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Life cycle assessment of seaweed biomethane, generated from seaweed sourced from integrated multi-trophic aquaculture 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 sustainability of seaweed biomethane, with seaweed sourced from an integrated seaweed and salmon farm 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 a transport biofuel. The baseline scenario shows high emissions in all impact categories. An optimal seaweed biomethane system can achieve 70% savings in GHG emissions as compared to gasoline with high yields per hectare, optimum seaweed composition and proper digestate management. Seaweed harvested in August proved to have higher methane yield. 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).

Highlights

1) Seaweed composition is the key factor in decreasing environmental impacts
2) Digestate handling and storage is a large contributor to impacts
3) Proper management of digestate offsets carbon emissions by 3 to 7 g CO_2 eq/MJ
4) Seaweed farming represents 53% of impacts in production of seaweed biomethane
5) Seaweed biomethane can deliver over 60% carbon savings as compared to fossil fuel

1. Introduction

1.1. Rationale for seaweed biomethane

The EU is committed to achieve at least 20% renewable energy share of gross energy consumption by 2020, rising to 27% by 2030 [1,2]. The Renewable Energy Directive (RED) requires 10% of renewable energy in transport by 2020 [1]. Biofuels have an important role in achieving transport targets but their sustainability must be ensured [1,3]. Land-based biofuels may compete directly or indirectly for land associated with food production [4]. An amendment to the RED sets a cap on first generation (land-based) biofuels to 7% of transport fuel and suggests an indicative target of 0.5% for advanced biofuels, such as algae [3,5]. The algal biofuel sector is immature, but does include for start-up companies and is supported by EU-funded projects [6,7]. Macro-algae (seaweed) do not compete for land and as such seaweed biomethane is considered an advanced (third generation) renewable gaseous fuel, which can be counted at twice its energy content in consideration of 2020 national renewable energy targets [5].

Of the 221 species of seaweed commercially used, 66% of use is for food [8], with the remaining in agrichemicals, fish feed, health and cosmetic sectors [9]. Cultivated seaweed production represents 95% of the seaweed market and produced 24 million tonnes wet weight (wwt) in 2013 [10,11]. The industry has a value of over €6 billion [12]. Farming of seaweed can be integrated with salmon farming in an integrated multi-trophic aquaculture (IMTA) system. This circular economy concept reduces the impacts from nutrient-rich waste released from fish farms, whilst enhancing the growth of seaweed [13–15]. Aquaculture produces over 2 million tonnes live weight each year of Atlantic salmon (*Salmo salar*) [10]. If an average price of €5 per kilogram is assumed [16], this gives a market worth €10 billion. The Food and Agriculture Organization (FAO) estimated that there will be a need to produce an extra 42 million tonnes of farmed seafood to feed the world by 2030, and salmon will play a key role in fulfilling this demand [17]. However, research shows that farmed salmon has a very high environmental footprint [18–21]. Implementation of efficient IMTA has the potential to increase the sustainability of aquaculture systems by minimising the risk of eutrophication in marine environments. Furthermore, the seaweeds become an additional product with additional revenues for the fish farmers [9,22]. The green seaweed *Ulva sp*. and the red seaweed *Gracilaria chilensis* were shown to have enhanced growth levels cultivated close to fish cages than in in control sites [13–15]. *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 [23]. It was also observed that *S. latissima* had a faster growth in an IMTA system than at a reference station [24]. *Saccharina latissima* and *Laminaria digitata* brown seaweeds native to northern Europe, are suitable for IMTA, due to their N uptake capacity and yield improvements in proximity to fish farms [9,25].

Experimental studies indicate the suitability of seaweed substrates for methane production [26–29]. In assessing yields of biomethane from seaweed, the fluctuation in the seaweed supply over the year and the seasonal variation in the chemical composition of seaweed for different species [30] must be assessed.

1.2. Life cycle assessment of seaweed biomethane associated with integrated seaweed and salmon farms

Life cycle assessment (LCA) is accepted as the most suitable tool for the sustainability assessment of algal projects [7,31]. Full comprehensive LCA studies of seaweed biofuels are scarce in the literature;

to date algal studies focused mainly on microalgal liquid biofuel systems [6,31,32]. Moreover, the majority of algal LCA papers only examine climate change as the impact category [32].

Taelman et al. [33] compared two off-shore cultivation systems of *S. latissima*; long-line (Ireland) and a raft system (France). 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. Results in Ireland show that about 81% of the impacts are related to 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.

The study of Langlois et al. [34] dealt with the environmental impacts of biomethane from the anaerobic digestion (AD) of the whole seaweed (*S. latissima*) and from alginate-extraction residues. Seaweed biomethane has important benefits for marine and freshwater eutrophication as seaweed removes eutrophying pollutants (N and P) from the surrounding seawater during growth. However, the study found that the overall environmental impact of seaweed biomethane could be higher when compared with natural gas, in terms of climate change, ozone depletion and human toxicity. The authors suggested that an optimal system including for 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 greatly improve the environmental footprint of seaweed biomethane.

Alvarado-Morales et al. [35] assessed the energy demands and environmental impacts of biofuel produced from *L. digitata* grown on long-lines in Nordic conditions for two seaweed biofuel systems. 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 biogas scenario performed better than bioethanol scenario for all the impacts categories considered. The difference between the two scenarios was linked 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) [36]. 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 water-to-wheel (well-to-wheel[1]) 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 is unknown and it is essential to assess if this third generation algal biofuel is actually sustainable and how the system could be optimised to ensure sustainability. 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.

[1] Feedstock is produced at sea; the ‘well’ is ‘water’ [61].

1.3. Aims and objectives

The aim of this paper is to assess the potential environmental impacts and benefits of integrated seaweed 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.

2. Methods

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 (Figure 1). The baseline scenario (Seaweed and Salmon farming system, SW-SF, Table 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. System expansion was applied; 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. 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 2.5).

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 [37]. 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 [38]. 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/tVS) at an organic loading rate of 2 kg VS/m^3 per day for feedstock collected in October [39] 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 studies by the authors were included as well as papers on LCA of seaweed biofuels [34,35,38,39]. GaBi database provided 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 reference fossil fuel-based systems (sections: 2.5 and 3.3), emissions from transport vehicles were included (well-to-wheels approach and FU of 1 km driven on biomethane).

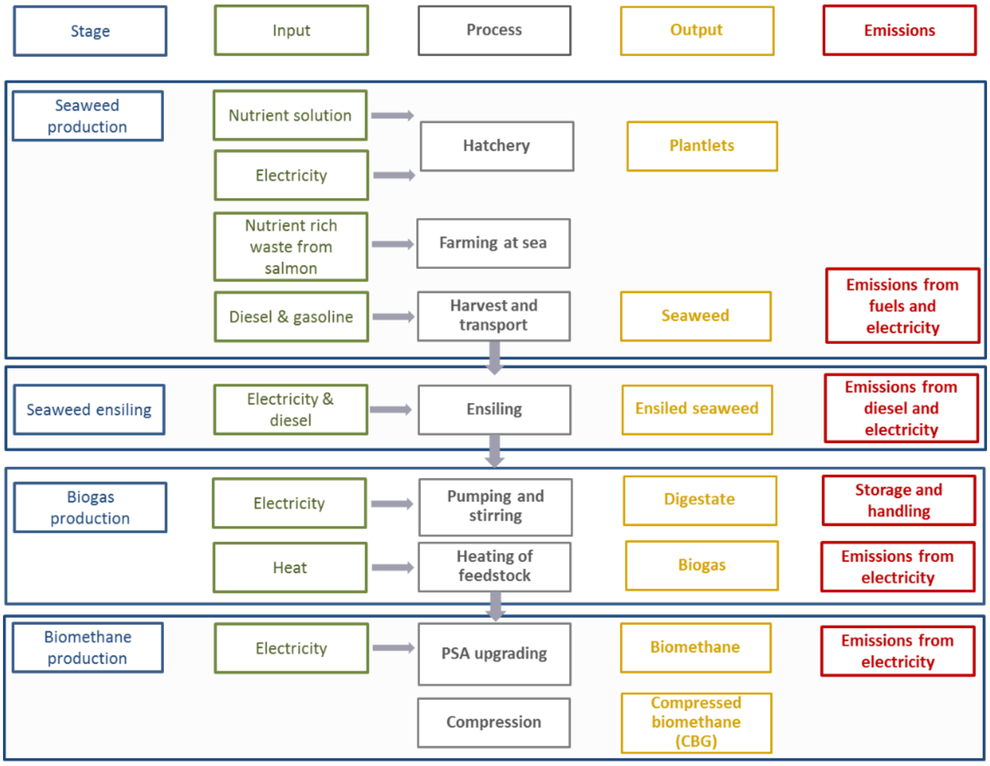

Figure 1 Integrated seaweed and salmon farming system for biomethane production (water-to-tank approach).

Table 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	Yield	DS content	VS content	BMP (batch)	SMY (CSTR)	Waste water treatment in hatchery	Digestate substitution of mineral fertiliser	Electricity grid mix	Additional comments
	(Yes/No)	(t wwt seaweed/ ha)	(%)	(%)	(m^3 CH_4/ t VS)	(m^3 CH_4/ t VS)		(%)	(Irish/renewable mix)	
• **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-SF$_{70\%}$	Yes	25.4	17.7	14.42	267	338[3]	UV-WWT	**70**	Irish mix	Increased yield: 20+27%= 25.42
SW-SF$_{NoWWT}$	Yes	25.4	17.7	14.42	267	338[3]	**No-WWT**	30	Irish mix	No water treatment
SW-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-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-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-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. [39]
[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 [38]

2.3. Life cycle inventory

2.3.1. Salmon farming

Salmon farming inputs and outputs were considered outside the system boundary. This decision is justified because it is assumed that increase demand for seaweed biomethane should not create an increase demand in 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 [23]. 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) [13], it was assumed that productivity of *L. digitata* is enhanced as much as *S. latissima*, when grown next to fish cages. The ability of nitrogen removal from seawater by cultivating (and enhancing yield of) seaweed for seaweed biomethane production as assessed by LCA is unique to this paper. For this purpose, the nitrogen excreted by salmon and absorbed by seaweed was calculated and assigned to the biomethane system in the form of 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 [39]. 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 [40]. Supplementary data and calculations related to this credit are presented in Box 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 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 [40]

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

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

2.3.2. Cultivation of *Laminaria digitata*

The outline of procedure for cultivating *L. digitata* is presented in Figure 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 for 20 hours 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 [41]. In this study, 35 days were assumed with aeration running full time and illumination for 12h per day.

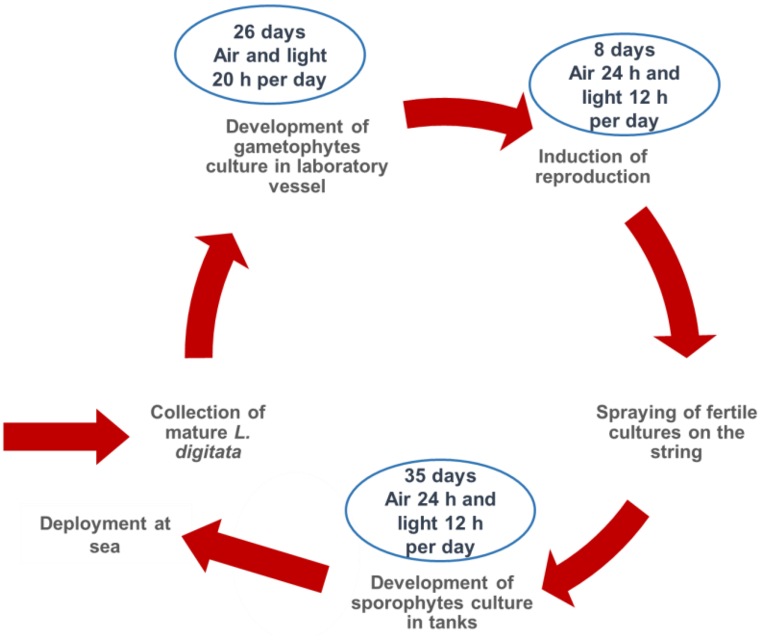

Figure 2 Overview of the cultivation procedure for L. *digitata* (based on Edwards and Watson [41]).

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 [41]. 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 [34]. Tanks need to be cleaned and the medium changed every 3 days (11 times in total); each time 50% of volume is exchanged [41]. 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 [41].

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 ($SW\text{-}SF_{NoWWT}$ Table 1).

2.3.3. Deployment at sea

In Ireland deployment occurs between October and December [41]. It was assumed that *L. digitata* is cultivated on the West Coast of Ireland in Galway Bay using long-lines, each 100 meters long (Figure 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.

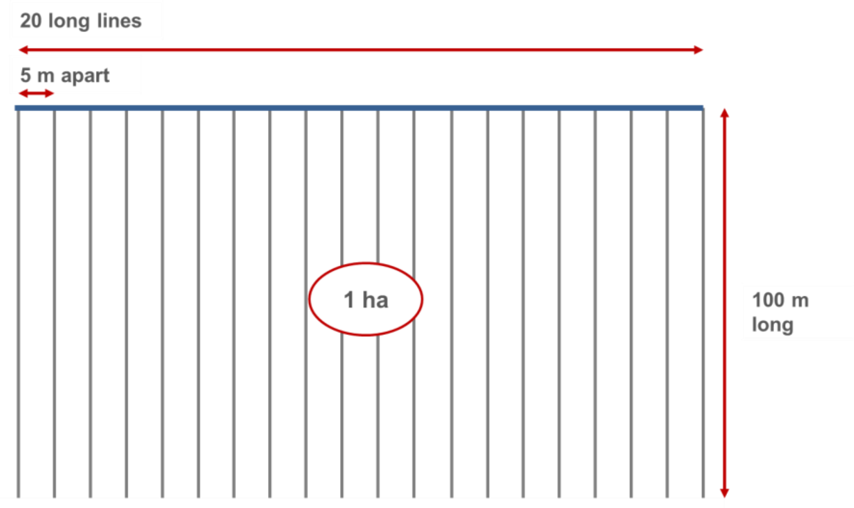

Figure 3 Distribution of long-lines in a hectare of water surface for seaweed cultivation.

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 [34]. 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).

2.3.4. 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 naturally lowers 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 [42]. The BMP of the ensiled seaweed is the sum of the BMPs of ensiled biomass and effluent produced during ensiling [42]; all the effluent is recirculated to the digester and the fugitive CH_4 emissions from ensiling were assumed to be nill. Energy input for the loading of seaweed into the tower pit for ensiling was assumed to be 7 MJ/t wwt, similar to that considered by Berglund and Börjesson [43] 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% [39].

Biogas from experimental data, typically has a 55% CH_4 content [39]. 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 [44]. Digester electrical demand was assumed at 10 kW_eh/ t wwt of substrate [45]. Thermal demand was calculated assuming specific heat capacity of water at 4.184 MJ/ t / °C, 85% boiler efficiency and 15 % heat losses [44]. 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 [46,47] (Table 2).

Table 2 Methane losses in the biomethane process.

Process	Value	Unit	Source
AD plant	1	% of produced CH_4	[46,47]
PSA upgrading	2	% of produced CH_4	[48]
Digestate storage	10.6	g CH_4/t wwt digestate (0.03% of produced CH_4)	[49]

2.3.5. PSA upgrading

Upgrading of biogas was modelled with a pressure swing adsorption (PSA) process, which is described in detail by Beil et al. [48]. The system operates at a pressure of 4-7 bars, with a methane recovery rate of 98%; methane losses to the atmosphere during the process are at most 2%, however the majority of this would be oxidised [48]. The analysis, which assumes 2% losses is conservative. 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 [50]. 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 [51].

2.3.6. 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 CH_4 losses from manure management [49] (Table 2). Digestate is stored in a gas-tight closed tank (Table 3), and the emissions of the closed storage system were assumed to be 2% of the open storage system [46]. Emissions from field application of digestate on cropland were based on literature values and were calculated according to Battini et al. (Table 4). Direct N-N_2O emissions were estimated at 1% of applied nitrogen [49] and N-NO at 0.55% [52]. Ammonia losses were estimated at 0.22 kg N/ t wwt digestate [53]. Nitrates leaching (N-NO_3) was assumed to be 30% of applied nitrogen [54]. CO_2 field emissions were considered negligible. Phosphorus losses in form of phosphate (PO_4^{3-}) run-off to freshwater were estimated at 1% of total P content in digestate and mineral fertiliser [55] (Table 4). P content in *L. digitata* was assumed to be 0.77 g P/kg DS [56].

Table 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-N_2O)	0.12[2]	g N/ t wwt digestate	[49]
Ammonia (N-NH_3)	0.16	g N/ t wwt digestate	[53]
Nitrogen oxides (N-NO)	0.002	% N in digestate	[53]
Nitrogen (N-N_2)	0.02	% N in digestate	[46]

[1] 2% of the emissions from open tank
[2] 0.13 g N/ t wwt digestate for seaweed harvested in August (SW-SF_{August})

Table 4 Losses of nitrogen and phosphate during field application of digestate and mineral fertiliser for seaweed collected in October and August.

<table>
<tr><th>Emissions</th><th colspan="2">Digestate losses</th><th>Source</th><th colspan="2">Mineral fertiliser losses</th><th>Source</th></tr>
<tr><td>Nitrous oxide direct (N-N_2O)</td><td>1</td><td>% N at field</td><td>[49]</td><td>1</td><td>% N mineral</td><td>[49]</td></tr>
<tr><td>Ammonia (N-NH_3)</td><td>220</td><td>g N/ t wwt digestate</td><td>[53]</td><td rowspan="2">10</td><td rowspan="2">% N mineral</td><td rowspan="2">[49]</td></tr>
<tr><td>Nitrogen oxides (N-NO)</td><td>0.55</td><td>% N at field</td><td>[52]</td></tr>
<tr><td>Nitrates (N-NO_3)</td><td>30</td><td>% N at field</td><td>[54]</td><td>30</td><td>% N mineral</td><td>[49]</td></tr>
<tr><td>Phosphate</td><td>1</td><td>% total P content</td><td>[55]</td><td>1</td><td>% total P content</td><td>[55]</td></tr>
</table>

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 the 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 N during digestate storage and field application are presented in Table 3 and Table 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 [57]. 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 [58]. 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 [37]). Avoided emissions from application of mineral fertiliser were also included.

2.4. Sensitivity analysis

2.4.1. 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 1). In the sensitivity analysis an optimistic approach was assumed with 70% replacement (SW-$SF_{70\%}$, SW-$SF_{August\ 70\%}$, SW-$SF_{40\ tAugust}$ and SW-$SF_{100\ tAugust}$). 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-N_2O, N-NH_3, N-NO and N-NO_3) (Table 4).

2.4.2. 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 [38] as opposed to seaweed collected in October (Table 1). The same inputs from hatchery and deployment at sea were assumed for both scenarios.

2.4.3. 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 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 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 [30].

2.4.4. 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% [59]. It was assumed that the hydropower is 2%, and fossil fuels are coal (19%), natural gas (34%) and peat (7.5%) [60]. 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%.

2.4.5. 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 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.

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 [37]. 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 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 [61]. 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 [62]. Biogenic CO_2 emissions from biomethane combustion were set to 0.

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 [63] as implemented in Gabi software [64]. 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 [65]. 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 [64]. 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. The 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 [64]. 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 [64]. Freshwater and marine eutrophication impacts are caused by waterborne emissions, such as nitrate, phosphate and other N and P compounds [64]. 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.

3. Results and discussion

3.1. Contribution analysis of baseline scenario

Digestate handling, storage and field application, is the largest contributor in all impact categories (Figure 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 these emissions are offset by digestate replacing mineral fertiliser and the benefit from capturing 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.

The impact of digestate handling comes from the emissions from storage (despite the closed tank) and field application of digestate (Table 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 due to the CH_4 and N losses assumed in this study (Table 2 and Table 3).

When considering scenario SW-SF without digestate fate (Figure 5), the largest contribution in all impact categories apart from marine eutrophication 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 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, 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). This is related to the significant energy inputs of fossil fuels in these processes.

3.2. Sensitivity analysis

3.2.1. Impact of digestate replacing mineral fertiliser

Figure 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.

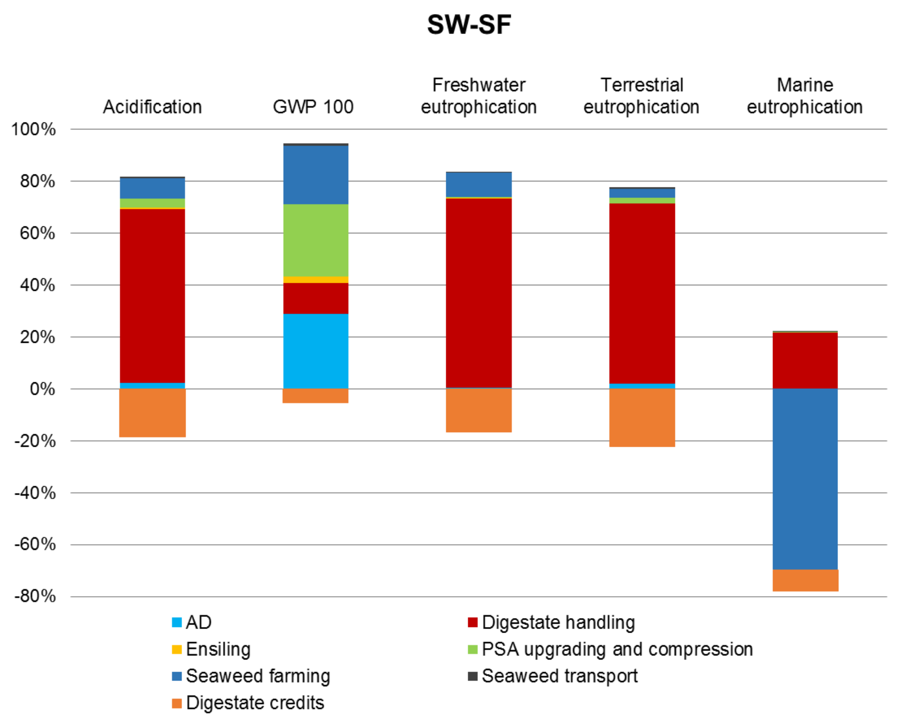

Figure 4 Cradle to gate environmental impacts of 1 MJ of biomethane from *L. digitata* in the baseline scenario SW-SF including both impacts and benefits from digestate (30% replacement).

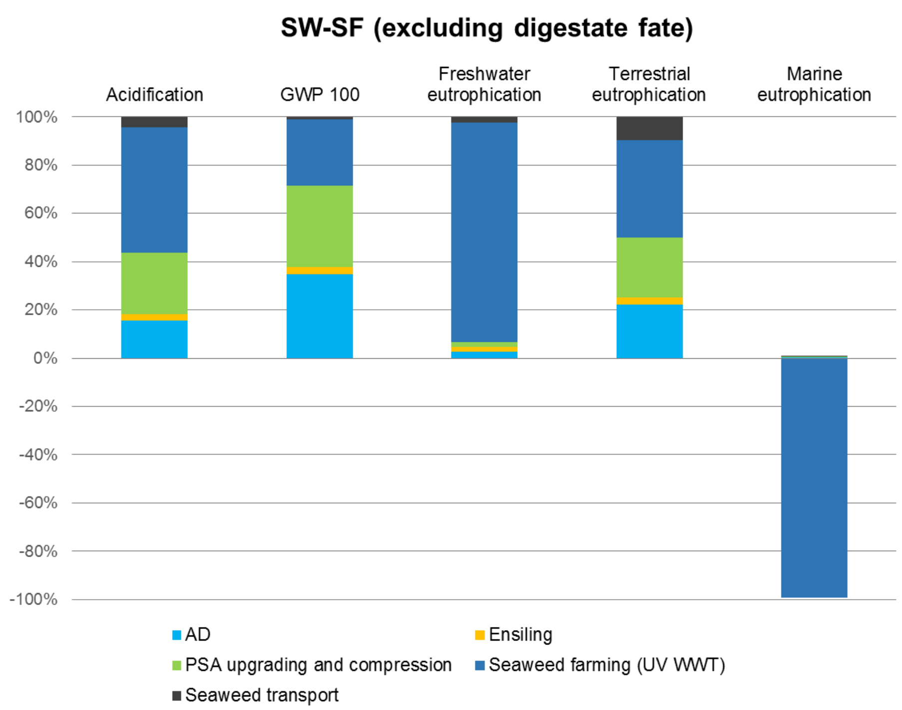

Figure 5 Cradle to gate environmental impacts of 1 MJ of biomethane from *L. digitata* excluding impacts and benefits from digestate in the baseline scenario SW-SF.

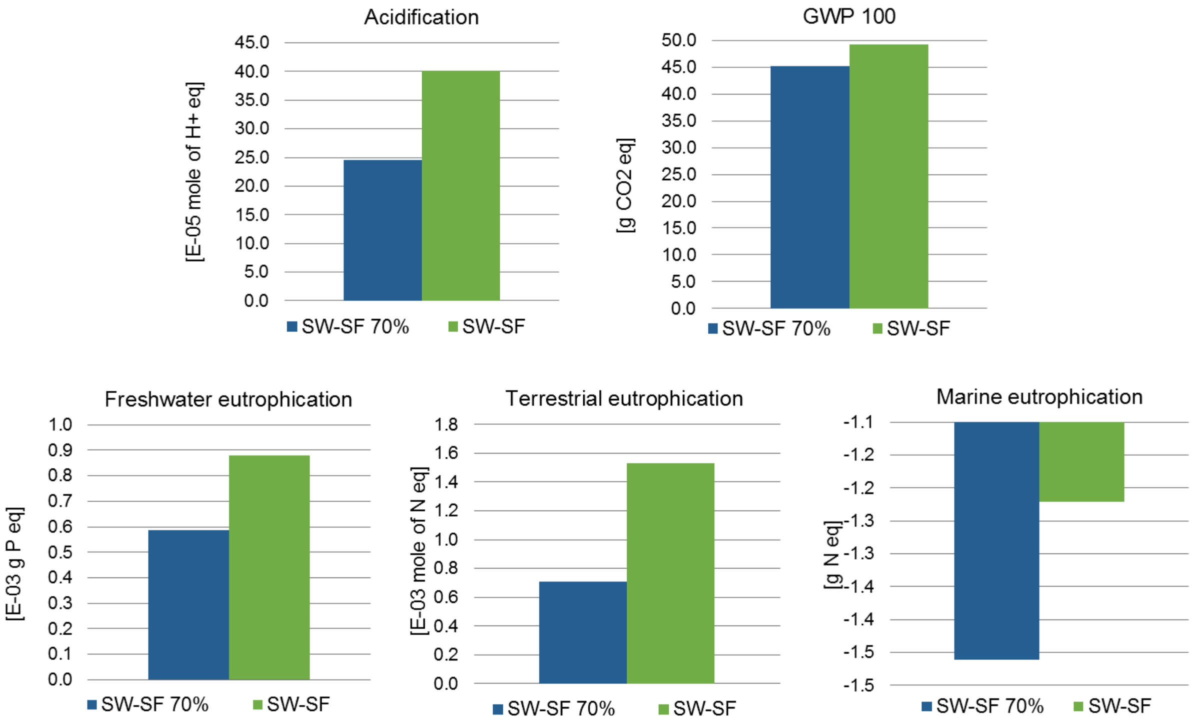

Figure 6 Comparison of environmental impacts of 1 MJ of biomethane from *L. digitata* for scenarios SW-SF$_{70\%}$ (70% replacement) and SW-SF (30% replacement of mineral fertiliser).

Table 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

3.2.2. Wastewater treatment in hatchery

The impact of wastewater treatment in the hatchery was tested (Figure 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.

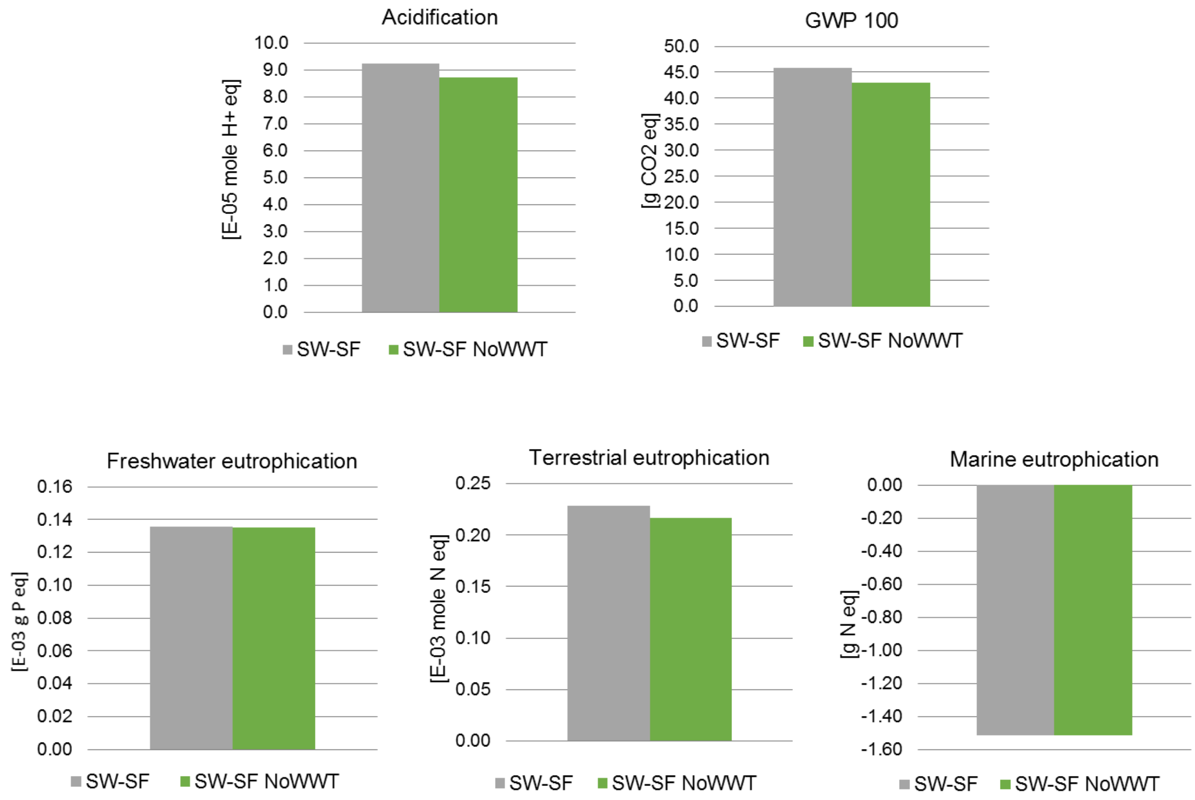

Figure 7 Comparison of environmental impacts of 1 MJ of biomethane from *L. digitata* for scenarios SW-SF (UV-WWT) and SW-SF$_{NoWWT}$ (no wastewater treatment in hatchery), excluding digestate handling and credit (these are the same for both scenarios).

3.2.3. 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 (Figure 8 and Figure 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 (Figure 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.

Scenario SW-A generates the worst case with higher impact in all categories as compared to SW-SF (Figure 8 and Figure 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-$SF_{100t\ August}$).

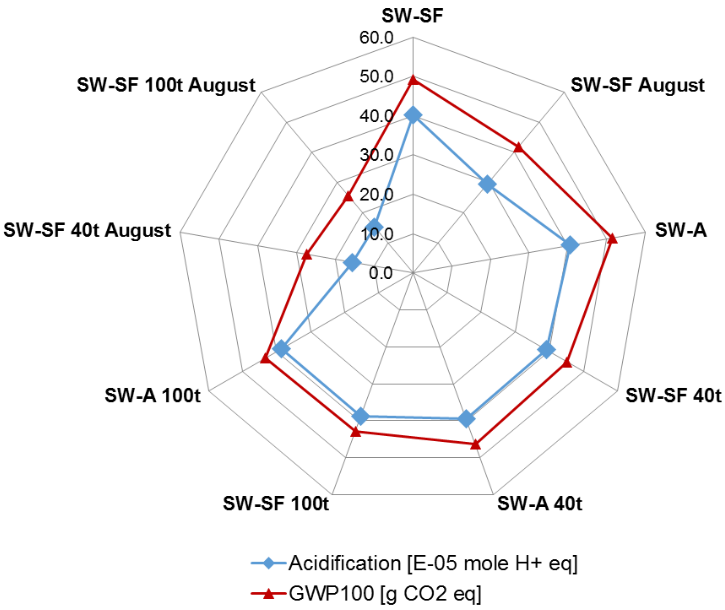

Figure 8 Acidification and climate change potentials of 1 MJ of biomethane from *L. digitata* in the sensitivity scenarios (as detailed in Table 1)

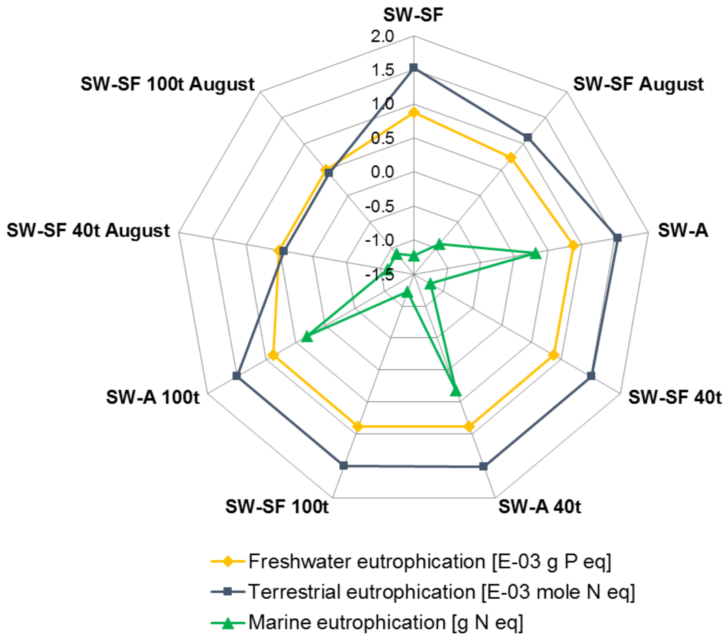

Figure 9 Marine, terrestrial and freshwater eutrophication potentials of 1 MJ of biomethane from *L. digitata* in the sensitivity scenarios (as detailed in Table 1)

3.2.4. 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 (Figure 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$_{\text{Wind}}$ showing a 34% lower impact and SW-SF$_{\text{2020 projection}}$ a 12% lower impact, as compared to SW-SF. Acidification potential is 26% lower for SW-SF$_{\text{Wind}}$ and 4% lower for SW-SF$_{\text{2020 projection}}$. 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$_{\text{Wind}}$, and 3% lower for SW-SF$_{\text{2020 projection}}$. Marine eutrophication varies only marginally for the three scenarios analysed. However, freshwater eutrophication is slightly higher for both SW-SF$_{\text{Wind}}$ (5%) and SW-SF$_{\text{2020 projection}}$ (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 [66].

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

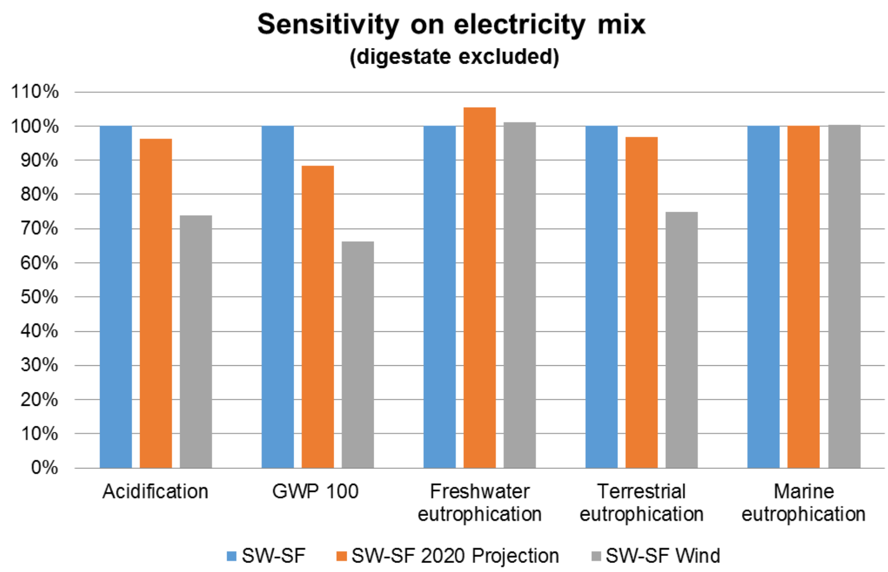

Figure 10 Comparison of environmental impacts of 1 MJ biomethane of 1) SW-SF scenario with current Irish electricity grid, 2) SW-SF$_{2020\ projection}$ based on 2020 projections, and 3) SW-SF$_{Wind}$ with 48% electricity coming directly from wind turbine and 52% from 2020 Irish grid (digestate handling and credit are excluded as these are the same for all scenarios).

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). 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. The large majority of the 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 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

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 [67]; 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 [68]. 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 [68]. In this analysis these losses appear to be significant. Data used follow the methodologies of Giuntoli et al. [69] and Battini et al. [46]. 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 [52]. 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 [34,35] 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 [33] assumed a complex pre- and post-treatment processes including drum filter, pump and UV disinfection unit. 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.

4. Conclusions

The future of energy requires renewable and sustainable energy systems. In particular biofuels need to decarbonise energy whilst minimising impact on the environment. The RED and its proposed amendments [1,3] require a minimum 60% greenhouse gas savings for transport biofuels as compared to the fossil fuel displaced on a whole life cycle analysis basis as of 2017; this rises to 70% beyond 2020 [70]. Questions that must be asked include: Can biofuels such as third generation algal biofuels meet these GHG thresholds? In meeting these decarbonising thresholds are there other impacts that are negative to the environment? In this assessment of seaweed biomethane the following pertinent points were uncovered:

Seaweed cultivated on its own in pristine waters, not associated with salmon farms, has a lower yield of seaweed per hectare than in a systems associated with a salmon farm. Seaweed performs a circular economy role of removing excess nutrients while yields of seaweed increase by over 27%. Seaweed biomethane associated with seaweed cultivated on its own has higher impacts than seaweed associated with fish farms across all categories assessed, especially marine eutrophication.

When analysing all the scenarios it was found that the optimal scenario is more closely related to optimal seaweed composition than to optimal yield per hectare. Thus higher dry solid content, volatile solid content, lower salt content and associated higher specific methane yield are more significant than an increase in seaweed yield per unit area. The higher specification of composition is associated with seaweed harvested when it is *ripened,* which is typically in August for *L. digitata*. This optimal harvest is more important that optimal yield per hectare.

A base case conservative non–optimised system, using unripened seaweed, and fossil electricity in the biogas system, with minimum replacement of mineral fertiliser can be deemed unsustainable generating 76.6 g CO_2 eq as compared to natural gas (105 g CO_2 eq.). This is only a 27% savings. However an optimised system (adjacent to a fish farm, using electricity with low carbon intensity, allowing for 70% effective substitution of the nitrogen in mineral fertiliser with the nitrogen in the digestate, harvested when seaweed is ripened, cultivated an a membrane yielding 100t wet weight (as opposed to cultivating on a long line) can be deemed sustainable generating 40.6 g CO_2 eq as compared to gasoline (136 g CO_2 eq.) This is a 70% savings.

It must be noted that when a biofuel is produced locally including for the proximity principle, then there will be impacts that would not be there if a fossil fuel were simply imported. Thus imported gasoline does not require application of digestate to land and as such does not cause eutrophication. However digestate from seaweed digestion does require land application of digestate and may lead to eutrophication. These impacts are minimised through optimal systems that manage the digestate spread process in such a way as to not over apply and to maximise the substitution of mineral fertiliser.

This study is based on oceanic waters in Ireland, but can be applied to any seaweed biomethane system in temperate oceanic climates including Northern Europe, Northern America, and Asia.

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Appendices

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 [1].

- 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} [2,3].

	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) $SW\text{-}SF_{2020\ projection}$, and 3) $SW\text{-}SF_{Wind}$. Digestate handling and credits are excluded.

Impact categories and units	**SW-SF**	**$SW\text{-}SF_{2020\ projection}$**	**$SW\text{-}SF_{Wind}$**
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) $SW\text{-}SF_{2020\ projection}$, and 3) $SW\text{-}SF_{Wind}$. Digestate handling and credits are included.

Impact categories and units	**SW-SF**	**$SW\text{-}SF_{2020\ projection}$**	**$SW\text{-}SF_{Wind}$**
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 $SW\text{-}SF_{NoWWT}$ (no wastewater treatment in hatchery). Digestate handling and credits are excluded.

Impact categories and units	**SW-SF**	**$SW\text{-}SF_{NoWWT}$**
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

References for the appendices

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