1}
Temperature	35 (+/-2)	°C
1st stage pH	5.5	pH (minimum)
1st stage Retention time	1.75	days
1st stage Hydrogen Yield	13.7	l kg^{-1} food waste
VS of 1st stage effluent	1.9	%
2nd stage pH	7.0	pH
2nd stage Retention time	30	days
2nd stage Methane yield	53.11	l kg-^{1} food waste
Food Waste (Single Stage Batch)		
TS content of food waste at input	12	% TS
pH	7.0	pH
Retention Time	30	Days
Methane Yield	62.67	l kg^{-1} food waste
Wheat Feed (Two Stage Semi Continuous)		
Total Solids of Wheat Feed	90.62	%
Volatile Solids of Wheat Feed	86.22	%
Hydration at Pre-treatment	17.22	Litres water / kg feed
Target pH for Alkali Pre-treatment	12	pH
Nutrient Solution Addition	0.048	Litres kg feed
Organic Loading Rate	64	g VS l^{-1}
Temperature	35	°C
Hydraulic Retention Time 1st stage	0.75	days
VS of 1st stage effluent	3.79	% VS
1st stage Hydrogen yield	6.2	l kg^{-1} wheat feed
2nd stage Hydraulic Retention Time	19.25	days
2nd stage Methane yield	310	l kg^{-1} wheat feed
Wheat Feed (Single Stage Semi Continuous)		
Organic Loading Rate	2.5	g VS l^{-1} d^{-1}
Hydraulic Retention Time	20	Days
Temperature	35	°C
Methane Yield	225	l kg^{-1} wheat feed
Typical Material Inventory (30,000 t/yr plant treating wheat feed)		
Concrete	6926.35	m^3
Reinforcing Steel	302.63	t
Construction Steel	78.23	t
Aggregate	7293.87	t

Dense concrete blocks	98.65	t
Rolled sheet steel	22.74	t
Polyisocyanurate	26.78	t
Zinc	6.37	t
Aluminium	21.98	t
HDPE	11.28	t
PVC	24.93	t
Parasitic Energy Provider		
CHP Electrical Conversion Efficiency	32	%
CHP Thermal Conversion Efficiency	50	%
Biogas Boiler Thermal Efficiency	98	%
Gas Upgrading		
Final methane concentration in gas	98	%
Compression pressure	200	Bar
Transport distance to retail point	20	km
Gas Loss during upgrading / compression	1.5	%
Chapter 7 – LCA of Electrolytic Production of Hydrogen and its use as a Vehicle Fuel		
Function Unit	100	Passenger km (pkm)
Primary Energy Source		
Photovoltaic Panel	3kW (peak) single crystal sloping panel (Ecoinvent)	
Solar Irradition	3.1	kWh m^{-2} d^{-1}
Wind Turbine	30 kW wind turbine (Ecoinvent)	
Windspeed (average)	5	m s^{-1}
UK Grid Electricity (2011)	Digest of UK Energy Statistics 2011	
UK Grid Electricity (2030)	Estimated for 2030 using Level 2-3 2050 Pathways Analysis Report	
Electrolyser		
Hydrogen flow rate	40	Standard litres / minute
Output pressure	15	Bar
Power rating	11	kW
Production efficiency	4.8	kWh / Nm^3 (peak)
Material Inventory	Scaled from 2kW PEM FC (Ecoinvent)	
Deionised water input	10	l m^{-3} hydrogen
Waste heat	1.29	kWh m^{-3} hydrogen
Oxygen output (atmosphere)	2.66	kg m^{-3} hydrogen
Unconverted deionised water	1.96	l m^{-3} hydrogen
Hydrogen Compression and Storage		
Compression pressure	400	Bar
Compression energy	3.37	kWh kg^{-1} hydrogen
Gas Loss	1.5	%
Compressor lifetime	15	years

Material Inventory	As per 4kW screw compressor (Ecoinvent)	
Total mass of storage cylinder (empty)	170.5	kg
Assumed mass of aluminium in cylinder	150	kg
Assumed mass of carbon fibre in cylinder	20.5	kg
Acrylonitrile Input per kg carbon fibre	2	kg
Epoxy resin input per kg carbon fibre	0.1	kg
Energy input to manufacture 1 kg carbon fibre	7.56	MJ (medium voltage elec.)
Transport distance to retail point	20	km
Fuel Cell Vehicle		
$LiMn2O_4$ battery mass	40	kg
Electric motors	104	kg
PEM Fuel Cell	456	kg
Fuel Cell lifetime	15	years
Chasis and body mass	800	kg
Fuel consumption	0.0104	kg H_2 km^{-1}
Unconverted hydrogen	0.5	%