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Holistic sustainability assessment of biomethane systems

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Thesis submitted for the degree of Doctor of Philosophy to the National University of Ireland, Cork

Supervisors: Professor Jerry Murphy & Dr. Paul Leahy

Head of School: Professor Liam Marnane

February 2017

“Jestem z tych, którzy wierzą, że nauka jest czymś bardzo pięknym”

“I am among those who think that science has great beauty”

Marie Curie-Skłodowska

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Declaration

I hereby certify that the work I am submitting is my own and has not been submitted for another degree, either at University College Cork or elsewhere. All external references and sources are clearly acknowledged and identified within the contents. I have read and understood the regulations of University College Cork concerning plagiarism.

Signature

Date: 13 February 2017

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I am finishing one project and I am embarking on another, very special one. Next March I am not going to be on my own for my PhD viva; and if some days my little one follows my footsteps, I can tell him that he already went through a PhD defence once, with me.

And to all that I might have forgotten: "*Before I go, I just want to tell you: you were fantastic. Absolutely fantastic! And you know what? So was I"*. The Doctor

Abstract

European states, including Ireland must ensure that an increasing portion of energy produced and consumed comes from renewable sources. This is a particular issue for transport, which in comparison to electricity and heat has very low levels of renewable penetration. Electric vehicles (EVs), liquid and gaseous biofuels are the most likely sources for future energy in transport. However, renewable does not automatically mean sustainable. For example the sustainability of biofuels sourced from food crops has been queried in the context of land use change emissions. This thesis has an ambition of assessing sustainable options for advanced biomethane production in Ireland, a country with a temperate oceanic climate, using various methodologies (life cycle assessment, energy system modelling and cost analysis). Biomethane is a versatile gaseous biofuel that is considered advanced when produced from second and third generation feedstocks such as wastes and residues, grasses, and seaweed, but a simplified and unified framework for biofuels LCA is required to compare different options. Under a low-level land use change emissions scenario, biomethane from grass could play a major role in the Irish energy system for transport in 2050, requiring only 5-11% of Ireland's agricultural land. With high land use emissions, however, the model would suggest using hydrogen, residues-based biodiesel, and EVs. Biomethane from seaweed could be deemed unsustainable if the system is not optimised. However in an optimal configuration it could achieve 70% greenhouse gases (GHG) savings as compared to gasoline. Such reductions in GHG emissions can be achieved in an optimal system: integrating seaweed cultivation with fish farming; using innovative growing techniques; ensuring optimal seaweed composition; reusing digestate; and using renewable electricity to power plant operations. Biomethane from landfill gas was shown to require a subsidy to allow financial sustainability. Thus in conclusion, biomethane can be a sustainable transport biofuel, but requires system optimisation and state subsidies.

Thesis output

Chapters published as papers or currently under review in peer-reviewed journals:

Chapter 3: Czyrnek-Delêtre MM, Smyth BM, Murphy JD, 2017. Beyond carbon and energy: The challenge in setting guidelines for life cycle assessment of biofuel systems, *Renewable Energy* 105: 436–448. doi:10.1016/j.renene.2016.11.043

Chapter 4: Czyrnek-Delêtre MM, Chiodi A, Murphy JD, Ó Gallachóir BP, 2016. Impact of including land use change emissions from biofuels on GHG emissions reduction targets - the example of Ireland. *Clean Technologies* and *Environmental Policy* 18: 1745-1758. doi:10.1007/s10098-016-1145-8

Chapter 5: Czyrnek-Delêtre MM, Ahern EP, Murphy JD, 2016. Is small-scale upgrading of landfill gas to biomethane for use as a cellulosic transport biofuel economically viable? *Biofuels, Bioproducts and Biorefining* 10: 139–149. doi:10.1002/bbb.1627

Chapter 6: Czyrnek-Delêtre MM, Rocca S, Agostini A, Giuntoli J, Murphy JD, 2017. Life cycle assessment of seaweed biomethane, generated from seaweed sourced from integrated multi-trophic aquaculture in temperate oceanic climates. *Applied Energy* 196: 34-50. doi:10.1016/j.apenergy.2017.03.129

Chapters presented on international conferences:

10th Conference on Sustainable Development of Energy, Water and Environment Systems, 27 September – 3 October 2015, Dubrovnik, Croatia. Presentation: *Impact of including land use change emissions from biofuels on GHG emissions reduction targets – the example of Ireland.*

24th European Biomass Conference and Exhibition, 6-9 June 2016, Amsterdam, the Netherlands Presentation: *Life Cycle Assessment study of integrated marine algae and salmon farming for biomethane production in Ireland.*

Invited talks:

Eco-design and LCA invited lectures to MSc students in Climate Change, Agriculture and Food Security. *Life cycle analysis of agricultural and food processes, and biofuel systems including direct and indirect land use changes.* 10-11 November 2014, National University of Ireland, Galway, Ireland.

Awards:

Awardee of the Climate KIC Pioneer into Practice programme, 2015, Climate KIC Innovation Birmingham, England.

Contribution to the papers

Chapter 3: I was the first author of the paper and was responsible for research design, data collection and writing.

Chapter 4: I was the first author of the paper jointly with Alessandro Chiodi. I was responsible for collection of data, and majority of writing and analysing of the results. The modelling in TIMES was conducted by my colleague Alessandro Chiodi.

Chapter 5: I was the first author of the paper jointly with Eoin Ahern. I was responsible for data collection and contacting the industry representatives. I made a significant contribution to the research design and data analysis together with the joint first author on this paper. I was responsible for the majority of the writing.

Chapter 6: I was the first author of the paper, and was responsible for research design and data collection that involved contacting the industry and academics working on salmon and seaweed farming, agriculture scientists and LCA experts. I was also in charge of the entire LCA modelling in GaBi software, results analyses and writing.

List of acronyms

AD: anaerobic digestion

AwR: alkaline with regeneration

BIM: Bord Iascaigh Mhara (Ireland's Seafood Development Agency)

BLUM: Brazilian Land Use Model

BMP: biomethane potential assay

BOC: Biofuel Obligation Certificate

CAPEX: capital costs

CBG: compressed biogas (biomethane)

CCS: carbon capture and storage

CH_4: methane gas

CHP: combined heat and power

CO_2: carbon dioxide gas

CNG: compressed natural gas

CSTR: continuously stirred tank reactor

DS: dry solids (or dry matter)

DLUC: direct land use changes

EAW: equivalent annual worth

EC: European Commission

EPA: Environmental Protection Agency

ETS: EU Emissions Trading Scheme

EU: European Union

EVs: electric vehicles

FU: functional unit

GHG: Greenhouse gases

GNI: Gas Network Ireland

GTAP: Global Trade Analysis Project

GWP: Global Warming Potential

ha: hectare

IEA: International Energy Agency

IEA-ETSAP: IEA Energy Technology Systems Analysis Program

ILUC: indirect land use changes

IMTA: integrated multi-trophic aquaculture

ktoe: thousand tonnes of oil equivalent

LCA: life cycle assessment

LFG: landfill gas

LFGE: LFG-to-energy

LUC: indirect land use changes

MSW: municipal solid waste

OFMSW: organic fraction of municipal solid waste

OPEX: operation and maintenance costs

pers. comm: personal communication

PET: Pan European TIMES

PSA: pressure swing adsorption

RED: Renewable Energy Directive (2009/28/EC)

RES: renewable energy sources

RES-E: renewable energy in electricity

RES-H: renewable energy in heating

RES-T: renewable energy in transport

RNG: Renewable Natural Gas

SEAI*:* Sustainable Energy Authority of Ireland

SMY: specific methane yield

TAN: total ammonical nitrogen

TIMES: The Integrated Markal-Efom System

TPER: total primary energy requirement

TPs: technology processes

UCO: used cooking oil

VOCs: volatile organic compounds

VPSA: vacuum pressure swing adsorption

VS: volatile solids

VSA: vacuum swing adsorption

wwt: wet weight

WWT: wastewater treatment

WEC: World Energy Council

Chapter 1. Introduction

1.1. Background

1.1.1. Climate and renewable energy in the EU policies

By 2050 global total primary energy consumption will increase by between 27% and 61% as compared to 2010, according to estimates of the World Energy Council (WEC 2013). The European Union (EU) plays a leadership role in global renewable energy development (EC 2016) with targets of 20% renewable energy in gross energy consumption by 2020 (EC 2009a), and 27% by 2030 (EC 2014a). In terms of decarbonisation policy, the EU proposed very challenging greenhouse gases (GHG) emissions targets aiming at a 20% GHG reduction by 2020 (EC 2009b), 40% by 2030 (EC 2014a), and between 80% and 95% by 2050 (EC 2011), relative to 1990 emission levels. Also, 10% of energy in transport must be generated from renewable sources by 2020 (EC 2009a).

In transport, the deployment of renewables is slower than in the electricity, heating and cooling sectors, and only reached 5.4% of total energy in EU transport in 2013 (EC 2015a). The "silver bullet" that could replace the fossil fuels in transport does not exist. Instead there are many technologies that will play a smaller or bigger role in the transport fuel future; these might vary between the countries, depending on available resources and created opportunities. Some of these solutions are liquid biofuels (bioethanol and biodiesel), some are gaseous (biomethane and hydrogen); electric vehicles (EVs) and hybrid cars using renewable electricity such as from solar or wind are also mooted (Murphy and Thamsiriroj 2011). Hybrid cars can also switch to gasoline if needed. While EVs will have a significant part in a green energy future, it is crucial to assure that all electricity for EVs is renewable, and this is a challenge. According to calculations performed for Ireland, by 2020 green energy in EVs will not exceed 2% of total energy in transport (Murphy and Thamsiriroj 2011). Another issue is the intermittent character of renewable sources of electricity such as wind and solar. Liquid and gaseous biofuels have a significant advantage over the present generation of EVs as they can be used in existing engines, can be produced on demand, can be stored and distributed in a similar way to the fossil fuels they directly replace;

also the vehicle technology allows long range of travel. Thus, these biofuels will have an important contribution towards the renewable energy in transport target.

1.1.2. Land use change emissions and biofuels

The sustainability of biofuels was questioned in the late 2000s due to concerns regarding the food vs fuel competition, and the emissions from direct and indirect land use changes (DLUC and ILUC) (Searchinger et al. 2008). DLUC occurs when a land use is converted to produce biofuels; ILUC is observed when biofuel feedstocks displace food or feed production on land previously not cultivated. According to some studies the impact from DLUC and ILUC could be so great that it would offset the savings from biofuels replacing the fossil fuels (Fargione et al. 2008; Searchinger et al. 2008). While incorporation of DLUC in the life cycle assessment seems to be more straightforward (Börjesson and Tufvesson 2011), and it is contained in the Renewable Energy Directive GHG calculation (based on carbon stock changes) (EC 2009a), the calculation of ILUC is much more controversial, and it might be difficult to include due to the methodological and scientific uncertainties, and complexity of the subject (Börjesson and Tufvesson 2011).

The food vs fuel dilemma applies typically to first generation biofuels, to biofuels from feedstocks produced on land that can be also consumed as food or feed for animals (such as corn, sugarcane, sugar beet and wheat). Second generation biofuels feedstocks do not compete directly with food or feed for land; these might be cultivated on land (non-food crop such as grasses, willow or Miscanthus) or be sourced from wastes and residues (food waste, agricultural and forestry wastes and residues). Third generation biofuels do not compete with food or feed for land. Typically these are algae based, both microalgae and seaweed, or may be gaseous fuel from non-biological origin such as from power to gas systems. The second and third generation biofuels are also termed advanced biofuels, as they have low carbon emissions on a whole life cycle basis, and should have low or zero ILUC emissions.

1.1.3. Life cycle assessment for evaluation of biofuels

Life cycle assessment (LCA) is deemed to be the most suitable tool for assessment of biofuels. LCA involves an evaluation of environmental impacts from a product or service life cycle. For biofuels, LCA might look only at the GHG or energy as

indicators, but a full LCA considering several environmental impact categories creates a much better picture of the overall impacts and benefits of biofuel, thus allows for more informed decisions regarding its production and use. LCA has been used extensively in assessment and comparison of biofuel systems with other biofuel systems or with the fossil fuel displaced. However, a challenge that many authors report is a lack of common and recognized guidelines for biofuel LCA that leads to discrepancies in the results, and does not allow for sound comparison between the studies (Gnansounou et al. 2009; Cherubini and Strømman 2011; Benoist et al. 2012; Brandão et al. 2012; Thomassen et al. 2015).

There are two main approaches in the LCA modelling: attributional and consequential. Attributional modelling can be termed 'descriptive' and 'retrospective'; a system is modelled as it is or was or is forecasted to be, including upstream supply-chain, and downstream use and end of life. Attributional modelling calls for historical and measurable data of known uncertainty (EC JRC 2010a). Contrary to this, consequential modelling is 'change-oriented' and 'market-based'. Consequential model does not reflect the actual specific or average supply-chain but depicts the generic supply-chain as it is theoretically expected in consequence of the analysed decision (EC JRC 2010a). This is the case of indirect land use change modelling, when the consequences of growing energy crop on other sectors of economy are forecasted (chapter 4). The attributional approach was applied in LCA of seaweed biomethane (chapter 6).

1.1.4. Biomethane

Biomethane is biogas cleaned and upgraded to a natural gas standard with high methane content (above 97% CH_4) (Thamsiriroj et al. 2011). Biogas is produced via anaerobic digestion (AD) using a range of feedstocks such as wastes and residues (from food and agriculture), grasses and seaweeds. Biomethane can be also produced from landfill gas collected at landfill sites (Czyrnek-Delêtre et al. 2016a). It can be used as transport fuel (compressed biogas abbreviated as CBG), as well as for electricity and heat generation. A huge advantage of biomethane is that it can be produced and stored until demand requires it, unlike intermittent sources such as solar or wind energy. Once produced the gas can be effectively stored for any period of time, in for example the natural gas grid (Murphy and Thamsiriroj 2011). A large array

of wet organic feedstocks can be used in the AD processes, which makes biomethane a very versatile fuel. This biomethane can be generated in countries without a significant area of arable land, such as Ireland whose agricultural land is dominated by grasses. Furthermore, it can be used as part of a waste management strategy; organic wastes such as agricultural slurry and manure, slaughter waste and organic fraction of municipal solid waste (OFMSW) are suitable substrates for biomethane production. Biomethane can be produced in small-scale plants such as on-farm or in municipal plants, and be used directly on-site satisfying farms or municipalities demand for renewable transport fuel, electricity and heat. All these benefits make biomethane a key element in green transport expansion (Thamsiriroj et al. 2011). To date, six European countries, Denmark, Sweden, Belgium, The Netherlands, France and Switzerland have committed to 100% CO_2-neutral green gas in the natural gas grid by 2050 (ONTRAS 2015). This will be satisfied in majority by biogas/biomethane from second and third generation feedstocks including manures and wastes, sewage sludge, grass, seaweed and landfill gas (Green Gas Forum 2014). In Ireland, Gas Network Ireland (GNI) proposed 5% and 20% targets of green gas in the Irish gas grid by 2020 and 2030, respectively (GNI 2015).

1.1.5. Ireland

Ireland is an island in the North Atlantic with a temperate oceanic climate. Ireland is an EU member state with the highest percentage (almost 70%) of agricultural land (CSO 2015), and grassland representing over 90% of the country's agricultural land (McEniry et al. 2013). Since the 1990s Ireland has shown an annual increase in energy use, with at present an 85% dependency on energy imports. Almost 90% of renewables were produced in Ireland (mainly from wind); however only 21% of biofuels were produced indigenously in 2014 (Dineen et al. 2016). The majority of renewable energy investments are in wind power. In terms of both climate mitigation and renewables in transport targets, Ireland has still a lot of ground to cover with renewable transport reaching 5.2% in 2014 (Howley et al. 2015a). With regard to the domestic resources for biomethane production, Ireland has significant potential in grasses and slurry (Wall et al. 2013; O'Shea et al. 2017), food waste (O'Shea et al. 2016) as well as marine algae species that can be cultivated in temperate oceanic climate (Tabassum et al. 2017).

1.2. Thesis aims and objectives

The main aims of this thesis were 1) to assess the sustainable solutions for advanced indigenous biomethane systems in Ireland, a country with a temperate oceanic climate; and 2) to provide the policymakers with a broader perspective on sustainable options for biomethane.

The specific thesis objectives were to:

- Discuss the existing frameworks providing guidelines for LCA of biofuels, and compare various approaches used in biofuel LCA in terms of functional unit, system boundaries, reference scenario, allocation methodology and impact categories (chapter 3);
- Make recommendations for improving the robustness and accuracy of the biofuel LCA framework (chapter 3);
- Assess direct and indirect land use changes emissions factors for bioenergy sources both domestic and imported for use in Ireland (chapter 4);
- Assess the implications of introducing land use change emissions on achieving Ireland's emissions and renewable energy targets, and identify key optimal biofuel sources (chapter 4).
- Examine the potential of landfill gas upgrading to biomethane for use as a transport fuel on existing landfill sites in Ireland (chapter 5);
- Analyse the costs of landfill gas upgrading and recommend the optimal technology (chapter 5);
- Generate a detailed LCA model of a seaweed biomethane system with integrated seaweed and salmon farming for biomethane production in Ireland (chapter 6);
- Identify the critical environmental impacts and benefits of the integrated seaweed and salmon farming for biomethane production (chapter 6);
- Assess the implication of several parameters on the overall LCA result of seaweed biomethane (such as seasonal variation in seaweed and increase in yield of seaweed per hectare, as well as using digestate to replace mineral fertilisers) (chapter 6);
- Identify ways of addressing and minimising the impacts, and maximising the sustainability of seaweed biomethane (chapter 6);

1.3. Methodology

The thesis was desktop-based. Collected data come from literature sources (both peer reviewed papers and reports), site visits, discussion and collaboration with industry representatives, field experts and policymakers. A number of techniques were used in the assessment process; these include life cycle approach for calculating land use change emissions and TIMES modelling to generate future energy system pathways, simple economic analysis with costs and incomes per m_n^3 of biomethane over a 15 year life time of the project, equivalent annual cost method, and a full life cycle assessment (LCA). In this thesis, when referring to the full LCA, it is understood that a range of environmental impact categories beyond climate change were assessed. The methodology specific for each chapter is contained in the second section of each chapter.

1.4. Thesis in brief

The thesis is comprised of seven chapters and two appendices. The common theme is the assessment of gaseous biofuels from first, second and third generation feedstocks using various methods including costs, environmental impacts and land use change emissions. The major focus is biogas/ biomethane produced through anaerobic digestion. This is a so called publication-based thesis. Chapter 2 presents the rationale for this thesis, explains the methods behind each chapter, provides a road map for the thesis and highlights links between the chapters. Chapters 3, 4, 5 and 6 are written for publication as academic peer review journal papers. Each chapter/paper comprises its own introduction, methodology, results and discussion and can be read independently, or alternatively may be read in sequence to provide a thesis (or argument) on the sustainability analyses of biomethane systems ranging from first to third generation. Chapters 3, 4, 5 and 6 are published as peer-reviewed journal papers and appear in the thesis as a word version of the published manuscripts with some minor modifications.

Chapter 2. Rationale for thesis

The thesis presents an applied approach to research, and involved extensive consultations with both industry (chapters 5 and 6) and policymakers (chapter 4 and 6). The focus of the thesis was on the methods for the assessment of gaseous biofuels (in particular on LCA), and on second and third generation biomethane. Typically renewable electricity production, such as from wind turbines and photovoltaic cells where the fuel is free and abundant (wind and sun), is automatically considered sustainable. This is not the case for biofuels as the fuel is typically a substrate, such as a residue (with associated energy input in collection) or a crop (which has an economic value and energy input in production). Thus all biofuels are renewable but not all can be labelled sustainable. According to the Renewable Energy Directive (RED) and its amendments (EC 2009a; EC 2015b), as of 2017 biofuels must demonstrate at least 60% GHG savings as compared to fossil fuel replaced on a whole life cycle analysis basis. These savings must rise to 70% beyond 2020 (EC 2016). There is a perspective abroad that first generation biofuel systems, such as from grains, which employ circular economy concepts (such as capture and reuse of CO_2 and use of stillage in biogas facilities) can be extremely effective in reducing greenhouse gas emissions (Murphy and Power 2008). However in the proposed amendments to the RED the methodology for dealing with land use change was to cap first generation biofuels, from sugary crops (sugar cane and sugar beet), starchy crops (maize and cereals) and oils (rapeseed and palm oil), at 7% of energy in transport. It is not yet definite that advanced biofuels (such as from algae) will automatically be sustainable.

The foregoing is an important challenge for EU Member states that have to comply with these obligations and targets when working towards the national targets. As such the aim of this thesis was set to assess the sustainability of possible solutions for biomethane and deliver a clear message to the policy makers.

Life cycle assessment (LCA) is deemed to be the most suitable tool for the environmental assessment of biofuels. The thesis starts with an evaluation of existing frameworks for LCA of biofuels. This was felt to be necessary since currently the only common guidelines for the assessment of biofuels are introduced in the RED (EC 2009a), the strengths and weakness of which are described in details in chapter 3. LCA

studies are not comparable if they do not follow unified guidelines for full LCA. In this thesis full LCA is understood as a study that includes impact assessment beyond carbon balance. In industry, there are Product Category Rules (PCR) offering a set of industry-specific rules and requirements that must be fulfilled for products/ services to be compared in terms of LCA (Subramanian et al. 2012). LCA results are published in the form of an Environmental Product Declaration (EPD), which is transparent and verified by an external party document (Fet et al. 2009).This is lacking in the environmental assessment of biofuels, as there are many approaches to LCAs and published results are not unified. Therefore, the first stage of this thesis involved screening of existing LCA studies, and providing a set of recommendations on how the guidance for biofuels LCA can be improved. The results of this work were used to conduct a full LCA of seaweed biomethane in chapter 6.

In chapter 3, the recommendations are made on impact categories that should be included in the LCA of biofuels; land use indicator is one of those. This was explored in further in chapter 4. The emissions associated with land use changes are of particular importance for land-based biofuels, as they may affect greatly the overall sustainability of the biofuel. While, calculating emissions from direct land use change (DLUC) is readily achievable, the causes behind indirect land use change (ILUC) are far more complex and interlinked to allow for a straightforward assessment of ILUC impact. However, low emissions from ILUC need to be ensured, and biofuels with low ILUC risk should be promoted. After the RED was published in 2009 (EC 2009a), the policymakers in EU sought to improve the legislation by including the ILUC emissions in biofuels assessment. In 2015, an amendment to the RED was published setting a cap on the first generation biofuels that indicates that those shall not exceed 7% out of the EU 10% renewable transport target (EC 2015b). Up to this, DLUC and ILUC were not considered when modelling the national energy systems. One of the objectives of this thesis was to estimate the impact of both DLUC and ILUC from biofuels on meeting the GHG and renewable energy targets beyond 2020. First generation feedstocks (such as wheat, rapeseed, corn sugar beet) and second generation feedstocks (such as wastes, residues, including recycled oil, landfill gas and grass), both imported and domestic were included in the study.

Biomass feedstocks with zero or low risk of land use change include for third generation substrates such as seaweed (considered in Chapter 6) and microalgae, but

also for what may be the most sustainable feedstocks, wastes and residues. These waste biofuel systems include for landfill gas (LFG), which is considered in Chapter 5, and recycled oil. It may be argued that grass from permanent grasslands has a minimum impact on land use change if there is no reduction or impact on beef or dairy output associated with the grass land. Some of the first generation feedstocks might have also a low land use impact under certain conditions (such as cultivation on fallow/marginal land and increase in yield per hectare). For the purpose of this thesis two very different biofuels were assessed in more detail: LFG upgraded to biomethane (chapter 5) and seaweed biomethane (chapter 6). LFG is already produced but not in a form useable by natural gas vehicles (NGV). Seaweed biomethane is an advanced biofuel, so far not produced on an industrial scale, but is of great interest as a potential fuel of the future with zero ILUC risk.

LFG is a gas produced from an existing waste infrastructure and has a composition very similar to biogas but with higher level of impurities and contaminants. LFG is considered a second generation biofuel when upgraded to biomethane, and has a potential to be a viable source of transport fuel. It is a readily accessible gas, which presently is either flared on landfill sites or converted to electricity. Chapter 5 is focussed on assessing the financial viability of small-scale upgrading of LFG in Ireland.

Seaweed is an example of a third generation biofuel feedstock native to Ireland; this is the ‘dream biofuel’, however its sustainability must be ensured through an optimisation of the system. Biomethane from seaweed can be counted at twice its energy content to renewable energy in transport targets, whilst playing a role as an indigenous transport fuel in Ireland. While LCA of microalgae are well published in scientific literature, the studies on LCA of seaweed (macroalgae or marine algae) are scarce. Also, there was no study so far that considered the sustainability of seaweed biomethane produced in the integrated multi-trophic aquaculture system, where seaweed is farmed adjacent to a salmon farm. The paper documented in chapter 6 is an outcome of an extensive 2 years long project. The methodological choices were based on recommendations developed earlier and presented in the chapter 3.

Each chapter is synthesised below including 1) a paragraph on rationale for the study, 2) a short summary of the paper and 3) a paragraph presenting the reason behind the study, the collaborations and undertaken actions.

2.1. Beyond carbon and energy: the challenge in setting guidelines for life cycle assessment of biofuel systems (chapter 3)

As there are no binding guidelines for LCA of biofuels, except the RED directive, a need was identified for a paper that proposes a framework for LCA conducted in this thesis. Therefore, an extensive review of peer-reviewed papers on LCA of 1) biogas/biomethane, 2) bioethanol and 3) biodiesel was conducted. Additionally, the guidelines as described in various existing LCA frameworks were compared.

Based on this data, recommendations were proposed on how to improve the robustness of LCA across the biofuel systems. A simple flowchart was proposed to facilitate the choice of assumptions by an LCA analyst. When conducting LCA the following should be clearly described: functional unit; system boundaries; reference scenario; and allocation and impact categories. If relevant, more than one functional unit should be applied. System boundaries should be expanded, sensitivity scenarios included, and to avoid burden shifting it is recommended to assess several impact categories. It is proposed that these should include at least global warming, acidification and eutrophication potentials, as well as land use and an energy indicator. This study was conducted in in collaboration with Queens University Belfast.

2.2. Impacts of including land use change emissions from biofuels on meeting GHG emissions reduction targets – the example of Ireland (chapter 4)

A huge drawback of first generation (food crops from agricultural land-based) biofuels are the emissions from direct and indirect land use change. As outlined in chapter 3 of this thesis, land use change (LUC) emissions are crucial in assessing the sustainability of biofuels. If accounted for, these increase substantially the impact of food crop-based biofuels. So far, the literature does not include the land use emissions in the TIMES modelling of the capability of a system to meet climate mitigation targets. This study

is the first of its kind and investigates the impact of land use change emissions from bioenergy on the entire Irish energy system as an example.

The approach applied in the study was based on life cycle assessment. An extensive list of biofuels and bioenergy feedstocks was created for both imported and domestic fuels. Values for DLUC were calculated using the BioGrace model (BioGrace 2015) or were based on literature values. ILUC values were much more difficult to estimate, and therefore for sensitivity analysis, two scenarios were tested: optimistic with low ILUC and conservative with high ILUC levels. The scenario analysis was conducted using the Irish TIMES energy systems model (Ó Gallachóir et al. 2012). Overall, the results showed that if LUC are included, there is a general shift towards zero-LUC feedstocks such as biofuels sourced from waste and residues. By 2050, in the optimistic scenario, biofuels come mainly from grass (to produce biomethane) and wastes; in the conservative scenario the majority of biofuels are sourced from wastes and residues. Also, the cost of abating the last tonne of CO_2 increases by up to 61% in ILUC conservative scenario as compared to the baseline without LUC.

This project was an outcome of close collaboration with the Energy Policy and Modelling Group (www.ucc.ie/en/energypolicy/) within MaREI (www.marei.ie). The study required an expertise in both life cycle assessment and in TIMES energy system modelling. The latter was provided by the experts in TIMES.

2.3. Small-scale upgrading of landfill gas to biomethane for use as a cellulosic transport biofuel (chapter 5)

With energy content between 11 and 23 MJ/m_n^3 (Kaparaju and Rintala 2013), landfill gas (LFG) can be a viable energy vector for either electricity generation or transport fuel when upgraded to biomethane. Landfill sites are still a major source of greenhouse gas emissions, if gas is not collected; methane, the main component of LFG has a global warming potential over 100 years 30 times higher than CO_2 (IPCC 2014), and therefore the capture of LFG is crucial to reduce the impact on climate change. Additionally, as gas produced from waste, LFG when upgraded to biomethane is considered a second generation biofuel, and can be an interesting solution for an indigenous transport fuel. In Ireland, LFG is eligible to receive Biofuel Obligation

Certificates (BOC), which provides additional revenue for gaseous transport biofuels projects (www.nora.ie/biofuels-obligation-scheme.141.html).

Four landfill sites in Ireland were assessed based on technical information from an industry partner. Cost analyses were conducted after a thorough data collection and several discussions with three commercial technology owners. The cost per m_n^3 of upgraded CH_4 was calculated for each technology process over a 15 year period of operation. Each project was also assessed for profitability using the equivalent annual cost method. The results showed that for LFG upgrading to biomethane to be financially viable, there is a need for a national support scheme, for example in a form of a biofuel certificate scheme.

The idea for the project came from an industry partner who was interested in exploring the potential of using the LFG produced on existing landfill sites in Ireland. The process of collecting data involved contacting industry representatives to enquire about the optimal methods for LFG cleaning and upgrading. Three out of seven agreed to share their commercial data but requested to stay anonymous. LFG is a particularly difficult gas as it contains contaminants such as halogenated compounds, H_2S and siloxanes that must be removed prior to the upgrading process. Moreover in the case presented in this thesis, the levels of nitrogen and oxygen in LFG were very high, 15-30% in case of nitrogen and 1-3% for oxygen. This was particularly challenging for technology providers, and made the upgrading process more complicated and therefore expensive. The best scenario was found to be an individual upgrading facility at each landfill site with an on-site gas service station. In essence a subsidy of at least €$0.55/m_n^3$ is required.

2.4. Life cycle assessment of integrated seaweed and salmon farming systems for biomethane production in temperate oceanic climates (chapter 6)

Seaweed biomethane is deemed a third generation biofuel as seaweed farming does not compete directly or indirectly for land with food or feed production. Seaweed farming takes place at sea, and as such has an additional advantage, which is an ability to clean the seawater from nutrient-rich fish excrement whilst simultaneously enhancing seaweed growth.

A full life cycle assessment of seaweed biomethane was conducted following the recommendations outlined in the chapter 3. This included not only global warming potential but also other environmental impacts such as: acidification; marine; terrestrial and freshwater eutrophication. The modelling was conducted using GaBi LCA software and GaBi Professional database (thinkstep 2016). The system boundaries comprised of an integrated salmon and seaweed (*Laminaria digitata*) farm, the biogas plant and the biogas upgrading unit. Additionally, the combustion of upgraded biomethane in a car engine was included when comparing seaweed biomethane with fossil transport fuel. As proposed in the recommendation for LCA of biofuels (chapter 3 of this thesis), two functional units were considered: 1) MJ of biomethane (energy content) in a water to tank analysis, and 2) km driven in a car in a water to wheel analysis. System boundaries were expanded to include the digestate replacement of mineral fertiliser. Several sensitivities scenarios were considered including: 1) seasonal variation and characteristics of *L. digitata*; 2) yield of seaweed per hectare; 3) increase of yield in seaweed due to the uptake of salmon excrements rich in nitrogen; 4) waste water treatment in the seaweed hatchery; and 5) electricity grid mix considered in LCA. Additionally, two scenarios that combined the most optimal production practices were assessed.

The project lasted two years and was a huge undertaking as it involved a detailed life cycle inventory for a novel biofuel system with integrated salmon and seaweed farming. The project was developed in close collaboration with colleagues in the Joint Research Centre (JRC) Sustainable Transport Unit in Petten, the Netherlands. This necessitated visits to JRC over a two year period. Data was collected and systems were discussed with: 1) academics working on the integrated multi-trophic aquaculture (pers. comm. with Dr. Thierry Chopin, University of New Brunswick, Canada and Dr Gregor K. Reid, Fisheries and Oceans, Canada); 2) consultants and academics in seaweed farming, including field trips to a seaweed farm in Ireland (pers. comm. Dr Maeve Edwards, Irish Seaweed Consultancy Ltd., and Lars Brunner, Scottish Association for Marine Science); and 3) agricultural scientists on mineral fertilisers use (pers. comm. Dr. Pádraig O'Kiely, Teagasc). Data on ensiling and anaerobic digestion of seaweed including the impact of seasonal variation were collected in close consultation with researchers assessing algal biofuels in the laboratory with researchers in the Bioenergy research area in MaREI. The LCA model in GaBi was

discussed with colleagues in JRC, as were the emissions from storage and application of digestate.

The results showed that an optimal seaweed biomethane system provides over 60% savings in global warming as compared to fossil fuels. However in terms of acidification, as well as terrestrial and freshwater eutrophication, seaweed biomethane can be worse than fossil fuels. Digestate handling is deemed responsible for 11% of global warming and over 80% of other environmental impacts.

Chapter 3. Beyond carbon and energy: the challenge in setting guidelines for life cycle assessment of biofuel systems

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Abstract

Life cycle assessment (LCA) is one of the most suitable tools for a uniform assessment methodology of biofuels' sustainability. However, there are no binding guidelines for LCA of biofuel systems. Published LCAs use a range of methodologies, different system boundaries, impact categories and functional units, various allocation approaches, and assumptions regarding by- and co-products, as well as different reference systems to which the biofuel system is compared. The European Renewable Energy Directive and the US Renewable Fuel Standard focus on greenhouse gas (GHG) emissions. However, previous LCAs of biofuel systems have shown that a reduction of GHG emissions does not lead automatically to a decrease in other environmental impacts, and might in fact be associated with an increase in impacts such as acidification, eutrophication, and land use change. In order to enable effective comparison of biofuel systems, the authors propose a framework for biofuel LCA. System boundaries should be expanded to include the life cycle of by- and co-products. Results should be reported using more than one functional unit. Burden shifting can be avoided by considering an array of impact categories including global warming potential and energy balance, along with eutrophication and acidification potential, and a land use indicator.

Keywords: life cycle assessment; sustainability guidelines; system boundaries, impact categories, biofuels.

3.1. Introduction

3.1.1. Background

Total world energy consumption is predicted to increase by 56% between 2010 and 2040 (U.S. Energy Information Agency 2013). The EU has set an overall mandatory target of 20% share of renewable energy in gross domestic consumption, and a target of 10% share of renewables in transport (RES-T), by 2020 (EC 2009a).The US Energy Independence and Security Act of 2007 requires 18% of transport fuel consumption to come from renewables by 2022 (Yacobucci and Bracmort 2009). The use of biofuels, biogas or bioliquids as a sustainable and efficient energy source has been explored for decades (Tyner 2008; Songstad et al. 2011). For some time, especially in the context of peak oil, biofuels were considered a reliable renewable energy source, but their overall sustainability has been questioned in recent years (Rathmann et al. 2010), leading to increased focus on methods for its evaluation. Life cycle assessment (LCA) is one of the most promising approaches for evaluating the sustainability of biofuels (Jensen et al. 1997) and for comparing biofuel systems. LCA is a multistage approach, which covers the full life cycle of a product. While LCA is an important tool for understanding biofuel systems, and supporting decision making, LCA procedures could be considerably strengthened in a number of key areas (Brandão et al. 2012). These include: i) methodology with goal and scope definition, ii) data collection based on real-life rather than lab scale facilities, and iii) multi-criteriality of LCA that considers all environmental impacts equally. These challenges are described in more detail below.

Goal and scope of the study, like functional unit, are often not clearly defined; sensitivity analyses are rarely performed (Reinhardt 2015). Thomassen et al. stated that there is a lack of a generic framework for sustainability assessment of algal biofuels (Thomassen et al. 2015). LCA studies of the same biofuel by different authors can give significantly different results, which can lead to the introduction of contradictory policies (Benoist et al. 2012). Such discrepancies are largely due to different methodological approaches and, although the need for a standardised methodology has been identified, there are as yet no guidelines for achieving this (Gnansounou et al. 2009; Cherubini and Strømman 2011; Benoist et al. 2012; Brandão et al. 2012; Thomassen et al. 2015). Moreover, many LCA studies do not specify the

methodological choices made, which makes it almost impossible to replicate the study (van der Voet et al. 2010). At the same time ISO (International Standard Organization) LCA norms require a transparency in documenting the assumptions made for LCA (ISO 2006a; ISO 2006b). LCA addresses the life cycle of produced and consumed product (ISO 2006b), and is largely used by industry to assess a real-life product system. Advanced biofuels are not yet deployed on a large scale and industry data might not be available. Therefore, often LCA is based on laboratory data which makes the results much less accurate (Raman et al. 2015; Thomassen et al. 2015).

A further weakness is that most existing LCA studies have focused only on energy and/or greenhouse gas (GHG) emissions. Reinhardt pointed out that majority of studies called LCA are in fact only GHG emissions and energy balances (Reinhardt 2015). Lazarevic and Martin analysing LCA and sustainability studies in Sweden found that GHG emissions dominate other environmental impacts (Lazarevic and Martin 2016). Ridley et al. assessed more than 1600 peer-reviewed papers on biofuels and found that the most discussed topics are production technologies, GHG emissions and agricultural production of substrates, whereas the impact of biofuels on biodiversity and human health was much less investigated (Ridley et al. 2012). This is in line with Raman et al. who argue that impact on human health and resources are understudied, when it comes to biofuel sustainability assessment (Raman et al. 2015). Also, water, land use and land use change are seldom found in the literature (van der Voet et al. 2010). GHG emissions and Global Warming Potential (GWP) are very important for LCA studies (EC 2009a). However, there are concerns that, although the use of biofuels may decrease GHG emissions, other detrimental environmental impacts, such as acidification, human toxicity, or land use change, may increase and should therefore also be taken into account in the LCA (Davis et al. 2009; Cherubini 2010; Lazarevic and Martin 2016). However, the inclusion of additional environmental impacts is hindered by the fact that some of the key parameters, such as indirect land use change, biodiversity or water use, are currently not well understood or lacking a transparent and mature calculation methodology (Cherubini 2010; Cherubini and Strømman 2011; Benoist et al. 2012; Lazarevic and Martin 2016).

3.1.2. Aims and objectives

Previous papers stressed a lack of guidelines for LCA of biofuels, and a need to propose a common framework (Gnansounou et al. 2009; Cherubini 2010; Cherubini and Strømman 2011; Brandão et al. 2012; Reinhardt 2015; Thomassen et al. 2015). With the overall goal of bridging this gap and assisting policy makers to make informed decisions based on solid scientific evidence, the aim of this paper is to investigate and make recommendations for improving the robustness and accuracy of biofuel LCA framework. The objectives of this paper are to:

- Discuss the existing frameworks providing guideline for LCA of biofuels;
- Compare various approaches used in biofuel system LCA in terms of functional unit, system boundaries, reference scenario, allocation methodology and impact categories;
- Discuss how extensive the evaluations should be in order to produce sound results;
- Make recommendations for improving the robustness of the biofuel LCA framework.

3.2. Methodology – construction of literature source database

A literature search was conducted in Science Direct, Research Gate and Google Scholar, using both "biofuel" and "LCA" as keywords. This includes: life cycle assessment, analysis and approach. Initially, 54 papers were selected, these were published between 2006 and 2014 (Fig. 3.1). Out of this initial trawl, 16 papers were retained as these papers met the criteria under investigation in this paper. Papers were excluded based on repetitiveness (in terms of system boundaries and subject of studies), and scope and aim of the studies (e.g. studies focusing on infrastructure such as LCA of anaerobic digester; LCAs of biofuels different that biodiesel, bioethanol and biomethane; LCAs with an incomplete scope, missing one of the following: functional unit, scope, impact categories, allocation). Similarly, reviews, methodological papers and cost analyses were excluded. Thus the initial 54 papers were reduced to 16 papers for detailed investigation. These 16 papers were selected in such a way that three biofuel systems (biodiesel, bioethanol and biomethane) are represented, and also under a condition that each discussed at least four of the five

headings in Table 3.1, namely: functional unit, system boundaries, reference system, allocation, and impact categories. Biogas was included as an energy vector that is very close to biomethane. A diverse range of substrates were selected from crops (rapeseed, palm oil, maize and grass), and from residues (tallow, used cooking oil, food waste, manures and straw). Third generation substrates were also selected (micro and macroalgae).

Additionally, to validate the focus of biofuel LCAs on GHG emissions a simple calculation was carried to evaluate the percentage of peer-reviewed LCAs according to impact categories assessed. These were classified into six groups: GHG or GWP, energy, both GHG and energy, other environmental impacts, land use and water. 39 papers were considered, which were published after 2008. These were screened for impact categories assessed (Fig. 3.5).

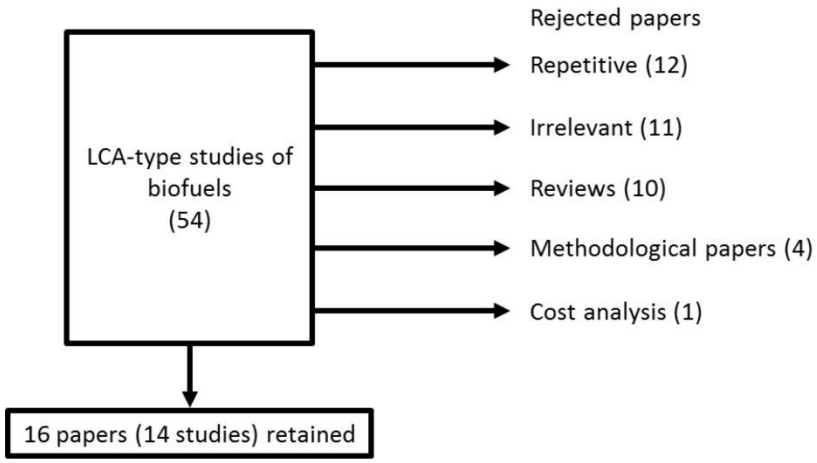


Fig. 3.1 Methodology for selecting reviewed papers.

3.3. LCA of biofuels- results

3.3.1. Existing LCA frameworks

LCA norms ISO (International Standard Organization) 14040 and 14044 set a general framework for LCA of any goods or services (ISO 2006a; ISO 2006b). Transparency is crucial, and all methodological choices and assumptions should be documented. Functional unit should reflect the function of the products; system boundaries and impact categories should stay consistent with the goal of the study; choices should be clearly stated and explained; in terms of allocation, ISO 14044 allows for any of the

allocation methodologies, provided that the choice is explained and fits into the scope of study; however it is recommended to avoid allocation whenever possible by 1) dividing processes into sub-processes and collecting data related to sub-processes; or 2) by expanding the system (substitution). It recommends also that a sensitivity analysis should be carried out to determine the importance of assumptions, data and methods applied in LCA (ISO 2006b). PAS (Publicly Available Specification for assessment of life cycle GHG of goods and services) follows ISO 14044 in terms of general guidelines, but concentrates only on GHG emissions (Defra et al. 2011). The UNFCCC (United Nations Framework Convention on Climate Change) through its MethPanel discusses only the issue of allocation and agrees partially with ISO, postulating that all allocation procedures can be justified (UNFCCC 2004). However, the International Energy Agency (IEA) in its BIOMITRE manual (BIOmass-based Climate Change MITigation through Renewable Energy) states that substitution may increase the complexity of the study, and therefore advocates in favour of allocation by market value or physical relationship, such as mass volume or calorific value (Horne and Matthews 2004). BIOMITRE also gives more detailed recommendations suggesting that the FU should be mass or volume or energy content. It gives guidelines and examples for biofuels' system boundaries. The tool focusses only on carbon emissions as an indicator. This is in line with the EU European Renewable Energy Directive (RED), which limits the allocation possibilities to allocation by energy content (lower heating value) and sets a MJ of fuel as the FU (EC 2009a). Sensitivity analysis are not required when following the directive. RED is applied in the BioGrace Excel tool (BioGrace 2015). BioGrace is a harmonised tool that can be used in a European context for life cycle GHG calculation of biofuels. The system boundaries are pre-set but there is a possibility to create new entries. The calculation includes also direct land use change emissions.

Table 3.1 Overview of sixteen selected papers on LCA.

Category References	Product studied	Functional unit	System boundaries	Reference system	Allocation methodology	Impact categories
• **Biodiesel**						
(Stephenson et al. 2008)	Rapeseed biodiesel	1 tonne of biodiesel	1. Crop production 2.Transport 3. Oil extraction 4. Biodiesel production 5. Distribution	Fallow set-aside (alternative land use)	1. Economic 2. Substitution	GWP and primary energy requirement based on EDIP 2003*
(Thamsiriroj and Murphy 2009)	Biodiesel from: 1. palm oil 2. rapeseed	1 GJ biodiesel or 1 ha per year	1. Crop production 2.Transport 3. Oil extraction 4. Biodiesel production 5. Distribution	Diesel reference system	No allocation (all burden to biodiesel)	GHG and energy balance
(Thamsiriroj and Murphy 2011)	Biodiesel from: 1. rape seed 2. tallow 3. Used cooking oil 4. grass biomethane	1 GJ biodiesel or 1 ha per year	Variable	Fossil fuel reference system	1. No allocation 2. Energy 3. Substitution	GHG and energy balance
(Yang et al. 2011)	Microalgae biodiesel	1 kg of biodiesel	1. Growing and harvesting 2. Drying 3. Oil extraction 4. Esterification	Alternative feedstocks	No allocation	Water footprint and nutrients balance
• **Bioethanol**						
(Kaufman et al. 2010)	Ethanol from: 1. corn-grain 2. from corn stover	1 MJ of ethanol	1. Corn production 2. Transport 3. Ethanol refining (co-product of corn-grain ethanol can be fed to cattle; co-product of corn-stover ethanol, electricity is injected into the grid)	Gasoline reference system	1. Mass 2. Economic 3. Energy 4. System expansion 5. Subdivision	GHG

(Kraatz et al. 2013)	Corn grain ethanol	1 kg of ethanol or 1 MJ of ethanol	1. Corn grain production 2. Transport 3. Ethanol refining 4. Use of stillage for biogas production or for animal fed production	Gasoline reference system	1. No allocation 2. Mass 3. Economic 4. Energy 5. Subdivision 6. System expansion	GWP, energy intensity, and net energy value
(Luo et al. 2009)	Corn stover ethanol	1 km driven in a midsize car	1. Agricultural production 2. Transport 3. Pretreatment 4. Fermentation 5. Distillation 6. Refining 7. Blending 8. Car driving	Gasoline reference system	1. Mass 2. Economic 3. Energy 4. System expansion	GWP, abiotic depletion, ozone layer depletion, photochemical oxidation, human and ecotoxicity, acidification and eutrophication based on CML 2001**
• **Biomethane**						
(Korres et al. 2010)	Grass biomethane	1 m^3 of biomethane per year, and 1 MJ energy replaced	1. Crop production 2. Biogas production 3. Upgrading and compressing of biomethane 4. Use in a bi-fuel car 5. Digestate use	Diesel reference system	No allocation	GHG and energy balance
(Smyth et al. 2009)	Grass biomethane	1 ha per year	1. Crop production 2. Biogas production 3. Upgrading and compressing of biomethane 4. Digestate use	First generation biofuels (palm oil biodiesel and others)	No allocation	Energy balance
(Wang et al. 2013)	Microalgal biomethane	1 GJ of biomethane	1. Algae cultivation (incl. production of photobioreactor) 2. Biogas production 3. Upgrading of biogas	Alternative biomethane production from ley crop	Substitution (avoided fertilizer)	GHG and energy balance

- **Biogas**

(Alvarado-Morales et al. 2013)	Biogas from brown seaweed (and bioethanol + biogas)	1 tonne of dry seaweed	1st scenario: 1. Seaweed production 2. Mechanical pretreatment 3. Biogas production 4. Energy production 2nd scenario: 3. Bioethanol production 4. Blending 5. Car driving 6. Use of stillage for biogas production	Coal-based electricity for biogas, and fossil gasoline for bioethanol	System expansion (avoided fertilizer and energy production, and fuel production	GWP, acidification and terrestrial eutrophication based on EDIP 2003* method, and energy consumption
(Börjesson and Berglund 2006) (Börjesson and Berglund 2007)	Biogas systems (ley crops, straw, sugar beet, manure, food waste, municipal organic waste)	1 MJ of biogas	1. Handling of raw materials (incl. crop cultivation) 2. Biogas production (farm-scale and large-scale biogas plants) 3. Biogas and digestate use	Various (fossil fuels and other bioenergy systems)	Based on dry matter content of the substrates	GWP, acidification, eutrophication, and photochemical oxidant creation potential
(Poeschl et al. 2012a) (Poeschl et al. 2012b)	Biogas system (various scenarios incl. single feedstock and co-digestion of manure, straw, corn, grass, whole wheat plant, MSW, food waste, pomace, slaugterhouse waste, grease sludge)	1 tonne of organic material (feedstock)	1. Feedstock supply 2. Biogas production 3. Biogas utilization 4. Digestate processing and handling; (system expansion: production of chemical fertilizers, electricity generation with Germany's fuel mix, heat generation using natural gas, and imports of natural gas and transportation fuel)	Typical biogas production and utilization pathways for Germany	System expansion	18 midpoint and 3 endpoint indicators based on ReCiPe*** method
(Tufvesson et al. 2013)	Biogas from industrial residues (distiller's waste, rapeseed cake, whey permeate, fodder milk, and bakery residues)	1 MJ upgraded and compressed biogas	1. Transport of feedstocks 2. Biogas production (including upgrading and compression) 3. Digestate use (no allocation and system expansion) 4. Replacement of mineral fertilizer 5. Animal feed production (system expansion)	Petrol and diesel reference system	1. No allocation 2. System expansion 3. allocation rules based on the sustainability criteria defined by the EU RED	GHG, energy balance, eutrophication and acidification potential

* EDIP 2003: Impact assessment method developed by the Institute for Product Development at the Technical University of Denmark (Dreyer et al. 2003).

** CML 2001: Impact assessment method developed by the Institute of Environmental Sciences at Leiden University in Netherlands (Dreyer et al. 2003).

*** ReCiPe: Impact assessment method developed by various actors: PRé Consultants, CML Leiden University, Radboud University Nijmegen and RIVM Bilthoven, (Goedkoop et al. 2013).

3.3.2. Functional unit

At the early stage of any LCA, the functional unit (FU) of product must be defined. The FU is a quantified description of the product system performance (Rebitzer et al. 2004; ISO 2006b). For biofuel, the product function might be the provision of fuel for transportation, or the processing of particular feedstock. Cherubini and Strømman (Cherubini and Strømman 2011) distinguished input- and output functional units. The FU can be expressed per mass of input substrate, such as one tonne of dry seaweed (Alvarado-Morales et al. 2013). However, a majority of studies used an output-related FU, typically expressed in MJ of energy generated from a given feedstock (Luo et al. 2010; Wang et al. 2013), or in kilogram of produced fuel (Yang et al. 2011). The EU RED recommends a FU of a MJ of fuel (EC 2009a). As the primary function of biofuel is to provide vehicle fuel, a commonly used FU is kilometres driven by a car or a truck transporting a given mass of freight (Luo et al. 2009; Zaimes et al. 2013). For land-based biofuels, the most relevant FU was identified as land area under a crop (Cherubini and Strømman 2011).

The impact of FU was investigated by Lettens et al. (2003). Low-input energy crops were compared with traditional energy crops, and it was found that if land surface was used as FU, conventional crops performed better in terms of GHG emissions, while if GJ of energy from crops was used as FU, then low-input crops performed better. Thamsiriroj and Murphy also explored the impact of choosing different FU, and came to contradictory conclusions when using GHG emissions per GJ or per ha as FU (Thamsiriroj and Murphy 2009; Thamsiriroj and Murphy 2011). In Thamsiriroj and Murphy (2011) grass biomethane performed better regardless the FU, but there were bigger discrepancies between rapeseed biodiesel and grass biomethane results if FU was set as GJ of fossil fuel displaced (Fig. 3.2 a.). In Thamsiriroj and Murphy (2009) if results were expressed using ha of land, rapeseed biodiesel performed better (less kg CO_2/ha), while if FU was switched to GJ of fuel, palm oil biodiesel was better (less kg CO_2/GJ) (Fig. 3.2 b). To better understand the findings of a LCA, Cherubini and Strømman (2011) recommended using more than one FU. However LCA studies that present results using several FU are seldom found, and there is a lack of guidance on the appropriate selection of multiple FUs.

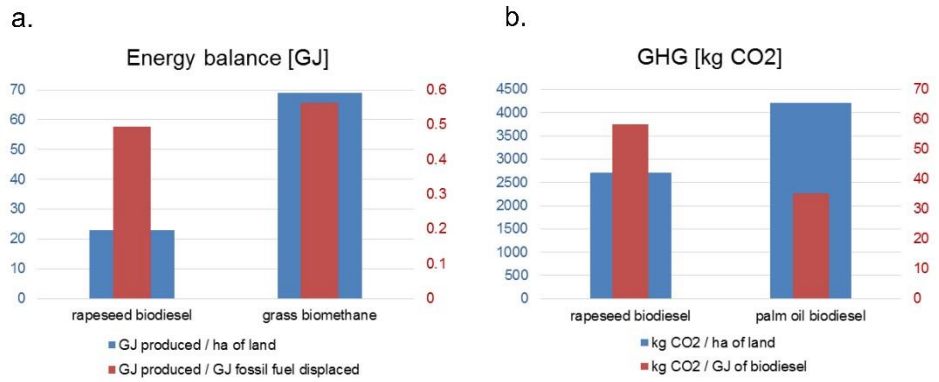


Fig. 3.2 The choice of functional unit and its impact on the results in cradle to gate (net) energy balance (GJ) and GHG balance (kg CO_2) of rapeseed biodiesel, grass biomethane and palm oil biodiesel.

3.3.3. System boundaries and reference system

The boundaries of the system must be defined together with the goal and scope of the LCA. Processes and flows should be listed to consider which should be included in a LCA. Usually, there is more than one product delivered to the market from the same system. In such a case, the system boundaries might be expanded to include the life cycle of co-products, by-products, and residues (Sarantakos and Opal 2008). Early LCAs of bioenergy systems did not include the life cycle of co- or by-products, thus giving a poor impression of biofuel systems environmental performance (Zaimes et al. 2013).

Some of the reviewed studies considered more than one system boundary (Thamsiriroj and Murphy 2011; Poeschl et al. 2012a; Poeschl et al. 2012b; Tufvesson et al. 2013). However, only three take into account the use of biofuel in a vehicle, a boundary expansion, which can have a considerable impact on the results. Korres et al. (2010) found that when use of grass biomethane in a car is included (well to wheel), the GHG savings are 18% lower than for well to tank analysis. Luo et al. (2009) reported that expanding boundaries to include car driving as well as food and fodder production, led to lower GWP but higher impacts in other environmental categories (Table 3.1).

The reference system for a biofuel is typically a fossil fuel system delivering the same service, thus having the same function. The EU RED sets a reference value based on fossil fuels. This value represents the actual average GHG emissions from petrol and

diesel within the EU that is set at 83.8 g CO_2 eq/MJ for both fuels. Several of the reviewed studies follow the RED recommendations to calculate GHG reduction. Thamsiriroj and Murphy (2009) used diesel as a reference system for biodiesel and Kaufman et al. (2010) used gasoline for bioethanol.

The choice of reference system is not always straightforward. If biomethane displaces natural gas (as might be the case in a country with a high penetration of NGVs), then the savings in displacing natural gas (at 50.3 g CO_2 eq/MJ natural gas) (EIA 2013) appear significantly less than if displacing petrol or diesel (at 83.8 g CO_2 eq / MJ), as might be the case in a country with a low penetration of NGVs. Generally, the reduction in GHG emissions will be higher if biofuel systems are compared with carbon intensive fuels, such as coal, and lower if compared with 'cleaner fuels', such as natural gas. In the majority of studies, the boundaries of the reference system did not extend beyond the production and use of fossil fuel (Thamsiriroj and Murphy 2009; Kaufman et al. 2010; Korres et al. 2010; Thamsiriroj and Murphy 2011; Kraatz et al. 2013). However, Börjesson and Berglund defined boundaries that included also the production of mineral fertilizers and alternative uses of the raw materials or land. By expanding the system's boundaries, Börjesson and Berglund detected potential indirect benefits derived from biogas systems linked to changes in handling of feedstocks and digestate (Börjesson and Berglund 2006; Börjesson and Berglund 2007).

3.3.4. Allocation methodology

If process chains deliver more than one product, all system flows must be divided between different products delivered by the system. This division procedure is called allocation whereby all flows are weighted and divided between the products of the system in proportion to the products' energy content, mass or market value (Fig. 3.3). Another approach is subdivision, in which multifunctional processes are sub-divided into sub-processes, and separate data are collected for each mono-functional process (Kraatz et al. 2009). No-allocation approach is the most conservative; all burdens are assigned to the main product.

The third major approach is system expansion, in which the boundaries of the system are expanded to include the functions and life cycles of co-products. System expansion and substitution are often used as synonyms. However, while the former approach is

only about expanding boundaries, the latter considers all products and/or functions that can be replaced by the co-products and by-products of the system under analysis. For example, digestate as a co-product of a biogas system can be used as fertilizer and therefore the system gets credits for reducing the use of mineral fertilizers (Brander 2012).

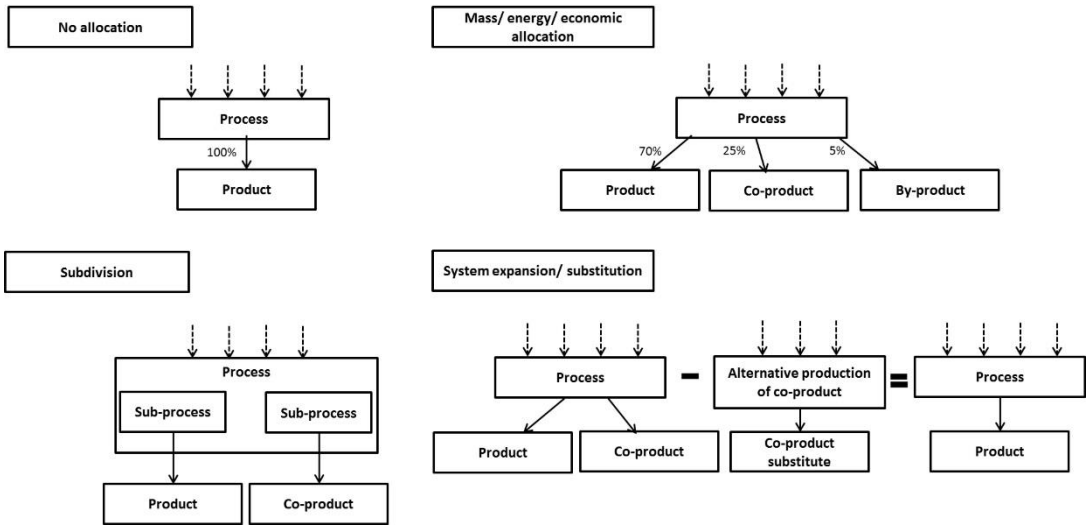


Fig. 3.3 Allocation approaches.

Despite the fact that choosing an allocation methodology is a fundamental step in LCA, different organisations recommend different approaches (Table 3.2) and several studies found that different allocation approaches can lead to completely different results (Kaufman et al. 2010; Thamsiriroj and Murphy 2011; Kraatz et al. 2013). Thus, it is important to use several scenarios when conducting sensitivity analysis.

Table 3.2 Allocation methodologies according to different sources.

Institution	Recommended Approach
• **Norm ISO 14044** (International Standard Organization)	whenever possible avoid allocation and instead use subdivision or system expansion (ISO 2006b)
• **RED and BioGrace tool**	allocation based on energy content (lower heating value) (EC 2009a)
• **BIOMITRE** (BIOmass-based Climate Change MITigation through Renewable Energy)	allocation by economic value (although not ideal since market prices often fluctuate) (Horne and Matthews 2004)
• **PAS** (Publicly Available Specification for assessment of life cycle GHG of goods and services)	dividing processes into sub-processes or system expansion to include co-products, by-products, and waste; when neither of these is feasible, then allocation based on economic value should be applied (Defra et al. 2011)
• **UNFCCC through Meth Panel** (Methodologies Panel)	all available approaches (UNFCCC 2004)

Comparing biodiesel and grass biomethane

Thamsiriroj and Murphy (2011) used three different allocation approaches to analyse rapeseed biodiesel, tallow biodiesel, UCO (used cooking oil) biodiesel, and grass biomethane. They found the biggest difference when assessing tallow biodiesel, with GHG savings varying between 33% (no allocation) and 150% (substitution approach). In the case of grass biomethane, GHG savings increased from 54% (no allocation) to 129% (system expansion). The simplest system, UCO biodiesel, seemed the least affected by the choice of the allocation method.

LCA comparison of rapeseed biodiesel system using various allocation approaches

Stephenson et al. (2008) considered that an allocation based on direct substitution was the most appropriate for rapeseed biodiesel system (scenario 2; Fig. 3.4). However, if the product being replaced is a by-product of another process, direct substitution becomes difficult to implement, and therefore Stephenson et al. applied allocation based on market prices (scenarios 1a and 1b; Fig. 3.4).

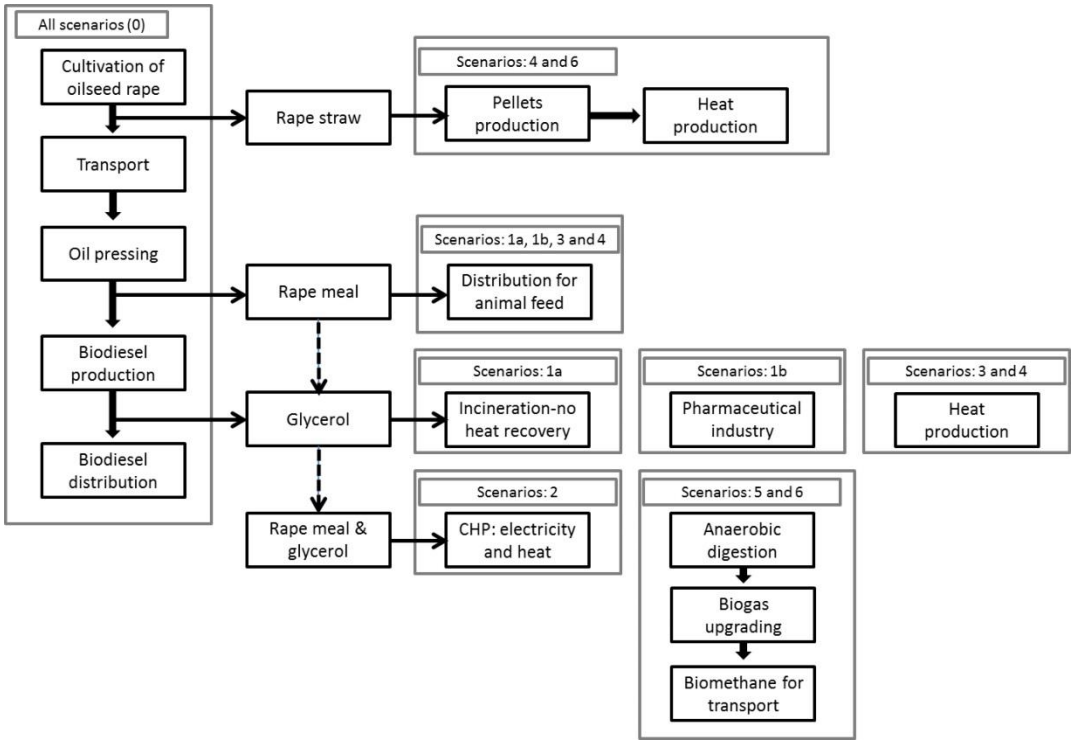


Fig. 3.4 Scenarios for LCA comparison of rapeseed biodiesel based on (Stephenson et al. 2008) and (Thamsiriroj and Murphy 2010).

Thamsiriroj and Murphy (Thamsiriroj and Murphy 2010) tested various scenarios, including no-allocation (scenario 0; Fig. 3.4), and substitution of various co-products (scenarios 3-6; Fig. 3.4). Compared to economic allocation (1a and 1b; Fig. 3.4), the direct substitution (scenario 2; Fig. 3.4) resulted in higher GHG savings and lower total energy consumption. If electricity and heat generated from rapeseed meal in CHP were used to substitute grid electricity and heat from coal or gas, the GWP decreased by 92% and total energy requirements by 216%. With the no-allocation method, however, the reduction in GHG emissions was only 28%. The highest GHG savings (135%) resulted from the use of rape cake for animal feed, glycerol for heat production, and straw for thermal energy (scenario 4; Fig. 3.4). Using rape cake as animal feed saved GHG emissions from the production and transport of soybean meal, usually imported to Ireland from South America. Stephenson et al. (2008) did not consider the burden of fodder production when rape meal is used to generate energy.

Assessing corn stover-based ethanol using system expansion, and allocation by mass, energy and economic value

In a case study of corn stover ethanol, Luo et al. showed that using an economic allocation gave much higher results for GWP of bioethanol in comparison to mass/energy allocation approaches (Luo et al. 2009). This is because the corn/stover allocation ratio shifted from 1.7 to 7.5 when switching from mass/ energy allocation to economic allocation.

Assessing bioethanol using six allocation methodologies

Kraatz et al. analysed ethanol production from corn, with dried distillers grains and solubles (DDGS) as co-product (Kraatz et al. 2013). Using no-allocation approach and system expansion resulted in the highest energy intensity and highest GWP (section 3.3.5). Conversely, mass, energy and economic allocation gave the lowest values for energy intensity and GWP of bioethanol produced.

3.3.5. Impact categories

Life cycle impact assessment (LCIA) methodologies model the pathway of substances and link them to effects. There is a large array of LCIA methodologies that propose diverse indicators or/and calculate the same indicator using different models. The International Reference Life Cycle Data System (ILCD) handbook reviews a wide range of methods for impact assessment, and provides LCA practitioners with recommendation on indicators and models used in LCIA (EC JRC 2010b; EC JRC 2011). The handbook was developed for LCAs in European context by the European Commission Joint Research Centre. EDIP 2003, ReCiPe or CML 2001 quoted in this study, are widely used methods (Goedkoop et al. 2013). The midpoint approach translates environmental impacts into mechanisms such as acidification, eutrophication, or climate change, while endpoint methodologies concentrate on damages and express impacts on the three following: human health, natural environment (biodiversity) and natural resources (Seppälä et al. 2006; EC JRC 2010c).

LCA based on energy and carbon balances

The majority of biofuel LCAs in the literature look only at GHG emissions or GWP, and/or energy balance (Cherubini and Strømman 2011). From 39 papers sampled in

Science Direct that used LCA in the title, about half examined both carbon and energy, while only 26% considered also other environmental impacts (Fig. 3.5).

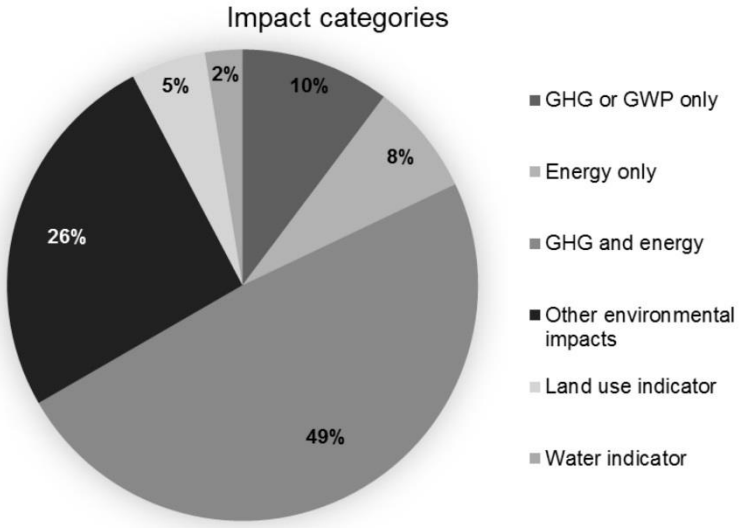


Fig. 3.5 Percentage of peer-reviewed LCA studies of biofuels by impact categories assessed (39 peer-reviewed papers sampled using Science Direct published between 2008 and 2015 using "LCA" and "biofuel" in title).

Stephenson et al. (2008) reported on GWP and primary energy requirement (EDIP 2003 (Dreyer et al. 2003)). Thamsiriroj and Murphy (2009) calculated GHG emissions and reduction according to EU RED recommendations, as well as the gross and net energy of both rapeseed and palm oil biodiesel. A similar approach was employed in studies assessing the sustainability the grass biomethane (Smyth et al. 2009; Korres et al. 2010). In a paper by Kraatz et al., corn grain ethanol was assessed using energy intensity and GHG based on average data (Fig. 3.6) (Kraatz et al. 2013). Electricity consumption, drying of DDGS and corn farming showed quite similar contribution to the overall energy intensity. This changed when looking at GHG, where 70% of impact comes from electricity alone. Also the discrepancies between various life cycle stages were much lower when energy intensity was used as measure. Drying of DDGS has a much higher impact on energy intensity than on GWP.

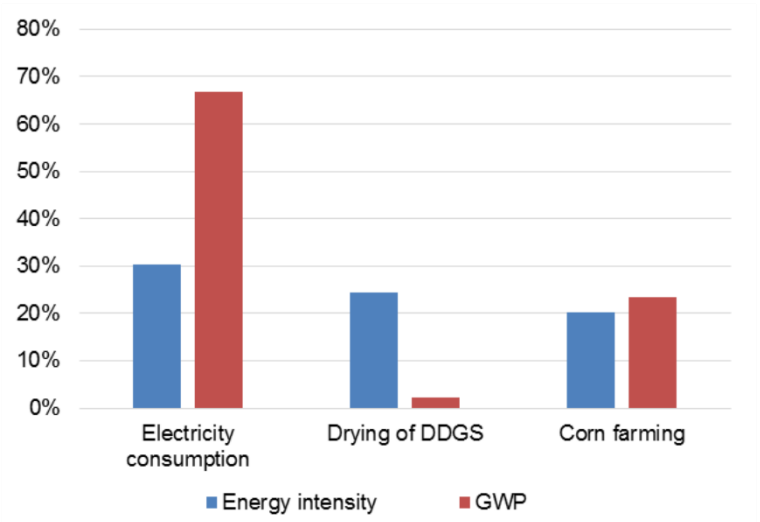


Fig. 3.6 Contribution of the three most impacting life cycle stages of corn grain ethanol to the total environmental impact measured in energy intensity (blue) and GWP (red) (Kraatz et al. 2013).

LCA based on impact factors beyond carbon and energy

Measuring of sustainability in terms of impacts beyond carbon and/or energy often gave different results to when just carbon and/or energy were assessed. Aguirre-Villegas et al. used four sustainability indicators: GWP; ammonia emissions; depletion of fossil fuel (DFF); and nutrient form and fate (Aguirre-Villegas et al. 2014). They found that the anaerobic digestion (AD) pathway in comparison to other manure utilisations had the lowest values for GWP and DFF, but had the highest NH_3 emissions. Tufvesson et al. in assessing biogas from industrial residues looked at GHG emissions, eutrophication, acidification, and energy balance (Tufvesson et al. 2013). While GHG emissions were reduced whatever the substrate, impacts of both eutrophication and acidification were higher for biogas than fossil fuel systems. Only Poeschl et al. (Poeschl et al. 2012a; Poeschl et al. 2012b) included land use change indicators (land transformation and occupation) as a part of the ReCiPe method (Goedkoop et al. 2013).

3.4. Discussion: overcoming the challenges

3.4.1. Existing LCA frameworks

From the frameworks listed in Table 3.2, BIOMITRE and RED are the only one specifically developed for biofuels and biomass. RED also provides the most detailed recommendations on LCA of biofuels. BioGrace is currently the only integrated tool that complies with RED and can be used by farmers, policy makers and consultants within Europe (Peter et al. 2016). It is an intuitive tool with simple interface that allows even an unexperienced LCA analyst to get a quick GHG calculation (BioGrace 2015). The tool provides also a liberty to change parameters and introduce more specific data. However, BioGrace can be applied only for GHG calculations. Moreover, the RED does not give any recommendation on extending the impact assessment beyond carbon; it does not encourage the system expansion approach nor the sensitivity analysis Also, despite the harmonisation, the RED still permits methodological choices that can lead to different results for same biofuel pathway. (Hennecke et al. 2013)

3.4.2. Functional unit

The choice of FUs should reflect biofuel life cycle stages (Fig. 3.7). Thus, if feedstock requires agricultural land, then LCA results should be reported on a per ha basis, and if biofuel is produced for transportation, then results should be reported on a per km basis. The advantage of using a ha of land as FU is that it allows an indirect comparison of land use impacts from biofuels. Low impacts with high energy output per ha indicate a reduced risk of emissions from land use changes. LCA results as per each life cycle stage should be available using different FU.

3.4.3. System boundaries and reference system

The definition of the system's boundaries and choice of the reference system are crucial, as the results of LCA vary according to the reference system chosen. In order to present a comprehensive understanding of the system, boundaries should be expanded to include co-products, by-products and residues (Fig. 3.7). The boundaries of the reference system should be the same as those of the primary system under analysis, and the choice of reference system should be informed by the goal of the analysis. It may be appropriate to define a reference system for each stage of the life

cycle process, such as alternative land use and LCA of products that are being replaced by co- and by-products. However, the authors recognise that expanding the reference system boundaries and using multiple reference systems can considerably increase analysis complexity; this can be addressed by using sensitivity analysis, for example by testing various allocation approaches.

3.4.4. Allocation methodology

From the assessed studies, LCA results depend heavily on type of allocation chosen. Van der Voet et al. stressed that substitution implies higher variability in the results (van der Voet et al. 2010). However, the authors believe that co- and by-products should be included in biofuel LCA. Complex processes should be sub-divided and data collected for each sub-process by linking of inputs and outputs to products, and co- and by-products. If this is not possible due to lack of specific data and/or multiplicity of co-/ by-products, then substitution should be applied (Fig. 3.7). A sensitivity analysis should be carried out to test the influence of the chosen allocation method.

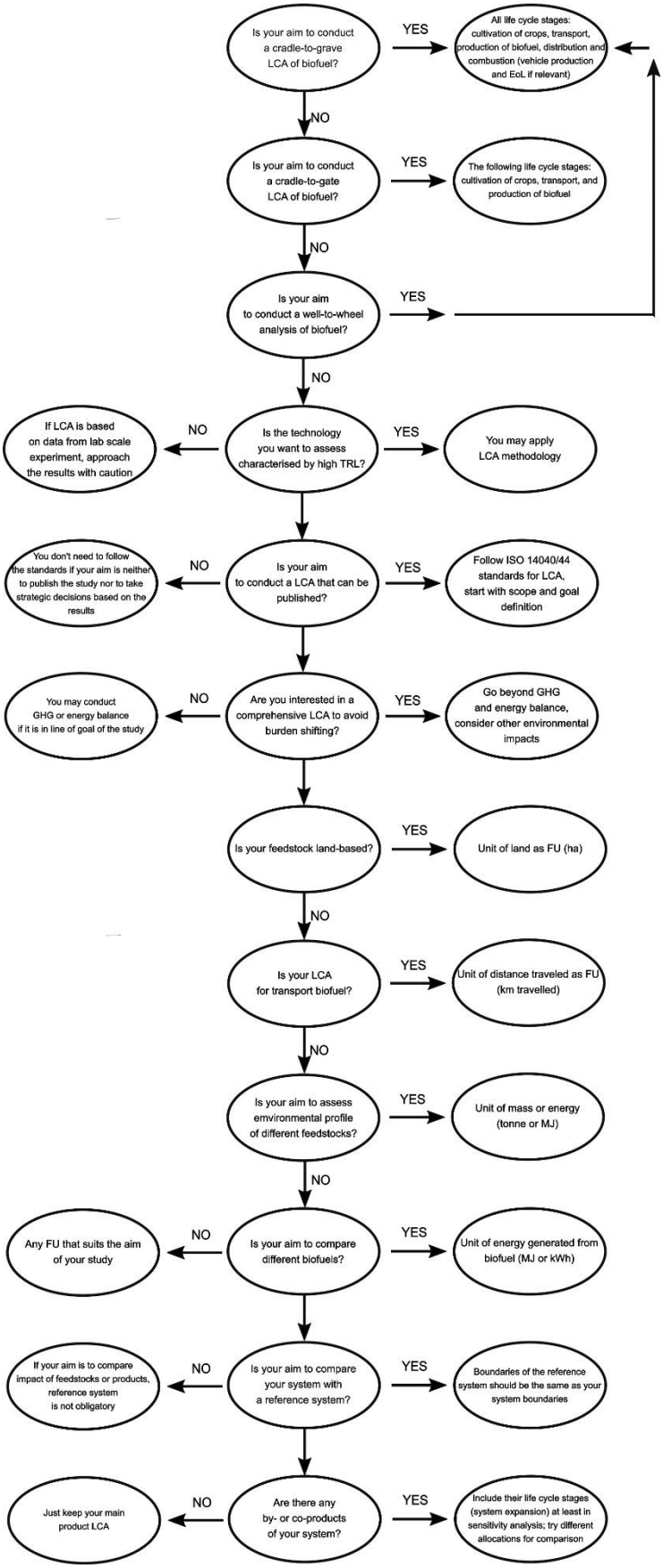


Fig. 3.7 Flowchart with recommendations for biofuels LCA.

3.4.5. Impact categories

Since the majority of LCAs look only at GHG and energy balance, this can lead to the problem of burden shifting; where a biofuel system might achieve a high level of GHG reduction but could also impact the environment in other ways, for example through acidification and eutrophication (Luo et al. 2009). In line with previous studies (van der Voet et al. 2010; Poeschl et al. 2012a; Poeschl et al. 2012b; Tufvesson et al. 2013; Aguirre-Villegas et al. 2014; Lazarevic and Martin 2016), the authors recommend broadening LCAs to include impacts other than just carbon and energy. The ISO does not set a list of recommended impact categories for life cycle impact assessment, but highlights the importance of choosing these in line with the goal and scope of the study (ISO 2006b). This choice can be affected by local and regional conditions.

A comprehensive LCA study would investigate a range of environmental impacts such as climate change, impact on human health, ecotoxicity, acidification and eutrophication of environment, ozone layer depletion. While assessment of all these factors would certainly be desirable to gain a full understanding of the system, it may not be practical due to time and resource constraints. A further difficulty is the current lack of knowledge of some parameters and/or their poor integration in LCA studies, particularly indirect effects such as land use change and nitrogen emissions (Cherubini et al. 2009; Davis et al. 2009).

The authors limited the number of recommended indicators to the five listed in Table 3.3. The ILCD handbook was employed to assess the existing indicators for climate change, acidification, eutrophication and land use. Only the midpoint categories were considered. Both mid- and endpoint approaches have their advantages and disadvantages, but midpoint categories are much more accurate and precise, and bring less uncertainty to the model, unlike the endpoint approach that requires weighting of the categories. Natural environment is an endpoint indicator that seems to be relevant for biofuels LCA; it can be measured in biodiversity loss or gain. However at the moment, this indicator is often overlooked (EC JRC 2010c).

Table 3.3 Recommendation for LCIA.

Impact categories	Indicators	Calculation and unit	Reasons
• Climate change	GWP or GHG	g or kg CO_2-equivalent	Following legislative requirements to calculate impact of biofuels on global warming and potential savings
• Energy	Energy balance, net energy, land use energy efficiency	Energy balance (output/input ratio), net energy (gross energy minus parasitic energy demand) land use efficiency (energy production per unit of land); kwh or MJ (per ha)	Traditional indicator related to biofuel energy efficiency
• Eutrophication (terrestrial and aquatic)	Accumulated exceedance (terrestrial), ReCiPe (aquatic)	Terrestrial: modelling following Seppala et al. (Seppälä et al. 2006; Posch et al. 2008); kg N eq. Aquatic: ReCiPe; kg P eq. (Goedkoop et al. 2013)	Terrestrial strongly correlated with agriculture and combustion (N compounds); aquatic with waterborne emissions (N and P compounds)
• Acidification	Accumulated exceedance	Modelling following Seppala et al. (Seppälä et al. 2006; Posch et al. 2008); moles of hydrogen ion (H+) eq.	Strongly correlated with transport and agriculture (N and S compounds)
• Land use	SOM or surface area of transformed and occupied land	mg SOM per year (deficit of SOM) (Milà i Canals et al. 2007a; Milà i Canals et al. 2007b) and m^2 of transformed and occupied land (Goedkoop et al. 2013)	Especially relevant for land-based biofuels

To assess climate change, all LCIA methodologies use the GWP midpoint indicator developed by the Intergovernmental Panel on Climate Change (IPCC) (EC JRC 2011). The GWP should be always based on the latest IPCC report, in this case the Fifth Assessment Report (Myhre et al. 2013). GWP can be calculated over a 20, 50 and 100-year timeframe. Well-mixed GHGs, such as CO_2, CH_4, and N_2O (including direct and indirect emissions from NH_3 and NO) are included. The GWP unit is kg CO_2 eq.

Acidification is mainly caused by the airborne acidifying substances, such as ammonia (NH_3) (after nitrification in the soil when nitrite is produced), nitrogen oxides (NO_x) and sulphur dioxide (SO_2) (largely from fossil fuels combustion) (EC JRC 2012). The

ILCD evaluated the accumulated exceedance (AE) model as the most suitable (Seppälä et al. 2006; Posch et al. 2008; EC JRC 2010c). This method is widely accepted and used by the European Commission and the United Nation Economic Commission for Europe for policy purposes. It uses critical load of nutrients to quantify the sensitivity of the ecosystem. It also provides characterization factors that are country specific. It is expressed in moles of hydrogen ion (H^+) eq.

Eutrophication potential examines the impacts of the surplus of nitrogen and phosphorus on the terrestrial and aquatic ecosystems (marine and freshwater). Terrestrial eutrophication is caused by deposition of airborne N emissions, such as NO_x (combustion processes), and NH_3 (agriculture). For terrestrial eutrophication, ILCD recommends the AE model (Seppälä et al. 2006; Posch et al. 2008; EC JRC 2010c). The indicator is expressed in kg N eq (EC JRC 2010c). Freshwater and marine eutrophication is induced by waterborne emissions, such as nitrate, phosphate and other N and P compounds (EC JRC 2010c). It is recommended to use the ReCiPe method, as it models best the aquatic fate of emissions; however it is restricted only to European countries. The indicator is expressed in kg P eq.

Land use indicators reflect the changes to ecosystems due to the effects of land occupation and transformation. To assess the impact of land use the ILCD handbook recommended the method by Milà i Canals et al., based on soil organic matter (SOM) (Milà i Canals et al. 2007a; Milà i Canals et al. 2007b); however the level of recommendation is a grade lower than for the other impact categories discussed above (EC JRC 2011). The drawback of this method is that the LCA practitioner must calculate the case-specific characterisation factors based on collected data, such as SOM value before and after the land occupation and SOM value of the reference system. Moreover, the method presents a limited impact indicator based on SOM that considers soil quality but does not include the impact on soil biodiversity (EC JRC 2010c). Alternatively, the midpoint ReCiPe method can be applied, but it accounts only for surface area of transformed and occupied land (expressed in m^2).

An energy indicator is typically associated with the assessment of energy vectors such as biofuels. It requires thorough data collection on energy inputs and outputs. It can take various forms, such as: 1) energy balance (energy output to input ratio), 2) net

energy (gross energy of the product minus parasitic energy demand of the processes), 3) land use energy efficiency (for land-based biofuels; energy produced per unit area).

Traditionally, LCA is a tool for assessment of global impacts (Finnveden and Nilsson 2005). This is still valid for climate change, but for impacts such as acidification and eutrophication that occur locally, there is a need for country- or site-specific characterisation factors. ReCiPe and CML methods include the European average factor (EC JRC 2011; Lazarevic and Martin 2016), while Seppala et al (Seppälä et al. 2006) and Posch et al (Posch et al. 2008) went even further to include EU country-specific factors for acidification and terrestrial eutrophication. Finnveden and Nilsson argue that the site-specific factors are needed to allow including the local conditions into the model (Finnveden and Nilsson 2005).

To conclude, there is a need for a framework on impact categories for biofuels assessment. These should include at least the categories described above: climate change, acidification, eutrophication, land use, and energy. Further research should provide LCA practitioners with site-specific factors for acidification, eutrophication and land use.

3.5. Concluding remarks

A sound evaluation of biofuel systems requires conducting a full cradle to grave LCA. Valid LCA studies should consider other sustainability indicators in addition to GHG emissions and energy balances. Biofuel LCA should be transparent, and standard requirements should include functional unit, system boundaries, allocation methodology, and environmental indicators. This unified methodology will allow comparing biofuel LCA studies, which currently is not possible as different studies follow different rules. Whereas it is preferable that full LCA is carried out in academia, industry should have access to a simplified and cost-effective version of LCA. LCA is a powerful tool but needs to continue to be refined as knowledge of the science grows. The recommendations outlined in this paper should go some way towards achieving this.

Chapter 4. Impact of including land use change emissions from biofuels on meeting GHG emissions reduction targets: the example of Ireland

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Abstract

The greenhouse gases (GHG) emissions from land use change are of particular concern for land-based biofuels. Emissions avoided by substituting fossil fuels with biofuels may be offset by emissions from direct and indirect land use changes (LUC). There is an urgent need to investigate what impact land use change emissions may have on the expansion of bioenergy and biofuels, in the context of EU mitigation policies. This paper focuses on Ireland, which faces a number of challenges in delivering its renewable energy and GHG reduction targets. The Irish TIMES energy systems model was used to assess the impact of a range of land use change emissions' levels on the evolution of Ireland's low carbon energy system. A reference scenario was developed where LUC is ignored and Ireland achieves a least cost low carbon energy system by 2050. If high ILUC emissions are included, this results in a decrease by 64% in biofuels and a 61% increase in marginal abatement costs by 2050. Hydrogen is used instead of biofuels in the freight sector in this scenario, while private cars are fuelled by renewable electricity. If GHG from ILUC were considered less severe, indigenous grass biomethane becomes the key biofuel representing 31% of total bioenergy consumption. This is in line with recent research in Ireland of the key role that grass biomethane can play.

Keywords: Bioenergy, land use change, climate mitigation, Renewable energy policy, Energy systems modelling, MARKAL-TIMES

4.1. Introduction

4.1.1. Policy context in Ireland

The European Commission has set ambitious emissions targets, aiming for a 20% GHG reduction by 2020 (EC 2009b), 40% by 2030 (EC 2014b), and between 80% and 95% by 2050 (EC 2011), all relative to 1990 emission levels. Moreover under the European Union (EU) the EU has committed to achieving at least a 20% renewable energy share of gross energy consumption by 2020 under the Renewable Energy Directive (RED), (EC 2009a) and 27% by 2030 (EC 2014a).

Ireland provides an interesting case study in terms of the implications of meeting GHG emissions reduction and renewable energy targets for a number of reasons. Under the EU Effort Sharing Decision on GHG emissions reduction (limited to emissions outside of the EU Emissions Trading Scheme), Ireland is obliged to reduce GHG emissions by 20% below 2005 levels by 2020. Ireland must also ensure that at least 16% of its gross energy consumption, and 10% of energy used in transport are generated from renewable sources by 2020 (EC 2009a). Ireland is facing significant challenges to meet these ambitious climate mitigation targets (Chiodi et al. 2013a), and has a lot of ground to cover to meet the 16% renewable energy target (Pye et al. 2014). The country experienced a continuous increase in energy use, particularly between 1990 and 2007. This is coupled with a recent growth in renewable energy deployment; as electricity from wind increased from 5% to 20% over the past ten years. Ireland has a relatively small synchronous power system, which poses specific challenges in integrating large levels of non-synchronous renewable generated electricity (Foley et al. 2013). Lastly, Ireland's energy system depends hugely on imported energy, with almost 85% of energy dependency in 2010 (Howley et al. 2015b).

4.1.2. Sustainability of bioenergy in the context of land use change

Energy from biomass and biofuels may play an important role in meeting EU RED targets, especially for renewable heat and transport. Bioenergy replacing fossil fuels has the potential to greatly decrease overall GHG emissions, in particular in the sectors outside of the EU Emissions Trading Scheme (ETS). However, biofuel sustainability was seriously questioned in the late 2000s with increasing levels of concern regarding direct and indirect land use changes (Searchinger et al. 2008). Direct land use change

(DLUC) occurs when a new crop replaces a prior land use, such as other crop, forest or grassland. Indirect land use change (ILUC) occurs when an energy crop replaces a food or feed crop, and the displaced land use occurs elsewhere, in order to compensate for the resulting gap in food or feed production (Gnansounou et al. 2009) (Fig. 4.1). Currently, biofuel sustainability in the EU is assessed under the RED Directive that prohibits the conversion of land with high biodiversity value for bioenergy cropping (EC 2009a). A recent directive amending the RED and Fuel Quality directives, goes further placing a cap on first generation land-based biofuels that implies these biofuels shall not exceed 7% out of the EU 10% renewable transport target (EC 2015b). However, in practice the EU RED methodology that accounts for emissions from the production and use of biofuels, including DLUC, does not factor in the ILUC emissions.

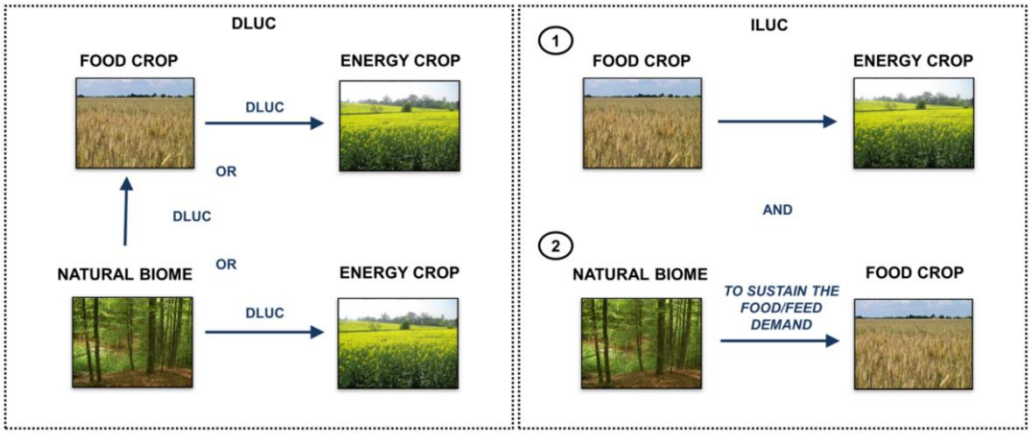


Fig. 4.1 Direct and indirect land use change (DLUC and ILUC).

4.1.3. Energy Systems Modelling

Energy systems modelling plays an important role in supporting policy makers (Chiodi et al. 2015d) TIMES (The Integrated Markal-Efom System) is a bottom-up model energy systems modelling framework developed and supported by the IEA Energy Technology Systems Analysis Program (IEA-ETSAP), which combines both technical engineering and economic approaches (Gargiulo and Ó Gallachóir 2013). It is used by 177 institutions across 70 countries. A number of studies involving TIMES and its predecessor MARKAL are summarised in the ETSAP Annex X (IEA-ETSAP 2008) and XI reports (IEA-ETSAP 2011), and in (Giannakidis et al. 2015). TIMES approaches energy as a system rather than as a set of elements. This has the advantage

of providing insights into the most important substitution options that are linked to the system as a whole and that cannot be understood when analysing a single technology, or commodity, or sector (Chiodi et al. 2015c). TIMES generates future energy system pathways that meet energy service demands at least-cost approach and subject to environmental and technical constraints, such as mitigation targets. The energy system costs include investment, operation and maintenance costs, plus the costs of imported fuels, minus the incomes of exported fuels, and the residual value of technologies at the end of the horizon (Loulou et al. 2005).

The Irish TIMES model (Ó Gallachóir et al. 2012) has been used to test a number of future energy and emissions' policy scenarios, such as impact of climate mitigation policy on Irish energy system by 2020 (Chiodi et al. 2013a) and 2050 (Chiodi et al. 2013b), energy security (Glynn et al. 2014), impact of limiting the bioenergy resources (Chiodi et al. 2015a), and integrated agricultural and energy systems modelling (Chiodi et al. 2015b).

4.1.4. State of the art in land use change associated with bioenergy

Land use impact is particularly important for sustainability evaluation of agricultural products such as energy crops (Bare 2014). However land use has proven to be difficult to quantify due to the complexity of the agricultural systems (Bare 2011). ILUC emissions are a controversial subject in discussion on bioenergy sustainability (Mathews and Tan 2009). Previous studies have shown that depending on the model and assumptions made, results might vary greatly and assign to biofuels, such as corn ethanol (Searchinger et al. 2008), palm oil and soybean biodiesel, and sugarcane ethanol (Fargione et al. 2008), much higher GHG net emissions than the fossil fuels they replace. In this case, savings from substituting fossil fuels can be offset by indirect emissions related to energy crop cultivation (Fargione et al. 2008). However some studies pointed out that biofuels should not be given all the blame, because they are clearly not the only driver of land use changes (Gawel and Ludwig 2011). More recent papers concluded that 'food versus fuel' debate did not address the complexity of the problem, providing only one solution: *"no more biofuels"* (Tomei and Helliwell 2015). In reality, feedstocks for bioenergy are flexible, and are in fact used by multiply markets; agricultural land is not only used for food production, but for a variety of other products, including biodiversity, cultural and other ecosystem services.

A recent study concentrating on four energy crop biofuels in selected countries (Poland, Romania, Hungary and Indonesia) suggested that biofuel production can be increased without inducing ILUC (Brinkman et al. 2015). The key strategies to prevent ILUC were: 1) increase of yields above-baseline and 2) use of underutilised land (such as contaminated, abandoned or fallow land). Similar studies conducted for India estimated that wastelands with potential for biomass and biofuel production represent around 12% of the total country area. It pointed out also that a policy framework is required to allow for sustainable intensification of wasteland (Edrisi and Abhilash 2016). Previous research in Sweden showed that cultivation of wheat for ethanol on excess grassland reduces the risk of ILUC but brings significant soil carbon losses; cultivation of lignocellulosic perennial crops instead reduces these soil C emissions and might even improve the GHG balance (Börjesson et al. 2015).

There is a strong interest in including both DLUC and ILUC emissions into policy making and targets (Panichelli and Gnansounou 2014). Some models do estimate ILUC emissions, usually based on equilibrium models that try to address three questions: i) how much of land is required for biofuel production, ii) what land types will be converted and where, and as a result iii) how much carbon will be released. Attempts to address these questions are subject to considerable uncertainty (Palmer and Owens 2015), and even if the existing models have been improved, there is still a need for a harmonised reporting (Panichelli and Gnansounou 2014). ILUC can be mitigated through consumption limitation and/or financial support by limiting the biofuels with high risk of ILUC and promoting biofuels with reduced ILUC risk (Tokgoz and Laborde 2014).

4.1.5. Innovation in the paper

The literature does not include modelling in TIMES or MARKAL addressing the possible impacts of land use changes associated with bioenergy and biofuels in delivering climate mitigation policies. This paper is the first paper to undertake such an analysis and to quantitatively assess how the impacts of land use changes from bioenergy may affect the capability of an energy system to meet challenging mitigation targets. This study is also the first to investigate the impact of direct and indirect LUC on the Irish energy system and Ireland's options for delivering climate mitigation targets. This is a preliminary study, and therefore findings should not be

taken as definitive. The main uncertainty lies in ILUC calculations as these are complex calculations, which are heavily based on assumptions. This was addressed using two sensitivity scenarios. Ireland is used as an example of an energy system but the approach developed and employed can be replicated in other countries.

4.1.6. Aims and objectives of the paper

The objectives of the paper are to:

- Assess DLUC and ILUC emissions factors for all bioenergy systems;
- Assess the implications of introducing land use change parameters on how emissions targets are met at least cost.
- Identify key bioenergy sources that perform well in an emissions constrained future that includes ILUC emissions.
- Assess via scenario analysis the projected marginal price of CO_2 emissions reduction.

4.2. Methodology

4.2.1. Emissions associated with DLUC and ILUC

The selection of bioenergy feedstocks for Ireland is based on current imports and consumption trends (NORA 2014), and projected bioenergy potential for the period up to 2050 (Chiodi et al. 2015a). Waste, residues and recycled oil were excluded as they do not induce any land use change (EC 2009a). An extensive literature review on DLUC and ILUC emissions for each commodity, including country of origin, was conducted.

The majority of current (2013) imported biofuels originates from Europe mainly from Spain, France, UK, Hungary and Poland (NORA 2014). The authors recognise that imported bioenergy may also originate from outside the EU. In 2013 the highest proportion of imported biofuels from outside Europe was from the US, in the form of biodiesel from used cooking oil (UCO), Guatemala (ethanol from sugarcane), Brazil (ethanol from sugarcane), Malaysia (palm oil biodiesel) and Costa Rica (ethanol from sugarcane) (NORA 2014).

Emissions from direct land use changes

Direct land use emissions were estimated either by using values from literature or using the BioGrace model (BioGrace 2015) that is based on life cycle assessment and evaluates European fuels pathway under the EU RED (EC 2009a).

Grass is the key agricultural crop for Ireland (Holden and Brereton 2002) and covers approximately 92% of the country's agricultural land (McEniry et al. 2013). This is permanent grassland that is not cultivated on arable land. A particularity and strong advantage of Irish grassland is that it can achieve very high yields, between 12 and 16 tonne dry solids (DS) per hectare (O'Donovan et al. 2011). Currently, grassland is used mainly for beef and cattle industries and these are projected to expand under the Food Harvest 2020 targets of increasing milk and beef production by 50% and 20%, respectively (DAFM 2014). However, there are 1.7 million tonne DS of grass available for alternative use other than livestock needs (McEniry et al. 2013), and this may significantly increase if applying changes in grassland management. In order to calculate Irish grassland DLUC in BioGrace, it was assumed that managed grasslands with medium inputs will be converted into managed high input grasslands (EC 2010). The authors considered that Ireland is situated in a cool temperate moist climate, and that Irish soils are mainly low activity clay soils (71%) with wetlands, spodic, sandy and high activity clay soil (EPA 2014a).

Under EU Common Agricultural Policy, the conversion of grassland to arable land is restricted and significant conversion rate is not allowed (Smyth et al. 2010). Therefore, it was assumed that domestic energy crops will be cultivated mainly only on existing croplands, such as barley, wheat and oats (CSO 2015). DLUC from oilseed rape and wheat was considered nil since the agricultural practices involved in their cultivation are similar to those of replaced cropland. For indigenous Miscanthus and willow, both perennial crops, DLUC emissions were calculated using BioGrace, and assuming that managed grassland (which involves tilling and reseeding on a regular basis as opposed to permanent grassland) was converted to either Miscanthus or willow (Styles and Jones 2007).

For energy crops such as sugar beet (Klenk et al. 2012), corn, wheat, and oilseed rape (Börjesson and Tufvesson 2011) imported from Europe, it was assumed that no DLUC is caused since they are grown on land that was already under cultivation prior to their

introduction. For both corn and wheat ethanol produced outside Europe, it was assumed that existing agricultural land was converted to corn and wheat cultivation, and DLUC is close to zero (Sinistore 2012). Results of studies conducted by Brazilian research groups suggest that sugarcane for ethanol in Brazil replaces mainly existing croplands and intensified pasture, and therefore DLUC emissions associated are very low (Moreira et al. 2012). The authors assumed that DLUC emissions from sugarcane ethanol are zero. The complexity of trading networks between Europe and other countries makes it very difficult to obtain sound data on DLUC of sugar beet ethanol, as well as oilseed and palm biodiesel imported from outside Europe. The authors adopted a conservative approach for estimating DLUC emissions related to these biofuels, and used the same values that were used for the ILUC optimistic scenario (see next section).

Emissions from indirect land use changes

Assessing ILUC emissions is still highly debatable and there is no widely accepted and sound methodology. The results of various studies might vary greatly. For sensitivity issues for each bioenergy pathway two scenarios were considered: optimistic (ILUC+) and conservative (ILUC-). In the ILUC- scenario, higher ILUC were assigned to all energy crops, both domestic and imported. Particularly high ILUC were found for grass, sugarcane ethanol (Brazil) and palm oil biodiesel (South East Asia).

Grassland is one of the key competitive advantages of Irish agriculture (O'Donovan et al. 2011). By using only 1.1% of grassland the 2020 target of 10% share of renewables in transport (RES-T) can be achieved through co-digestion with dairy slurry (Wall et al. 2013). Grass biomethane is also considered a second generation biofuel. Therefore in case of ILUC+ scenario, ILUC is not going to be an issue if grass is used for biomethane production. Competition for land does not occur, as both livestock requirements and biofuel production can be satisfied through efficient land management (McEniry et al. 2013). For ILUC-, a far more conservative approach was adopted (Smyth and Murphy 2011), in which the grass biomethane industry might indirectly affect the beef sector. The United Kingdom is the largest importer of Irish beef, accounting alone for 52% of Irish beef exports in 2014 (Bord Bia 2014). Thus any changes in beef production in Ireland are expected to impact on the origins of beef

consumed in the UK. As Brazilian beef production is the second largest on the planet (Ferraz and Felício 2010); the required substitute beef is assumed to be produced in Brazil and tropical rainforest to be converted into pasture (Smyth and Murphy 2011). However, it should be borne in mind that there is a huge uncertainty in this assumption because of the complexity of the relation between grass biomethane, global meat demand and supply, and demand for land. Increased demand for beef can be alone the main cause of Amazon deforestation (Smyth and Murphy 2011).

Both Searchinger et al. (2008) and Fargione et al. (2008) claim that ILUC emissions from Brazilian sugarcane ethanol are very high. However Brazilian research groups came out with a different conclusion suggesting that ILUC induced by sugarcane for ethanol is much lower (Moreira et al. 2012). Expansion of crop and pasture land in Brazil is occurring despite of sugarcane expansion (Meloni Nassar et al. 2008), and there is no evidence that deforestation is induced by sugarcane industry (Walter et al. 2011). Calculations by Moreira et al. (2012) were followed to estimate ILUC+ for sugarcane ethanol. This calculation is based on partial equilibrium economic model, Brazilian Land Use Model (BLUM) which includes geospatial data on land use and land availability in Brazil (ICONE 2012).

Following recent burning of forest for palm oil in Indonesia (Lamb 2015), it was considered that there is a risk of high indirect emissions from palm oil. The methodology of Fargione et al. (2008) was applied to estimate ILUC emissions from 1) Malaysian palm oil biodiesel, and 2) Brazilian sugarcane ethanol (conservative scenario). Conversion of natural ecosystems into energy crops or other crop causes CO_2 emissions from soils and from aboveground and belowground biomass. The amount of CO_2 released during the first 50 years after the conversion is called the carbon debt. Assuming that similar practices are needed to clear land for 1) palm oil plantation, and any other cropland in Indonesia or Malaysia, and 2) for sugarcane plantation and pasture in Brazil, the same debt was expected respectively for conversion 1) from peatland or lowland tropical rainforest to cropland (ILUC+ and ILUC-), and 2) from tropical rainforest to pasture (ILUC-). The emissions for each scenario were annualized.

To estimate the ILUC from corn ethanol both from and outside Europe, two scenarios were proposed: 1) converting grassland to cropland (Fargione et al. 2008), and 2)

converting forest and grassland worldwide to cropland (Searchinger et al. 2008). The conversion appears outside EU.

The ILUC estimates for domestic Miscanthus, willow, wheat ethanol and oilseed rape biodiesel were based on the assumption that adopting these energy crops in Ireland would lead to a need for replacing existing cropland. This was based on Danish calculation and modified GTAP model (Global Trade Analysis Project) estimating how much land and where (worldwide) will be converted, and what kind of biomes will be converted (Tonini et al. 2012). This methodology was applied also to imported biomass (Miscanthus and willow).

For EU and non-EU oilseed rape biodiesel, sugar beet and wheat ethanol, ILUC was estimated based on a global model simulation modelling biofuels consumption in Europe (Overmars et al. 2011). In this study ILUC emissions from various substrates sourced worldwide were averaged into two main types: bioethanol and biodiesel. These values were applied for the purpose of this study.

All the numerical assumptions as implemented in the Irish TIMES model are summarized in Table 4.1.

Table 4.1 Assumed domestic and imported LUC emission factors (g CO_2/MJ).

Origin	**Commodity**	**DLUC**	**ILUC+**	**ILUC-**	**Unit**
Domestic	Wheat ethanol	0.0[a]	70.0[b]	130.0[b]	g CO_2/MJ
	Miscanthus crop for biomass	-11.3[c]	70.0[b]	130.0[b]	g CO_2/MJ
	Willow crop for biomass	-28.5[c]	70.0[b]	130.0[b]	g CO_2/MJ
	Forestry residues	N/A	N/A	N/A	g CO_2/MJ
	Grass biogas	-16.8[d]	0.0[e]	625.9[f]	g CO_2/MJ
	Oilseed rape biodiesel	0.0[a]	70.0[b]	130.0[b]	g CO_2/MJ
	Agricultural waste and residues (dry)	N/A	N/A	N/A	g CO_2/MJ
	Landfill gas	N/A	N/A	N/A	g CO_2/MJ
	Municipal waste (MSW)	N/A	N/A	N/A	gCO_2/MJ
	Recycled vegetable oil biodiesel	N/A	N/A	N/A	g CO_2/MJ
	Agricultural slurries	N/A	N/A	N/A	g CO_2/MJ
	Wood processing residues	N/A	N/A	N/A	g CO_2/MJ

Imported	EU imported corn ethanol	0.0[a]	28.1[g]	104.0[g]	g CO_2/MJ
	EU imported sugar beet ethanol	0.0[a]	26.0[h]	154.0[h]	g CO_2/MJ
	EU imported wheat ethanol	0.0[a]	26.0[h]	154.0[h]	g CO_2/MJ
	non-EU imported corn ethanol	0.0[i]	28.1[g]	104.0[g]	g CO_2/MJ
	non-EU imported sugar beet ethanol	26.0[h]	26.0[h]	154.0[h]	g CO_2/MJ
	non-EU imported sugarcane ethanol	0.0[j]	6.5[j]	246.5[g]	g CO_2/MJ
	non-EU imported wheat ethanol	0.0[i]	26.0[h]	154.0[h]	g CO_2/MJ
	EU imported oilseed rape biodiesel	0.0[a]	26.0[h]	154.0[h]	g CO_2/MJ
	EU imported tallow biodiesel	N/A	N/A	N/A	g CO_2/MJ
	EU imported used cooking oil biodiesel	N/A	N/A	N/A	g CO_2/MJ
	non-EU imported oilseed rape biodiesel	26.0[h]	26.0[h]	154.0[h]	g CO_2/MJ
	non-EU imported palm oil biodiesel	117.4[k]	117.4[k]	577.3[k]	g CO_2/MJ
	non-EU imported tallow biodiesel	N/A	N/A	N/A	g CO_2/MJ
	non-EU imported used cooking oil biodiesel	N/A	N/A	N/A	g CO_2/MJ
	EU imported biomass (Miscanthus and willow)	-14.0*[l]	70.0[b]	130.0[b]	g CO_2/MJ
	non-EU imported biomass (Miscanthus and willow)	-14.0*[l]	70.0[b]	130.0[b]	g CO_2/MJ

* Average of Miscanthus and willow

[a] (Börjesson and Tufvesson 2011; Klenk et al. 2012); [b] (Tonini et al. 2012); [c] (BioGrace 2015), in line with (Styles and Jones 2007); [d] (BioGrace 2015); [e] (McEniry et al. 2013; Wall et al. 2013); [f] (Fargione et al. 2008; Smyth and Murphy 2011); [g] (Fargione et al. 2008; Searchinger et al. 2008); [h] (Overmars et al. 2011); [i] (Sinistore 2012); [j] (Walter et al. 2011; Moreira et al. 2012); [k] (Fargione et al. 2008); [l] (Tonini et al. 2012)

4.2.2. Irish TIMES

This paper is based on scenario analysis using the Irish TIMES energy systems model (Ó Gallachóir et al. 2012). The Irish TIMES model provides a range of energy system configurations for Ireland. Each of them delivers projected energy service demand requirements optimised to least cost and subject to a range of technical and policy constraints for the period out to 2050. It provides a means of testing energy policy choices and scenarios, and assessing the implications i) for the Irish economy (including technology choices, prices, output), ii) for Ireland's energy mix and energy dependence, and iii) for the environment, with a particular focus on GHG emissions. It is used both to examine baseline projections, and to assess the implications of

emerging technologies and of mobilising alternative policy choices such as meeting renewable energy targets and carbon mitigation strategies.

The Irish TIMES model was originally extracted from the Pan European TIMES (PET) model and then updated with improved data based on much extensive local knowledge. Extensive description and details on modelling structure and approach may be found in (Ó Gallachóir et al. 2012).

4.2.3. Key model sets and assumptions in TIMES

The Irish TIMES model used in this analysis has a time horizon of 65 years that ranges from 2005, the base year, to 2070, with a time resolution of four seasons with day-night time resolution, the latter comprising day, night and peak time-slices (Ó Gallachóir et al. 2012). Energy demands are driven by a macroeconomic scenario, which is based on the ESRI HERMES macroeconomic model of the economy (FitzGerald et al. 2013), with key drivers extended to the period 2070. Fossil fuel prices are based on IEA's current policy scenario (IEA 2012). Based on work undertaken by Ireland's transmission system operator EirGrid (EirGrid 2010)[1], the level of variable and non-synchronous renewable generation (wind, solar and ocean energy) is limited here to a maximum share of 70% of electricity generation within each time slice, and to 50% at annual level to account for operational issues associated with such high levels of variable generation in the power system. Regarding policies, investment subsidies and feed-in-tariffs for renewables based on policies currently in practice are assumed here to continue until 2030 and no trading of green certificates is assumed. The installation of new coal power plant capacities are limited to the replacement of current capacity levels, while for wind a maximum installation rate is set at 750 MW_e per year.

The domestic bioenergy resources are represented in the model by 12 different commodities. The total resource capacity limit for domestic bioenergy – considering both available and technical potential – has been set at 2887 ktoe (120.9 PJ) for the year 2030 and at 3805 ktoe by 2050, based on the estimates from SEAI (2010) for agricultural dry residues, algae and municipal wastes; Smyth et al. (2010) and SEAI

[1] Ireland's Transmission System Operator (TSO).

(2010) for biogas from grass; Phillips (2011) for forestry; Clancy et al. (2012) for agricultural wet residues, recycled vegetable oils, oil seed rape and wheat.

The cost assumptions for domestic bioenergy commodities are based on McEniry et al.(2011) for biogas from grass, Kent et al. (2011) for forestry, Clancy et al. (2008) for willow and Miscanthus crops and delivery costs, and Clancy et al. (2012) for wheat crops, oil seed rape and recycled vegetable oil. For the remaining commodities, the cost assumptions used in the PET model within the RES2020 project were used (RES2020).

The import locations of bioenergy are not explicitly modelled in Irish TIMES. However for the purpose of this paper the model formulation has been expanded to distinguish between import locations (i.e. within the EU and outside the EU) and feedstocks. Cost projections for these sources are based on international trends in Clancy et al. (2012). Both the potentials and costs assumptions for each individual commodity are summarized in Chiodi et al. (2015a). Non-energy GHG emissions related to growth of energy crops (although not considered in this paper) are also modelled, based on inputs from the FAPRI-Ireland model (Donnellan et al. 2013). Additional information regarding the main input assumptions may be found online (Energy Policy and Modelling Research Group 2015).

For the purpose of this study a cluster of four scenarios was tested. Each mitigation scenario assumes identical emissions reduction trajectory in which the energy system is required to achieve at least 80% CO_2 emissions reduction below 1990 levels by 2050. The pathway includes interim targets in line[2] with the EU 2020 climate energy package (EC 2009b; EC 2009c), i.e. 20% CO_2 emissions reduction by 2020 relative to 2005 levels. Non-energy GHG emissions are assumed to be in line with EPA projections (EPA 2014b), while over the period 2030-2050 they are assumed to be constant. The effects of renewable targets and caps on first generation biofuels (EC 2015b) have not been considered in this analysis. Thus this model allows ethanol and biodiesel from food crops. Each scenario introduces different levels of life cycle emissions related to bioenergy production and land use changes. The impact of

[2] Not with the ETS / non-ETS split

transportation of imported biofuels from various source countries was not included in the analysis, while domestic transportation system is explicitly described in the model.

The CO2-80 scenario includes all emissions generated in the energy system from combustion and industrial processing, while land use change emissions generated from cultivation of bioenergy sources are excluded. The CO2-80 DLUC scenario simulates a policy development, where all direct land use change emissions generated from cultivation of bioenergy sources are included in the mitigation targets. For imported bioenergy sources, DLUC emissions are applied to the country, which uses these fuels. The CO2-80 ILUC+ and CO2-80 ILUC- scenarios simulate a policy scenario in which DLUC and ILUC emissions generated from both domestic and imported bioenergy are included in Ireland's mitigation targets.

4.3. Results

4.3.1. Implications of including land use changes towards GHG targets

Four mitigation scenarios were tested: CO2-80, CO2-80 DLUC, CO2-80 ILUC+ and CO2-80 ILUC-. In terms of sectorial emissions shares, by 2030 all scenarios indicate similar mitigation pathways, while by 2050 differences in "effort sharing" between sectors are more pronounced (Fig. 4.2). The CO2-80 DLUC indicates high net reduction from agriculture, which is driven by production of bioenergy crops, such as grass, with negative emissions factors. This leaves room for electricity generation sector to emit more (over 30% of total emissions). The two ILUC scenarios show higher emissions shares from industry (up to 18%) and transformation sectors (up to 6%), while deeper reductions as compared to CO2-80 are found in transport (up to 15% of total emissions).

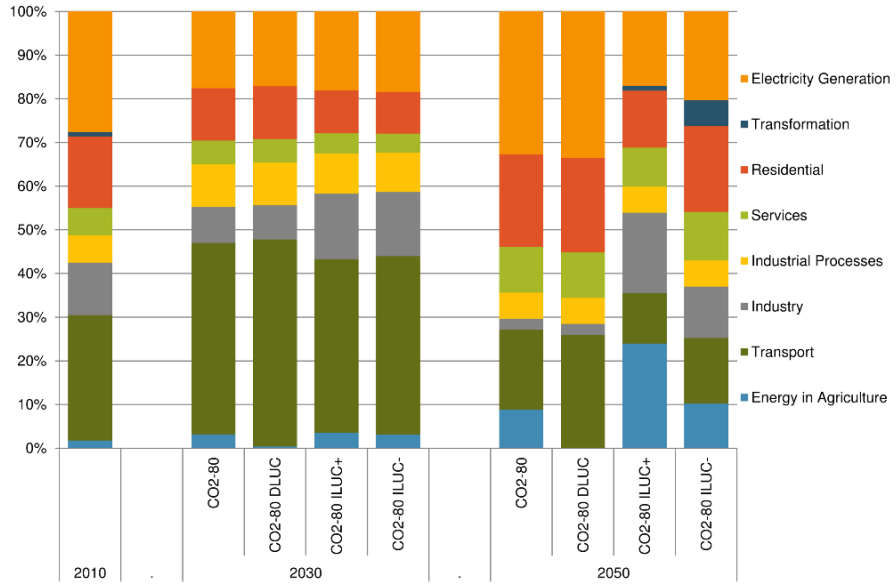


Fig. 4.2 Sectorial emissions shares in four mitigation scenarios: CO2-80, CO2-80 DLUC, CO2-80 ILUC+, CO2-80 ILUC- (%).

Total primary energy requirement (TPER) was compared for each scenario (Fig. 4.3). Results show that by 2050 ILUC emissions lead to a decrease in the overall bioenergy share (biogas, bioliquids and solid biomass). The introduction of ILUC emissions affects mainly biomass and bioliquids. Biogas share increases (8% of TPER for ILUC+) or decreases (0.4% for ILUC-) depending on the specific assumptions made on ILUC emissions. As the bioenergy share drops the role of other renewables increases (18% for ILUC-). The role of fossil fuels also decreases, although variably depending on fuels. By 2050 oil use will drop by two-thirds for all mitigation scenarios as compared with 2030, whereas coal will be used only in combination with carbon capture and storage (CCS) facilities in the cement sector, and to produce hydrogen. Gas consumption will remain almost the same until 2030, and then will slightly decline by 2050 in the CO2-80 and CO2-80 DLUC scenarios. In contrast, ILUC scenarios indicate a further increase in gas use up to 43% by 2050, mostly to meet an increase in electricity demand. 80% of used gas is equipped with CCS.

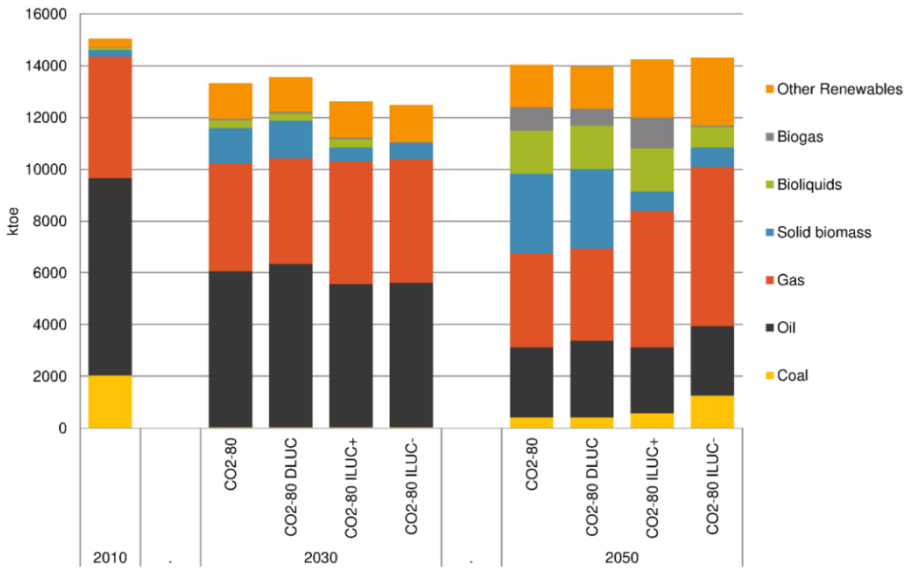


Fig. 4.3 TPER in the four mitigation scenarios (ktoe).

Biofuels consumption decreases drastically for ILUC- scenario with highest input coming from imported waste and residues based biofuels, mainly tallow and used cooking oil biodiesel (Fig. 4.4). Indigenous grass biomethane plays an important role for three other mitigation scenarios. Table 4.2 provides additional insights on the projection of bioenergy sourcing. The inclusion of high LUC emissions leads to strong replacement of land-based feedstocks by waste and residues. In CO2-80, DLUC and ILUC+ scenarios, biofuels are sourced mainly from a mixture of imported sugarcane, sugar beet and oil seed rape, and domestic grass. By 2050 grass biomethane represents up to 31% of total bioenergy consumption (ILUC+). Domestic waste and residues such recycled oil, agricultural residues, MSW, slurries and landfill gas also play a role. Miscanthus and willow use is reduced to zero for both ILUC scenarios, and their role in heat generation is replaced by forest and agricultural residues, and electricity.

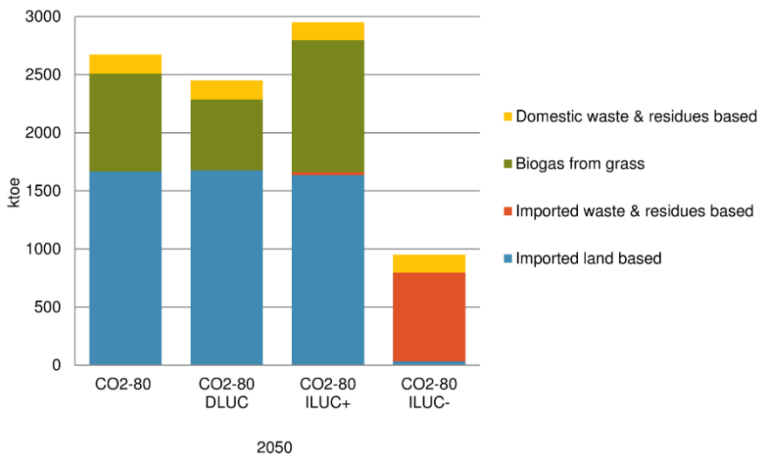


Fig. 4.4 Biofuels consumption by origin in 2050 (ktoe).

Table 4.2 Projected primary bioenergy consumption by origin resulting from modelling in Irish TIMES (ktoe).

	Commodity	Origin	2010	CO2-80		CO2-80 DLUC		CO2-80 ILUC+		CO2-80 ILUC-	
				2030	2050	2030	2050	2030	2050	2030	2050
Imported	Ethanol	Corn	0	71	0	267	0	0	0	0	34
		Sugarbeet	71	204	0	0	906	0	0	0	0
		Sugarcane	22	0	906	0	0	285	906	0	0
		Wheat	0	0	0	0	0	0	0	0	0
	Biodiesel	Oil Seed Rape	0	0	762	24	771	29	730	0	0
		Tallow	0	0	0	0	0	0	0	0	764
		Used Cooking Oil	0	0	0	0	0	0	24	0	0
		Palm Oil	0	24	0	0	0	0	0	0	0
	Biomass	Miscanthus/Willow	0	877	1898	877	1882	0	0	0	0
Domestic	Biodiesel	Recycled Oil	0	2	2	2	2	2	2	2	2
		Oil Seed Rape/Algae	1	0	0	0	0	0	0	0	0
	Biomass	Miscanthus	6	160	353	157	353	0	0	0	0
		Willow	6	5	79	85	79	0	0	0	0
		Forestry/Agri Residues	131	305	651	305	651	517	651	517	651
	Biogas	Landfill Gas and Other	26	57	57	57	57	57	57	57	57
		Grass	0	0	841	0	610	0	1136	0	0
	Biogas	MSW and Slurries	40	31	104	31	104	29	94	91	92
Total			**304**	**1736**	**5654**	**1805**	**5415**	**919**	**3601**	**668**	**1601**

Renewable energy consumption was compared by mode: electricity (RES-E), transport (RES-T) and heat (RES-H) (Fig. 4.5). In all scenarios bioenergy is the main renewable energy source for both transport and heat, with an important input from bioethanol and biodiesel. Wind energy dominates the electricity generation sector and increases up to 86% of RES-E, if ILUC emissions are included. By 2050 renewables represent 55%[3] of Irish gross energy consumption for CO2-80 scenario, and 53% for CO2-80 DLUC. For CO2-80 ILUC+ and CO2-80 ILUC- this value decreases to 48% and 35% respectively, as the overall bioenergy contribution drops.

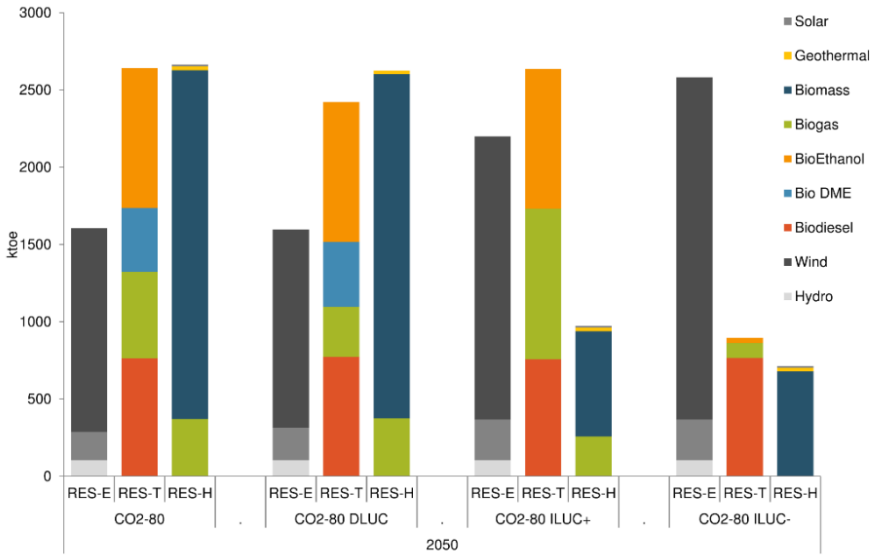


Fig. 4.5 Renewable energy consumption by mode (ktoe).

ILUC scenarios show a marked increase in electricity generation with an increased electrification in industry, residential (heat) and transformation sector (Fig. 4.6). Centralized electricity (from wind and gas CCS) is used in the transformation sector to produce hydrogen through electrolysis, making up 11% of total electricity consumption (ILUC-). This may be termed power to gas and may also enable reduced wind energy curtailment, i.e. be viewed as a storage mechanism for intermittent electricity produced when demand is low, whilst simultaneously changing the energy vector from electricity to gas (Persson et al. 2014). Together with renewable fuels such as biofuels and renewable electricity, hydrogen becomes a key fuel in transport by 2050 and is used mainly in freight sector (Fig. 4.7). Renewable electricity dominates

[3] Including international aviation

private transport (hybrid cars and EVs). Technically, this is possible, however such a high penetration of wind in electricity generation might pose practical problems and will be very challenging.

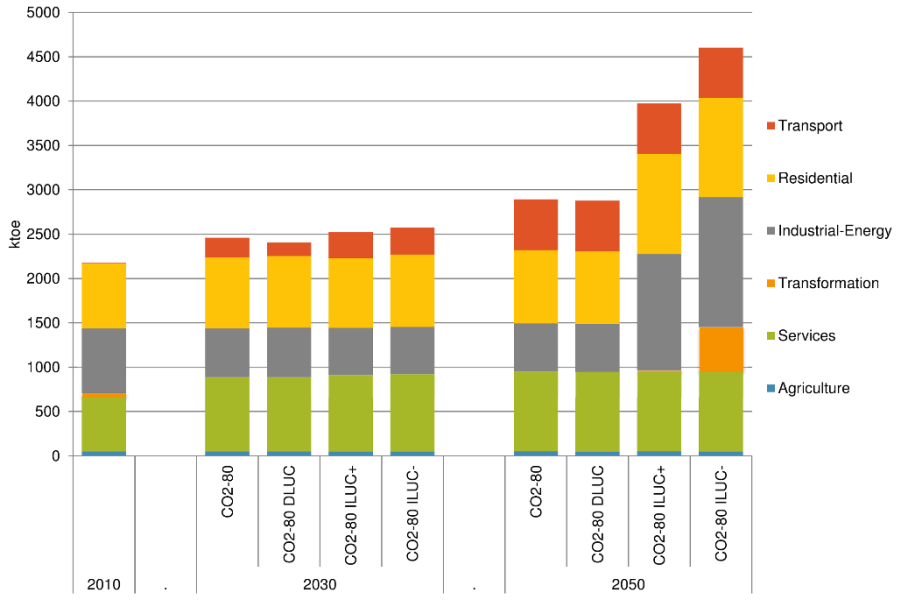


Fig. 4.6 Electricity consumption by sector (ktoe).

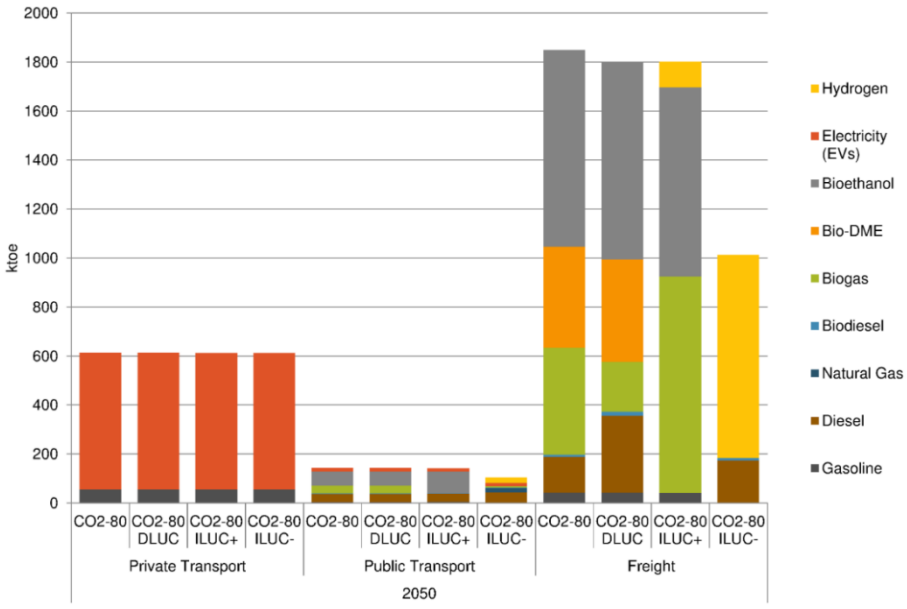


Fig. 4.7 Transport final energy consumption by mode (private, public and freight) in 2050 (ktoe).

4.3.2. Cost of CO_2 abatement with LUC-preventing policy in place

One of the main insights gained from energy system modelling such as TIMES is in quantifying the impact of different mitigation targets on marginal CO_2 abatement costs, i.e. indicative costs of abating the last tonne of CO_2.

Under the CO2-80 scenario, the marginal cost increases from €58/tonne to €213/tonne between 2020 and 2040 (Fig. 4.8). By 2050, the marginal abatement cost will grow to €469/tonne, which underscores how challenging the mitigation targets are. The CO2-80 DLUC scenario indicates slightly lower CO_2 abatement costs of €448/tonne by 2050. The CO2-80 ILUC+ and CO2-80 ILUC- both show higher CO_2 abatement prices due to reduced availability of low emissions bioenergy resources already by 2030. For these scenarios, the 2050 marginal CO_2 abatement cost reaches €706/tonne and €754/tonne respectively, illustrating how ILUC emissions influence the achievement of this challenging target.

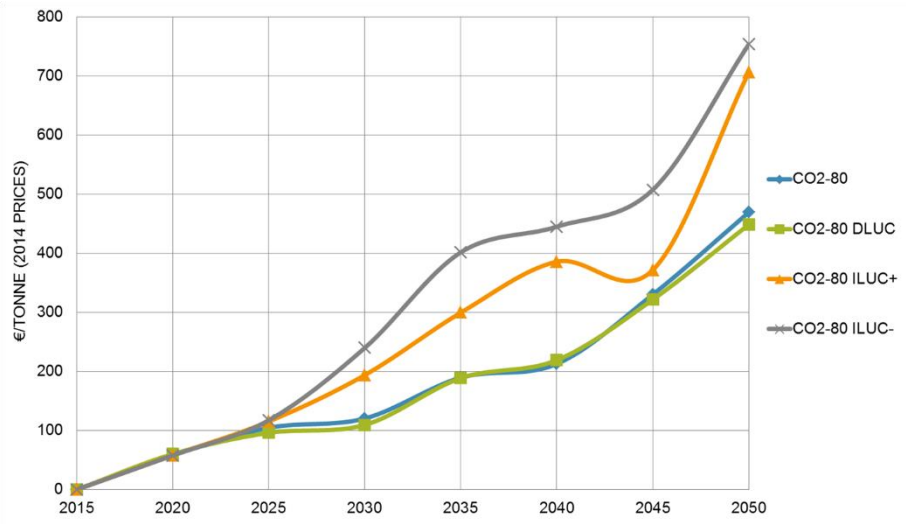


Fig. 4.8 CO_2 shadow prices (€/tonne (2014 prices)).

4.4. Discussion and conclusions

With little surprise, the impact of incorporating land use change emissions on Ireland's energy transition is high. The most constraining scenario appeared to be the conservative CO2-80 ILUC-. Applying high ILUC emissions to the Irish energy system results in a drop of 72% in primary bioenergy consumption (biogas, bioliquids and solid biomass) and switching to imported tallow biodiesel (50% of total

bioenergy) by 2050. 66% of transport fuel[4] comes either from hydrogen produced through electrolysis (power to gas) or residues-based biodiesel and renewable electricity (in form of hybrid cars and EVs). High penetration of wind in electricity generation will be very challenging as a secure and stable operation of the power system must be ensured. There is also a need for a strong incentive to encourage customers switching to hybrid cars and EVs (McCollum et al. 2015). Decrease in bioenergy also causes a rise in fossil fuels, principally coupled with CCS.

These advanced technologies are not mature yet, and their future development and deployment are considered very costly and uncertain. This makes the most conservative scenario also the most expensive. By 2050, the marginal CO_2 abatement cost increases by up to 68% as compared to the CO2-80 DLUC scenario. Yet this is still much below the abatement costs as in Chiodi et al. (2015a); in which a scenario with a strong cap on imported bioenergy, led to abatement cost in the range of €1400/tonne, twice the price obtained for ILUC- in this paper.

Results are very different if the optimistic scenario is considered. Domestic grass biomethane becomes the major biofuel with 31% of total bioenergy consumption projected for 2050. Growing grass for biomethane would require only 5-11% of Ireland's agricultural land, depending on grass yield per hectare, to deliver biomethane projected in ILUC+ scenario (yields according to Murphy et al. 2011). This is in line with recent works in Ireland. Permanent grasslands are abundant in Ireland and McEniry et al. (2013) and Wall et al. (2013) argue that there is more than enough grass to sustain demands for both agriculture and biomethane production. As little as 1.1% of grassland would generate 10% of renewable energy in transport, if co-digested with dairy slurry (Wall et al. 2013). Biomethane is deemed to be a solution for an indigenous advanced biofuel and achieving the RES-T target (Singh et al. 2010). Biomethane potential from feedstocks such as the organic fraction of municipal waste, agricultural and slaughter waste, and excess grass is such that together they can fuel approximately 25% of Irish private cars (around 0.44 million cars) (Thamsiriroj et al. 2011).

DLUC emissions applied in this study are not constraining bioenergy development and lead to almost 40% of bioenergy in primary energy by 2050, with the majority

[4] Excluding international aviation

coming from imported land-based feedstocks. Future work should include calculation of specific DLUC emissions for each energy crop.

The main limits of the present study lie in the calculation methodology for ILUC. At this point, it is difficult to apply a sound ILUC calculation method since a common methodology does not yet exist. The numbers presented in this paper are not definitive and further work will be required to refine modelling and calculation methodologies. In addition, it would be interesting to apply this approach using consumption-based emissions accounting rather than production-based emissions accounting. This is particularly relevant for the inclusion of ILUC emissions associated with imported biofuels.

The preliminary results presented in this paper provide interesting insights for policymakers. Direct and indirect land use changes have a clear impact on energy systems and how to achieve 2050 GHG reduction target. Policymakers will have a huge impact on development of future ILUC methodology. The amendment to the RED put a 7% cap on first generation biofuels towards 2020 targets. An adequate measure to prevent ILUC seems to put a cap on land-based bioenergy and biofuels also beyond 2020, and to provide a stronger financial support for advanced biofuels that do not compete for land with food or feed. However, such a cap removes the technical equality between biofuels. Not all biofuels are equal if LUC impacts are considered, although the local and regional factors should be factored in, especially as some countries have large potential of unused marginal and degraded land with low ILUC risk. The efforts should be in targeting this land that can be recovered for cultivation of bioenergy crops.

If ILUC impact is to be counted in, there is a need for a sound calculation methodology, though achieving this might not be possible in near future This must stay in line with country bioenergy and agricultural potentials; like for example in Ireland, there is sufficient permanent grassland to sustain both biofuel and the livestock sector. High ILUC leads also to very expensive technologies, such as hydrogen and CCS that are still under development. In the input assumptions used here in Irish TIMES, they are considered to be available for deployment by 2050, however there is no guarantee for this to happen

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Chapter 5. Small-scale upgrading of landfill gas to biomethane for use as a cellulosic transport biofuel

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Abstract

Landfill gas (with methane content of 35-55%) adds significantly to global warming if released to the atmosphere. Under the EU Landfill Directive, all landfill sites are obliged to collect this gas if supplied with biodegradable municipal waste. Landfill gas can be i) flared, ii) combusted to produce electricity, or iii) upgraded to biomethane. The last scenario is of special interest: in the US landfill gas is now classified as a cellulosic biofuel; in the EU there is a mandatory target of 10% share of renewables in transport by 2020. A significant challenge for upgrading landfill gas (LFG) to biomethane is the high nitrogen content resulting from negative pressure in the landfill. Cost analyses were conducted to compare three technology solutions for landfill gas upgrading with accessible landfill gas flow of 250-500 m^3/h. If injection to the transmission grid is considered, then a single step PSA system may be viable. The optimal solution suggested is an on-site service station; the cost of this system (including propane addition and service station) was assessed as €0.84/m_n^3 LFG biomethane. This required a subsidy of €0.55/m_n^3; this is available in Ireland under the Biofuel Obligation Certificate scheme.

Keywords: landfill gas, upgrading to biomethane, cost analysis, small-scale landfill sites

5.1. Introduction

5.1.1. Rationale for collection of landfill gas

Landfill sites are a significant source of anthropogenic greenhouse gas (GHG) emissions. Landfill gas (LFG) is formed by the microbial decomposition of the biodegradable fraction of waste. The gas may contain CH_4 (35-55%), CO_2 (15-50%), N_2 (5-40%), O_2 (0-5%), H_2 (0-3%), H_2S (0-100 ppm), as well as a number of trace gases, including volatile organic compounds (VOCs), halogenated compounds and siloxanes (Deed et al. 2004; Rasi et al. 2008b; Petersson and Wellinger 2009). The energy content of LFG is typically in the range 11-23 MJ/m_n^3 (Kaparaju and Rintala 2013).

If released to the atmosphere, CH_4 has a global warming potential 30 times higher than that of CO_2 (IPCC 2014). Therefore, the capture of landfill gas is essential to help limit the effects of climate change. Under the EU Landfill Directive, landfill operators are obliged to collect landfill gas from all landfills receiving biodegradable municipal waste (EC 1999). In 2009, with the exception of one site, all open landfills in Ireland either employed flaring or utilised the landfill gas for energy (McCarthy et al. 2010).

The number of landfill facilities receiving waste continues to decline in Ireland, from 30 active facilities in 2009 to 7 in 2015 (McCarthy et al. 2010; Murphy 2015). There are over 400 closed landfills (McCarthy et al. 2010), however a number of these still continue to produce useful gas. Gas Network Ireland (GNI) are the owners of the gas infrastructure in Ireland. They proposed targets of 5% and 20% of Renewable Natural Gas (RNG) in the Irish gas grid by 2020 and 2030 respectively (GNI 2015). According to their calculations LFG can represent as much as 20% (880 GWh per annum) of total renewable gas available in Ireland in 2014 (GNI pers. com.).

Landfill gas collection systems typically comprise a series of pipes penetrating into the landfill. The gas is collected in these pipes and directed to a central collection point by a blower for further treatment (Li et al. 2014). Table 5.1 presents typical CH_4 recovery values. Some of the CH_4 escapes to the atmosphere, migrates laterally, is stored internally in the landfill or is oxidised. An engineered landfill site with 1 million tonnes of biodegradable waste, may produce several hundred m_n^3 of LFG per hour (Spokas et al. 2006).

Table 5.1 CH_4 recovery rates from landfills (Spokas et al. 2006).

Recovery system type	**CH_4 recovery %**
Operating cell with active LFG recovery system	35
Temporarily covered cell with active LFG recovery system	65
Cell with final clay cover and active LFG recovery system	85
Cell with geomembrane final cover and active LFG recovery system	90

There are three main ways of using landfill gas. Flaring is the simplest way. CH_4 is combusted ($CH_4 + 2O_2 = CO_2 + 2H_2O$) and CO_2, with a far lower global warming potential, is released to the atmosphere. Landfill gas may be used to generate electricity via internal combustion engines, gas turbines or steam turbines. Alternatively, LFG can be upgraded to biomethane and used as a transport fuel.

Emissions need to be limited to meet environmental regulations, including for odour and methane emission (Jaramillo and Matthews 2005). A negative pressure gradient should be maintained to minimise gas migration; however, this pulls air into the landfill, reducing the energy content of the LFG (Cronin et al. 2008).

For energy generation, a "cleaning" step must be carried out to remove contaminants, including O_2 and N_2, halogenated compounds, H_2S and siloxanes. Halogenated compounds, H_2S and other sulphur gases can cause chemical corrosion of the engine and result in the emission of acidic gases. Siloxanes can cause a build-up of silicon deposits on critical components in the engine, such as pistons, cylinder heads and valves (Deed et al. 2004). In order to use LFG as a transport fuel, a further "upgrading" step must be applied to remove CO_2 (Fig. 5.1). There are at least 27 LFG upgrading facilities in operation globally (IEA 2014). Eight of these upgrade to transport fuel standard; the remainder inject to the gas grid.

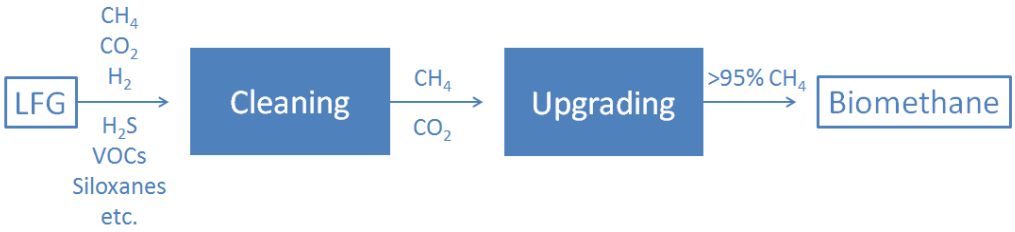


Fig. 5.1 Route from LFG to biomethane.

5.1.2. LFG cleaning and upgrading techniques available

LFG Cleaning

Activated carbon adsorption is the most commonly used approach for LFG cleaning (Ajhar et al. 2010). However activated carbon tends to have a relatively low capacity for the most harmful components in landfill gas, including siloxanes. Siloxanes form microcrystalline silicon dioxide during combustion, which forms abrasive layers on engine parts (Rasi et al. 2008a). Siloxanes are removed by adsorption (on aluminium or silica gel), absorption (in liquid mixtures of hydrocarbons) and deep-chilling (Petersson and Wellinger 2009; Ajhar et al. 2010).

VOCs in LFG have a high impact on human health when released in exhaust gas (Kampa and Castanas 2008). During combustion, organosulfur and organochlorine compounds form H_2SO_4 and HCl which cause high rates of corrosion of the combustion chamber (Allen et al. 1997). Techniques for VOC and H_2S removal include adsorption (on activated carbon or zeolite), chemical absorption, membrane separation as well as biological destruction using microorganisms (Khan and Kr. Ghoshal 2000; Petersson and Wellinger 2009).

Both nitrogen and oxygen are difficult and expensive to remove. Techniques for nitrogen removal include activated carbon, molecular sieves or membranes (Petersson and Wellinger 2009). The most promising technology is pressure swing adsorption (PSA) (Mitariten 2007).

LFG Upgrading

CO_2 has higher solubility in water than methane, which allows separation in the adsorption column. CO_2 is removed using: scrubbing systems such as water and organic physical scrubbing; membranes; adsorption units such as PSA and vacuum pressure swing adsorption (VPSA); and combinations of membrane and PSA or VPSA. Membrane and PSA systems are the most commonly used (Mitariten 2007; Lokhandwala et al. 2010).

Organic physical scrubbing uses an organic solvent such as polyethylene glycol instead of water. In chemical scrubbing, CO_2 is not only absorbed but also reacts with the amine solutions. However none of these methods can remove nitrogen from LFG. Membrane systems can produce methane streams with purity in excess of 97%,

however do not allow for nitrogen to be separated from methane (Petersson and Wellinger 2009).

Adsorption systems can produce high purity methane streams and are suitable for LFG with high nitrogen and oxygen content (Sircar et al. 1988; Mitariten 2007). They can also remove impurities such as siloxanes and VOCs (Mitariten 2007; Xebec 2007). In an adsorption process, the contaminant gas (such as nitrogen) is physically removed from a gas (such as LFG) by spontaneous adhesion to the surface of a very porous adsorbent (Xebec 2007; Grande 2012). There are three adsorption techniques worth mentioning for LFG upgrading: PSA, VSA (vacuum swing adsorption) and VPSA. PSA always operates at pressures greater than atmospheric. Adsorption of gas molecules is performed under elevated pressure until the adsorbent is saturated, then the pressure is decreased in order to regenerate the adsorber (Petersson and Wellinger 2009; Grande 2012; Hedin et al. 2013). In VSA units, adsorption is conducted under ambient or near-ambient pressure, and a blower is used to regenerate the saturated adsorber as it draws out the adsorbed gas molecules (Hedin et al. 2013). VPSA is a hybrid system that uses both elevated pressure during the adsorption phase and vacuum pump for regeneration (Linde AG).

The VPSA initial capital investment seems to be higher than PSA or VSA since it requires both a compressor and a blower, however, companies offering installation of VPSA systems claim that payback can be as low as one year depending on scale (Adsorptech Inc. 2014).

Post-upgrading removal of nitrogen may be achieved by cooling, as the boiling point of nitrogen is significantly lower than that of methane.

5.1.3. Cost analysis in literature

There are few examples of cost analysis of LFG projects available in the literature. Of these, the majority concentrate on the environmental benefits of LFG recovery (Chaya and Gheewala 2007; Wanichpongpan and Gheewala 2007) and at sites with high gas flows (Jaramillo and Matthews 2005). Starr et al. compared various applications of biogas from three landfills in Spain (250 m_n^3/h, 1000 m_n^3/h and 5000 m_n^3/h) (Starr et al. 2015). A new upgrading technology using carbon mineralization, called alkaline with regeneration (AwR) was considered for upgrading of pre-cleaned landfill gas. For the smallest landfill, the burning of LFG to produce electricity brings a profit of

€214,000 per annum, while upgrading to biomethane brings losses in range of 2-3 million € per annum. The feed-in tariffs were not factored in.

Jaramillo et al. analysed LFG-to-energy projects (LFGE), in which LFG was used to generate electricity, from social and economic perspectives (Jaramillo and Matthews 2005). Three technologies were compared: internal combustion engines; gas turbines; and steam turbines. These options were examined for three different landfills, with gas flows of 750, 3525 and 6213 m_n^3/h respectively. The study included: cost of installation and maintenance of gas collection and electricity generation equipment; revenues from electricity generation; valuation of GHG emissions prevented by LFG recovery system; and emissions of pollutants from electricity generation. The authors found that the breakeven price of electricity is less than €0.036/kWh if provided with a government subsidy of €0.0077/kWh (Jaramillo and Matthews 2005).

Boodhan analysed potential LFGE projects in Trinidad and Tobago based on three landfill sites, with gas flows ranging from 142 to 464 m_n^3/h (Boodhan 2014). The study used LFGcost–Web, an energy cost model for economic feasibility analysis of LFGE projects (US EPA 2014). The analysis included capital and operational costs from three electricity generating technologies (internal combustion engines, gas turbines and micro-turbines). Using the internal rate of return to assess the investments only the smallest site with either internal combustion engines or gas turbines was economically viable (Boodhan 2014).

R.W. Beck Engineers produced a study comparing three LFG systems: electricity generation; direct thermal application; and upgrading to biomethane (R.W. Beck 2010). Capital costs (CAPEX) of biomethane production were estimated at €1,932 per m_n^3 of LFG per hour; operation and maintenance costs (OPEX), including for power consumption, were €0.14/m_n^3 of biomethane (62% of CAPEX). The additional capital costs to process as a transport fuel, including for compression, a fuelling station and biomethane storage, was assessed as €1,800,000 regardless of the flow. The additional OPEX would amount to €360,000 per annum (20% of CAPEX). The study concluded that for the landfill sites without a gas collection system already in place, the net present value of all options examined was negative. If a gas collection system was already in place, upgrading to biomethane was feasible, although the return would be lower than if the gas was to be used for electricity generation on-site. The study also

underlined the importance of well-designed LFG collection systems to prevent oxygen and nitrogen from slipping into the landfill.

The Environment Agency in Britain concluded that the economics of LFG upgrading are "marginal at best" (Deed et al. 2004). This is in line with findings by Starr et al. and R.W. Beck Engineers (R.W. Beck 2010; Starr et al. 2015). However, while LFG-to-energy projects costs are high, they can be mitigated by investment tax credits, sales tax exemption, low-interest rates and grants (Li et al. 2014); such measures have led to an increase in LFGE projects in the US.

5.1.4. Rationale for research

The upgrading of LFG to biomethane is still considered an emerging technology by the US Environmental Protection Agency (EPA) however as of 2014 LFG is considered a cellulosic transport biofuel in the US (Bracmort 2014; Foody 2014). This should see a major impetus in LFG biomethane industry in the US. There is still little scientific data published on the economic viability of LFG systems producing transport biofuel. In Ireland, in 2013, 97.5% of total oil demand (dominant use in transport) was served by imported oil products (Dineen et al. 2014). LFG biomethane could play a role as an indigenous renewable transport fuel.

5.1.5. Aims and objectives

This paper examines the potential of LFG upgrading to biomethane for use as a transport fuel at four small existing landfill sites in Ireland. Currently, the LFG at three of the four sites is flared, with a CHP unit at the fourth site. The aim of this paper is to highlight optimum methods and costs of upgrading LFG. The objectives are to:

- Analyse the costs of upgrading LFG from existing small scale landfills;
- Recommend the optimal LFG upgrading technology.

5.2. Methodology

5.2.1. Description of landfill sites

The four landfill sites assessed are now closed and no longer accepting waste. These sites are sites are middle-sized in comparison to other landfills in Ireland (McCoole et al. 2014). The projected drop-off in gas production over the coming years is presented

in Fig. 5.2. Site 1 stopped accepting waste in the summer of 2014 and has 77 gas wells. The spike in available LFG flow seen in Fig. 5.2 is a result of the sealing of previously active cells when the site closed. It is currently the only site generating electricity from the LFG collected. A 330 kW generator is used, with the electricity exported to the national grid. Sites 2, 3 and 4 have 56, 64 and 62 gas wells each respectively and all were permanently closed in the last five years. The LFG collected from these sites is currently flared.

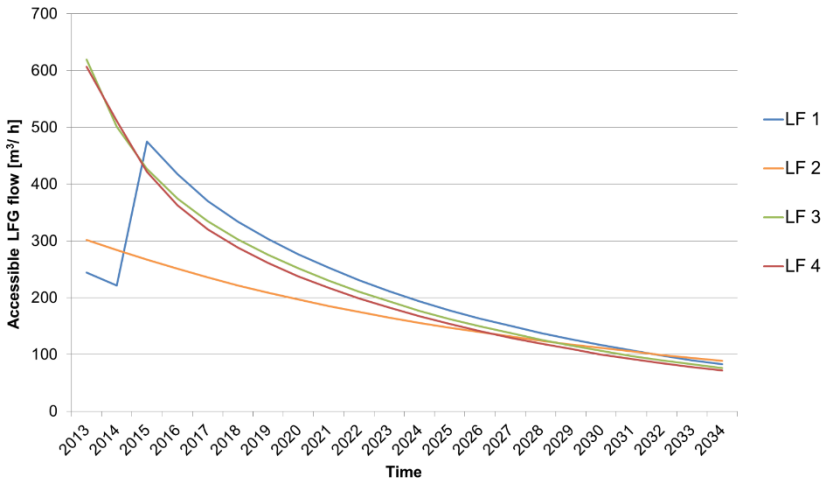


Fig. 5.2 LFG production rates over the lifetime of the facilities.

Table 5.2 Typical gas composition from sites in 2014.

Component	**Content**
CH_4	43-48%
CO_2	35-45%
N_2	15-30%
O_2	1-3%
Organic fluorine	<1.0 mg/m^3
Organic chlorine	<1.0 mg/m^3
Organic silicon	2-5 mg/m^3
Total sulphur	<2.0 mg/m^3

Nitrogen and oxygen levels in the LFG at these sites are very high (Table 5.2). The collection systems were only installed after the sites had been operational for some time, and as a result the gas quality is relatively poor.

5.2.2. Evaluation of upgrading options

Initially three different upgrading options were considered for these landfill sites:

- A mobile membrane-based upgrading unit shared between the four sites
- A centralised upgrading system
- Individual upgrading facilities at each site

Mobile membrane upgrading unit

The implementation of a mobile membrane upgrading system would allow the cost of the upgrading plant to be shared between the four sites. LFG could be upgraded using a mobile upgrading unit at each site and be transported to a centralised grid injection facility or gas service station via a "virtual pipeline". A virtual pipeline is a substitute to physical pipeline and replicates the continuous flow of gas in pipeline via transport such as trucks or ships (Verdek Solutions 2013; GE Power & Water 2014). Small scale membrane upgrading units are often designed to fit inside shipping containers for ease of transport and installation on-site at the biogas plant. In correspondence with technology providers upgrading facilities can be relatively easily modified to a mobile system. However, high levels of nitrogen present in LFG would be a great challenge for the upgrading process. The raw gas needs to be dried and cleaned of nitrogen prior to delivery to the upgrading facility. Due to the limited space available in a shipping container, a nitrogen removal plant would need to be located permanently at each landfill site. The high cost of nitrogen removal facility would make this option unsuitable for the analysed sites.

Centralised upgrading facility

As with a mobile upgrading system, a shared centralised upgrading facility would allow the capital cost of the upgrading plant to be split between the four landfill sites. Such a system would require raw LFG to be compressed and transported via virtual pipeline to the centralised site. However, the volume of gas which would need to be transported would be far higher than in the case of a mobile system, due to the fact that the LFG is ca. 45% CH_4. This would require at least twice as many trucks to

transport the gas to the central upgrading site. Furthermore, a significant amount of energy would be wasted in compressing the CO_2, N_2 and other gases in LFG which are of no value. Added to this, the H_2S present in the LFG would need to be removed before compression to avoid corrosion of the compressor and storage tanks. Given these constraints, a centralised LFG upgrading facility must be ruled out.

Individual upgrading facilities

Finally, LFG can be upgraded in individual upgrading facilities at each landfill site. Produced biomethane could then be either i) used locally in a service station or ii) be transported by truck to a centralised facility and injected into gas grid. The centralised facility could be located at one of the landfill sites. If the biomethane were to be transported, a truck would make return trips from the injection site to each landfill site individually. The total distance to be driven is 650 km. This would require a total driving time of approximately 10.5 hours; the gas could potentially be collected daily from each of the sites using one truck.

5.2.3. Choice of technology

Three out of seven contacted industry representatives agreed to cooperate on this study. Data presented in this section came from personal communication with industry and cannot be disclosed further. Energy and cost data were obtained for three different upgrading systems (Table 5.3) for three technology processes (TPs):

- TP: a two-step solution, in which membranes are used to remove CO_2 and other impurities, while N_2 removal is conducted using VPSA with almost 90% methane recovery. TP1 is adaptable to smaller size sites starting from 280 m_n^3/h;
- TP2: a one-step PSA for cleaning and upgrading of LFG. Its capacity ranges from 150 to 5000 m_n^3/h gas flow;
- TP3: two-step system with membranes and PSA. TP3 is usually designed for larger sites with at least 600 m_n^3/h gas flow as for smaller sites it might not be economically feasible.

Capital costs vary greatly with the highest for TP3. Operating costs are 5-10% of CAPEX depending on technology.

Table 5.3 Technologies assessed for LFG upgrading (pers. comm. with industry).

	TP1	**TP2**	**TP3**
Steps	2 step	Single step	2 step
Type of technology	1) Membranes (removal of CO_2 and impurities) 2) VPSA (N_2 removal)	PSA (fast-cycle)	1) Membranes (CO_2 and impurities removal) 2) PSA (N_2 removal)
Methane recovery	89.5%.	70%	70%
Minimal feed flow	280 m_n^3/h	150 m_n^3/h	600 m_n^3/h
CAPEX	€1,900,000 - €2,800,000	€1,125,000	€2,700,000
OPEX (per annum)	8-10% of CAPEX	5-6% of CAPEX	8% of CAPEX
Power consumption	0.32 kWh/m_n^3 feed gas	0.33 kWh/m_n^3 feed gas	0.33 kWh/m_n^3 feed gas

5.2.4. Calculation methodology

As all four sites were quite similar in LFG output and composition, one site (Site 4) was selected for economic analysis. The cost per m_n^3 of upgraded CH_4 was calculated and compared for each of the technology processes over a 15 year operational life (2015-2029). The equivalent annual cost was calculated for each case, using equation 1:

$$R = \frac{P(1+r)^N r}{[(1+r)^N - 1]} + OPEX \qquad \text{(Equation 1)}$$

where R = equivalent annual cost; P = total capital cost; N = number of years; r = discount rate; OPEX = annual operational costs

The equivalent annual cost method is based on discounted cash flow analysis and spreads the capital cost over the lifetime of the project. The equivalent annual worth (EAW) was then calculated for each year by subtracting the income from the sale of upgraded LFG from the equivalent annual cost for that year. Summing these gives an indication of the overall profitability of the project over its lifetime. A negative EAW indicates that the project is not profitable.

5.2.5. Asset value of LFG

From discussions with industry, the market value for the upgraded LFG was taken as €28/MWh. In Ireland there is a Biofuel Obligation Certificate (BOC) scheme, which provides further income to transport biofuel (National Oil Reserves Agency;

www.nora.ie/biofuels-obligation-scheme.141.html); 1 BOC is priced to be equivalent to the difference between the price of diesel and biodiesel. Two BOCs are available for biofuels from residues (such as landfill material). Three BOCs are available if the biofuel is gaseous with an energy value in excess of 35 MJ/m_n^3. A BOC trades between 15 and 35c/L diesel; a conservative approach is initially applied here in which upgraded LFG receives 2 BOCs at €15/MWh each. Thus, the total value of upgraded LFG is estimated at €58/MWh or €0.61/m_n^3, assuming biomethane energy content of 37.78 MJ/m_n^3. This assumes that the biomethane produced will be injected into the high pressure transmission network. If, however, the biomethane is to be used directly as a transport fuel or injected into the distribution network, propane must be added to bring the Wobbe index up to requirements. The cost of this propane addition is taken at €0.13 per m_n^3 of biomethane (around 9% of propane per m_n^3 of biomethane) (DECC UK 2014). Two cost analyses were carried out: one including propane addition and one without it. Assumptions made in the calculations are presented in Box 5.1.

Box 5.1 Assumptions for LFG Calculations.

- Electricity cost based on UK Department of Energy and Climate Change projections (UK DECC 2013), extended to 2034. Average rate of 17.7 p/kWh including VAT which equates to 20.9 c/kWh assuming £1 = €1.18
- 8322 operational hours per annum (95% availability) (Beil and Beyrich 2013)
- 43% CH_4 present in feed gas
- 8% cost of capital rate

5.3. Results and discussion

5.3.1. Transmission network injection with no propane addition

The results (Table 5.4) showed clearly that if the gas is to be directly injected to the transmission network, the only financially viable technology is TP2. TP3 performed worst, with a negative EAW from year one. High capital costs are the main reason for the poor returns from TP1 and TP3 (Table 5.5).

Table 5.4 Equivalent annual worth for each proposed system (no propane addition; 2 BOCs and low value per BOC).

	Equivalent Annual Worth		
	TP1	**TP2**	**TP3**
2015	€171,292	€214,308	-€104,698
2016	€92,795	€156,821	-€162,185
2017	€35,913	€115,164	-€203,843
2018	-€7,317	€83,504	-€235,502
2019	-€43,721	€56,843	-€262,163
2020	-€74,438	€34,348	-€284,658
2021	-€101,741	€14,353	-€304,654
2022	-€126,769	-€3,977	-€322,983
2023	-€148,384	-€19,806	-€338,813
2024	-€168,862	-€34,803	-€353,810
2025	-€187,064	-€48,133	-€367,140
2026	-€204,128	-€60,631	-€379,637
2027	-€220,055	-€72,295	-€391,301
2028	-€233,707	-€82,292	-€401,299
2029	-€246,221	-€91,457	-€410,464
Cumulative EAW	-€1,462,407	€261,948	-€4,523,150

Table 5.5 Production costs per m_n^3 upgraded LFG without propane addition for a 15 year project.

	TP1	**TP2**	**TP3**
CAPEX /m_n^3	€0.34	€0.24	€0.57
OPEX /m_n^3	€0.21	€0.12	€0.37
Power/m_n^3	€0.18	€0.22	€0.22
Total Cost /m_n^3	€0.73	€0.58	€1.15
Asset value /m_n^3	€0.61	€0.61	€0.61
Potential for profit /m_n^3	-€0.12	**€0.03**	-€0.54

5.3.2. Distribution network injection/gas service station – including propane addition

Upgrading of LFG to transport fuel standard was very expensive across all the systems examined and it would require significant economy of scale to be feasible (Table 5.6). With reference to Table 5.5 the addition of propane would cost €0.13/m_n^3 yielding all systems as loss making: TP1 at €0.25/m_n^3; TP2 at €0.10/m_n^3; TP3 at €0.67/m_n^3.

Table 5.6 Equivalent annual worth for each proposed system (including propane addition).

	Equivalent Annual Worth		
	TP1	**TP2**	**TP3**
2015	€660	€77,019	-€241,988
2016	-€54,099	€38,630	-€280,376
2017	-€93,781	€10,812	-€308,194
2018	-€123,938	-€10,329	-€329,336
2019	-€149,334	-€28,132	-€347,139
2020	-€170,762	-€43,154	-€362,161
2021	-€189,809	-€56,506	-€375,513
2022	-€207,269	-€68,746	-€387,753
2023	-€222,347	-€79,317	-€398,324
2024	-€236,633	-€89,331	-€408,338
2025	-€249,331	-€98,233	-€417,240
2026	-€261,235	-€106,578	-€425,585
2027	-€272,346	-€114,367	-€433,374
2028	-€281,869	-€121,044	-€440,050
2029	-€290,599	-€127,164	-€446,170
Cumulative EAW	-€2,802,690	-€816,441	-€5,601,539

5.3.3. Potential for financial viability

Only one technology proved to be financially viable over the 15-year lifetime assessed: TP2 (single step PSA) injected to the gas grid with no propane addition. It was viable on an annual basis only for the first seven years of the project. This is a result of high capital costs, the small size of the sites and the sharp decline in LFG

flow expected (Fig. 5.2). The gas cleaning and upgrading systems are sized to accommodate higher gas flows at the start, however with time they will operate at significantly reduced capacity. Below a certain flow rate these systems will not be able to function. A gas buffer will be required, and systems will have to be cycled more frequently. The cost of increased wear and tear on the system due to cycling was not included in this analysis, but should not be neglected.

5.3.4. Environmental revenues for biofuels – sensitivity analysis

In the baseline scenario (BL) above it was assumed that LFG is assigned 2 BOCs and each BOC is of value of €15/MWh. However, LFG is eligible to receive 3 BOCs if the energy content of the gaseous fuel is above 35 MJ/m_n^3; this should be the case at CH_4 levels in excess of 97%. As discussed in section 4.2.5, the price of one BOC trades between 15 and 35 c/L diesel equivalent; an upper value of €30/MWh. Two additional sub-scenarios are considered:

- A: 3 BOCs at €15/ MWh (€0.77/m_n^3);
- B: 3 BOCs at €30/ MWh (€1.24/m_n^3).

The increased environmental revenue associated with 3 BOCs at the low value (A) still does not allow TP3 A to be profitable. However TP1 A and TP2 A are profitable (Table 5.7).

The increased environmental revenue associated with 3 BOCs at the high value (B) allows all technologies to be financially viable, with TP2 B allowing earning of €0.66 per m_n^3 biomethane produced. Propane addition reduces all profits by €0.13 per m_n^3 biomethane.

Table 5.7 Financial analysis of 1 m_n^3 upgraded LFG without propane addition for a 15 year project.

	TP1 BL	**TP1 A**	**TP1 B**	**TP2 BL**	**TP2 A**	**TP2 B**	**TP3 BL**	**TP3 A**	**TP3 B**
Total Cost /m_n^3	€0.73	€0.73	€0.73	€0.58	€0.58	€0.58	€1.15	€1.15	€1.15
Asset value /m_n^3	€0.61	€0.77	€1.24	€0.61	€0.77	€1.24	€0.61	€0.77	€1.24
Profit/m_n^3	-€0.12	€0.04	€0.51	€0.03	€0.19	**€0.66**	-€0.54	-€0.39	€0.08

5.3.5. Cost of the virtual pipeline

Costs for a virtual pipeline and grid injection facility for distribution network injection or a gas service station were not accounted for in the above analysis. A virtual pipeline consists of a number of elements: compressed gas trailer, truck, gas storage and compressors. Costs for a standard 40 ft compressed gas trailer vary significantly depending on construction material and storage capacity. A trailer with a high capacity carbon fibre composite tank system costs between €436,500 and €576,000 (Pacific Northern Gas Ltd. 2013; Verdek Solutions 2013). The cost of a truck to pull the trailers is ca. €100,000 (Das Magazin fur Fernfahrer Trucker 2011). Industrial suppliers suggest that each landfill site should be provided with two trailers (pers. comm.). This ensures that there is always one trailer on-site which eliminates the need for any intermediate biomethane storage facilities on-site. Once the trailer is full, it is replaced with an empty trailer returning from the grid injection site. The gas is stored at 250 bar, so compressors will be required. The capital cost of compressors was calculated at €137,000 per unit, with an operational cost of €0.10/m_n^3 (Johnson 2010).

5.3.6. Cost of grid injection facility

In Ireland, no grid injection facilities for biomethane exist at the moment, and therefore the cost of building such a facility is uncertain. In Britain, the cost of the first transmission grid entry unit was €1 million in 2010, but the cost of subsequent facilities dropped to ca. €400,000 by 2014. With an easing of gas quality requirements, this cost is soon expected to drop to €300,000 (Baldwin 2014). This fixed one off capital cost applies to injection facilities of all capacities with the majority planned in the range of 800-1200 m^3/h.

In Germany, a fixed gas grid connection fee of €250,000 is paid by the biogas plant operator. The grid operators are in charge of gas quality control, compression and metering; they also pay the CAPEX and OPEX for the injection point and connection pipeline (Stephanblome 2011).

5.3.7. Gas service station at landfill site

Selling biomethane directly at a gas service station, will require capital expenditure of €400,000 per station with maintenance cost at 6% (Johnson 2010). This is cheaper than the virtual pipeline in combination with gas grid injection, which requires a

capital investment in excess of €1,000,000. Waste collection lorries can utilise the fuel as well as local taxis, cars and buses. The potential for profit for this system is shown in Table 5.8, including cost of service station on site and addition of propane. Scenario TP2 A is close to financial sustainability with a financial support in the form of 3 BOCs, at minimum price per BOC (at €15/ MWh). Scenario TP2 B with 3 BOCs at €30/ MWh (biomethane value €1.24/m_n^3) may be considered too lucrative. In essence a subsidy of a minimum €0.55/m_n^3 is required. The asset value of LFG upgraded to biomethane is a crucial factor; for the project to be economically viable the price per BOC must therefore be higher than €17/ MWh (total revenue from BOCs above €52/ MWh). The market value of upgraded LFG (natural gas price) is also subject to variations (€28/ MWh in all scenarios considered) and may affect the project profitability.

Table 5.8 Minimum cost per m_n^3 upgraded LFG, including for propane addition and annualised cost of service station (maintenance at 6% per year).

	TP2 BL	**TP2 A**	**TP2 B**
Total Cost /m_n^3	€0.84	€0.84	€0.84
Asset value /m_n^3	€0.61	€0.77	€1.24
Profit/m_n^3	-€0.23	-€0.07	**€0.40**

An on-site service station might not be the most realistic solution for industry. For the Glenmore project in County Donegal (Ireland), biogas produced on-site will be upgraded to biomethane, compressed to 250 bars and road transported for use at other industrial sites owned by manufacturing companies such as Bombardier and Montupet (CEHL 2016; Macauley 2016; WIS Group 2017). This can be seen as an alternative to on-site upgrading and use.

5.3.8. Support schemes for LFG

Success of LFG projects depends hugely on natural gas prices (Li et al. 2014), and therefore in order to make LFG upgrading economically viable, the costs should be mitigated using different subsidy schemes including grants, low-interests loans, feed-in tariffs and tax reliefs (Urban 2013). These ensure biomethane production and utilization. Currently direct feed-in tariffs for biomethane are applied in France,

Denmark, and the U.K. as well as in the Netherlands (Thrän et al. 2014). This paper suggests that the minimum feed-in tariff to allow financial sustainability is €0.55/m_n^3 whether in Ireland or in the USA.

5.4. Conclusions

Upgrading of LFG to biomethane at small landfill sites can be feasible but requires economies of scale. Small-scale projects might struggle for profitability. Profitable systems must be simple with a cheap technology adapted to LFG characteristics. A single step system based on fast-cycle PSA was most cost efficient. Due to the environmental revenue termed biofuel obligation certificates, gas grid injection to the transmission grid was profitable in Ireland, if upgraded biomethane is to be used as a transport biofuel. Injection to the distribution network requires addition of propane adding €0.13/m_n^3 reducing potential for profit. An on-site service station can be a simple cost effective solution for transport biofuel production and is the recommended option for Ireland. The cost was assessed as €0.84/m_n^3 LFG biomethane. This required a subsidy of €0.55/m_n^3, which is available in Ireland under the BOC scheme. However, for a project involving several industrial partners the more realistic solution could be to transport the upgraded biomethane by road to other sites for further use. The approach and findings presented in this paper can be replicated in other countries.

Chapter 6. Life cycle assessment of integrated seaweed and salmon farming systems for biomethane production in temperate oceanic climates

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Abstract

Biomethane produced from seaweed is a third generation renewable gaseous fuel. The advantage of seaweed for biofuel is that it does not compete directly or indirectly for land with food, feed or fibre production. Furthermore, the integration of seaweed and salmon farming can increase the yield of seaweed per hectare, while reducing the eutrophication from fish farming. So far, full comprehensive life cycle assessment (LCA) studies of seaweed biofuel are scarce in the literature; current studies focus mainly on microalgal biofuels.

The focus of this study is an assessment of the potential environmental impacts and benefits of integrated seaweed and salmon farming for biomethane production in a north Atlantic island, namely Ireland. With this goal in mind, an attributional LCA principle was applied to analyse a seaweed biofuel system. The environmental impact categories assessed are: climate change, acidification, and marine, terrestrial and freshwater eutrophication.

The seaweed *Laminaria digitata* is digested to produce biogas upgraded to natural gas standard, before being used as fuel. The baseline scenario shows high emissions in all impact categories. An optimal seaweed biomethane system can achieve 60% savings in GHG emissions as compared to fossil fuels with high yields per hectare, optimum seaweed composition and proper digestate management. Seaweed harvested in August

proved to have higher methane yield, thus August seaweed biomethane delivers 22% lower impacts than biomethane from seaweed harvested in October. Seaweed characteristics are more significant for improvement of biomethane sustainability than an increase in seaweed yield per unit area.

Keywords: Seaweed; biomethane; anaerobic digestion; life cycle assessment (LCA); wastewater; integrated multi-trophic aquaculture (IMTA).

6.1. Introduction

6.1.1. Rationale for seaweed biomethane

Under the European Union (EU) Renewable Energy Directive (RED), the EU is committed to achieve at least 20% renewable energy share of gross energy consumption and 10% of renewable energy in transport by 2020 (EC 2009a). A further increase of renewables penetration must be ensured beyond 2020 yielding 27% of renewable energy share by 2030 EU-wide (EC 2014c). Biofuels have an important role in achieving those targets but their sustainability must be ensured (EC 2009a). Land-based biofuels are of particular concern since they may compete directly or indirectly for land associated with food and feed production (Searchinger et al. 2008). An amendment to the RED was approved in 2015 to mitigate the potential impacts of indirect land use change (Czyrnek-Delêtre et al. 2016b). The amended directive sets a cap on first generation (land-based) biofuels to 7% of renewable transport fuel. Also, an indicative target of 0.5% for advanced biofuels, derived from biomass other than food and feed, such as wastes and algae, was proposed (EurActiv.com 2015; EC 2015c). The algal biofuel sector is still at a very early stage of development but there are start-up companies and EU-funded projects (mainly dealing with microalgae) examining commercialisation of algal biofuels (Chisti and Yan 2011; Chiaramonti et al. 2015). Marine algae (seaweed) are of special interest for this study since seaweed produced at sea, do not compete directly or indirectly for land associated with food or feed production. Therefore, biomethane produced from marine algae is considered an advanced (third generation) renewable gaseous fuel. Also, algal biomethane can be counted at twice its energy content in consideration of 2020 national renewable energy targets (EC 2015c).

Seaweeds are macroscopic and multicellular organisms. They are part of a very diverse group, which colonize various marine environments. As many as 221 species of seaweed are commercially used for a range of applications; 66% are used for food (Cox 2012), and the remaining in agrichemicals, fish feed, health and cosmetic sectors (Soto 2009). Cultivated seaweed production reached 24 million tonnes wet weight (wwt) in 2013 (FAO 2016). This represents over 95% of the seaweed market (Murphy et al. 2013) with a value over €6 billion (FAO 2014). The remaining 5% are wild

seaweeds collected onshore. The major producers of seaweed are China and Indonesia (Murphy et al. 2013).

Farming of seaweed can be integrated with salmon farming in an integrated multi-trophic aquaculture (IMTA) system. These present a way to reduce the impacts from nutrient-rich waste released from fish farms (mainly in the form of ammoniacal excrement), whilst enhancing the growth of seaweed (Soto 2009; Holdt and Edwards 2014). However, existing studies focus rather on an increase in seaweed growth than on modelling of nitrogen flow between fish farm and seaweed farm (Halling et al. 2005; Dalsgaard and Krause-Jensen 2006; van den Burg et al. 2013). Therefore in this study, the positive impact of integrated farming was included as an increase in seaweed yield per hectare.

Aquaculture produces over 2 million tonnes live weight each year of Atlantic salmon (*Salmo salar*) (FAO 2016). If an average price of €5 per kilogram is assumed (Index mundi 2015), this gives a market worth €10 billion. The Food and Agriculture Organization (FAO) estimated also that there will be a need to produce an extra 42 million tonnes of farmed seafood to feed the world population by 2030, and salmon will play a key role in fulfilling this demand (BIM Bord Iascaigh Mhara 2012). However, research shows that farmed salmon has a very high environmental footprint (Pelletier and Tyedmers 2008; Pelletier et al. 2009; Buchspies et al. 2011; Taranger et al. 2015). Implementation of efficient IMTA has the potential to increase the sustainability of aquaculture systems by reducing the nutrient load release (N and P) and by minimising the risk of eutrophication in marine environments. Furthermore, the seaweeds resulting from IMTA become an additional product to be exploited in the energy and/or food sectors with additional revenues for the fish farmers (Chopin et al. 2001; Soto 2009).

It has been long recognized that brown seaweeds, such as *Saccharina latissima* and *Laminaria digitata*, which are native to northern Europe, are suitable for IMTA, due to their N uptake capacity and yield improvements in proximity to fish cages (Soto 2009; Holdt and Edwards 2014). Previous research investigating the green seaweed *Ulva sp*. and the red seaweed *Gracilaria chilensis*, showed that the growth level of seaweed cultivated close to fish cages is higher than in control sites (Halling et al. 2005; Dalsgaard and Krause-Jensen 2006; van den Burg et al. 2013). In a more recent

study, *S. latissima* and a red algae, *Palmaria palmata* produced respectively 27% and 63% higher yields when grown close to fish farms than on reference sites (Sanderson et al. 2012). It was also observed that *S. latissima* had a faster growth in an IMTA system than at a reference station (Handå et al. 2013).

Experimental studies indicate the suitability of seaweed substrates for methane production under anaerobic condition (Dave et al. 2013; Allen et al. 2015; Alvarado-Morales et al. 2015; Montingelli et al. 2016). However, there are technical and economic challenges for seaweed based production systems such as, the fluctuation in the seaweed supply over the year and the seasonal variation in the chemical composition of seaweed that was observed for different species (Murphy et al. 2015). These can become an asset if addressed properly to identify optimal system configurations and potential environmental risks.

6.1.2. Life cycle assessment of integrated seaweed and salmon system

Life cycle assessment (LCA) is accepted as the most suitable tool for the sustainability assessment of algal projects (Bradley et al. 2015; Chiaramonti et al. 2015). So far, full comprehensive LCA studies of seaweed biofuels are scarce in the literature; to date algal studies focused mainly on microalgal liquid biofuels systems Moreover, the majority of algal LCA papers only examine climate change as the impact category (Collet et al. 2015).

Taelman et al. compared two off-shore cultivation systems of *S. latissima;* long-line (Ireland) and raft system (France) (Taelman et al. 2015). The study focused on the assessment of the environmental impacts of seaweed farming (hatchery and deployment at sea) based on the total consumption of resources, such as fossil, nuclear and marine resources. Results in Ireland show that about 81% of the impacts are related to the transport (between hatchery and sea site) and infrastructure; diesel used for transport contributed 44.3% of impacts, while production of materials used in the processes contributed 36.6%. The impacts of both systems could be lowered if biomass yields per unit area were increased (Taelman et al. 2015).

The study of Langlois et al. (Langlois et al. 2012) dealt with the environmental impacts of biomethane from the anaerobic digestion (AD) of the whole seaweed (*S. latissima*) and alginate-extraction residues. Macro-algal biomethane has important benefits for

marine and freshwater eutrophication, because seaweed removes eutrophying pollutants (N and P) from the surrounding seawater during its growth. However, the study found that the overall environmental impact of seaweed biomethane was higher when compared with natural gas, in terms of climate change, ozone depletion and human toxicity, among others. The authors suggested that eco-design (materials recycling, heat recovery), technical improvements (increased biomass yield per unit area and lowered fuel consumption), and use of renewable energy (from offshore wind farms) could improve greatly the environmental footprint of seaweed biomethane.

Alvarado-Morales et al. assessed the energy demands and environmental impacts of biofuel produced from *L. digitata* grown on long-lines in Nordic conditions for two (hypothetical) seaweed biofuel systems (Alvarado-Morales et al. 2013). Biogas production from digestion of seaweed was compared with bioethanol production via saccharification and fermentation. They found that seaweed biogas has the potential to deliver beneficial impacts for climate change (Global Warming Potential), acidification and terrestrial eutrophication. These are related to both the production of electricity from biogas (displacement of coal-based electricity) and use of digestate (displacement of mineral fertilisers). The seaweed production phase was documented as the highest contributor to the impacts. The study suggests that increasing the biomethane potential of seaweed could improve the impacts of the system. The biogas scenario performed better than bioethanol scenario for all the impacts categories considered, and the difference between the two scenarios was linked mainly to the energy consumed for bioethanol downstream and purification process.

In an LCA study of biomethane from *Ulva lactuca* grown in an open pond in southern Italy, seaweed was co-digested with poultry manure and agricultural waste (citrus pulp) (Cappelli et al. 2015). The biomethane produced was used for electricity and heat generation. Compared with a fossil fuel scenario, the seaweed system performed better if total electricity inputs to the systems are supplied by electricity generated from biogas using an onsite CHP system, and digestate is assumed to replace mineral fertilisers.

The gap in the state of the art, and the corresponding innovation in this paper, is that this is the first paper to undertake a full comprehensive well-to-wheel LCA study of gaseous seaweed biofuel associated with an integrated multi-trophic aquaculture

system including for consideration of a range of impact factors. The overall sustainability of such systems has remained unknown until now. The advantage of these systems is based on the use of nutrient rich salmon waste to increase yields in biomass per hectare and the improvement in water quality brought about by growth of seaweed. The systems described are pre-commercial, and as such an extensive sensitivity analysis is undertaken to assess the major sources of impacts and how to maximize sustainability of such systems.

6.1.3. Aims and objectives

The aim of this paper is to assess the potential environmental impacts and benefits of integrated marine algae and salmon farming for biomethane production in a country with a temperate oceanic climate. The specific objectives are to:

- Generate a detailed LCA model of biomethane from seaweed grown near a salmon farm;
- Identify the critical environmental impacts;
- Assess the implication of using the salmon waste to increase the seaweed yield per hectare;
- Assess the influence of assumptions over critical parameters such as using digestate as a replacement for mineral fertilisers;
- Identify ways of addressing and minimising the impacts and maximising the sustainability of seaweed biomethane.

6.2. Methods

6.2.1. Scope of the study and boundaries of the system

An attributional approach was applied in a cradle-to-gate LCA, which includes release of nutrient rich waste from salmon farming, seaweed hatchery and deployment at sea, harvesting and subsequent ensiling of seaweed, biogas production through anaerobic digestion, and upgrading to biomethane (Fig. 6.1). The baseline scenario (Seaweed and Salmon farming system, SW-SF, Table 6.1) was compared with two alternative fossil fuel systems based on gasoline and natural gas. The model included a credit assigned to biomethane that comes from removal of nitrogen-rich waste during the seaweed growth and as a consequence of this, an increase of seaweed yield per unit

area. The impacts and benefits from digestate management (displacement of mineral fertilisers) were included in all the scenarios, and analysed in a sensitivity analysis.

The functional unit (FU) considered was one MJ of compressed biomethane (CBG) at the gate of the production plant. However, when CBG was compared with fossil fuels (natural gas and gasoline), the combustion emissions were included, and the FU used was the kilometre driven in a vehicle under specific assumptions (section 6.2.5).

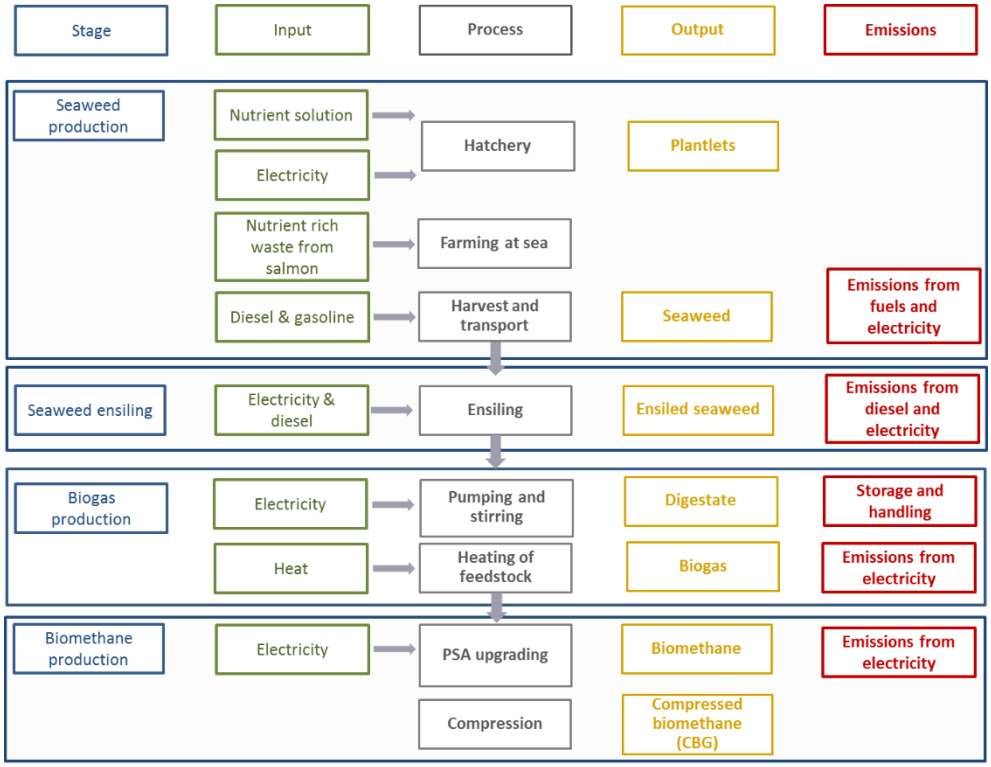


Fig. 6.1 Integrated seaweed and salmon farming system for biomethane production (well-to-tank approach).

6.2.2. Data collection

There were four main sources of data used for the analysis: primary data from experiments and personal communications, and secondary data from literature and GaBi Professional database (thinkstep 2016). The results from laboratory experiments carried out at University College Cork on continuous digestion of *L. digitata* were used to determine the biomethane potential and seaweed characteristics (total and volatile solids, nitrogen and carbon content). The composition of *L. digitata* changes substantially with season, with the most suitable characteristics (highest volatile solids, VS) and the highest biomethane potential (BMP) resulting in 327 m^3 CH_4/t VS

in August (Tabassum et al. 2016b). An acclimatization period of microbial groups within a continuous anaerobic reactor improved significantly the specific methane yield (SMY) of seaweed. This led to an increase of 26.5% (from 267 to 338 m^3 CH_4/t VS) at an organic loading rate of 2 kg VS/m^3 per day for feedstock collected in October (Tabassum et al. 2016a) as opposed to August when seaweed composition is more optimal. The Irish Fisheries Board and Irish Seaweed Consultancy Ltd. provided information on salmon farming and IMTAs. Relevant previous studies by the authors were included as well as papers on LCA of seaweed biofuels (Langlois et al. 2012; Alvarado-Morales et al. 2013; Tabassum et al. 2016a; Tabassum et al. 2016b). GaBi database provided the background data. Emissions associated with infrastructure, buildings and equipment used in the processes, as well as waste production and disposal were not included in this LCA. For the contribution and major part of the sensitivity analysis life cycle inputs and outputs from the use of CBG in transport vehicles were considered outside the system boundaries (well-to-tank approach and FU of 1 MJ of biomethane). However, when biomethane was compared to fossil fuels (sections: 6.2.5 and 6.3.3), emissions from transport vehicles were included (well-to-wheels approach and FU of 1 km driven on biomethane).

Table 6.1 Baseline and sensitivities scenarios analysis (in bold the parameters/data for which variations were considered as compared to the baseline)

Scenario ID[1]	Near salmon farming (Yes/No)	Yield (t wwt seaweed/ ha)	DS content (%)	VS content (%)	BMP (batch) (m^3 CH_4/ t VS)	SMY (CSTR) (m^3 CH_4/ t VS)	Waste water treatment in hatchery	Digestate substitution of mineral fertiliser (%)	Electricity grid mix (Irish/renewable mix)	Additional comments
• **Baseline**										
SW-SF	Yes	25.4	17.7	14.42	267	338[3]	UV-WWT	30	Irish mix	Increased yield: 20+27%= 25.4[2]
• **Sensitivities**										
$SW\text{-}SF_{70\%}$	Yes	25.4	17.7	14.42	267	338[3]	UV-WWT	**70**	Irish mix	Increased yield: 20+27%= 25.42
$SW\text{-}SF_{NoWWT}$	Yes	25.4	17.7	14.42	267	338[3]	**No-WWT**	30	Irish mix	No water treatment
$SW\text{-}SF_{August}$	Yes	25.4	**19.72**	**16.12**	**327**	**410[4]**	UV-WWT	30	Irish mix	Highest DS, VS, BMP and SMY (August)
$SW\text{-}SF_{August\ 70\%}$	Yes	25.4	**19.72**	**16.12**	**327**	**410[4]**	UV-WWT	**70**	Irish mix	Highest DS, VS, BMP and SMY (August)
$SW\text{-}SF_{40t}$	Yes	**50.8**	17.7	14.42	267	338[3]	UV-WWT	30	Irish mix	Increased yield: 40+27%= 50.8[2]
$SW\text{-}A_{40t}$	**No**	**40**	17.7	14.42	267	338[3]	UV-WWT	30	Irish mix	Basic yield: 40

SW-SF$_{100t}$	Yes	**127**	17.7	14.42	267	338[3]	UV-WWT	30	Irish mix	Increased yield: 100+27%= 127[2]
SW-A$_{100t}$	**No**	**100**	17.7	14.42	267	338[3]	UV-WWT	30	Irish mix	Basic yield: 100
SW-SF$_{2020\ projection}$	Yes	25.4	**19.72**	**16.12**	**327**	**410[4]**	UV-WWT	30	**2020 projection mix**	Highest DS, VS, BMP and SMY (August)
SW-SF$_{Wind}$	Yes	25.4	**19.72**	**16.12**	**327**	**410[4]**	UV-WWT	30	**Wind mix**	Highest DS, VS, BMP and SMY (August)
SW-SF$_{40t\ August}$	Yes	**50.8**	**19.72**	**16.12**	**327**	**410[4]**	UV-WWT	**70**	**Wind mix**	Increased yield (40+27%)[2]; Highest DS, VS, BMP and SMY; Renewable mix
SW-SF$_{100t\ August}$	Yes	**127**	**19.72**	**16.12**	**327**	**410[4]**	UV-WWT	**70**	**Wind mix**	Increased yield (100+27%)[2]; Highest DS, VS, BMP and SMY; Renewable mix

[1] SW-SF – seaweed and salmon farming system, SW-A seaweed alone system; '70%' – 70% replacement of fertiliser; 'NoWWT' - no water treatment in hatchery (release of water back to the sea); 'August' - seaweed in August (all remaining harvested in October); '40t' and '100t' - increased yields per hectare.

[2] Increase of 27% in yield per hectare is included for scenarios with combined seaweed and salmon farming (SW-SF)

[3] Acclimatization effect from Tabassum et al. (Tabassum et al. 2016a)

[4] The value was obtained by the same pro-rata increase on August yield as October yield; this value is less than the theoretical yield of *L. digitata* in August, which is 452 m^3 CH_4/t VS (Tabassum et al. 2016b)

6.2.3. Life cycle inventory

Salmon farming

Salmon farming inputs and outputs were considered outside the system boundary. This decision is justified because it is assumed that increase in demand for seaweed biomethane should not create an increase in demand for salmon farming; instead, it will provide a solution to decrease the impact of existing salmon farms. In SW-SF it was assumed that the basic yield per hectare of seaweed cultivated near salmon cages increased by 27% as compared to control sites (Sanderson et al. 2012). This value was found for farmed *S. latissima* (farm located in Badcall, UK). Since both *L. digitata* and *S. latissima* are brown algae species (kelps) with similar growth conditions and characteristics (VS, DS, ash, C:N ratio) (van den Burg et al. 2013), it was assumed that productivity of *L. digitata* is enhanced as much as *S. latissima*, when grown next to fish cages.

The ability of removing nitrogen from seawater by growing seaweed is unique and it was considered in this LCA of IMTA system. For this purpose, the nitrogen excreted by salmon and absorbed by seaweed was calculated and assigned to the biomethane in form of the emission credit. This value was calculated for the modelled system with *L. digitata* as described below.

From the modelling, it was known that 0.13 and 0.10 kg DS of *L. digitata* was required to produce 1 MJ biomethane in October and August, respectively. The nitrogen content of both seaweeds was known from laboratory analysis: 12.2 and 11.4 g N/kg DS (Tabassum et al. 2016a). Based on these, the credit values were calculated at 1.54 g N/MJ and 1.16 g N/MJ in October and August, respectively. These values are very close to the literature values, which assumed that the mean ratio of wet weight *S. latissima* (kg wwt) necessary to sequester nitrogen excreted by salmon (kg) is 12.9:1, and that 1 kg wwt of salmon produces 29.49 g N (Reid et al. 2013). Supplementary data and calculations related to this credit are presented in Box 6.1. The credit values were deduced from the marine eutrophication potential for all scenarios with salmon and seaweed integrated farming (all SW-SF scenarios).

Box 6.1 Calculation of the credit to biomethane from removal of nitrogen by seaweed growth (example for SW-SF).

1. Based on N content

When N content in seaweed is 12.2 g N/kg DS

$$12.2\ \frac{g\ N}{kg\ DS} \times 0.13\ \frac{kg\ DS}{MJ} = 1.54 \frac{g\ N}{MJ}\ removed$$

2. Based on literature data (Reid et al. 2013)

When seaweed to salmon ratio is at 12.9:1

$$0.71\ \frac{\text{kg wwt seaweed}}{MJ} \div 12.9\ = 0.05\ \frac{\text{kg salmon}}{MJ}$$

When the amount of nitrogen produced by salmon is at 29.49 g N/kg of salmon

$$0.05\ \frac{\text{kg salmon}}{MJ} \times 29.49\ \frac{g\ N}{kg\ salmon} = 1.63\ \frac{g\ N}{MJ}\ removed$$

Cultivation of *Laminaria digitata*

The outline of procedure for cultivating *L. digitata* is presented in Fig. 6.2. Mature *L. digitata* is collected at low tide, cleaned and prepared in laboratory for spore release. The gametophytes culture is set in a vessel with an appropriate quantity of nutrients for culture development. It was assumed that culture is aerated and illumination is necessary over 20h per day for 26 days. Next, the induction of reproduction takes place when female and male reproductive structures are developing. This process is assumed to take up to 8 days and requires both air (24 h per day) and light (12 h per day). Once large quantities of reproductive structures are observed, the fertile cultures are sprayed onto strings. Cultures must be allowed to develop in the laboratory tanks for at least a month before deployment at sea, but can be held in the laboratory for up to 2 months if weather conditions are not suitable for deployment (Edwards and Watson 2011). In this study, 35 days were assumed with aeration running full time and illumination for 12h per day.

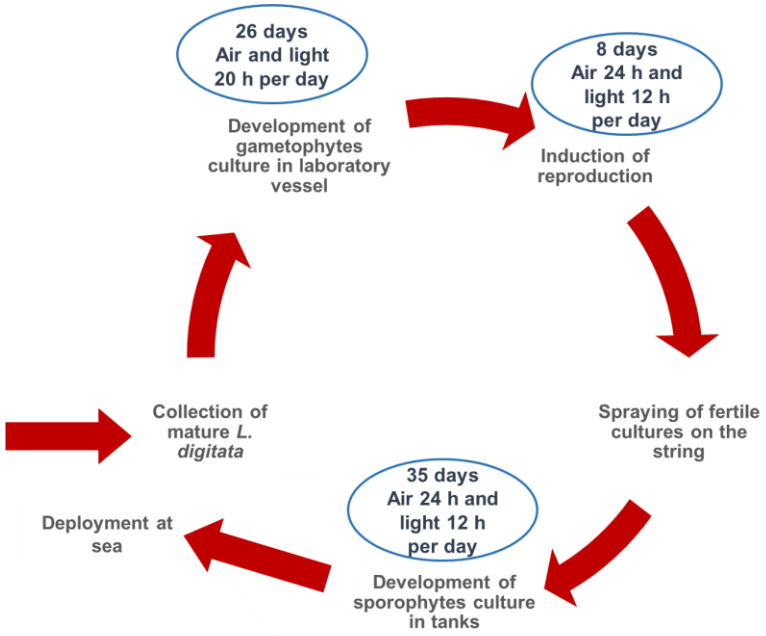


Fig. 6.2 Overview of the cultivation procedure for L. *digitata* (Edwards and Watson 2011).

It was assumed that a 14W LED lamp is used for illumination and 2.1 W air pump for aeration in all hatchery processes. One LED tube and one air pump were assumed to be needed to produce enough seedlings to yield 1 tonne dry solids (DS) of seaweed; the emissions related to their production were considered outside the system boundary. The quantities and type of substances used as nutrient solution in the hatchery processes were calculated based on Edwards and Watson (Edwards and Watson 2011). Water used in all the hatchery processes is sterilized seawater. The seawater is pumped from the sea, filtered using a sand filter and sterilised using ultra-violet light (Langlois et al. 2012). Tanks need to be cleaned and the medium changed every 3 days (11 times in total); each time 50% of volume is exchanged (Edwards and Watson 2011). A 500 L tank is filled with water up to 95% of volume (Box A.1, Appendix A).

The same growth medium is used for both developments in laboratory flasks and in the tanks. Three solutions are used: Miquel A (MA), Miquel B (MB) and Provasoli 6 (P6), 2 ml of MA, and 1 ml of each MB and P6 mixed with seawater is required per litre of seaweed culture (Edwards and Watson 2011).

In scenario SW-SF waste seawater from the hatchery was assumed to be treated using UV light before being released to the sea (UV-WWT). Wastewater should be treated due to the potential presence of algal DNA material that might not be genetically

similar to the seaweeds in the wild (pers. comm. Dr Maeve Edwards, Irish Seaweed Consultancy Ltd.). However, there is a point of view that since the seaweed species used for deployment is indigenous, the contamination is improbable (pers. comm. Lars Brunner, Scottish Association for Marine Science). An additional scenario was considered, in which used water is not treated (SW-SF_{NoWWT} Table 6.1).

Deployment at sea

In Ireland deployment occurs between October and December (Edwards and Watson 2011). It was assumed that *L. digitata* is cultivated on the West Coast of Ireland in Galway Bay using long-lines, each 100 meters long (Fig. 6.3). The 'traditional' long-line is strong and durable, and has enough flexibility to deal with heavy seas. The comfortable distance between lines is 10 m (pers. comm. Dr Maeve Edwards, Irish Seaweed Consultancy Ltd.). For this LCA purpose, 5 m distance is assumed.

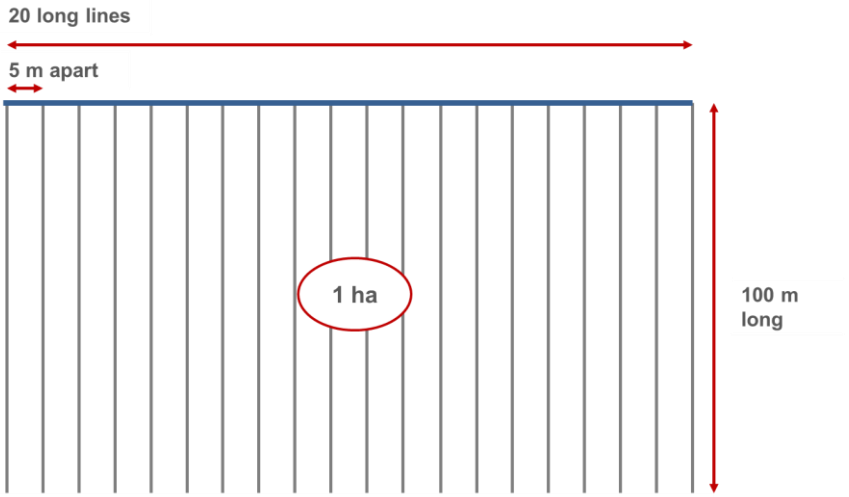


Fig. 6.3 Distribution of long-lines in a hectare of water surface for seaweed cultivation.

After the culture is deployed, the site should be visited for maintenance and monitoring once a month. It was considered that over 5 months, a small boat travels out once a month for necessary maintenance.

Harvesting of seaweed is labour intensive and costly if conducted manually. A stable boat such as a polar circle aquaculture work boat is necessary (pers. comm. Dr Maeve Edwards, Irish Seaweed Consultancy Ltd.). Mechanization is possible and much more practical if the seaweed is cultured in large quantities and needs to be harvested quickly for less quality critical purpose such as for biofuel production rather than food

(pers. comm. Lars Brunner, Scottish Association for Marine Science). For this study it was assumed that harvesting is mechanized using 18 l of diesel/t DS of seaweed (Langlois et al. 2012). Harvesting methods are still subject of research to find the most optimal and energy efficient method. The yield of seaweed in baseline scenario SW-SF was 25.4 t wwt per ha (10 kg wwt per 1 m of long-line and with 27% increase in baseline yield; 20 t wwt per ha +27%). Additional details on data used in the LCA model are presented in the Appendix A (Box A.2).

Anaerobic digestion and biogas production

L. digitata was considered to be transported for 5 km by road to a coastal anaerobic digestion facility. First the seaweed is ensiled in a tower pit. During ensilage the pH is naturally lowered to 4 and production of methane and any degradation is inhibited. The volatile solids (VS) losses occurring during storage were assumed to be compensated by the increase in methane yield of ensiled seaweed. As a result, fresh and ensiled *L. digitata* showed very similar biomethane potential (BMP), with differences in the range of 4% which were not deemed statistically significant (Herrmann et al. 2015). The BMP of the ensiled seaweed is the sum of the BMPs of ensiled biomass and effluent produced during ensiling (Herrmann et al. 2015); all the effluent is recirculated to the digester and the fugitive CH_4 emissions from ensiling were assumed to be null.

Energy input for the loading of seaweed into the tower pit for ensiling was assumed to be 7 MJ/t wwt, similar to the one considered by Berglund and Börjesson (Berglund and Börjesson 2006) for the loading of the solid fraction of the digestate. Seaweed was assumed to be macerated using a heavy duty 15 kW mixer. The dry solids content of 1 tonne wwt of *L. digitata* was assessed by Tabassum and co-workers as 17.7% (Tabassum et al. 2016a).

Produced biogas is at 55% CH_4 content. It was assumed that biogas production is effected through a continuously stirred tank reactor operating in the mesophilic temperature range at 38°C. The temperature of incoming feedstock is typically 10°C (Smyth et al. 2009). Digester electrical demand was assumed at 10 kW_eh/ t wwt of substrate (Murphy and Power 2009). Thermal demand was calculated assuming specific heat capacity of water at 4.184 MJ/ t / °C, 85% boiler efficiency and 15 %

heat losses (Smyth et al. 2009). The source of thermal energy is identified as natural gas used in Ireland as based on national energy career mix.

Fugitive methane emissions/losses come from accidental emissions due to digester cover permeability, eventual flank leakages and maintenance operations, and were assumed at 1% of produced biomethane (Liebetrau et al. 2010; Battini et al. 2014) (Table 6.2).

Table 6.2 Methane losses in the biomethane process.

Process	Value	Unit	Source
AD plant	1	% of produced CH_4	(Liebetrau et al. 2010; Battini et al. 2014)
PSA upgrading	2	% of produced CH_4 (oxidised to CO_2)	(Beil and Beyrich 2013)
Digestate storage	10.6	g CH_4/t wwt digestate (0.03% of produced CH_4)	(IPCC 2006)

PSA upgrading

Upgrading of biogas was modelled with a pressure swing adsorption (PSA) process (Czyrnek-Delêtre et al. 2016a). Operation pressure is 4-7 bars, methane losses to the atmosphere during the process were assumed at 2% of the produced methane, thus the methane recovery rate is 98% (Beil and Beyrich 2013). However, the majority of CH_4 is then oxidised to CO_2 in a burner (Beil and Beyrich 2013). The end product, biomethane, is composed of 97% methane and 3% of CO_2 and other gases. Power consumption was assumed at 0.3 kW_eh per m^3 biogas (Czyrnek-Delêtre et al. 2016a). Produced biomethane is then compressed and injected into gas grid. The electrical input of biomethane compression to 250 bar was estimated at 0.23 kW_eh/ m^3 of gas introduced into compression unit (Bauer et al. 2013).

Digestate storage and use as fertiliser

Digestate was assumed to be transported for 5 km by a tanker with an actual payload of 3.3 tonnes. Methane emissions from digestate storage were estimated based on the IPCC calculation methodology for CH4 losses from manure management (IPCC 2006)

(Table 6.2). Digestate is stored in a gas-tight closed tank (Table 6.3), and the emissions of closed storage were assumed to be 2% of the open storage (Battini et al. 2014). Emissions from field application of digestate on land were based on literature values and were calculated according to Battini et al. (Table 6.4). Direct N-N2O emissions were estimated at 1% of applied nitrogen (IPCC 2006) and N-NO at 0.55% (Stehfest and Bouwman 2006). Ammonia losses were estimated at 0.22 kg N/t wwt digestate (Amon et al. 2006). Nitrates leaching (N-NO3) was assumed to be 30% of applied nitrogen (Perego et al. 2012). CO2 field emissions were considered negligible. Phosphorus losses in form of phosphate (PO43-) run-off to freshwater were estimated at 1% of total P content in digestate and mineral fertiliser (van der Werf et al. 2009) (Table 6.4). P content in L. digitata was assumed as 0.77 g P/kg DS (Yanik et al. 2013).

Table 6.3 Losses of nitrogen during digestate storage for seaweed collected in October and August.

Emissions	Closed tank[1]	Units	Source (open tank)
Nitrous oxide direct ($N\text{-}N_2O$)	0.12[2]	g N/ t wwt digestate	(IPCC 2006)
Ammonia ($N\text{-}NH_3$)	0.16	g N/ t wwt digestate	(Amon et al. 2006)
Nitrogen oxides (N-NO)	0.002	% N in digestate	(Amon et al. 2006)
Nitrogen ($N\text{-}N_2$)	0.02	% N in digestate	(Battini et al. 2014)

[1] 2% of the emissions from open tank

[2] 0.13 g N/ t wwt digestate for seaweed harvested in August ($SW\text{-}SF_{August}$)

Table 6.4 Losses of nitrogen and phosphate during field application of digestate and mineral fertiliser for seaweed collected in October and August.

Emissions	Digestate losses		Source	Mineral fertiliser losses		Source
Nitrous oxide direct (N-N_2O)	1	% N at field	(IPCC 2006)	1	% N mineral	(IPCC 2006)
Ammonia (N-NH_3)	220	g N/t wwt digestate	(Amon et al. 2006)	10	% N mineral	(IPCC 2006)
Nitrogen oxides (N-NO)	0.55	% N at field	(Stehfest and Bouwman 2006)			
Nitrates (N-NO_3)	30	% N at field	(Perego et al. 2012)	30	% N mineral	(IPCC 2006)
Phosphate	1	% total P content	(van der Werf et al. 2009)	1	% total P content	(van der Werf et al. 2009)

When digestate is used as an organic fertiliser, it can be considered as a co-product of the biomethane plant. To resolve the multi-production system, a system expansion approach was applied to test the impact of three scenarios for mineral fertiliser substitution. In GaBi plan, the link between biogas plant main plan and credits from digestate was provided by using the so called global parameter (N_dig). This parameter represents the NET N available for the plant absorption as provided by digestate according to the following equation:

$$Total\ NET\ N = \left(N_{dig} \times Total_{dig}\right) - (N_{vol} + N_{leach})$$

Where:

Total NET N is the total NET N available for the plant;

N_{dig} is the N content in digestate;

$Total_{dig}$ is the total amount of digestate produced by the system;

N_{vol} is the N losses by volatilization;

N_{leach} is the N losses by leaching.

Based on seaweed characteristics (N content, VS in feedstock and digestate), the total nitrogen content of digestate was calculated at 2.44 kg N/t wwt of digestate (93% water

content) for seaweed produced in October and 2.58 kg N/t wwt of digestate for August seaweed. It was assumed that all nitrogen in seaweed passes to the digestate. Total ammonia nitrogen (TAN) was considered at 1.5 kg N t/wwt of digestate. The losses of nitrogen during digestate storage and field application are presented in Table 6.3 and Table 6.4.

A parameter (substitution_ef) was included to test the sensitivity to the probability that digestate may replace mineral fertiliser. A value of 100% indicates that the farmers consider that all N in the digestate replaces the same amount of N from mineral fertiliser, and the corresponding quantity of N in mineral form is not going to be produced. A value of 0% means the opposite; the digestate is still disposed of on farmland but no mineral fertiliser is actually replaced. In the baseline scenario SW-SF, 30% replacement was assumed.

Irish agricultural land comprises 81% grassland, of which 56% are permanent pastures. Similarly 56% of Irish farms are beef production farms (Central Statistics Office 2013). Irish farm surveys show that on an average farm 65 kg of N, 3 kg of P and 9 kg of K in the form of mineral fertiliser are applied per hectare per year (Lalor et al. 2010). This makes up 19.3% of N, 8.2% of P_2O_5 as P, and 6.4% of K_2O as K in a unit of mineral fertiliser (based on a simplified NPK mixer (thinkstep 2016)). Avoided emissions from application of mineral fertiliser were also included.

6.2.4. Sensitivity analysis

Digestate replacement of mineral fertiliser

While digestate is a good replacement for mineral fertilisers, the substitution does not always happen. This can be due to poor awareness by farmers, when both artificial and organic fertiliser may be applied at the same time. In the baseline scenario SW-SF, it was assumed that digestate only replaces 30% of the fertiliser. This value was applied for majority of the scenarios (Table 6.1). In the sensitivity analysis an optimistic approach was assumed with 70% replacement ($SW\text{-}SF_{70\%}$, $SW\text{-}SF_{August\ 70\%}$, $SW\text{-}SF_{40\ tAugust}$ and $SW\text{-}SF_{100\ tAugust}$).

It was assumed that if mineral fertiliser is not produced and is not applied on field, this automatically saves the emissions from P (PO_4^{3-}) and N losses ($N\text{-}N_2O$, $N\text{-}NH_3$, N-NO and $N\text{-}NO_3$, Table 6.4).

Seasonal variation in L. digitata

In scenario SW-SF_{August} it was assumed that *L. digitata* collected in August has a higher DS content (19.7%), higher VS content and a higher specific methane yield (SMY) as evaluated by Tabassum and co-workers (Tabassum et al. 2016b) as opposed to seaweed collected in October (Table 6.1). The same inputs from hatchery and deployment at sea were assumed for both scenarios.

Salmon waste and increased yields in L. digitata

In the SW-SF it was assumed that *L. digitata* can yield 10 kg wwt per meter of long-line (20 t wwt/ha); however, the total yield is increased by 27% due to nutrient rich waste from salmon farms, giving yields of up to 25.4 t wwt per ha. Sensitivity analysis was performed to understand how results can be affected by changes in the yields of seaweed. In SW-A (seaweed alone), it was assumed that seaweed yields 20 t wwt per ha without the 27% increase (Table 6.1).

Additional scenarios were introduced with higher yields farms, again stand-alone (SW-A_{40t} and SW-A_{100t}), and associated with fish farms (SW-SF_{40t} and SW-SF_{100t} with 27% increase in yields, Table 6.1). This may be possible if an advanced technology for seaweed cultivation is applied, such as textiles investigated in the European AT SEA project. In this case, the yields are expected to be at 200 t wwt seaweed per hectare, however, as the entire hectare cannot be covered by textiles; this decreases the overall yield. For this study 100 t wwt/ha was assumed as the maximum possible yield was assumed (Murphy et al. 2015).

Electricity grid mix

The impact of including more renewable electricity in the electricity mix was tested. In scenario SW-SF it was assumed that electricity used throughout the life cycle is the current Irish electricity mix, which is dominated by fossil fuels, and has a carbon intensity of 172 g CO_2 eq/MJ (Table A.1, Appendix A). Two renewable scenarios were created: 1) SW-$SF_{2020\ projection}$ (carbon intensity of 137 g CO_2 eq/MJ), and 2) SW-SF_{Wind} (carbon intensity of 70 g CO_2 eq/MJ). The SW-$SF_{2020\ projection}$ was based on forecasting published by the Sustainable Energy Authority of Ireland (SEAI) on the expected electricity mix by 2020. The target is 40% of renewable electricity in electricity consumption with the largest contribution from wind (32%) and with

biomass contributing 6% (SEAI 2012a). It was assumed that the hydropower is 2%, and fossil fuels are coal (19%), natural gas (34%) and peat (7.5%) (SEAI 2012b). The SW-SF_{Wind} is a theoretical scenario assuming that 48% of electricity is sourced from a nearby wind turbine and 52% from the 2020 Irish grid. Since the Irish grid is projected to be 32% wind based, the net wind energy contribution in the wind scenario is 66%.

Combination of the most sustainable practices

Scenarios SW-$SF_{40t\ August}$ and SW-$SF_{100t\ August}$ were created to examine the most sustainable production methods of seaweed biomethane (Table 6.1). In both scenarios it is assumed that seaweed is harvested at the most suitable time of year (August) to assure the highest SMY, VS and DS content. Modern technology to grow seaweed is applied, and therefore high yields per unit area were assumed (40 and 100 t wwt per ha for SW-$SF_{40t\ August}$ and SW-$SF_{100t\ August}$, respectively). Moreover, the seaweed farm is situated adjacent to a salmon farm, and therefore it benefits from nutrients increasing the yield of algae per hectare by 27%. In these optimum processes, the renewable electricity mix is the wind mix (as in SW-SF_{Wind}). 70% replacement of mineral fertiliser is assumed for the by-product digestate. Scenarios SW-$SF_{40t\ August}$ and SW-$SF_{100t\ August}$ were compared with other scenarios and with fossil fuel.

6.2.5. Fossil fuel comparison and reference system

The SW-SF was compared in terms of environmental impacts with a fossil fuel reference system in which the energy function is covered by gasoline or natural gas. The process for both gasoline and natural gas production is taken from the GaBi Professional database (thinkstep 2016). The datasets for gasoline/ natural gas in GaBi include the entire supply chain: well drilling, crude oil/ natural gas production and processing, transportation of crude oil by tanker/ of natural gas via pipeline, and refinery processing. Natural gas, similar to biomethane was assumed to be compressed from 1 bar to 250 bars with an energy input at 0.23 $kW_eh/\ m^3$. Combustion of fuel in a car engine was included based on a Tank-to-Wheels Technical report by Joint Research Centre (Hass et al. 2014a). The emissions of CO_2, CH_4 and N_2O and consumption of fuel per km driven were considered as for conventional vehicles (not hybrids) with Port Injection Spark Ignition (PISI) engine modelled for beyond 2020

(Hass et al. 2014b). Biogenic CO_2 emissions from biomethane combustion were considered 0.

6.2.6. Life cycle impact assessment

The study considered the following impact categories: climate change, acidification, and terrestrial, marine and freshwater eutrophication. These were calculated using the methods recommended by the ILCD (International Reference Life Cycle Data System) Handbook for LCAs in a European context (EC JRC 2010b) as implemented in Gabi software. (EC JRC 2010c).

The climate change impact category was determined using the Global Warming Potential (GWP) over a time horizon of 100 years, and is based on the latest data presented in the IPCC Fifth Assessment Report (Myhre et al. 2013). The impact is limited to well-mixed greenhouse gases (GHG): CO_2, CH_4, and N_2O (including direct and indirect emissions from NH_3 and NO). The GWP unit is kg CO_2 eq.

The acidification impact was calculated using the Accumulated Exceedance (AE) model. It addresses the impacts caused by the atmospheric deposition of acidifying substances, such as nitrogen oxides (NO_x), sulphur dioxide (SO_2) (the largest source is combustion of fossil fuels) and ammonia (NH_3) (contributes to acidification after it undergoes nitrification in the soil). These substances cause the acidity of water and soil systems by increasing the hydrogen ion (H^+) concentration (EC JRC 2010c). This impact category is expressed in moles of H^+ eq.

Eutrophication assesses the impacts from an excess of macro-nutrients such as nitrogen and phosphorus in bio-available forms on terrestrial and aquatic ecosystems. Consequences of eutrophication typically involve significant alterations of flora and fauna, such as increased productivity of phytoplankton and suspended algae, and oxygen depletion in the bottom strata of lakes and coastal waters (EC JRC 2010c). Terrestrial eutrophication is caused by deposition of airborne emissions of N-compounds, such as NO_x from combustion processes, and NH_3 from agriculture, and it is expressed in mole N eq (EC JRC 2010c). Freshwater and marine eutrophication impacts are caused by waterborne emissions, such as nitrate, phosphate and other N and P compounds (EC JRC 2010c). Phosphorus has been identified as a key growth-limiting nutrient for eutrophication in freshwater ecosystems; therefore freshwater

eutrophication impact category is expressed in kg P eq. Similarly, nitrogen is the limiting nutrient for eutrophication in marine systems, and this impact is expressed in kg N eq.

6.3. Results and discussion

6.3.1. Contribution analysis of baseline scenario

Digestate handling, storage and field application, is the largest contributor in all impact categories (Fig. 6.4) representing 11% of GWP 100, and over 80% in all other impact categories. The contribution from biogas plant operation, PSA upgrading, and compression and seaweed farming is very high in GWP100, but much lower in other impact categories (34%, 31% and 21%, respectively for GWP 100). Part of the emissions is offset by digestate replacing mineral fertiliser and benefit from capturing of N-rich salmon excrements by growing seaweed. Digestate replaces 30% of mineral fertiliser that would be otherwise produced to sustain agricultural demand; the production of which is based on fossil fuels. The digestate credit for GWP 100 potential is -3.00 g CO_2 eq/MJ of biomethane, while the emissions for all life cycle stages are 49.26 g CO_2 eq/MJ of biomethane. For marine eutrophication, 89% of the credit comes from nitrogen credit, and 11% from the digestate replacing mineral fertiliser.



Fig. 6.4 Cradle to gate environmental impacts of 1 MJ of biomethane from *L. digitata* in the baseline scenario SW-SF including both impacts and benefits from digestate (30% replacement).

The impact of digestate handling comes from the emissions from storage (despite the closed tank) and field application of digestate (Table 6.5). The latter is responsible for over 97% of the environmental impacts in all impact categories. The storage is responsible for 2.8% of the impact in GWP100 mainly due to the methane leakage, and nitrogen losses assumed in this study (Table 6.2 and Table 6.3). A decrease in CH_4 slippage during storage would decrease the overall impact from digestate.

Table 6.5 Environmental impacts from digestate handling (storage and field applications) in scenario SW-SF.

Impact categories	Storage emissions %	Field emissions %
Acidification	0.1	99.9
GWP100	2.8	97.2
Freshwater eutrophication	0.0	100.0
Terrestrial eutrophication	0.1	99.9
Marine eutrophication	0.003	99.997

When considering scenario SW-SF without digestate fate (Fig. 6.5), the largest contribution in almost all impact categories apart from marine eutrophication and GWP100 comes from seaweed farming with UV-WWT in the hatchery. Seaweed farming has the highest impacts in freshwater eutrophication accounting for 91% of total impact, which is mainly due to the use of diesel (64%) and gasoline (32%) for the deployment, maintenance and harvest. Also, seaweed farming contributes to the climate change potential (12.55 g CO_2 eq/MJ biomethane) due to the use of fossil electricity in water sterilization and treatment, as well as aeration and illumination processes in hatchery. However, the seaweed farming resulted in the highest emission benefits for marine eutrophication potential, due to the uptake of N-rich waste from salmon farming (-1.53 g N eq/MJ of biomethane) during seaweed growth. Seaweed transport and ensiling show negligible potential impacts in all categories considered.

The GWP 100 of biomethane is dominated by the operation of the biogas plant (15.96 g CO_2 eq/MJ of biomethane) and PSA upgrading and compression (15.36 g CO_2 eq/MJ of biomethane) (results detailed in Table B.1, Appendix B). Over 30% of carbon emissions in AD plant comes from methane leakage. In the upgrading and compression stage, the major contributor is the fossil-based electricity, while the emissions from oxidised methane represent 15% of the GWP100.

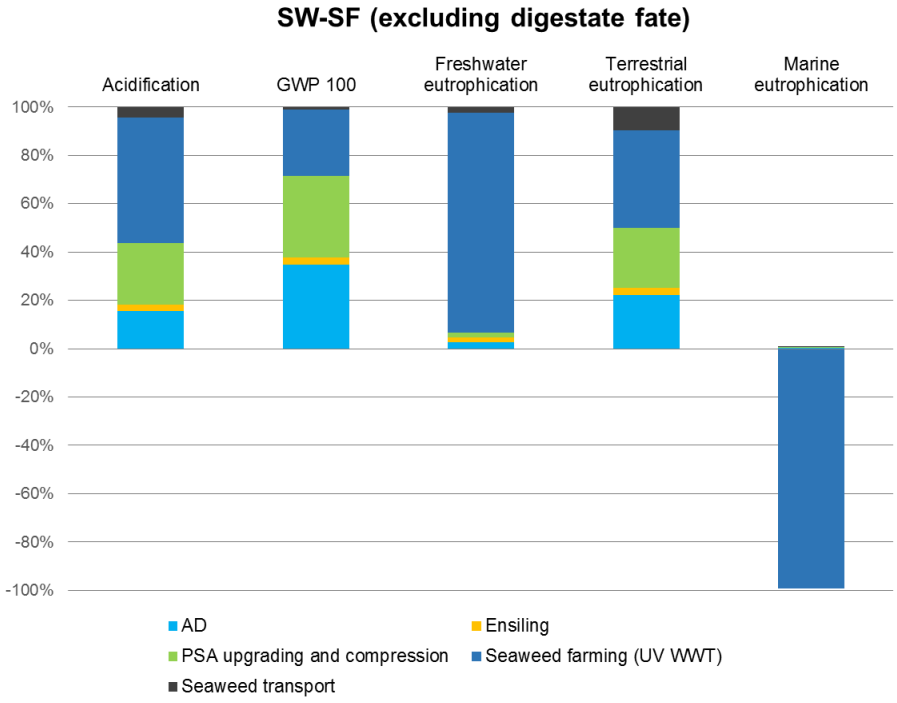


Fig. 6.5 Cradle to gate environmental impacts of 1 MJ of biomethane from *L. digitata* excluding impacts and benefits from digestate in the baseline scenario SW-SF.

6.3.2. Sensitivity analysis

Impact of digestate replacing mineral fertiliser

Fig. 6.6 presents the two approaches to substitution of mineral fertiliser, in which different rates of replacement were assumed (more details in Table B.2, Appendix B). These are related to several factors, including the awareness of farmers of the fertilising value of the digestate, and at what rate they are willing to replace mineral fertiliser with digestate. Savings from scenario SW-$SF_{70\%}$ (70% replacement) are between 8% (GWP 100) and 54% (terrestrial eutrophication) as compared to SW-SF (30% replacement), depending on the impact considered. Scenario SW-$SF_{70\%}$ shows considerably lower emissions for acidification and eutrophication potentials. In terms of climate change, the GWP 100 drops to 45.27 g CO_2 eq/MJ as compared to SW-SF (49.26 g CO_2 eq/MJ). If 100% replacement would be assumed this would lead to a further decrease in the overall impact of biomethane.

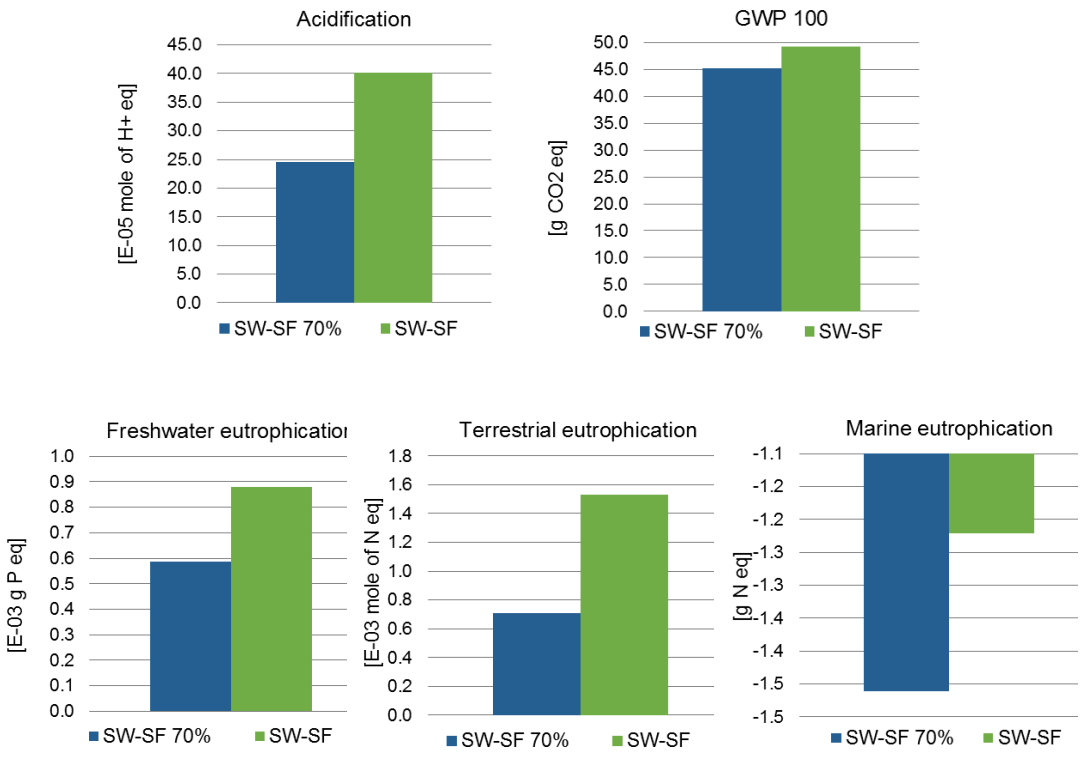


Fig. 6.6 Comparison of environmental impacts of 1 MJ of biomethane from *L. digitata* for scenarios SW-SF$_{70\%}$ (70% replacement) and SW-SF (30% replacement of mineral fertiliser).

Wastewater treatment in hatchery

The impact of wastewater treatment in the hatchery was tested (Fig. 6.7). The principal reason for water treatment is to remove the DNA material, and not the nutrients in waste water. Both scenarios show a very similar range of environmental impacts with SW-SF$_{NoWWT}$ impacts being marginally lower than SW-SF for most impacts categories (the results for SW-SF$_{NoWWT}$ are up to 6% lower than those for SW-SF). Marine eutrophication potential is entirely offset by the nitrogen uptake in seaweed farming and is almost the same for both scenarios.

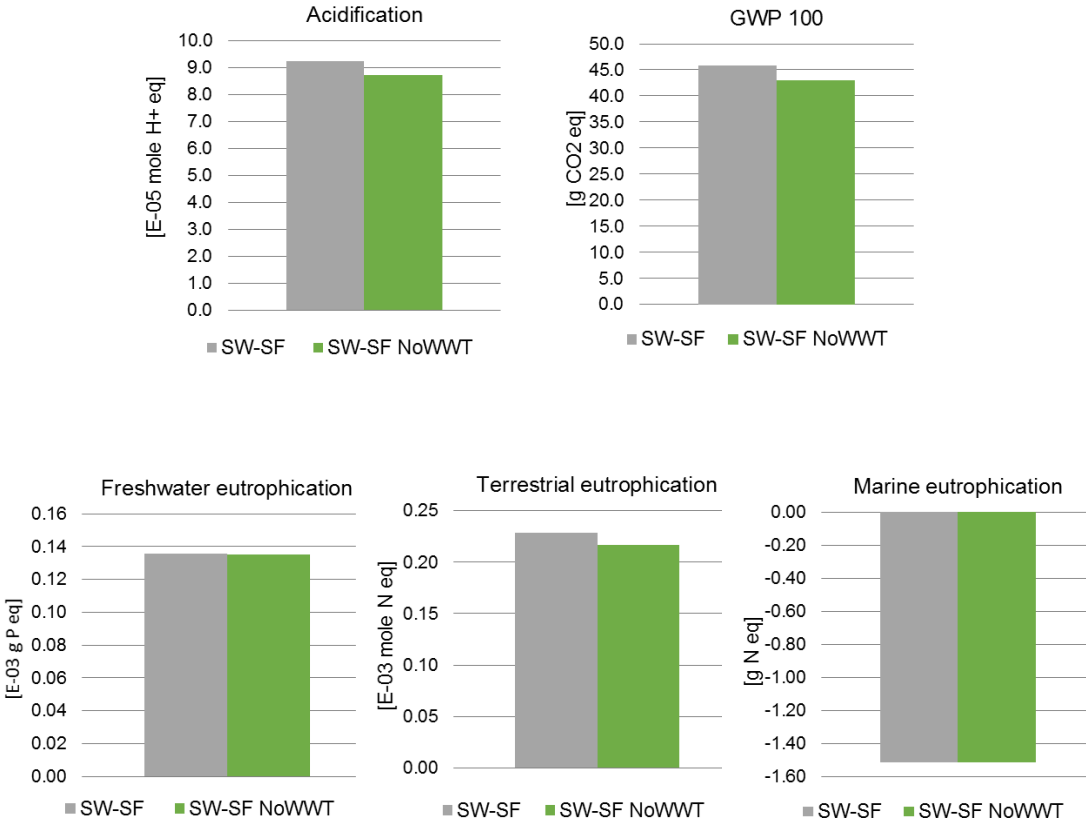


Fig. 6.7 Comparison of environmental impacts of 1 MJ of biomethane from *L. digitata* for scenarios SW-SF (UV-WWT) and SW-SF_{NoWWT} (no wastewater treatment in hatchery), excluding digestate handling and credit (these are the same for both scenarios).

Impact of seasonal variation and increase yields in *L. digitata*

Improvement in characteristics of *L. digitata* as a consequence of seasonal variation, and an increase in SMY by 21% (SW-SF_{August}) led to a decrease in overall environmental impacts, except for marine eutrophication as compared to the baseline (Fig. 6.8 and Fig. 6.9, detailed results in Appendix B, Table B.5). Lower impacts are observed for all scenarios with August seaweed (SW-SF_{August}, SW-$SF_{40t\ August}$ and SW-$SF_{100t\ August}$). As compared to SW-SF, the savings in GWP 100 are between 15% (SW-SF_{August}) and 48% (SW-$SF_{100t\ August}$); in acidification between 26% (SW-SF_{August}) and 62% (SW-$SF_{100t\ August}$); in freshwater eutrophication between 17% (SW-SF_{August}) and 43% (SW-$SF_{100t\ August}$); and in terrestrial eutrophication between 27% (SW-SF_{August}) and 72% (SW-$SF_{100t\ August}$). In case of marine eutrophication (Fig. 6.9), all scenarios with IMTA are emissions negative, with the highest emissions cut for SW-SF, SW-SF_{40t} and SW-SF_{100t}. The emissions savings are also slightly lower for SW- SF_{August} and other August scenarios if compared

with SW-SF. This is due to the lower N content in August seaweed (1.14% DS) as compared to October seaweed (1.22% DS), and lower demand for feedstock per MJ of biomethane produced from August seaweed (0.126 kg DS/MJ produced) as compared to October seaweed (0.102 kg DS/MJ produced). When analysing all the scenarios, higher DS, VS and SMY appear to be more significant than an increase in seaweed yield per unit area.

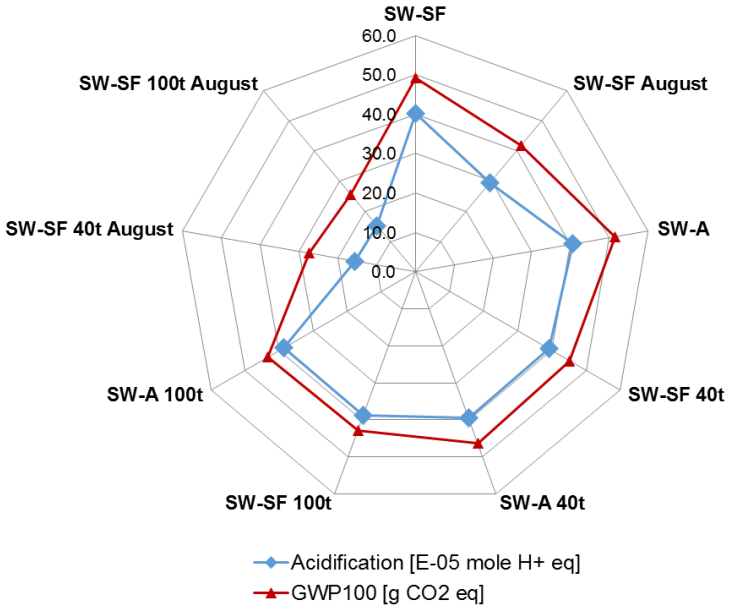


Fig. 6.8 Acidification and climate change potentials of 1 MJ of biomethane from *L. digitata* in the sensitivity scenarios (as detailed in Table 6.1).

Scenario SW-A generates the worst case with higher impact in all categories as compared to SW-SF (Fig. 6.8 and Fig. 6.9). All scenarios with stand-alone seaweed farm (SW-A, SW-A_{40t} and SW-A_{100t}) have the highest impact in marine eutrophication (0.32 g N eq/MJ) since they do not benefit from uptake of nitrogen from salmon farm during seaweed growth.

Scenarios SW-$SF_{40t\ August}$ and SW-$SF_{100t\ August}$ which combine very good seaweed characteristics (optimum harvest in August), increased yields due to proximity to fish farm, higher renewable electricity input in production chain, and 70% replacement of

mineral fertiliser show a strong decline in all environmental impacts (15.13 E-05 mole H^+ eq, 25.62 g CO_2 eq., 0.50 E-03 g P eq., 0.43 E-03 mole N eq. and -1.11 g N eq. per MJ of biomethane for SW-$SF_{100t\ August}$).

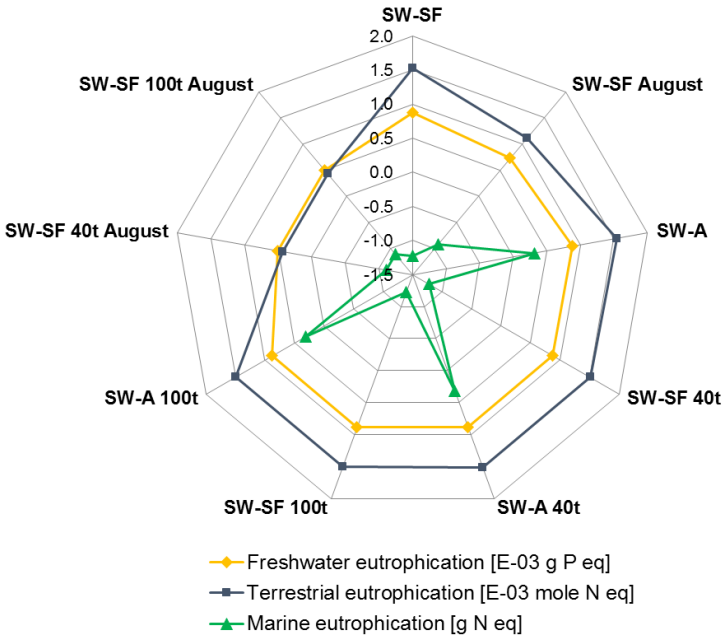


Fig. 6.9 Marine, terrestrial and freshwater eutrophication potentials of 1 MJ of biomethane from *L. digitata* in the sensitivity scenarios (as detailed in Table 6.1).

Replacement of fossil fuel electricity with the renewable electricity mix

A sensitivity analysis was conducted, in which the electricity mix used throughout all the life cycle was replaced with more renewable mixes (Fig. 6.10 and Table B.3a, Appendix B). The results are presented as percentage of the baseline SW-SF (100%), and do not include digestate impacts and credits, as these are the same for all three scenarios. The difference between the scenarios is especially visible for the GWP 100, with SW-SF_{Wind} showing 34% lower impact and SW-$SF_{2020\ projection}$ 12% lower impact, as compared to SW-SF. Acidification potential is 26% lower for SW-SF_{Wind} and 4% lower for SW-$SF_{2020\ projection}$. This is due to a decrease in acidifying gases from fossil fuel combustion which increase soil and water acidity by accumulation of hydrogen ions. Pressure on terrestrial eutrophication decreases with an increase of renewable inputs to the electricity grid; it is 25% lower for SW-SF_{Wind}, and 3% lower for SW-

$SF_{2020\ projection}$. Marine eutrophication varies only marginally for the three scenarios analysed. However, freshwater eutrophication is slightly higher for both SW-SF_{Wind} (5%) and SW-$SF_{2020\ projection}$ (1%). This is because of higher biomass and biogas input in the system. Bioenergy electricity is based on a mix of feedstock, and includes also farmed crops that are associated with use of fertilisers and pesticides (Agostini et al. 2015).

When digestate fate was included, the major difference between scenarios is between the GWP 100 results with 31% savings in SW-SF_{Wind} as compared to SW-SF (Table B.3b, Appendix B).

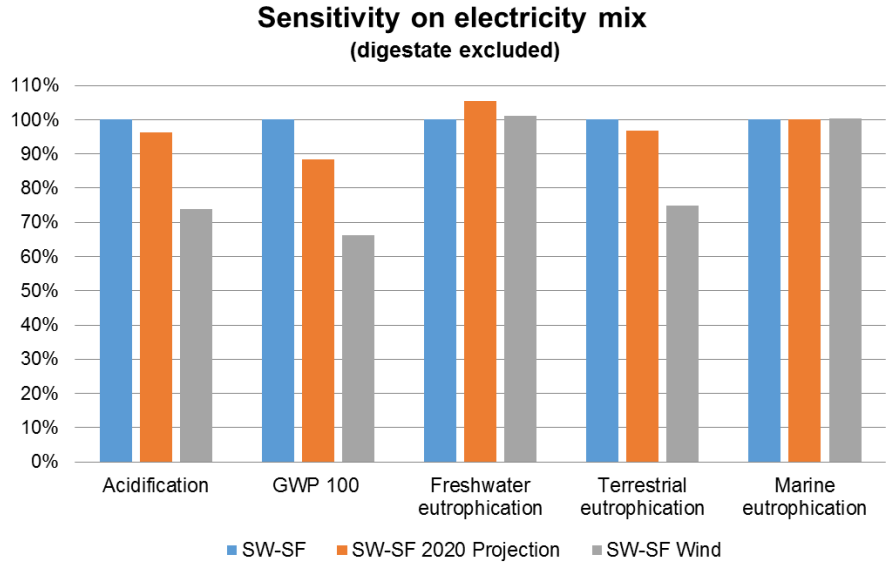


Fig. 6.10 Comparison of environmental impacts of 1 MJ biomethane of 1) SW-SF scenario with current Irish electricity grid, 2) SW-$SF_{2020\ projection}$ based on 2020 projections, and 3) SW-SF_{Wind} with 48% electricity coming directly from wind turbine and 52% from 2020 Irish grid (digestate handling and credit are excluded as these are the same for all scenarios).

6.3.3. Comparison with fossil fuel

The results of the LCA of 1 km driven on biomethane from seaweed were compared with results for 1 km driven on generic compressed natural gas (CNG) or gasoline consumed in the EU with both upstream and combustion emissions included (Table 6.6). The baseline scenario SW-SF performs worse than natural gas and gasoline in almost all impact categories, except for GWP 100 and marine eutrophication. In terms

of GWP 100, this scenario provides 27% carbon savings when compared to CNG, and 44% when compared to gasoline. Seaweed scenarios are always better in terms of marine eutrophication generating an environmental benefit (between -1.40 and 2.22 g N eq/km driven) in all scenarios considered. When 70% replacement of mineral fertiliser was assumed (SW-$SF_{70\%}$), the carbon savings in relation to CNG and gasoline increase (33 and 48% respectively), but seaweed biomethane is still worse in other environmental impacts such as acidification and freshwater and terrestrial eutrophication. With SW-SF_{August}, there is further decline in GWP 100 and savings are 38 and 52% as compared to CNG and gasoline, respectively.

Carbon savings exceed 60% for both SW-$SF_{40t\ August}$ (68%) and SW-$SF_{100t\ August}$ (70%) as compared to gasoline (59% and 61% respectively as compared to CNG). The two scenarios perform also much better than the baseline in other impact categories; however both CNG and gasoline are still better in acidification, and freshwater and terrestrial eutrophication. In terms of marine eutrophication, both scenarios are emissions negative.

In case of both fossil fuels, 80% of carbon emissions come from the use of fuel, while for biomethane the combustion emissions represent only between 2 and 5% of total GWP 100 potential. Large majority of biomethane impact is in the production phase. The remaining 20% of carbon emissions from fossil fuel comparators are primarily related to the refining activities (energy input, refining technology, gaseous emissions, and leaks of crude oil and hazardous substances), and transport of crude oil by tanker (from combustion). For the other environmental impact categories, both in case of fossil fuels and biomethane, the potential impacts come only from the production and use stage.

Table 6.6 Comparison of the environmental impacts of 1 km driven on biomethane with 1 km driven on natural gas or gasoline.

Impact categories and units	**SW-SF**	**SW-SF$_{70\%}$**	**SW-SF$_{August}$**	**SW-SF$_{August\ 70\%}$**	**SW-SF$_{40t\ August}$**	**SW-SF$_{100t\ August}$**	**Natural gas**	**Gasoline**
Acidification [E-05 mole H+ eq.]	60.95	37.23	44.81	26.72	23.64	23.00	4.91	16.15
GWP 100 [g CO_2 eq.]	76.55	70.49	65.29	60.67	43.33	40.62	105.07	135.94
Freshwater eutrophication [E-03 g P eq.]	1.34	0.89	1.10	0.77	0.76	0.76	0.01	0.39
Terrestrial eutrophication [E-03 mole N eq.]	2.33	1.08	1.69	0.74	0.67	0.65	0.09	0.28
Marine eutrophication [g N eq.]	-1.86	-2.22	-1.40	-1.67	-1.68	-1.68	0.01	0.03

6.3.4. Limitations of study

Digestate is a crucial by-product of biogas production with potential to reduce application of mineral fertilisers. However, the exact fertilising potential of digestate from various substrates are still being investigated (Möller 2015); furthermore, there is also a lack of awareness and established practices among farmers which leads to a sub optimal reduction in mineral fertiliser application. This may be improved through educational programmes and cooperation between agricultural authorities and farmers to assure high replacement rate.

If considering only the impacts from digestate handling, the majority of impacts come from the nitrogen field emissions. Based on studies to date, nitrogen emissions from digestate field application may play an important role in the environmental footprint of biogas systems, and have a significant contribution to its GWP potential (Cherubini and Strømman 2011). Nitrogen losses occur via nitrification and denitrification processes in the soil (N-N_2O emissions), volatilisation of ammonia during spreading (N-NH_3 and N-NO) and nitrate loss via leaching to groundwater (N-NO^{3-}). While existing studies usually include the direct N_2O emissions as part of the nitrogen balance, the indirect emissions from volatilisation of ammonia and leaching of nitrates are assessed in less detail (Cherubini and Strømman 2011). In this analysis these losses appear to be significant. Data used follow Giuntoli et al. (Giuntoli et al. 2015) and Battini et al. (Battini et al. 2014). The nitrogen modelling in this study could be improved by including an array of specific factors; these include crop type, fertiliser type (organic vs. mineral), soil characteristics such as organic C and N content, and climate (Stehfest and Bouwman 2006). The model proposed in this paper may be therefore improved by including more specific modelling of nitrogen in digestate life cycle in specific Irish conditions.

The analysed LCA model includes the advantage of coupling salmon and seaweed farming in two forms; 1) by increasing the yields of algae per unit area, and 2) by including the nitrogen credit from fish waste uptake by seaweed. When comparing stand-alone farms (SW-A scenarios) with integrated farms (SW-SF scenarios), it appears that the reduction of the pollution from fish farming benefits the system significantly by increasing the environmental benefits.

Seawater treatment used in the hatchery processes appeared important for an overall LCA result. So far this aspect of hatchery was understudied with some sources omitting this stage (Langlois et al. 2012; Alvarado-Morales et al. 2013) or assuming that waste seawater can be safely released to the environment without treatment (pers. comm.). It is a foreign DNA contamination issue rather than typical waste water/ nutrients issue. There is a point of view that if waste seawater from hatchery would be released uncontrollably in large quantities, this might alter the habitats of native algae species present in given location (pers. comm. Dr Maeve Edwards, Irish Seaweed Consultancy Ltd.). One way to prevent that is to release water only in locations populated with the farmed species. Taelman et al assumed a complex pre- and post-treatment processes including drum filter, pump and UV disinfection unit (Taelman et al. 2015). Hence, it seems sensible to maintain a model requiring a treatment of wastewater. The exact electricity inputs in current model might be studied in further details.

6.4. Conclusions

The baseline scenario (SW-SF) analysed in this study showed relatively higher environmental impacts than natural gas and gasoline, in agreement with previous studies (Langlois et al. 2012). In terms of GWP 100, SW-SF emitted 49.26 g CO_2 eq/MJ of biomethane produced; however, if seaweed was harvested in August, the emissions were reduced to 41.85 g CO_2 eq/MJ of biomethane produced. The most sustainable scenarios, SW-$SF_{40t\ August}$ and SW-$SF_{100t\ August}$ showed a low GWP 100 at 27.40 and 25.62 g CO_2 eq/MJ, respectively. These two scenarios show also low impacts in the other environmental categories addressed. When combustion emissions were included, SW-$SF_{40t\ August}$ and SW-$SF_{100t\ August}$ demonstrated 59% and 61% savings in terms of GWP 100 as compared to CNG, and 68% and 70% when compared to gasoline.

A proper management of digestate can offset the carbon emission by 3.0 to 7.0 g CO_2 eq/MJ biomethane at plant gate. In such case, an assumed quantity (30 or 70%) of mineral fertiliser is not produced, and losses from P and N during application of fertilisers are avoided. This suggests that good management practices related to the storage and application of digestate on fields are therefore important to ensure the competitiveness of biomethane with fossil fuels.

When digestate fate is excluded, the highest impacts in seaweed biomethane production come from seaweed farming (27% of GWP100 and 53% on average in all impact categories), which is due mainly to the use of fossil fuels in transport at sea. This is in line with previous studies that found that the majority of impacts comes from seaweed production, and mainly from transport and infrastructure (Alvarado-Morales et al. 2013; Taelman et al. 2015).

In terms of marine eutrophication, seaweed biomethane proved to be emission negative in all scenarios with the integrated system with seaweed cultivation placed next to a salmon farm (IMTA system). The sea pollution from fish was reduced by seaweed accumulating the nitrogen-rich excrements for its growth.

Seaweed composition seems to be the key factor in decreasing the overall impacts, and was more significant than an increase in yield per hectare. The results suggest that, to ensure the lowest emissions, the seaweed should be harvested at the best time possible, when its characteristic are the most suitable for anaerobic digestion with high DS, VS and SMY. These are influenced by seasonality as the optimal characteristic are to be found for *L. digitata* harvested in August. Results showed that high DS and VS content, and SMY of harvested seaweed have much more impact on decreasing the net environmental impacts of biomethane than increasing the yield per unit hectare. The seasonality of seaweed should be an important component in development of seaweed biofuel systems. Furthermore, the electricity mix used in processes can be improved by increasing the input from renewable sources such as wind. These are the main recommendations on how to enhance the sustainability of algal biomethane as assessed in scenarios SW-$SF_{40t\ August}$ and SW-$SF_{100t\ August}$. This study was conducted for Ireland, but the findings can be applied to any country with a temperate oceanic climate with similar access to marine resources

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Chapter 7. Conclusions and recommendations

7.1. Introduction

This thesis set out to explore sustainable solutions for advanced indigenous biomethane systems for Ireland, a country with a temperate oceanic climate. This is an important subject of research in the context of the challenging EU 2020 targets for climate mitigation and renewable energy in transport, which puts an obligation on each member state to develop a strategy for developing green energy. Ireland has still a lot to do to achieve the 2020 10% RES-T target (in 2014, renewables amounted only to 5.2% of transport energy). Ireland also needs to establish an indigenous renewable transport fuel industry, as currently the country imports nearly 90% of the energy it consumes. Moreover, in the light of EU legislation to be deemed sustainable biofuels should demonstrate a minimum 70% savings in GHG as compared to fossil fuels after 2020. Therefore, there is a need for research that supports policymakers and industry with evidence on available domestic and sustainable transport fuels.

This thesis sought to answer the following questions:

- identify the guidelines for environmental assessment of biofuels and the indicators crucial to assess the biofuels sustainability;
- evaluate the impact of land use change emissions on meeting the renewable and GHG targets; and
- identify key optimal indigenous biomethane options for Ireland.

First, LCA as the most suitable tool for the evaluation of biofuels was assessed. Proposed guidelines for full LCA were followed later in the assessment of seaweed biomethane. The land use was found to be one of the crucial impact categories; therefore, the impact of land use change emissions from biofuels on meeting the 2020 targets was assessed further. More detail was given to two biofuel systems: a simple and easily accessible landfill gas, and a ‘biofuel of the future’, seaweed biomethane, an advanced biofuel with a potential for successful development in Ireland.

7.2. Overview of findings

7.2.1. Guidelines for LCA of biofuel (chapter 3)

So far, the only available integrated tool for LCA of biofuels is BioGrace that was developed by JRC and is based on the RED (EC 2009a). However, BioGrace allows only for GHG calculation, and allocation is based on energy content. In this thesis, it is proposed that LCA of biofuels should use more than one FU, in particular in the case of land-based biofuels (*per ha* basis) and transport fuels (*per km* basis). It is also advocated to report results per energy content of biofuel (*per MJ*). Lower environmental impacts and higher energy outputs per ha indicate a reduced risk of land use change emissions. RED recommends to use the allocation by energy content (EC 2009c); however, a comprehensive understanding of a system requires expanding the system boundaries in order to include the system's by- and co-products (such as digestate). Also, in as much as possible a complex system should be divided and data should be collected for smaller sub-processes. If possible, LCA studies of biofuels should include more than solely GHG/GWP calculation. Direct land use changes should be included, as well as the eutrophication and acidification potentials, alongside with the energy balance when assessing biofuels. A sensitivity analysis should be conducted to test the uncertainty of the data, assumptions and methods used in the LCA. These guidelines were followed in Chapter 6 when conducting the LCA of an integrated salmon-seaweed for biomethane system.

7.2.1. Land use change emissions in biofuels (chapter 4)

The emissions from land use changes were found crucial for assuring the sustainability of biofuel in the long term; this is particularly valid for the land-based feedstocks. The implications of including ILUC in the modelling of energy systems are very severe, and would require the development of technologies that are so far still very expensive, such as hydrogen fuel and Carbon Capture and Sequestration (CCS). When factoring in high emissions from ILUC in the Irish energy system, the modelled contribution of biofuels drastically decreases by 2050; 66% of transport fuel would come from hydrogen, residues-based biodiesel (tallow) and hybrid cars/EVs, and the system would remain largely dependent on imported energy. Moreover, the CO_2 abatement cost would increase by 61% compared to the baseline scenario with no LUC, as this scenario involves new advanced technologies still under development. Assuming

lower emissions from ILUC, indigenous grass biomethane from permanent grassland would become a major source of biofuel. This scenario would require only between 5 and 11% of Irish agricultural land to be converted to biofuel production. This is in line with previous studies on the potential for biomethane in Ireland, which showed that 25% of private cars in Ireland could be fuelled by biomethane from excess grass together with OFMSW and agricultural and slaughter waste (Thamsiriroj et al. 2011). Finally, the impact of DLUC emissions on the system is not very high, and would allow for almost 40% of bioenergy share in the total primary energy by 2050.

7.2.2. Economic potential of small-scale LFG upgrading to biomethane (chapter 5)

LFG is an easily accessible gas in Ireland. For small-scale landfill sites, LFG upgrading to biomethane can be economically viable, under certain conditions. Cleaning and upgrading technology should be simple, cheap and adapted to the LFG composition. The most optimal system is a single step PSA system, with an on-site service station for direct use of produced biomethane. The cost of this system (including propane addition and service station) is €0.84/m_n^3 biomethane. This is cheaper than the virtual pipeline with gas grid injection. However, for the investment to be profitable, there is a need for financial support; in Ireland, such support is provided under the BOC scheme. In essence, a subsidy of €0.55/m_n^3 of biomethane is required if the market value of biomethane is €0.29/m_n^3. This financial incentive can be lowered depending on the LFG composition. In the case analysed in chapter 5, a very high nitrogen content was assumed; that made the process of LFG upgrading much more challenging and expensive than when the nitrogen content is kept as low as possible through better design of landfill sites and LFG collection system. More realistic scenario could be to transport by road the upgraded biomethane to industrial sites for further use.

7.2.3. LCA of the integrated salmon and seaweed farming for biomethane system (chapter 6)

Among the alternatives for advanced biofuels, Chapter 6 studied a system based on one of Ireland's staple industries, salmon farming, integrated with seaweed cultivation for biomethane production. Seaweed biomethane does not create increase demand for salmon, but offers a solution to decrease environmental impacts from salmon farming.

Under the baseline scenario with seaweed yielding 25.4 t wwt/ha in an IMTA system, the results of the LCA showed relatively high environmental impacts, with the highest impact in digestate handling accounting alone for 11% of GWP, and over 80% (on average) in all other impact categories; this is due to the digestate impact on the nutrient cycle. AD plant operation, PSA and seaweed farming contribute also largely to GHG emissions (86% in total of GWP). However, the benefit from integrated salmon and seaweed farming is high for marine eutrophication, and seaweed biomethane can be emission negative when the uptake of nitrogen from salmon farming is considered. Seaweed composition seems to be the key factor for decreasing the overall impacts, and has a more significant impact than an increase in yield of seaweed per hectare. In terms of GWP 100, the baseline scenario produced 49.26 g CO_2 eq/MJ, while biomethane from seaweed harvested in August was 41.85 g CO_2 eq/MJ of biomethane produced. Seaweed therefore should be harvested at the best time possible, which in case of *L. digitata* in Ireland is in August.

The way to improve seaweed biomethane sustainability is through optimizing the system, which can be done by 1) ensuring proper management of digestate with high replacement of mineral fertiliser; 2) optimizing harvest time of seaweed to ensure high DS, VS and SMY; 3) cultivating seaweed in an integrated system with fish farming; 4) achieving high yields of seaweed per unit area; 5) using highly renewable electricity mix. Under the optimal scenario, and accounting for biofuel combustion in a car engine, the system achieved a 61% and 70% saving in GHG emissions compared to CNG and gasoline, respectively.

7.3. Policy and industry implication

There is no wrong way of conducting a LCA, as long as LCA transparency is assured, and all hypotheses and assumptions are documented. However, if studies are to be compared, there is a need for setting up a unified methodology. LCA can be time consuming and expensive to conduct, but a simplified LCA framework could be developed and shared with the industry.

Chapter 4 showed that land use change emissions are of great importance for assessing the sustainability of biofuels. These findings are crucial for policymakers, as LUC should be included in strategies for meeting EU renewables and climate mitigation

targets. So far, the EU put a 7% cap on first generation biofuels, and stressed the importance of investing in advanced biofuels (EC 2015b). This cap might be extended to 3.8% by 2030 with advanced biofuels making up at least 3.6% of fuels in transport (EC 2016). However, there is a point of view that first generation biofuels demonstrating high emissions reduction and low ILUC should not be excluded from policy strategies, as the development pace of advanced biofuels is still very slow. In fact, the efforts should be in setting the national goals taking into account local and regional conditions in determining the country biofuel potential. The risk of ILUC can be considerably decreased if high productivity per ha is assured and underutilised land is used for cultivation of bioenergy crops.

There is still a lot of ground to cover to meet the 2020 renewable transport target in Ireland. Biomethane from landfill gas and seaweed (*L. digitata*) as well as grass, MSW and slurries, can be part of the solution especially as these are advanced zero or low ILUC and locally sourced feedstocks. Biomethane from those systems can make up to 30% of total energy in transport by 2050 (in the optimistic ILUC scenario, excluding aviation).

These are important considerations for: 1) policymakers at a national level; for small-scale production to be financially viable there is a need for an appropriate financial incentive; 2) and natural gas providers; although there is a potential for locally sourced, advanced biofuels, investments and collaboration with potential biogas providers will be needed to develop a national green gas network.

7.4. Recommendations and future research

Based on the research presented in this thesis, some recommendations can be made:

- There is a need for a unified and simplified biofuel LCA framework that will allow comparisons between different biofuel systems. Recommendations for choosing the functional unit, defining the system boundaries and reference system, and choosing the allocation and impact categories, are outlined in Chapter 3, section 4;
- Land use change emissions should be included in national policy strategies for achieving the renewable transport targets; underutilised land can be considered for low ILUC risk cultivation; more attention should be given to advanced

biofuels based on indigenous feedstocks, for example through the development of support schemes.

- A financial incentive is necessary for making advanced biofuels economically viable. A subsidy of minimum €52/MWh (€0.55/m_n^3) biomethane is required for LFG project to breakeven. Studies outlined in Chapter 5 suggest that a single step PSA system with on-site service station as the most economically viable option.
- Biomethane from seaweed is a possible option for green gas growth; the most optimal system integrated with salmon farming demonstrated significant reduction in impact on climate change and marine eutrophication, and should be developed further in collaboration with industries and local communities, as outlined in Chapter 6, section 3.

Future research should concentrate on delivering a full LCA of biomethane from other second and third generation feedstocks that will play a major part in meeting Ireland's renewable transport targets. These include grass, agriculture wastes (slurries and manures), food waste and landfill gas. Seaweed can be co-digested with any of those to create an optimal mix of substrates for AD. While feedstock options for biomethane production are highly country-specific, the research findings of this thesis can be transposed to other countries with similar climatic conditions and land/agricultural resources.

Future research on seaweed biomethane from IMTA systems should focus on the detailed energy balance and economic viability of the RES. While seaweed biomethane commercialisation is still at an early stage of development, an initial cost analysis would bring additional value to future work on these systems, and possibly help attract industry partners.

7.5. Final statement

Despite their presence in the current energy market, the share of first generation biofuels in total energy in transport by 2050 will greatly decrease when ILUC is factored in, while hydrogen, hybrid cars and EVs, and advanced biofuels will gain in importance.

Biomethane from second and third generation feedstocks, such as grass, seaweed, landfill gas and waste and residues, can play a key role in expanding the contribution of indigenous green gas in transport. But for this to happen, there is a need for 1) optimization of the specific system in order to maximize benefits and minimize impacts, and 2) development of a financial support scheme to allow small-scale projects to be commercialized.

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Appendix A. Details on data used in the modelling of LCA in GaBi

Box A.1 Requirements for development of seaweed culture in laboratory tanks (Arbona and Molla 2007).

- 1 m seeded long-line will give 10 kg of wet weight (wwt) *L. digitata* (pers. comm.)
- One (seeded seaweed) collector can be used to seed 30 m of long line
- One litre of culture media (nutrients and water) is needed for 8 collectors
- One 500 L tank can hold 15 collectors
- 2 led pipes per tank
- One pump per tank
- dw content is 17.66%

1000 kg x 1m/ 10 kg = 100 m / t wwt and 566 m/ t dw

100m/30m = 3.33 collectors / t wwt and 19 collectors / t dw

3.33 collectors x 1 tank/15 collectors = 0.22 tanks / t wwt and 1.26 tank / t dw

2 tubes x 0.22= 0.44 led pipes / t wwt and 2.5 led / t dw

0.22 pumps / t wwt and 1.26 pumps / t dw

Box A.2 Yield of seaweed per hectare explained.

- 1 m seeded long-line will give 10 kg wwt *L. digitata* (pers. comm.)
- 20 longlines each 100 m long in a hectare

Basic yield (seaweed alone scenario SW-SA)

10 kg wwt/1m of longline x 100 m x 20 longlines/ha = 20 000 kg= 20 t wwt/ha

Increased yield by 27% (baseline scenario SW-SF)

20 t wwt/ha + 27% = 25.4 t wwt/ha

Table A.1 Electricity mixes used in the GaBi modelling for the sensitivity analysis1) SW-SF (and all other scenarios unless specified), 2) SW-$SF_{2020\ projection}$, and 3) SW-SF_{Wind} (based on: SEAI 2012a and SEAI 2012b).

	GaBi Irish electrcity mix	**Assumptions for Gabi** 2020	**Theoretical** if 50% from wind turbine, 50% from 2020 grid
Biomass	0.65%	3%	1.50%
Biogas	0.72%	3%	1.50%
Hard coal	19.93%	18.91%	9.46%
HFO (heavy fuel oil)	0.90%	0%	0%
Hydro and ocean	3.67%	2%	1.00%
Natural gas	49.78%	33.59%	16.79%
Peat	9.43%	7.50%	3.75%
Wind	14.53%	32%	66.00%
Waste to energy	0.39%	0%	0%

Appendix B. Detailed results of LCA analysis

Table B.1 Cradle to gate environmental impacts of 1 MJ of biomethane from *L. digitata* excluding the benefits from digestate in the baseline scenario SW-SF.

Impact categories and units	Digestate handling	AD	Ensiling	PSA upgrading and compression	Seaweed farming (UV WWT)	Seaweed transport
Acidification [E-05 mole H^+ eq.]	**42.58**	1.42	0.27	2.34	4.77	0.42
GWP 100 [g CO_2 eq.]	6.513	**15.96**	1.36	**15.36**	12.55	0.51
Freshwater eutrophication [E-03 g P eq.]	**0.962**	0.004	0.003	0.003	0.124	0.003
Terrestrial eutrophication [E-03 mole N eq.]	**1.916**	0.051	0.007	0.057	0.092	0.022
Marine eutrophication [g N eq.]	**0.473**	0.005	0.001	0.005	-1.527	0.002

Table B.2 Comparison of environmental impacts and benefits of 1 MJ of biomethane from *L. digitata* for scenarios SW-$SF_{70\%}$ (70% replacement) and SW-SF (30% replacement of mineral fertiliser).

Impact categories and units	SW-$SF_{70\%}$	SW-SF
Acidification [E-05 mole H^+ eq.]	24.49	40.10
GWP 100 [g CO_2 eq.]	45.27	49.26
Freshwater eutrophication [E-03 g P eq.]	0.588	0.880
Terrestrial eutrophication [E-03 mole N eq.]	0.710	1.530
Marine eutrophication [g N eq.]	-1.461	-1.221

Table B.3a Sensitivity analysis of electricity mix used in the processes for scenarios: 1) SW-SF (and all other scenarios unless specified), 2) SW-$SF_{2020\ projection}$, and 3) SW-SF_{Wind}. Digestate handling and credits are excluded.

Impact categories and units	**SW-SF**	**SW-$SF_{2020\ projection}$**	**SW-SF_{Wind}**
Acidification [E-05 mole H^+ eq.]	9.22	8.87	6.82
GWP 100 [g CO_2 eq.]	45.77	40.47	30.35
Freshwater eutrophication [E-03 g P eq.]	0.136	0.143	0.137
Terrestrial eutrophication [E-03 mole N eq.]	0.229	0.221	0.171
Marine eutrophication [g N eq.]	-1.515	-1.515	-1.520

Table B.3b Sensitivity analysis of electricity mix used in the processes for scenarios: 1) SW-SF (and all other scenarios unless specified), 2) SW-$SF_{2020\ projection}$, and 3) SW-SF_{Wind}. Digestate handling and credits are included.

Impact categories and units	**SW-SF**	**SW-$SF_{2020\ projection}$**	**SW-SF_{Wind}**
Acidification [E-05 mole H^+ eq.]	40.10	39.74	37.70
GWP 100 [g CO_2 eq.]	49.26	43.99	33.87
Freshwater eutrophication [E-03 g P eq.]	0.880	0.887	0.881
Terrestrial eutrophication [E-03 mole N eq.]	1.530	1.522	1.472
Marine eutrophication [g N eq.]	-1.221	-1.222	-1.227

Table B.4 Comparison of environmental impacts of 1 MJ of biomethane from *L. digitata* for scenarios SW-SF (UV-WWT) and SW-SF_{NoWWT} (no wastewater treatment in hatchery). Digestate handling and credits are excluded.

Impact categories and units	**SW-SF**	**SW-SF_{NoWWT}**
Acidification [E-05 mole H^+ eq.]	9.22	8.72
GWP 100 [g CO_2 eq.]	45.77	42.92
Freshwater eutrophication [E-03 g P eq.]	0.136	0.135
Terrestrial eutrophication [E-03 mole N eq.]	0.229	0.216
Marine eutrophication [g N eq.]	-1.515	-1.516

Table B.5 Comparison of the environmental impacts of 1 MJ of biomethane in various scenarios that include changes in seasonal variation and increase yields in *L. digitata.*

Impact categories and units	**SW-SF**	**SW-SF$_{August}$**	**SW-A**	**SW-SF$_{40t}$**	**SW-A$_{40t}$**	**SW-SF$_{100t}$**	**SW-A$_{100t}$**	**SW-SF$_{40t}$ August**	**SW-SF$_{100t}$ August**
Acidification [E-05 mole H+ eq.]	40.10	29.48	40.61	39.14	39.40	38.67	38.56	15.56	15.13
GWP 100 [g CO_2 eq.]	49.26	41.85	51.44	45.23	46.32	42.82	43.25	27.40	25.62
Freshwater eutrophication [E-03 g P eq.]	0.880	0.726	0.881	0.878	0.878	0.877	0.876	0.503	0.502
Terrestrial eutrophication [E-03 mole N eq.]	1.530	1.110	1.541	1.510	1.516	1.501	1.498	0.438	0.429
Marine eutrophication [g N eq.]	-1.221	-0.919	0.317	-1.223	0.314	-1.224	0.313	-1.105	-1.106

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