Environmental Life Cycle Assessment (LCA) of algae production in North West Europe (NWE)

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Environmental Life Cycle Assessment (LCA) of algae production in North West Europe (NWE)

Content

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1. INTRODUCTION

The work presented in this report was undertaken within the context of the EnAlgae project; a 4-year Strategic Initiative of the INTERREG IVB North West Europe (NWE) programme which took place between 2011 and 2015.

1.1. Sustainability assessment of algae production systems

Although algae are claimed to be a sustainable resource, there has been an increasing awareness of the possible impact of algae production on the natural environment. Life cycle assessment (LCA) can be used as a tool to quantify all relevant emissions and resources consumed, as well as the related environmental impacts and resource depletion associated with a product's life cycle. LCA takes into account the full lifecycle: from the extraction of resources, through production, use, recycling, to disposal of the remaining waste (Rebitzer et al. 2004). Hence, LCA makes it possible to identify opportunities to improve the environmental footprint of products at different phases of their life cycle. It can be used for decision makers in industry and (non-) governmental organizations.

In this report, the framework of the International Standards Organization (ISO) 14040 and 14044 was followed to assess the environmental sustainability of several algae production systems developed within the EnAlgae project (ISO, 2006a,b). The first phase of an LCA study consists of defining the goal and scope of the study, followed by a thorough inventory analysis, a life cycle impact assessment (LCIA) step and an interpretation phase (see Figure 1).
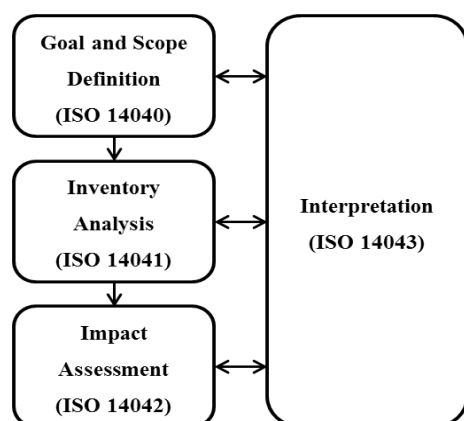
Figure 1: LCA as a 4-phase process according to the ISO standards 14040: goal and scope definition, inventory analysis, impact assessment and interpretation.

During the first phase, the purpose of the study has to be clearly defined. In addition, the scope should describe the detail and depth of the study, and show that the goal can be met with the actual extent of the limitations. The following aspects should be considered and described: the product system, the functions of the product system, the functional unit and reference flow, the system boundaries, possible allocation procedures, the environmental impact assessment methodology, the data (quality) requirements and the

assumptions and limitations. Through the iterative character of the LCA, adjustments can easily be made during the LCA analysis.

The system boundaries describe which processes within the life cycle of a product are included to provide the function of the system, defined by the functional unit. The functional unit represents a reference to which all inputs and outputs of the inventory data are related and normalized (Hou et al. 2011; Roy et al. 2009).

To deal with processes that produce more than one product (e.g. algal meal and oil), it is important to partition all inputs and outputs (other than products, e.g., emissions) to the co-products under study. Several types of allocation already exist; e.g., based on energy or exergy content, based on mass or costs. However, according to ISO 14041, wherever possible, allocation should be avoided. As a result, system expansion (with substitution) was given a prominent place in LCA, i.e. the product or function that can be replaced by the co-product/co-function is being studied (quantification of emissions and resources). Nevertheless, expanding the product system to include additional functions related to the co-products is not always feasible (Weidema, 2001). System expansion may involve processes that also generate multiple products, i.e. it would involve an unending regression. Another example is when a by-product does not substitute for another alternative product, system expansion may be regarded as incompatible with the requirement that compared systems must have identical functions.

As a result, two types of LCAs are described in the literature: attributional LCA (ALCA) and consequential LCA (CLCA). ALCA is defined by its focus on the environmental burdens (average data) that are associated with the life cycle of the good and/or service produced within the system. The environmentally relevant physical flows of a past, current or potential future product system are described. The CLCA is a more market-oriented approach and is defined by its goal to identify the environmental consequences of changes that are based on a decision. A CLCA ideally includes marginal data to be able to include the marginal technologies that contribute to the environmental consequences of a change. Allocation is typically avoided through system expansion by using substitution (Finnveden et al., 2009). Because it is not convenient to know which processes are affected by a certain change in the time and/or space, the CLCA concept is usually more complex than attributional LCA. Moreover, the results obtained are highly sensitive to the assumptions that are made, which can result in a poor analysis (Sokka and Soimakallio, 2009).

All LCA studies as discussed in this report used a common environmental impact assessment method; the Cumulative Exergy Extraction from the Natural Environment (CEENE) method (Dewulf et al., 2007). This method accounts for the consumption of natural resources (e.g. fresh water and minerals) and is based on thermodynamics through quantification of these resources in terms of their exergy content. Exergy is a thermodynamically based measure and is defined as the maximal amount of work a system can deliver in equilibrium with its environment via a reversible process. The use of the unit 'exergy' instead of e.g., 'energy' has some advantages: (1) not only the quantity but also the quality of a resource can be assessed, (2) all of the resources can be expressed in the same unit, which enhances comparisons, and (3) because exergy is not conserved, it exposes inefficient processes, which indicate the loss of work potential. An important strategy for improving the sustainability of real processes is to reduce the rate of exergy loss, i.e., entropy production, or to increase the exergy efficiency (De Meester et al., 2009; Dewulf et al., 2008). The resources considered in the CEENE method are divided in 8 categories: renewable resources, fossil fuels, nuclear energy, metal ores, minerals, water resources, land occupation and atmospheric resources (Dewulf et al., 2007). Therefore, the CEENE method is more consistent than the Cumulative Energy Demand (CED) and the Cumulative Exergy Demand (CExD) methods by accounting for both non-energetic resources as well as land use (Dewulf et al., 2007). Alvarenga et al. (2013) provided an enhanced approach to the CEENE method: instead of using available photosynthetic solar exergy as a proxy for areal land occupation, the net primary production (NPP) of the

potential natural vegetation is used because it better represents the exergy that is deprived from the natural environment. Furthermore, a site-specific approach was developed taking into consideration the spatial differentiation of land resources. Taelman et al. (2014) developed an extended version of the CEENE 2013 method to account for marine resources when determining the life cycle resource footprint. Spatially differentiated characterization factors to account for sea surface occupation were calculated based on the potential net primary production (NPP) available in the photic zone.

To conduct the different LCA studies, the software SimaPro 8.0.3.14, openLCA 1.3.4 and/or Umberto NXT have been used. To provide information on the background processes, the database ecoinvent version 2.2 was selected.

Specific for the Enalgae pilots, different goals were set up according to the main interest of the pilot operators. Because the choice of the functional unit, the system boundaries and allocation methods depend on each specific goal made, a direct comparison between all LCA studies was not possible. In addition, each study had its own limitations and assumptions. However, this should not be a drawback to make overall conclusions on the environmental sustainability of algae production in NWE.

1.2. Objectives

Environmental concerns regarding natural resource depletion have led to the cultivation of more renewable resources such as algal biomass. Algae are a very rich type of biomass and offer great potential as an ingredient for the chemical, cosmetic, pharmaceutical, energy and food or feed industry. Algae can also play an important role in waste stream mitigation. However, the development of algal biomass production technology faces several technical, economic and environmental barriers.

This report is part of the EnAlgae Work package 2, Action 11, dedicated to the sustainability analysis of algae production. As the cultivation in Europe is still in its early stages, an estimation of the environmental sustainability may boost further development of this sector by highlighting its competitiveness. Hence, this report contains three objectives:

1)   Calculate the environmental resource footprint of algae cultivation (and processing into valuable products) in NWE for a few detailed case-studies (cfr. EnAlgae pilots)

2)   Based on these LCA results, draw conclusions on the environmental sustainability of algae production in NWE.

3)   Identify the way forward: how can we improve our algae production systems in NWE in terms of environmental resource footprint and what are the main LCA challenges related to quantifying impacts of algae production.

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2. LCA CASE STUDIES: ALGAE PRODUCTION IN NWE

2.1. Microalgae production in the Netherlands as a livestock feed ingredient[1]

2.1.1. Introduction

According to the FAO agricultural outlook for 2015-2030, the consumption of animal products in the European Union (EU) will continue to increase, which is associated with a higher demand for vegetable protein feed sources. The supply of proteins in the EU for animal feed applications relies mainly on the import of soybean crops, containing approximately 40% proteins and 18% lipids (fresh weight) (Dutch Soy Coalition, 2008). The Netherlands in particular accounts for 26% of the EU soy import and plays a crucial role in the soybean market. The soybeans originate mainly from Brazil (57%), followed by the United States (22%) (Eurostat, 2012). Most of the imported soybeans are crushed into soybean oil (20%) and meal (80%); the latter commonly used in animal feed for dairy cattle, chickens, pigs and calves. However, lately, questions have arisen about the environmental sustainability of soybean production because of amongst others, threats to biodiversity in countries such as Brazil where deforestation of the Amazon rainforest and the Cerrado biome (tropical forest savannah) is occurring (Fearnside, 2001).

Consequently, there is an increasing interest in alternative, more sustainable sources of proteins. Some algal strains contain large amounts of nutritional proteins (6-52%), carbohydrates (5-23%) and lipids (7-23%) with essential ω-3 and ω-6 polyunsaturated fatty acids (PUFA), minerals, carotenoids and vitamins. Because of scarce land availability in North West Europe, algae production can be a good alternative because marginal land is adequate to grow this type of aquatic (marine or freshwater) biomass. Thus, competition with food crop cultivation can be avoided. In addition, algae can grow on waste streams that provide a sufficient amount of carbon dioxide, nitrogen, and phosphorous. Therefore, it is clear that algae have a large feed potential compared to terrestrial plants (Becker, 2004; Borowitzka and Moheimani, 2013). According to a study of Pulz and Gross (2004), it has been proven that some algae species such as Chlorella, Scenedesmus and Spirulina have beneficial effects on the health of animals: improvement in their immune response system and fertility, a healthier coat and better weight control. For this reason, protein-rich algal meal is a promising alternative to soy meal, which is now the main protein source in the animal feed sector (Lum et al., 2013).

2.1.2. Algae production facility

In The Netherlands (Lelystad), a mixture of photo-autotrophic microalgae *Scenedesmus* and *Chlorella* are cultivated in two separated ponds: each pond had a surface area of 250 m² and a depth of 0.6 meter. These ponds are located at the ACRRES (Application Centre of Renewable Resources) facility, which is part of Wageningen University and Research Centre (WUR). On the ACRRES site, two digester units processed cattle manure, silage maize, maize straw and feed residues (silage maize and grass) to produce an energy-rich biogas (Figure 2). The biogas is recovered and burned in a combined heat and power (CHP) installation, generating green electricity that is delivered to the national grid. The remaining product of the digester unit, digestate, is used as a fertilizer on nearby fields to decrease the use of mineral fertilizers. The CO_2-rich flue gases (400°C) produced during combustion in the CHP are cooled down in two condensers that are placed in sequence. In the first condenser, the gas is cooled to 120°C, and in the meantime, heat (hot water) is provided. This heat is one of the end products of the biorefinery, which is used in a nearby bio ethanol installation. In the second condenser, the temperature of the gas

drops to 50°C. Again, hot water is produced, which is used for heating the algae ponds. The carbon dioxide of the flue gas is used as well in the algae cultivation systems as a carbon source.

The inoculum is prepared in a sequence batch system that consists of three separate successive tanks with a volume of 1 m³ (T6), 20 m³ (T3) and 60 m³ (T4), respectively. Necessary nutrients (mainly nitrogen and phosphorous), flue gas sparging and artificial lighting were used to create optimal growth conditions. On average, the inoculum procedure took place four times a year (12-day periods) to offer a start-up of the ponds when necessary.

One of the algae ponds is located in a greenhouse, and the other is located outside (for experimental research). This arrangement affected some of the parameters, such as the temperature of the ponds, rain input in the outdoor pond and extra infrastructure (glass of the greenhouse) for the indoor pond. Apart from these small differences, both ponds operated very similarly. Every hour, approximately 6 m³ (0.27% dry weight (DW)) per pond was withdrawn and pumped to a coalescer (each pond had one coalescer), where the algae could sink due to specific flow patterns that induced auto flocculation, i.e., no energy is consumed at this stage (except for pumping). In total, 2 m³ (0.29% DW) is retained in the coalescers on a daily basis which results in an annual average productivity of 1 g m^{-2} day^{-1}. The part of the biomass that did not sink is recycled back to the respective algae pond. The harvested fractions are then dewatered using an SMB Apeldoorn centrifuge (one device per pond is available), where approximately 90% of the algae are harvested. In total, an average of 3241 kg DW ha^{-1} year^{-1} is harvested. The concentrate (10.2% DW) is sent to the drying equipment, and the centrate is discharged into the sewer. Both of the coalescers, similar to the centrifuges, are located in the greenhouse. A natural gas-fired dryer is used to dehydrate the algae stream to 89% DW, which is a good percentage for a stable conservation (Sander and Murthy, 2010). Afterwards, the dried algae paste is sent to an extraction unit. This pilot setup leads to a yearly production of 2453 kg meal (DW) and 591 kg oil (DW) per hectare. Figure 2 represents a schematic overview of the full process chain of the integrated algal biorefinery.

2.1.1. Material and methods

Goal and scope

To steer this development in a sustainable way, this study determines the natural resource footprint of protein-rich algal meal for livestock feed applications in The Netherlands. Microalgae are cultivated at a pilot scale (500 m² open ponds) and are integrated in a biorefinery (anaerobic digester, combined heat and power (CHP), condensers), making use of waste heat and flue gases. The final products are electricity, digestate, the heat available for a nearby bio-ethanol facility and algae oil and meal. The sustainability of this rather new biomass source for animal feed is compared with the more traditional route of soybean crop production. The soybeans are cultivated, dried and crushed in Brazil, and the soy meal, which is commonly used as a protein-rich animal feed ingredient, is transported to The Netherlands.
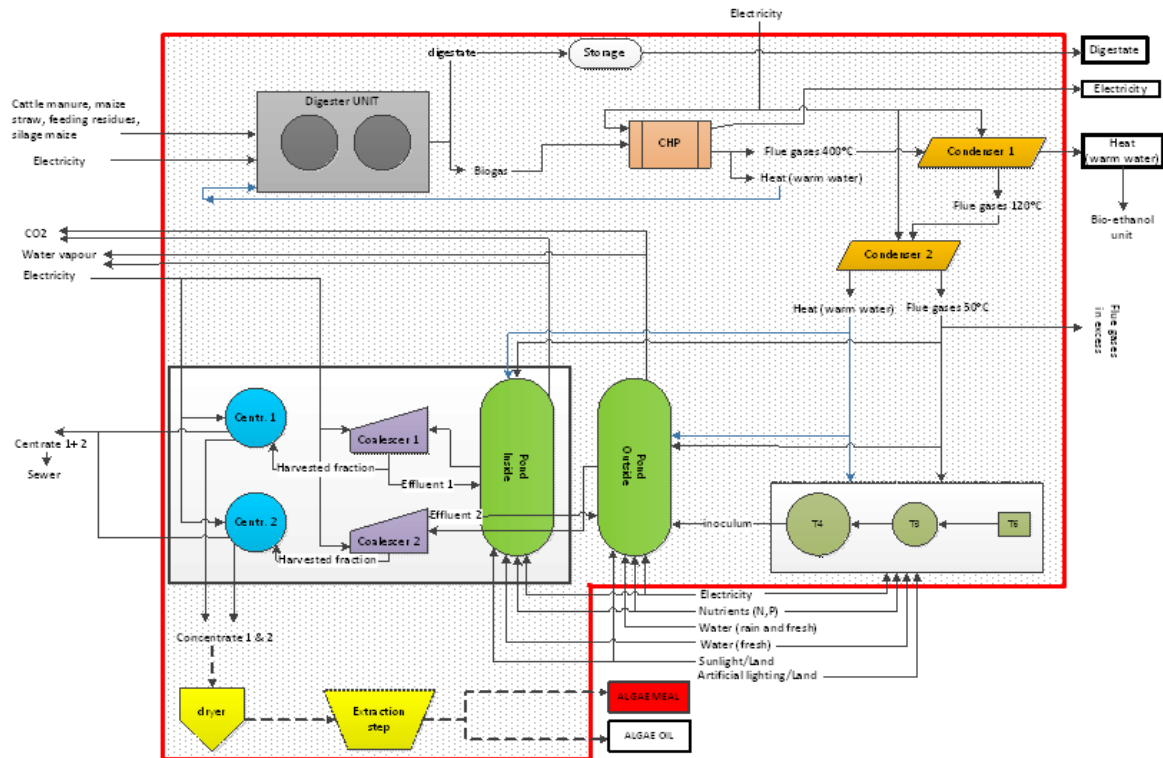
Figure 2: *Process flow scheme of the integrated algal biorefinery in Lelystad, The Netherlands. The end products (digestate, electricity, heat, algal oil and algal meal) are marked with a black frame. The system boundary of the foreground processes is indicated by a red frame.*

A cradle-to-gate (exergetic) resource footprint of both product systems, i.e., the soybean-based linear economy and the integrated algal biorefinery, is examined. In the integrated algal biorefinery, several products are produced. To make a fair comparison with the soybean product system, it is necessary to deliver the same functionalities. System expansion is used at a black box level, which implies several functional units (FU). In this study, the basket of products to which the environmental impact is assigned to is 1 MJ_{ex} (algal/soy) meal, 0.51 MJ_{ex} (algal/soy) oil, 986.62 MJ_{ex} electricity, 139.15 MJ_{ex} heat and 1015.76 MJ_{ex} digestate. Protein-rich meal is chosen to be the reference product (Figure 3). The system boundary of the LCA study is cradle-to-gate.

Figure 3: *Comparison between the integrated algal biorefinery and the soybean based linear economy where the same (amounts of) functionalities were produced. System expansion was used to avoid allocation.*

Life cycle inventory and impact assessment (LCI(A))

It is estimated that at least 80% of all of the data are collected directly on-site, and other essential data are computed through mass and energy balancing or are found in the literature. Especially for the drying and extraction step, data was used from literature and/or databases because these processes were not operational yet. Data about the energy consumption of the natural gas-fired dryer is found in the study of (Sander and Murthy, 2010) and data about soybean processing into oil and meal is used from the inventory database ecoinvent version 2.2 (Frischknecht and Rebitzer, 2005). For a complete data inventory of the soybean production in Brazil and transport to The Netherlands, we refer to the study of Prudêncio da Silva et al. (2010) and Taelman et al. (2015a).

The CEENE method (2013) is used to calculate the environmental resource footprint of the five different functionalities. To assess the resource footprint of cultivating soybeans in Brazil, a CF of 38.8 MJ_{ex} m^{-2} $year^{-1}$ is used, and for algae production in The Netherlands, a CF of 25.3 MJ_{ex} m^{-2} $year^{-1}$ is obtained (supporting information (Alvarenga et al., 2013)).

2.1.2. LCA results and discussion

Environmental resource footprint of the integrated algal biorefinery

The total resource footprint of the biorefinery is 3033.72 MJ_{ex} to produce the basket of products. The largest contributor (73%) to the total footprint is the digesting step in which biogas and digestate is produced. The biomass inputs to the digesters (especially silage maize) had a large resource footprint due to the large daily request for digestible material. Furthermore, in the dosage system, mixing and pumping required a large amount of electricity (339 kWh day^{-1}). The cultivation steps in the outdoor and indoor ponds are the second and third contributors. Especially the electricity consumption for the different blowers and mixing devices had a high impact on the resource use; it contributed 89% of the resource footprint for the algae cultivation. The CHP process had a relative contribution of 6.14% to the total resource footprint of the biorefinery, mainly because of the electricity consumption of the corresponding emergency gas cooling system and pumps as the CHP itself did not consume any energy. The other processes (e.g., dewatering, condensing, drying and crushing) contributed less than 1% to the total environmental resource footprint.

The most exploited natural resources at the cradle are land resources (47.16%) and fossil fuels (43.44%) (Figure 4). Both of these resources are most often used within the digesting process; the cultivation of agricultural crops requires high land use, and the electricity production in The Netherlands is highly dependent on natural gas and coal (fossils). Mainly, electricity is consumed for mixing, pumping and proportioning (dosage system).
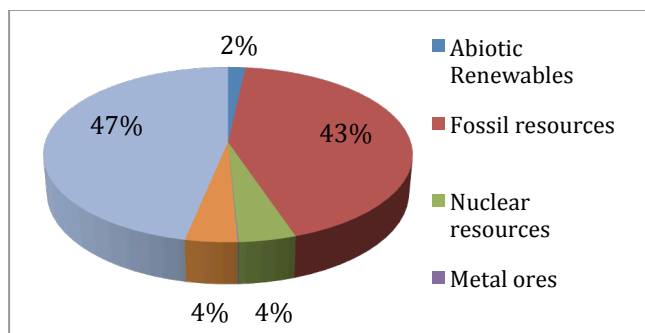

Figure 4: *Most exploited natural resources (%) to produce the basket of products.*

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Comparison with the environmental resource footprint of the linear economy with soybean production

It is assumed that the heat, electricity and digestate could be produced as a first part of the linear economy, which can take place at the same location (ACRRES site in Lelystad). Therefore, the resource footprint for this part is the same as for the linear economy and the biorefinery and further discussion are not relevant in a comparison. The second part of the linear economy is the Brazilian soybean cultivation and oil and meal production. In this study, it is assumed that soy-based or algae-based oil and meal have, besides the same exergy content, the same functionalities.
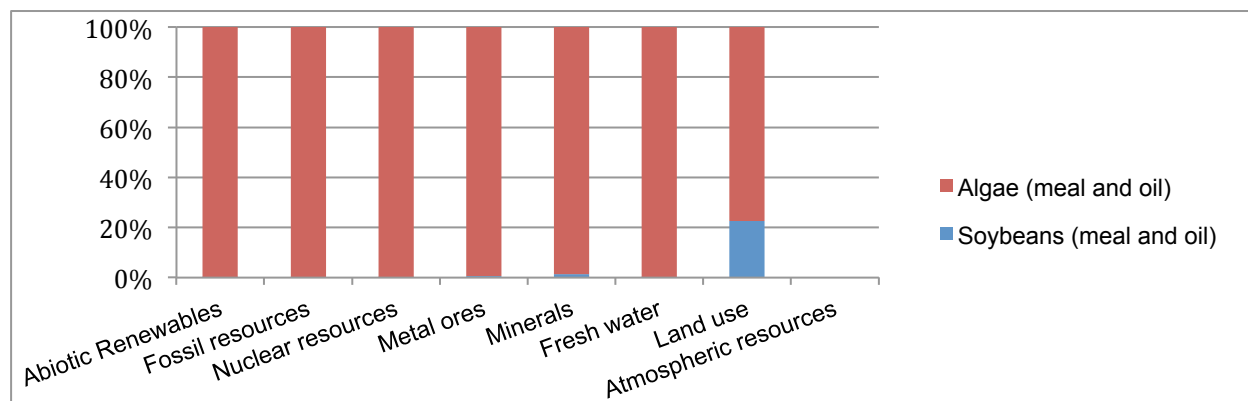

Figure 5: *Comparison between the integrated algal biorefinery and the soybean based linear economy where the same (amounts of) functionalities were produced. System expansion was used to avoid allocation. The relative contribution for each impact category is presented.*

Under the current situation, a difference in the resource footprint of a factor of 10^2 can be detected, in favour of soybean production. Soybean cultivation requires a large amount of land resources (Figure 5), which entail a relative contribution of 93.47% to the total resource footprint of soy meal (1 MJ_{ex}). The second most used raw materials are fossil resources, which is mainly due to the fuel use of the agricultural machinery and because of the energy used for the production of chemical fertilizers. To have the soybean meal available in The Netherlands, it had to be transported over several kilometres. However, this transportation makes only a minor contribution (3.58%). In contrast, during algae cultivation, mainly electricity is used for blowing and mixing (28.96 MJ kg DW algae^{-1}). The supply chain of producing electricity includes the use of several natural resources such as fossil resources, land and water, implying much higher resource consumption for algal meal production. Furthermore, the algal biomass (10.2% DW after dewatering) requires more drying than the harvested soybeans (82% DW).

First results of a sensitivity analysis

The LCA results reveal that significant improvements in terms of energy efficiency must be realized for algal meal to become competitive with commonly used soy meal.

Tested parameters:
- Source of electricity production: from fossil-based to wind-based energy
- Operating hours of the air blowers: from 18 h d^{-1} to 12 h d^{-1}
- Type of blower: from BOSA ventilators to fancom blowers (more energy-efficient)
- Biomass yield: from 1 g.m^{-2}.day^{-1} to 6 g.m^{-2}.day^{-1} (e.g., genetic and metabolic engineering)

The sensitivity of the LCA results toward changing each proposed parameter is examined, i.e. a modified sensitivity test of a hypothetical algal cultivation scenario for livestock feed applications is performed. The sensitivity test reveals the possibility of achieving a resource footprint that is 20 times less. When

compared with soy meal and oil production, it appears that the total footprint is still a factor of six too high to be competitive from an environmental sustainability point of view. However, when abiotic renewable sources are considered to be inexhaustible, it changes the footprint of both systems: 5.60 MJ_{ex} and 5.61 MJ_{ex} for soybean and algae production, respectively. Even some fresh water requirements could be reduced when the centrate is recycled back (perhaps after filtration to remove the possible bacteria) to the ponds. At that point, the algal meal could be produced with a similar or even lower (non-renewable) resource footprint profile compared with soy meal.

2.1.3. Conclusion

The LCA study revealed that to produce the same functionalities, it is better to feed the livestock in The Netherlands with protein-rich soy meal instead of algal meal from a resource point of view (factor 100) because algae cultivation is, for now, much more energy intensive. Especially the electricity use of blowing flue gases into the ponds has a major contribution to the total footprint, and more efficient equipment must be used.

A note should be made to clarify that a comparison is performed between a relatively immature small-scale (500 m² meters) algae cultivation and the large-scale, technically mature soybean production (several hectares) and processing. It is clear that further progress must be made to lower the resource footprint of the algae production, but because this technology is rather new, many improvements are possible. A sensitivity test revealed that it is possible to meet a resource footprint of the same order as the footprint of soy meal and oil production. Therefore, electricity should be produced making use of renewables (wind energy), more energy-efficient blowers must be installed that have fewer operating hours, and biomass productivity should increase (genetic modification could be an option) together with the amount of biomass that can be harvested.

2.2. Wastewater treatment coupled with microalgae production in Belgium[2]

2.2.1. Introduction

From 2006 to 2011, the world aquaculture production increased by 34% (FAO, 2014), leading to an increasing production of nutrient-rich waste and wastewater that need to be treated. To enhance the sustainability of intensive aquaculture systems, recirculating aquaculture systems (RASs) which include a water treatment system have been developed. These RASs offer advantages in terms of reduced water consumption, and improved opportunities for waste management and nutrient recycling compared to conventional flow through aquaculture systems (Martins et al., 2010). In line with the current paradigm shift towards resource recovery in wastewater technology, the sludge and the dissolved organic matter and nutrients in aquaculture backwash wastewater should be valorized. Despite the presence of free ammonia, anaerobic digestion of fish sludge to produce biogas is a promising approach to reduce the environmental impacts of fish sludge treatment (Mirzoyan et al., 2010). The remaining backwash supernatant needs further treatment, especially to remove dissolved organic matter and nutrients. Therefore, a sunlight-based microalgal bacterial floc (MaB-floc, a bioflocculating consortium of bacteria and microalgae) technology was developed (Van Den Hende, 2014). In this system, costly mechanical aeration is replaced by photosynthetic aeration by the microalgae present in the MaB-flocs. In situ bioflocculation of MaB-flocs is obtained via operation as sequencing batch reactor (SBR). This enables the easy separation of MaB-flocs from the treated wastewater.

A possible pathway for further valorisation of the MaB-flocs is the use as feedstock for anaerobic digestion to produce biogas. Nevertheless, the anaerobic digestion conversion efficiency of MaB-flocs grown on pikeperch backwash supernatant is below 40 % (Van Den Hende et al., 2014c). This is also a low-value valorisation pathway, in the order of 30-60 € per ton MaB-floc VSS (Van Den Hende, 2014). An alternative MaB-floc pathway is using MaB-flocs as pigment-enhancing feed for herbivorous aquaculture species. Recently, it was shown that dried MaB-flocs can replace 8% of the ingredients (mainly wheat) of diets of Pacific white shrimp *Litopenaeus vannamei* (Boone, 1931) while enhancing their pigmentation (Van Den Hende et al., 2014b).

Switching from linear fish aquaculture and separated aquaculture sludge and wastewater treatment to an integrated MaB-floc-based aquaculture waste treatment system could be a key strategy to mitigate the environmental footprint of the aquaculture sector; e.g. by valorizing fish sludge into biogas and recovering nutrients through MaB-floc cultivation.

2.2.2. Algae production facility

The studied system was a 28 m^2 MaB-floc raceway pond treating backwash supernatant from pikeperch culture (Van Den Hende et al., 2014a). On average, 2.59 m^3 day^{-1} of backwash supernatant was pumped into the raceway pond stirred by 2 propeller pumps. Bottled flue gas was sparged into the pond to regulate the pH. Below the pond, copper tubes were used to conduct hot water to maintain a minimum pond temperature of 12°C. To maintain a concentration of 0.5 g TSS L^{-1} in the pond, MaB-flocs were harvested as previously described. On average, 387 g day^{-1} MaB-floc TSS was harvested (Figure 6A).
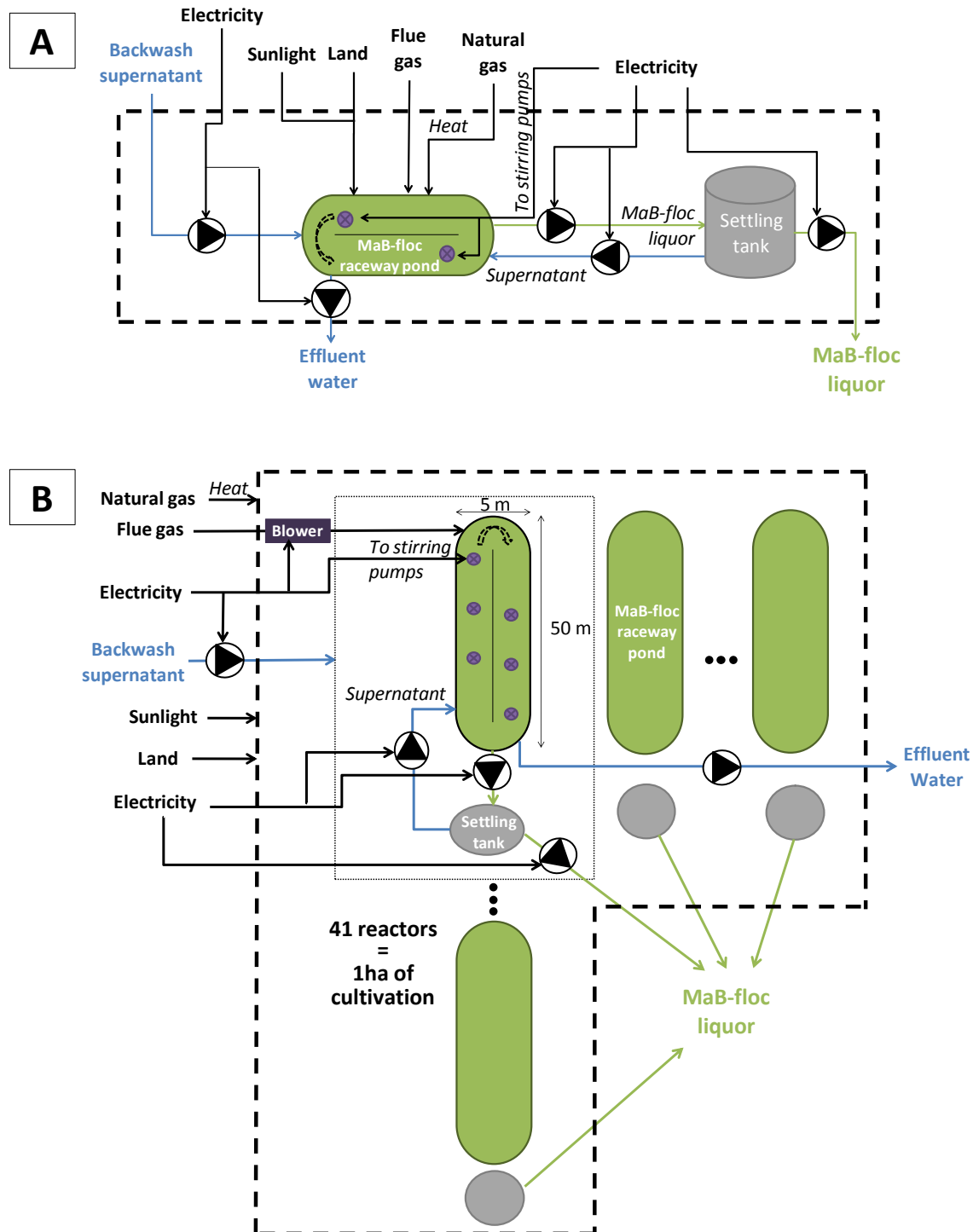
Figure 6: *Process flows of the MaB-flocs cultivation pilot plant operated in Roeselare, Belgium (A) and the modeled linearly up-scaled plant (B).*

Up-scaling is an important step to evaluate the potential environmental impact of the process when applied to industry. Thus, a linearly up-scaled MaB-floc-based wastewater treatment plant was modeled (Figure 6B). This plant was designed to treat 1000 m³ day⁻¹ of pikeperch backwash supernatant. On average, 155 kg MaB-floc TSS is pumped in the settling tank per day for the entire plant (Table 1). In addition, some improvements to the up-scaled plant are proposed. More details regarding the main

differences between the pilot plant and the (improved) up-scaled plant can be found in the study of Sfez et al. (2015)

Integrating the described MaB-floc-based wastewater treatment plant in a broader aquaculture waste treatment system can be an option to reduce the environmental impact of the aquaculture systems. In this study, three integrated scenarios with their respective valorisation scenario are considered (Figure 7):

- Baseline scenario: the pikeperch aquaculture system releases backwash supernatant in the sewage system. To reduce inhibition by free ammonia, fish sludge is co-digested with maize silage to produce biogas. A Combined Heat and Power system (CHP) converts the biogas into heat (used to heat the digester) and electricity which is delivered to the grid.

- Scenario 1: the pikeperch aquaculture system releases backwash supernatant treated by a MaB-floc pond. The fish sludge is co-digested with silage to produce biogas and MaB-flocs are dried to add in shrimp feed (Van Den Hende et al., 2014b). Biogas is converted to heat and electricity through a CHP. Electricity is delivered to the grid and heat is used to dry the MaB-flocs and to heat the raceway pond and the digester.

- Scenario 2: the pikeperch aquaculture system produces backwash supernatant treated by a MaB-floc pond but fish sludge, MaB-flocs and silage maize are co-digested to produce biogas which is converted into heat and electricity through a CHP. Electricity is delivered to the grid and heat is used to heat the raceway pond and the digester.

For the three scenarios, the digestate is used as soil conditioner.

Table 1: Main differences between the designs of the MaB-floc-based pilot plants and up-scaled plants.

	Pilot scale[a]	Linearly up-scaled plant[b]	Unit
Incoming water	2.6	24.5	m^3 day^{-1} $pond^{-1}$
Ponds			
Number	1	41	ponds
Pond area	12	244.6	m^2
Pond volume	28	97.9	m^3
Length	11.7	50	m
Width	2.5	5	m
Distance between each pond	-	2	m
Material	Steel + polyuretane	Pit in the ground + HDPE foil	-
HRT[c]	4 to 8 (average)	4	days
Number of stirring pumps	2	6	pumps $pond^{-1}$
Flue gas injection			
Concentration	89 to 214	89	g Nm^{-3}
Injection method	Gas valve and bottle pressure	Electrical blower	-
Power of appliance	-	0.05[c]	kW
Heating system	Copper tubes	Steel tubes	-
Settling tank	1 m^3 cubitainer	8 m^3 settling tank per raceway pond - covered with HDPE foil	-

[a] *Van Den Hende et al., 2014a;* [b] *Collaboration with experts;* [c] *Bosa Ventilatoren bv - SER-8, The Netherlands;* [c] *At pilote scale, the HRT varied between 4 and 8 days for experimental reason*

2.2.3. Material and methods

Goal and scope

This study first evaluates the environmental sustainability of a pilot MaB-floc SBR raceway pond treating backwash supernatant from a pikeperch RAS in Belgium (Van Den Hende et al., 2014a). The pilot plant was then compared to an up-scaled plant modeled as a linear projection of the pilot plant (called linearly up-scaled plant) and to three improved up-scaled plants in which some parameters were modified. To determine the potential of impact reduction associated with system integration, the improved plants were implemented into two industrial scale scenarios in which MaB-flocs were valorized into biogas or into shrimp feed. These two scenarios were compared to the baseline scenario, in which aquaculture backwash supernatant is released in the sewage system without any treatment by MaB-flocs.
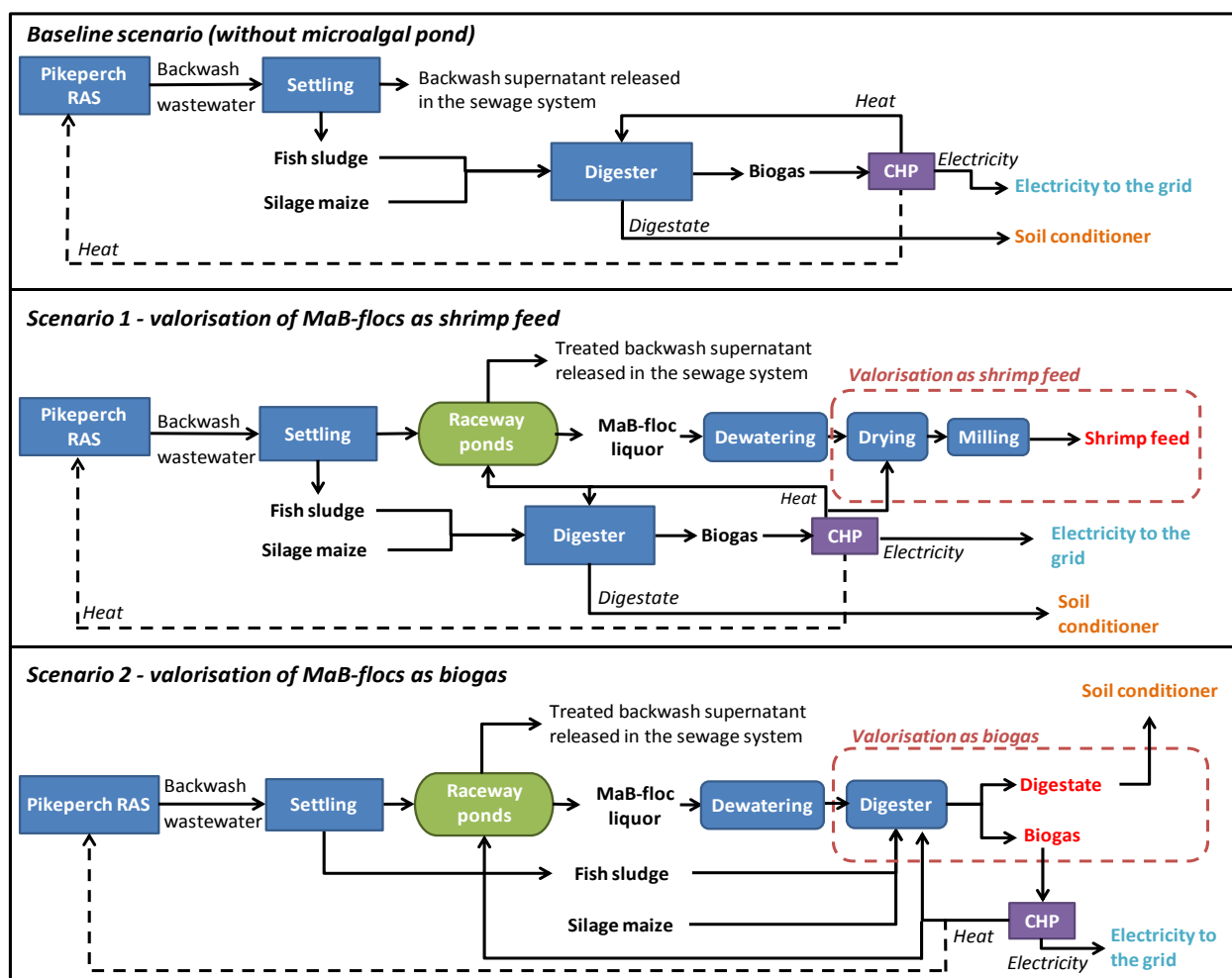
Figure 7: Process flows of the three studied scenarios based on a system without any microalgal treatment (Baseline scenario)and a treatment of the wastewater by MaB-flocs which are then valorized into shrimp feed (scenario 1) or biogas (scenario 2).

Goal 1: evaluation of the improvement potential of the MaB-floc-based wastewater treatment plant for treatment of aquaculture backwash supernatant

A cradle-to-gate LCA was conducted, from the entrance of water in the raceway ponds to the release of water in the natural environment (with the zero burden assumption). To compare the pilot and up-scaled plants, the production of 1 kg MaB-floc TSS was chosen as functional unit for the system, as the goal is

to analyze how the products can be produced efficiently in the context of the treatment of aquaculture wastewater.

Goal 2: evaluation of the sustainability of integrating MaB-floc-based wastewater treatment systems into the aquaculture systems
A cradle-to-gate LCA was conducted, from the entrance of water in the raceway ponds to the release of water in the natural environment and the biomass valorisation as (1) production of shrimp feed and (2) heat and electricity from biogas. The functional unit of the studied system is the treatment of 1 m^3 of aquaculture backwash supernatant as the goal is to analyze how this water can be used as bioresource in the most sustainable way.

For both goals, processes not included in the study are the construction work (excavation work and transport of material to the construction site), the release of MaB-flocs losses and press filtrate in the environment, the end-of-life of buildings, material and appliances and the transport and application of the digestate to the field.

The different scenarios generate several other products such as bio-based electricity and shrimp feed and all the impacts cannot only be allocated to the outgoing treated water. Therefore, system expansion with substitution was used in this study. It is assumed that compost can be replaced by the digestate, the electricity from the Belgian grid and heat from a natural gas boiler can be replaced by the electricity and heat produced by the CHP and wheat-based shrip feed by MaB-floc powder.

Life cycle inventory and impact assessment (LCI(A))

Data for the foreground processes of the pilot plant operated in Roeselare was collected from Van Den Hende et al. (2014a), site visits and direct discussion with the author. Data for the foreground processes of the up-scaled plants and the integrated scenarios was estimated in collaboration with experts in the field and collected in literature. To model the avoided products, data was used from the database ecoinvent version 2.2. The CEENE (2014) method is used to evaluate the environmental footprint of all scenarios.

2.2.4. Results and discussion

Starting from the pilot plant (P), a linearly up-scaled plant (L) was modeled. In addition, possible improvements were proposed:

Tested parameters:
- Stirring efficiency: from propeller pumps to paddle wheels (plant S)
- Electricity mix: from Belgian electricity mix to 100% wind energy (plant E)
- MaB-flow productivity: increase of 30% (plant M)

At pilot scale, 848 MJ$_{ex}$ was required from the natural environment to produce 1kg MaB-floc TSS (Figure 8B). Electricity consumption to stir the raceway pond contributes the most (93%) for all CEENE impact categories except for metal ores (Figure 8A). For the latter, the production of steel used to build the pond contributes most. Note that at pilot scale, steel was chosen to facilitate the mobility of the raceway pond to conduct experiments on different sites, i.e. it was not intended for an industrial scale wastewater treatment facility. As the electricity needed from the Belgian grid for stirring contributes the most to the total resource consumption of the pilot plant, mostly nuclear (50%) and fossil fuels (40%) are consumed.

The linearly up-scaled plant consumes 278 MJ$_{ex}$ kg^{-1} MaB-floc TSS, which is 3 times less than the pilot plant (Figure 8B). Up-scaling is especially beneficial for stirring and infrastructure from a resource consumption point of view. In the up-scaled plant, the number of pumps per volume of pond is lower than

at pilot scale, as the length to width ratio of the raceway pond is more beneficial to stirring in the up-scaled pond. As a result, up-scaling decreases land resources consumption by 57% and water resources, fossil fuels and nuclear energy by 68%. Infrastructure of the up-scaled plant consumes 64% less resources compared to pilot scale (Figure 8A). This is explained by the replacement of the steel tank by a dug pond and the copper heating tubes by steel tubes.

Now the results for the different improvement options (pilots S, E and M). The total CEENE of scenario E increases by 29% compared to the linearly up-scaled plant (Figure 8B), mainly because of the consumption of abiotic renewable resources such as wind energy. However, abiotic renewable resources are freely available in the environment and can be withdrawn from the total CEENE. Without these renewable resources, improvement E is the option which reduces most the amount of resources consumed (plant E consumes 93% less resources than the linearly up-scaled plant), which shows the importance of increasing the energy efficiency of the system.
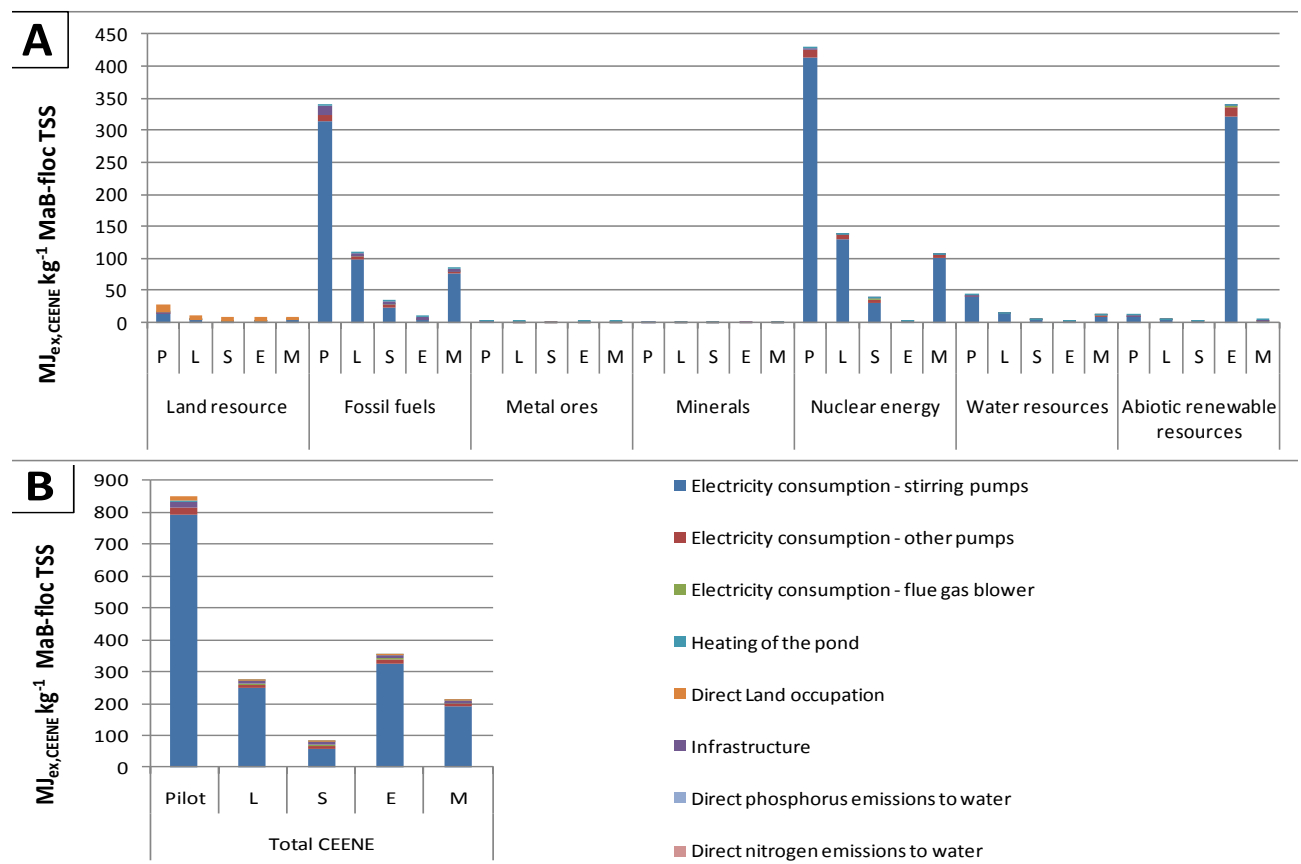
Figure 8: Resource footprint (CEENE 2013 method), of the pilot (P), linearly up-scaled algae-based wastewater treatment plant (L), up-scaled with paddle wheels (S), up-scaled with 100% wind power supply (E) and up-scaled with a MaB-floc productivity increased by 30% (M).

For the integration of the MaB-floc-based wastewater treatment plant in the integrated aquaculture system, the environmental impact was calculated for four modeled systems:

- *UpL,shrimp feed*: the up-scaled plant is integrated in the system and the MaB-flocs are valorized into shrimp feed
- *UpL,AD*: the up-scaled plant is integrated in the system and the MaB-flocs are valorized into biogas
- *UpSEM,shrimp feed*: the up-scaled plant integrates the three improvement options and the MaB-flocs are valorized into shrimp feed

- *UpSEM,AD*: the up-scaled plant integrates the three improvement options and the MaB-flocs are valorized into biogas

The baseline scenario has an environmental footprint of -1.2 MJ_{ex} m^{-3} water treated. However, the two valorisation scenarios avoid more resource consumption, i.e., scenario 1 avoids mainly land resources required for the production of wheat which is replaced by MaB-flocs and scenario 2 avoids mainly the consumption of electricity from the Belgian grid. Additionally, abiotic renewable resources are freely available in the natural environment and they can thus be excluded from the total CEENE. As a result, the resource footprint of *UpSEM,shrimp feed* and *UpSEM,AD* decrease to -10.9 MJ_{ex} m^{-3} and -0.5 MJ_{ex} m^{-3}, respectively (Figure 9), i.e. both valorisation pathways become competitive with the baseline scenario.

The main contributor to the (improved) valorisation scenarios is the algae-based wastewater treatment plant itself due to the high electricity consumption required to stir the pond. Anaerobic digestion is the second contributor. As less feedstock is digested in scenario 1, the impact of anaerobic digestion is lower than for scenario 2. Moreover, the consumption of energy from an external source to dry the MaB-flocs is low due to the on-site consumption of heat produced from biogas. Thus, overall the shrimp feed valorisation pathway has a lower environmental footprint than the anaerobic digestion pathway because scenario 1 consumes less resource types and a lower amount of each resource types compared to scenario 2 (Figure 9).
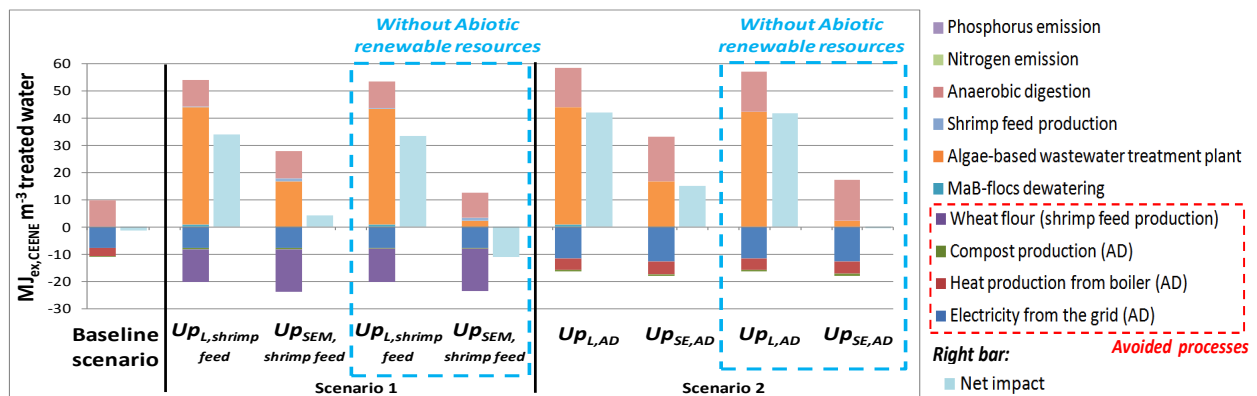
Figure 9: Comparison of MaB-floc valorisation scenarios for shrimp feed production (Scenario 1) and biogas production via anaerobic digestion (Scenario 2).

2.2.5. Conclusion

The comparison of the pilot MaB-floc raceway pond with an up-scaled scenario shows a high potential of impact reduction associated with up-scaling. When the up-scaled system is integrated in a broader aquaculture wastewater treatment facility, valorizing MaB-flocs into shrimp feed had a lower resource footprint than when using MaB-flocs for biogas production. However, these scenarios are not competitive with the baseline scenario (no microalgal treatment) in terms of resource efficiency mainly because the MaB-floc-based wastewater treatment plant is too energy intensive (stirring of the ponds!). On the contrary, when the proposed improvement options are implemented, the benefit of delivering products to the market, shrimp feed for scenario 1 and heat and electricity for scenario 2, outweighs the gross impact of the plant itself.

2.3. Seaweed production on the West coast of Ireland and the Northern coast of France[3]

2.3.1. Introduction

Global environmental concerns about the depletion of natural resources and industrial pollution have led to the cultivation of more renewable resources such as seaweed biomass, which is translated in high aquaculture production rates of 20.8 million tonnes (wet weight) in 2012 compared to 6.4 million tonnes in 2000 (FAO, 2012). Due to the high demand for food and phycocolloid products, seaweed farming has become economically important for many countries, especially in Asia.

For European countries, seaweeds have traditionally been used by the pharmaceutical, cosmetic and food industry for their useful extracts (e.g., phycocolloids such as agar) or as products for agriculture (fertilizer, animal feed) and are less commonly used for direct human consumption (Ngo et al., 2011). However, these seaweeds are imported from Asian countries. Lately, in an attempt to become self-sufficient and to reduce pressure on land and its resources, efforts have been made to develop suitable seaweed cultivation techniques which are adapted to cold-temperate conditions. Worldwide, there are at least three near-shore seaweed cultivation methods demonstrated; the off-bottom method, a raft system and single longline ropes (Andersen, 2005). Although the off-bottom monoline method is most used because of its simplicity, cheapness, easy installation and maintenance, this method is not operational in Europe because of the associated high labour costs and exposed coastlines.

As seaweed cultivation and the search for markets in Europe is still in its infancy, and data regarding processing of the biomass towards a final application is scarce, focus should be first on optimizing the cultivation processes. An estimation of the environmental sustainability of current seaweed production in NWE may assist in their further development. Therefore, in this study, the environmental resource footprint of cultivating seaweed in the Atlantic Ocean on the West coast of Ireland was compared to seaweed farming on the Northern coast of France (Brittany). Different seeding procedures and nearshore cultivation systems were considered. To be able to quantify all resources used (also marine resources such as occupying sea surface), the occupation factor α was calculated for both production sites. Additionally, a comparison with the resource footprint of microalgae and different types of terrestrial biomass (maize, potatoes and sugar beet) was made.

2.3.2. Algae production facilities

Both EnAlgae pilot sites cultivate the large brown seaweed *Saccharina latissima*.

Seaweed cultivation in Ireland

In the West of Ireland, the National University of Ireland, Galway (NUIG) operates an aquaculture research facility (The Ryan Institute Carna Research Station) in Carna, Co. Galway. The facility has a complex water treatment system installed that can supply seawater for use in, amongst other facilities, the seaweed hatchery.

The production of seedlings starts with collecting fertile Saccharina latissima in the lower intertidal and subtidal coastal zones. In the hatchery, the reproductive sori, which are clusters of sporangia containing many millions of zoospores, are cut out from the blade of seaweed, cleaned and air-dried before being placed in small flasks (6 L) with autoclaved water. The numerous flagellated male and female zoospores (haploid) that are released after this process develop into male and female gametophytes (also haploid). Under laboratory conditions, the gametophytes produce gametes and a fusion between a male and female gamete leads to a diploid zygote that develops into juvenile sporophytes, or seedlings. The sporophyte culture is sprayed on 12 collectors (each containing 50 m of polyvinyl alcohol fibers or culture string). These collectors are placed in culture tanks filled with UV-sterilized seawater and nitrates, phosphates, vitamins and trace elements are added. The cultures are kept at 10 °C using a room air chiller. Agitation and aeration is provided by a blower device. Approximately 9000 m of seeded cultivation string is produced per year.

The seaweed grow out phase is located in the South West of Ireland within Ventry Harbour, Co. Kerry and is owned and operated by the commercial seaweed farm Dingle Bay Seaweeds (Castletownbere, Co. Cork). In total, 18 hectares of sea site is licensed for seaweed aquaculture. Therefore, a van is used to transport the seeded collectors between the hatchery and Ventry Harbour. Other equipment (e.g., culture rope, anchor chains, buoys) is transported from Castletownbere to Cuan Pier, Ventry Harbour by lorry and from Cuan Pier to the seaweed site by several boats. These boats were also used for maintenance and harvest of *Saccharina latissima*. In December, the seeded seaweed collectors are wrapped around the culture rope (280 m per longline) at deployment. Figure 10 gives a schematic representation of the single longline structure used to cultivate the biomass. Manual harvesting takes place in May, when the quality of the seaweed is optimal. About 25 kg seaweed (9.7% DW) per m longline can be harvested.
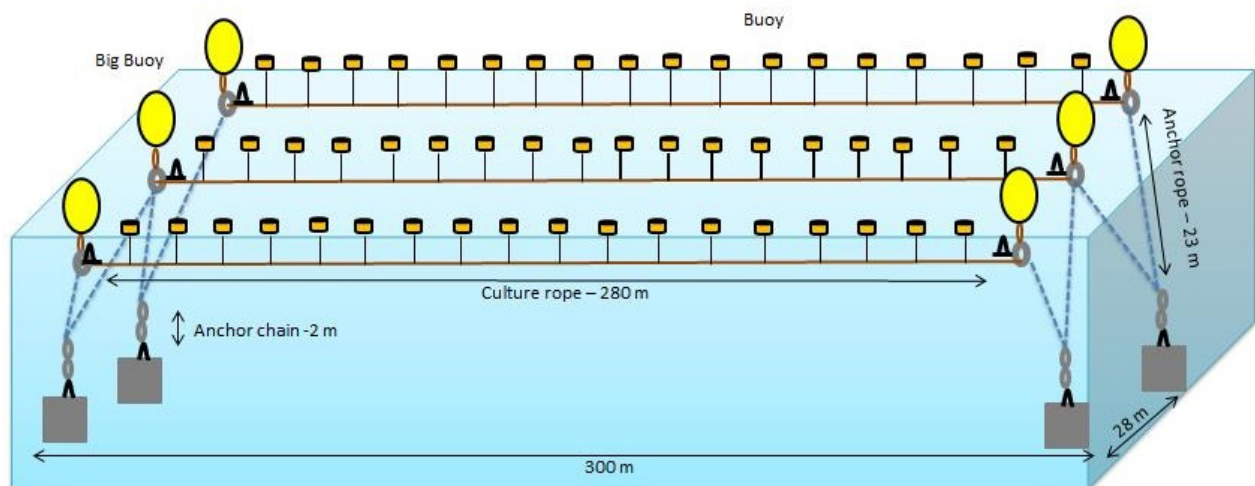
Figure 10: Equipment for nearshore seaweed cultivation in Ventry Harbour, Co. Kerry, Ireland. A cultivation unit contains 3 longlines, 4 anchors and anchor chains, 8 anchor ropes and buoyancy is required to maintain the longlines at 0.5-1m below the water's surface.

Seaweed cultivation in France

The seaweed hatchery is located in Pleubian, France, at the CEVA (Centre d'Etude et de Valorisation des Algues) facility. Seawater is pumped to a storage tank, from where water is delivered to the hatchery. Seedling breeding starts with the collection of local fertile Saccharina latissima. Sori are cut from the blades, the pieces are dehydrated and then submerged in autoclaved seawater to release the spores. The spore solution is poured into cultivation tanks where a blower device is used to provide mixing and

aeration. Nitrogen and phosphorous are added to the tanks. Compared to the seeding procedure at NUIG, there is no gametophyte development in CEVA. Direct seeding of mobile zoospores is cheaper (no maintenance of the immobile gametophyte phase), but, as a disadvantage, more fertile sporophyte is necessary. On a yearly basis, 4000 m of cultivation string is seeded. Each collector contains 60 m of string, which is wrapped around 50 m of culture rope. This procedure takes place in the hatchery, not at sea.
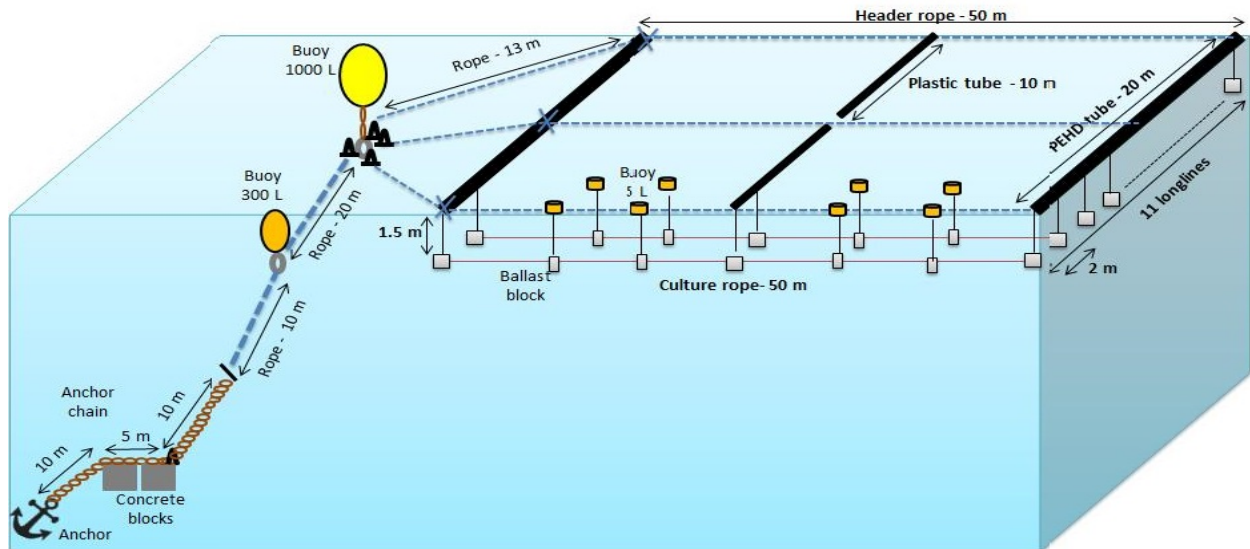

Figure 11: Schematic representation of a raft system (20 m x 50 m, type 1) and anchoring used at the sea farm (6000 m²) near the coast of Lézardrieux, France. Only 2 culture ropes (1.5 m below the sea surface) are shown.

Around mid-December, the nursery culture is transferred to the sea. The sea farm is located 2 km from shore and 8 km from the nearest harbour of Lézardrieux (use of boat and lorry is required). Because of the many surface currents at the sea site, a raft system was chosen over a longline system (Figure 11). In total, about 6 hectares of sea site is licensed for CEVA to use. There 4 raft units available, which occupy 6000 m² of sea surface and about 2.85 ha of sea site (taking into account anchoring and space between the units). In total, 3300 m culture rope is harvested manually in May, keeping in mind that harvesting in June yields more biomass but, due to more epiphytes, food and feed applications are limited. A maximum yield of 20.3 kg fresh weight (FW) m^{-1} could be achieved with this system but due to friction of culture ropes with buoys and tubes the yield drops to an average of 5 kg seaweed (9.7% DW) per m longline.

2.3.3. Material and methods

<u>Goal and scope</u>

Comparing the environmental sustainability in terms of natural resource use for two different seaweed production scenarios in North West Europe requires a common functional unit and system boundaries. In this study, results are expressed per MJ$_{ex}$ *Saccharina latissima*. Processes related to the seaweed hatchery and grow-out phase at sea are included in the foreground system (Figure 12). This LCA study applied a cradle-to-gate boundary.

In the case of Ireland, the impact of using electricity and infrastructure during pre- and post-treatment of seawater is allocated to the volume of water used within the hatchery. In France, the blower in the hatchery supplies air for microalgae (330L) and macroalgae (600L) cultivation, i.e. the same allocation method as in the case of Ireland (volume-based) was applied.
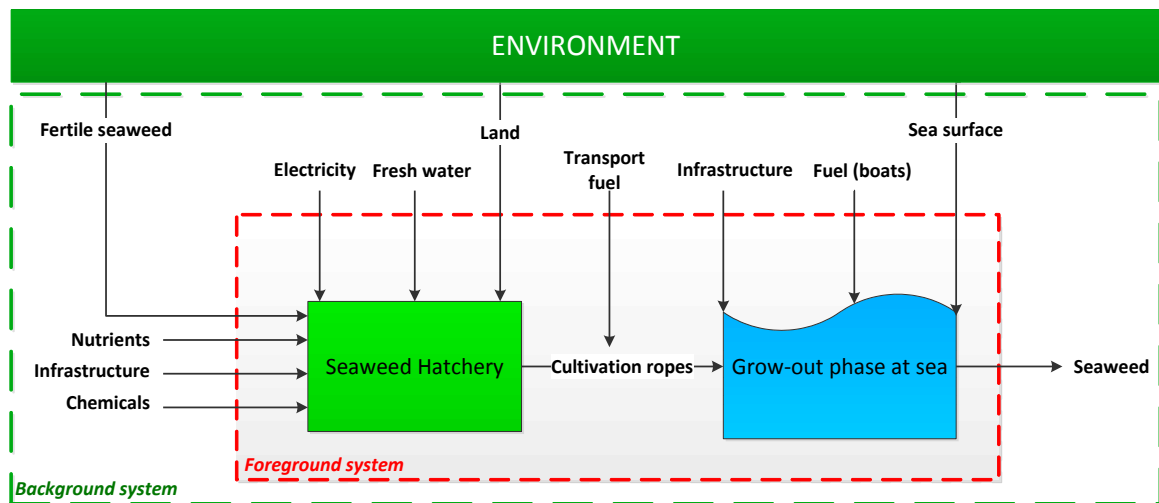
Figure 12: *Schematic representation of the foreground (red dotted line) and background system (green dotted line) of both seaweed production systems located in Ireland and France.*

Life cycle inventory and impact assessment (LCI(A))

Data on the chemical composition of seaweed was obtained from CEVA and was assumed to be applicable for both cases. Data related to the foreground system is collected at the production sites. Potential emissions at the foreground system were not quantified. In section 2.3.4, the resource footprint of seaweed production is compared to the cultivation of marine microalgae and some terrestrial plants (sugar beet, maize, potatoes). For a complete data inventory of the Nannochloropsis sp. production, we refer to the study of Taelman et al. (2013). Database ecoinvent version 2.2 provided data for the terrestrial plants (Frischknecht and Rebitzer, 2005).

As explained in section 1.1, the CEENE method accounts for the occupation of land or sea area in human-made systems based on the exergy content of potential net primary production (NPP). This potential NPP represents the NPP that an area would produce without human intervention. Spatially differentiated characterization factors (CF) were calculated for both types of occupation. The CFs for direct land occupation in Ireland and France are 25.7 MJ_{ex} m^{-2} yr^{-1} and 28.0 MJ_{ex} m^{-2} yr^{-1}, respectively, and for direct marine sea surface occupation 22.7 MJ_{ex} m^{-2} yr^{-1}, as both nearshore regions are located in the Celtic Sea. When compared to the production of microalgae and some terrestrial plants, different CF's for land occupation were used (26.9 MJ_{ex} m^{-2} yr^{-1} for Belgium and for 24.4 MJ_{ex} m^{-2} yr^{-1} for Switzerland). For the upstream processes (background system), about 95% of the occupied land has its origin in Europe and no sea surface occupation is considered. Therefore, the impact of land occupation in the background system is calculated according to the average CF of Europe (23.2 MJ_{ex} m^{-2} yr^{-1}). In the study of Taelman et al. (2014), an occupation factor α was introduced to deal with the possibility of occupying only part of the photic zone. For example, nearshore seaweed farming on one hectare sea surface still allows sunlight to penetrate the waterbody due to the longline/grid structure used for cultivation. For this case study, the α factor is calculated for the infrastructure used at the West coast of Ireland and the Northern coast of France. For more information about the calculation of this factor, we refer to the study of Taelman et al. (2015b).

2.3.4. Results and Discussion

First, the annual average sea surface occupation factor α of the seaweed pilots in Ireland and France was determined; 2% and 18%, respectively. This factor was then used to calculate the sea surface occupation of both systems, part of the impact category marine resources of the CEENE 2014 method.

Resource footprint of seaweed cultivation

Ireland

The environmental resource footprint of *Saccharina latissima* cultivation in Ireland is 1.7 MJ_{ex} extracted to produce 1 MJ_{ex} seaweed biomass (10%DW). Due to the long distance between the hatchery and grow-out phase at sea, the diesel consumption for transport makes a large contribution to the overall footprint (44.3%). Furthermore, the production of materials (especially culture and anchor ropes) used at the hatchery and sea site requires a considerable amount of raw resources and contributes to 36.6% of the footprint. Surface occupation has a relative contribution of 11.9% (6% due to land occupation and 94% due to sea surface occupation). The direct electricity consumption of machinery has a contribution of 6.8% with the majority of electricity used for lighting the cultivation bottles and tanks in the hatchery. The impact of fresh water, nutrients and chemicals consumed during the seedling production is less than 1.0%.

It can be concluded that fossil resources are mainly consumed during seaweed cultivation (contribution of 75.1%); diesel is produced on the basis of crude oil and power production in Ireland relies mainly on natural gas, hard coal and peat (Itten et al., 2012). This has an implication on the use of nuclear resources, which is lower than in countries such as France having a large share in nuclear power. Furthermore, the extraction of marine resources, i.e. sea surface occupation by a human made system, contributes to 11.2% of the overall resource demand (Figure 13).

France

The cultivation of seaweed in France has a resource footprint of 8.7 MJex MJex-1, which is a factor 5 more than the footprint in Ireland. The impact of infrastructure used at the hatchery (CEVA) and for deployment at sea has the biggest contribution (54.7%). Compared to the seaweed facility in Ireland, this could be expected due to material-intensive cultivation system (plastic tubes) used in France. This is related to the impact of direct surface occupation (15.6%), of which 84% is due to sea surface occupation and 16% due to land occupation. The production of electricity and use in the hatchery represents 16.2% of the overall footprint (especially due to operation of the blower). Furthermore, direct gasoline consumption (approximately 550 L yr^{-1}) used for transport by boat during deployment of the equipment, maintenance and harvest of the biomass contributes to 13.4%. Less transport fuel is used in France compared to Ireland, as the hatchery and sea site are situated much closer to each other. Similar to the life cycle of seaweed production in Ireland, the impacts of fresh water, nutrients and chemicals are negligible.

A major impact is identified for fossil resources due to the consumption of gasoline and energy during the production of equipment (61.0%) (Figure 13). According to the International Energy Agency (IEA) statistics, the electricity production in France relies mainly on nuclear resources (75%), e.g., uranium (Itten et al., 2012), which represents 17.1% of the total resource footprint. Moreover, the demand for marine resources of raft systems that have a higher average sea surface occupation factor α than a single longline system cannot be ignored (13.1%).
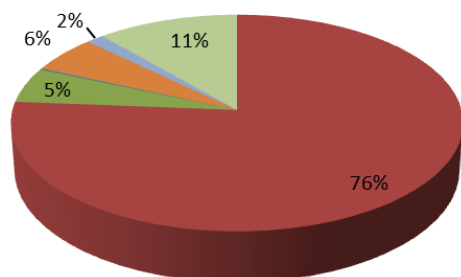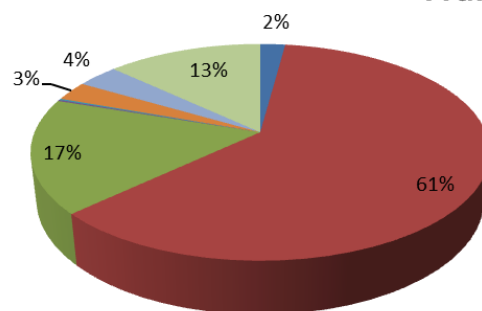
Figure 13: *Most exploited natural resources (%) to produce 1 MJ$_{ex}$ Saccharina latissima, in Ireland and France.*

Resource footprint of seaweed compared to microalgae and terrestrial plants

The total resource demand of seaweed production depends mainly on fossil fuels, a trend which is similar to microalgae production as this biomass can only be cultivated and harvested using energy-intensive processes. For the terrestrial plants, more than 90% of all required resources are land resources, especially for organically produced crops. Direct arable land occupation for cultivating the biomass and indirect land occupation for the production of manure are the biggest contributors. Interestingly, organic production requires more natural resources (especially land) than inorganic production as more green manure (organically produced) and more direct land is used to achieve the same biomass yield.
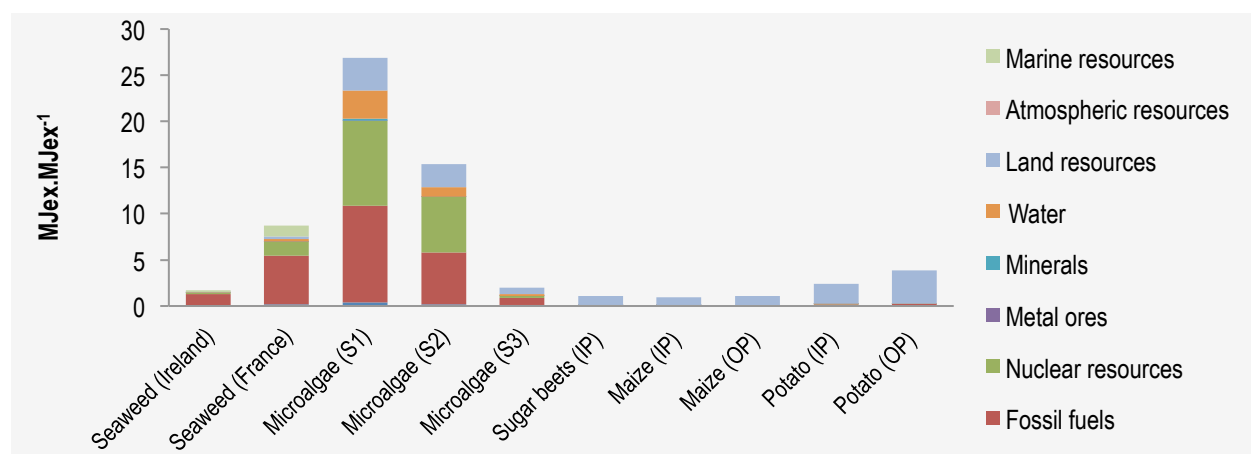
Figure 14: *Environmental resource footprint (expressed in MJex.MJex-1) of aquatic and terrestrial biomass production; seaweed from Ireland and France, microalgae from Belgium (S1 = 240m², S2 = 1320m², S3 = 2.5 ha) and sugar beet, maize and potatoes from Switzerland. IP = inorganic production, OP = organic production.*

Compared to the production of terrestrial plants (0.9-3.9 MJ$_{ex}$ MJ$_{ex}^{-1}$) seaweed production in Ireland is already quite efficient in terms of natural resource demand and is even more efficient than the third (hypothetical) scenario of microalgae cultivation (1.9 MJ$_{ex}$ MJ$_{ex}^{-1}$), as described in Taelman et al. (2013). However, in France, the resource footprint exceeds the footprint of all terrestrial plants (Figure 14). At present, attempts are being made to modify the raft systems to reduce friction and improve the yield.

Possible environmental improvements

In this study, the biggest potential to improve the footprint of seaweed production in Ireland relies on reducing the fuel demand for transport, i.e. benefits could be obtained by locating the hatchery and grow-out facility in the same area. In France, the resource footprint is five times as large, mainly because of the lower biomass yield of the system. As the raft structure occupies more sea surface than the single longline system, it is interesting to have a look at the effect of having a greater distance between the culture ropes. Furthermore, the use of plastic tubes at sea is resource demanding, so a scenario with an alternative floating material is analysed. The aeration device used in the hatchery is also over-sized and thus more efficient equipment could be used. Detailed information about the assumptions made is available in the study of Taelman et al. (2015a).

Tested parameters:
Ireland
  - Distance between facilities: from 490 km to 100 km

France
  - Power blower device: from 1.4 kW to 0.11 kW
  - Material floating tubes: HDPE to softwood
  - Distance between culture ropes: 2 m to 5 m
  - Biomass yield: 5 kg FW m^{-1} to 25 kg FW m^{-1}

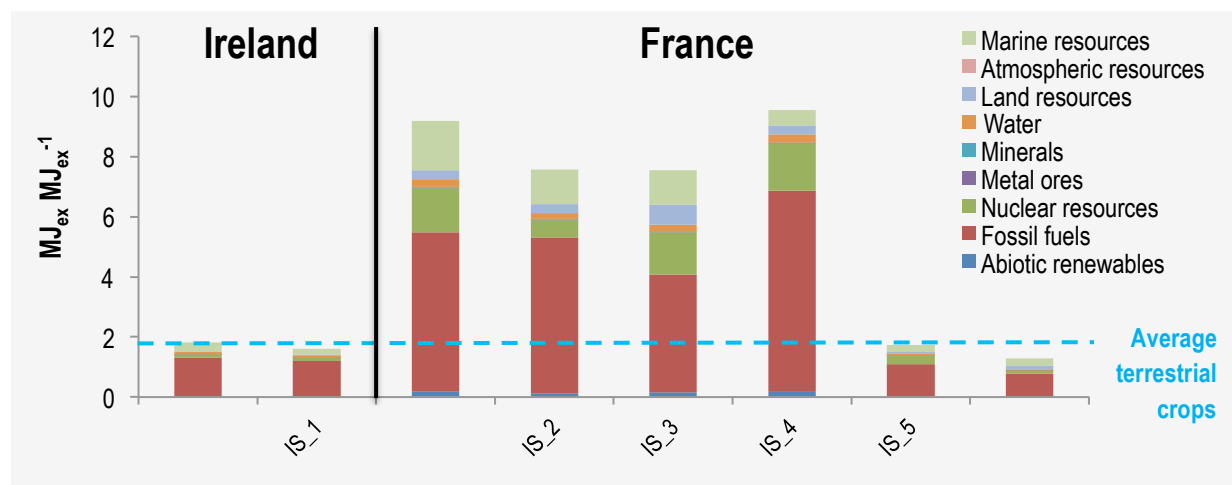***Figure 15****: Environmental resource footprint (expressed in MJex MJex-1) of seaweed production (Saccharina latissima) in Ireland and France. Results of the base cases as explained in Sections 3.2.1 and 3.2.2 are shown next to 5 improvement scenarios (IS); (IS_1) distance between facilities, (IS_2) power of blower device, (IS_3) floating tubes, (IS_4) distance between culture ropes, (IS_5) biomass yield. IS_2+3+5 represents the resource footprint of 3 improvements.*

In the case of Ireland, limiting the distance between the facilities up to 100 km improves the footprint by 11.4% compared to the footprint of the base case (IS_1). In the case of France, reducing the power consumption of the air blower reduces the footprint by 17.7% (IS_2). When HDPE tubes are replaced by wooden planks, the original resource footprint drops by 17.9% (IS_3). Most notably, increasing the distance between the culture ropes of the raft system to 5 m increases by 4% the life cycle demand for resources during seaweed production, i.e. despite the fact that the impact of sea surface occupation has been reduced, the demand for more HDPE because of the longer tubes per cultivation units results in a higher footprint. From Figure 15, it is clear that the environmental impact reduces considerably when productivity increases; the footprint decreases from 8.7 MJ$_{ex}$ MJ$_{ex}^{-1}$ to 1.7 MJ$_{ex}$ MJ$_{ex}^{-1}$ (comparable footprint as the Ireland base case) which emphasizes the importance of achieving a high yield. Ultimately, it is possible to achieve a life cycle resource footprint of 1.6 MJ$_{ex}$ MJ$_{ex}^{-1}$ (IS_1) and 1.3 MJ$_{ex}$ MJ$_{ex}^{-1}$

(IS_2+3+5) for Ireland and France, respectively, which is comparable to the footprint of the terrestrial plants.

2.3.5. Conclusion

This study highlights the usefulness of quantifying the total resource footprint (including marine resources) of seaweed production in NWE in a life cycle perspective. Mainly fossil resources (75.1%) are consumed in Ireland because of the high fossil-based fuel demand for transport between the different facilities. In France, fossil resources also take the largest share of the resource footprint, albeit at a lower rate than Ireland, followed by nuclear resources and marine resources. Fossils are used for gasoline production (transport fuel) and, together with nuclear resources, for electricity production. The raft system occupies more sea surface than single longline structures, which increases the consumption of marine resources (higher α factor). Compared to the footprint of microalgae and several terrestrial plants (sugar beets, maize and potatoes), seaweed production in North West Europe (especially in Ireland) is relatively resource-efficient. Moreover, the potential to improve the resource footprint of seaweed production is investigated; in the short-term, seaweed can be cultivated with a comparable life cycle resource demand as several terrestrial plants when less transport and electricity is used and the biomass productivity increased.

3. SUMMARY AND MAIN CONCLUSIONS

Because of the emerging interest in renewable energy production, efforts are currently being made to produce microalgae for fuel applications such as biodiesel, bio(syn)gas, bioethanol and biohydrogen. Seaweeds cannot be used for biodiesel production because of their low amount of extractable lipids but as an alternative, biogas could be produced via anaerobic digestion or ethanol via fermentation with yeasts, the latter production pathway still being in its infancy (Thi Hong Minh and Van Hanh, 2012). However, the production of these low value algae-based products involves worse performance than the existing (petroleum based) alternatives because the energy embodied in the algal fuel is lower than the energy required to produce it. Consequently, the focus has shifted toward higher value applications, such as animal feed, whether or not in combination with waste stream mitigation.

Insight into the potential of microalgae has led to the development of different cultivation and processing pathways. The most used cultivation systems are open (raceway) ponds as these systems are rather inexpensive, are based on a simple technology and generally consume less energy compared to closed systems (photobioreactors). However, the main drawback of these systems is the lower biomass productivity, which leads to higher harvesting and downstream processing requirements. For seaweed production in Europe, the most used cultivation systems are nearshore single longlines. However, for some marine regions, these longlines are not an option due to highly turbulent conditions. As a result, research is carried out to develop other cultivation systems such as raft systems. After the cultivation phase, a series of harvesting and processing methods are often necessary to deliver biomass with a desired moisture content. The most well-known methods are centrifugation, filtration, flocculation, flotation, gravity sedimentation and electrophoresis (Wang et al., 2015). Most of these methods are very energy-intensive and/or have high capital costs (Aitken and Antizar-Latislao, 2012). Often, multiple separation steps are used to concentrate the biomass, e.g., first microfiltration to retain the biomass followed by centrifugation (Weschler et al., 2014). Furthermore, often a drying method is used to conserve the biomass for a longer period of time. Several types of drying are applicable for algae: spray drying, drum drying, freeze-drying, belt drying and sun or wind drying, although it is stated that the latter is not very effective (Mata et al., 2009). Apart from these steps, cell disruption and extraction techniques can be used to release the metabolites of interest (Steriti et al., 2014; Pragya et al., 2013). Most technologies for algal growth, harvesting, and conversion are operational at a pilot scale (several squared meters). Pilot scale plants demonstrate the robustness and scalability of the technology, providing the degree of confidence that is required to secure the investment to take the technology to the next level. However, limited data exists about the feasibility of these technologies being able to operate at a commercial scale (Handler et al., 2014).

In this early phase of development, it is essential to consider the sustainability of the algae production to have minimal environmental and social impacts combined with maximal economic value. Probably the best way to reach a low environmental footprint is to switch from a linear economy to an integrated biorefinery concept in which algae are produced close to an industrial facility, which delivers its own products, and where waste streams (heat, waste water and flue gases) are used to stimulate the algal growth (Aitken and Antizar-Ladislao, 2012). This concept is beneficial for the joining industries: on the one hand, the required products are made, and on the other hand, sustainability-related issues such as land occupation, fossil fuel use and greenhouse gas emissions can be mitigated. In addition, when algae cultivation and processing are positioned near sale market, e.g., livestock production, it creates a win-win situation for all of the associates.

However, according to the LCA results of the case study on microalgae production as a livestock feed ingredient (section 2.1), the linear soybean economy still performs better (factor 100) than the integrated algal biorefinery when the total resource footprint is considered. The main reason is the high resource (energy)-demanding algal cultivation stages, especially the blowing of flue gases into the ponds. The

share of energy consumption, i.e. fossils, nuclear and abiotic resources, during soybean cultivation is much lower, as mainly land resources are used for soy production (relative percentages). The rather high differences in the resources consumed could be expected because a comparative study is conducted between a mature, large-scale technology (soybean meal/oil production) and a young, pilot-scale process chain (microalgae meal/oil production) that is still under development.

The same trend appears when microalgae are used to treat wastewater, e.g., backwash supernatant from pikeperch aquaculture systems (case study 2.2). The high contribution of the energy intensive stirring process of the MaB-floc based wastewater treatment plant is in line with the results previously found. However, the potential of impact reduction associated with up-scaling is highlighted; it is predicted that 66% less resources will be consumed when linearly up-scaled from 12 m² microalgae production area to 1 ha. In addition, possible improvements which can be made on this up-scaled plant (stirring efficiency, source of electricity production, and MaB floc productivity) were investigated; the potential to reduce the total resource footprint could not be ignored. After harvesting the MaB-flocs, a suitable valorisation pathway has to be chosen. A possible pathway is the use of wastewater-grown algae as feedstock for anaerobic digestion to produce biogas and later on, electricity and heat through a CHP system (Collet et al., 2011). However, according to the LCA results, this valorisation pathway is not the best option. In the search for higher value applications of the MaB-flocs, it seems that the use of dried MaB-flocs as pigment-enhancing feed for herbivorous aquaculture species is a more promising route. The LCA results reveal that valorizing MaB-flocs into shrimp feed is the best alternative from an environmental sustainability point of view.

As explained previously, microalgae have enormous potential but further progress must be made to lower the resource footprint, especially by increasing the biomass productivity and energy-efficiency of the cultivation stages and harvesting steps. Algae based technology in NWE must overcome these hurdles before it can compete with the currently existing alternatives.

Compared to the results from the microalgae case-studies (section 2.1 and 2.2), it can be concluded that, from an environmental resource point of view, seaweed production has more potential than microalgae production in NWE (section 2.3). A note should be made to grasp the fact that this conclusion is based on current pilot scale knowledge and hypothetical up-scaled plant scenarios. In addition, the LCA results of the case-study revealed that the resource footprint of seaweed cultivation in Ireland is already comparable with the footprint of several terrestrial plants (sugar beets, maize and potatoes). However, a careful interpretation is required as the composition (incl. dry weight content) and functionality of the different biomass types are not identical. However, this could not be concluded for the French case. As the raft systems used near the coast of France are subject to friction, a large amount of seaweed biomass is lost which results in a much lower biomass yield (factor 5) and a higher demand for infrastructure compared to the single long lines used near the coast or Ireland. This, of course, has a significant impact on the overall LCA results.

With respect to the type of resources used, more fossil resources are consumed during marine biomass production while more land resources are used for terrestrial biomass production. It seems that marine biomass meets the requirements to reduce pressure on land. As it is expected that the energy mix will become more renewable, it is anticipated that the footprint of seaweed production will be even lower in the future. At that point, seaweed could be cultivated as a sustainable feedstock in (North West) Europe as it avoids much of the competition for land and fresh water.

4. THE WAY FORWARD

4.1. Research gaps related to algae production for several applications

Because algae production in Europe is a young technology, many steps within the production chain can still be optimized, e.g., strain isolation, nutrient sourcing and utilization, production management, harvesting, coproduct development, pigment extraction, refining and residual biomass utilization. The major challenges include increasing the biomass concentration and reducing the energy requirements, especially at the cultivation stages. Improved engineering solutions will have a significant impact on algae production. These improvements can include efficient strategies for nutrient circulation and light exposure in combination with the development of low-cost scalable cultivation systems. Parallel to this research, the harvesting, dewatering and processing steps need further optimisation. Many technologies have already been successfully demonstrated but are relatively expensive, either in terms of equipment needed or energy required.

On top of the knowledge on the main challenges involving sustainable algae production in NWE, some specific examples of research gaps could be identified from the case studies.

Section 2.1
To increase to algae productivity in the ponds, which is required to reduce the overall footprint, it is possible to increase the temperature as this would have a positive effect on the algal growth (Aleya et al., 2011; Chalifour and Juneau, 2011). However, when more hot flue gases (C-source) and hot water is pumped to the ponds, more energy use, evaporation losses and pH fluctuations will occur (which can hamper algal growth). Therefore, more experiments/trial-and-error studies are necessary to achieve optimum biomass yield at a minimal environmental impact.

To reduce the consumption of water, energy, nutrients and land, some improvements to the integrated algal biorefinery can be made: the centrates can be recycled and pumped into the ponds, marginal land can be used to produce the biomass and further integration of the algae production in the livestock industry by using manure and urine as nutrients to feed the algae would be a step forward. In this study, the liquid fraction of the digestate that is produced within the biorefinery contains all necessary nutrients for stimulating microalgae growth. However, inducible from a small-scale experiment at the site, more research is necessary to determine the effect of these nutrient sources on the growth rate, as the variability and turbidity of these flows may distorts the light penetration and the possible presence of heavy metals could negatively affect the algal growth. Additionally, it appears possible at first sight to skip the energy-intensive drying process and directly feed the cattle with wet algal paste (after centrifugation). However, the risk of instability, contamination and non-availability of the biomass will increase. Additionally, the electricity and additional heat that is produced within the biorefinery could be used for algae cultivation and processing, and it would lower the resource footprint of the system. However, this approach leads to wrong conclusions when making a comparison with soybean meal production because the functionalities obtained from the basket of products would no longer be comparable.

It has been proven that some algae species have beneficial effects on the animal health. However, to substitute soybean meal which is the main protein source in the feed sector, algae should meet some other requirements: high protein levels, good amino acid quality, digestibility and low price (De Visser, 2013). Several algae species have a dry matter crude protein content of approximately 50%, which is higher than soybeans (44%), and the amino acid profile appears to be well balanced and is (similar to the digestibility) comparable with soybeans (Polprasert, 2007; Becker, 1994). However, due to the scarce availability of experiments and conflicting results, more research is necessary to determine whether the amino acid profile and protein efficiency ratio (i.e., digestibility) is similar to or better than soybeans. Moreover, one should wonder what the capability of livestock to tolerate (a full) replacement of soybean

meal with algal meal is. Factors causing intolerance might include the high ash content of the algae or amino acid imbalances (Lum et al., 2013). Furthermore, it is possible to feed the livestock with dried algal biomass without the extraction of oil. However, recent studies have indicated that the total amount of fat should be limited to ± 6% in the feed (dry matter). Otherwise, too much fat (especially unsaturated fatty acids) can cause problems for the digestive system.

Despite the potential of algae being used in the feed industry, it is not cost-effective yet to replace soybean meal with algal meal in cattle diets (Spolaore et al., 2006; Lum et al. 2013). As a result, current research must focus on increasing the efficiency along the microalgae production chain.

Section 2.2
The energy efficiency of the MaB-floc raceway pond should be improved and a possible solution is using paddle wheels instead of propeller pumps. This stirring system should be tested to know if it fits with Belgian conditions (possible freezing temperatures, especially during night when the pond is not stirred) as well as the conditions required for MaB-floc cultivation (necessity of a deep stirring due to the high settling speed of the MaB-flocs and possible need for high shear stress to induce bioflocculation). For both stirring systems, other improvements are possible to reduce electricity consumption. For example, changing the blade shapes of a paddle-wheel can reduce its shaft power consumption up to 50% (Li et al., 2013). Changing the design of the pond, such as adding baffle boards in the channel, can also participate to decrease the energy consumed for stirring (Chiaramonti et al., 2013).

When MaB-flocs are valorized as shrimp feed, a more efficient drying system could allow delivering additional heat to the market and increasing the associated environmental benefits. One bottleneck when using MaB-flocs as shrimp feed in Europe is that algae grown on wastewater are not allowed entering the European feed market (Van Den Hende et al., 2014b). This restricts the use of MaB-flocs as shrimp feed ingredient at the industrial sites where they are produced.

The environmental sustainability assessment may be improved in several ways. First, data on the direct GHG emissions from the raceway pond is needed as they may have a significant contribution to the total global warming potential of the system. Second, the nutrients content of the digestate should also be investigated in order to assess its potential of replacing synthetic fertilizers. Third, the estimation of the amount of organic carbon available in the digestate could also be refined as some carbon can be present in the digestate in the form of CH_4 or CO_2. Fourth, in this study, the remaining heat produced by the CHP is assumed to heat the fish tanks, which will be the case most of the year in Belgium, except during hot summers. Nevertheless, valorizing MaB-flocs as shrimp feed is still expected to be the most sustainable pathway as in this case even during summer time, heat can be valorized to dry the MaB-flocs.

Section 2.3
As the raft systems used near the coast of France are subject to friction, a large amount of seaweed biomass is lost. Despite this, previous experience suggests that the use of single longlines is not an option due to the turbulent marine conditions close to the harbour of Lézardrieux. Therefore, further research focusing on enhanced environmentally suitable cultivation techniques in France is necessary, especially because an improved biomass yield has the most significant impact on the overall LCA results.

When comparing the resource footprint of nearshore seaweed cultivation in Ireland with the footprint of several terrestrial plants (e.g., maize), it could be concluded that marine biomass production is already quite efficient and competitive. However, this comparison assumes an equal functionality of both types of biomass. Though, seaweed biomass has at the time of harvest a higher moisture content (10% DW) than the terrestrial crops (approx. 24%). This implies effects later on in the process chain, when e.g., more drying of the seaweed biomass is required. When valuable data of potential processing steps become available, more research will be required to fully quantify the life cycle footprint of the whole process

chain. Furthermore, additional effects of seaweed production on the environment (emissions, biodiversity, nutrient bioremediation etc.) and the economic feasibility should be assessed and compared to the existing alternatives.

4.2. LCA challenges related to quantifying environmental impacts of algal products

4.2.1. Overall LCA challenges

LCA is an increasingly important tool for environmental policy, and even for industry. When performing an LCA, methodological choices such as allocation and defining the system boundaries, time scope and functional unit have to be made. This may hinder a fair comparison of the results and makes it more difficult to draw overall conclusions. In addition, data availability, data quality and types of data used in the assessment are the focus of discussion for many LCA studies. Due to the lack of data from large scale operative plants, the studies are often based on conceptual designs and assumptions and in the best case on pilot scale data, which adds to uncertainty. LCA is a viable screening tool that can pinpoint environmental hotspots in complex value chains. However, the fact that we have to make a choice out of the range of existing LCA methods implies the incompleteness of grasping all relevant environmental impacts. Future advances in LCA in enhancing regional detail and accuracy as well as broadening the assessment to economic and social aspects will make it more relevant for the end-users.

4.2.2. LCA challenges associated with algae production

The fact that algae can be cultivated on marginal, non-arable land or in the sea is an enormous advantage because (fertile) land is today a scarce resource. However, accounting for the environmental benefit of using marginal land instead of fertile land used by other biomass types (e.g., food crops) or the use of sea surface is not aphoristic. The assessment of terrestrial land use (or land occupation) has gained already wide attention in LCA. Nevertheless, efforts still have to be made to account for the environmental impact of different land uses in a consistent way. Several questions arise: should we account for resource depletion related to land use or focus of the impact on ecosystem functioning? What is the best way to grasp these impacts, i.e., what kind of data do we use to calculate these impacts? Would it be feasible to develop one method that accounts for all land use impacts or do we let experts develop LCA methods that account for specific land use impacts? These questions still need to be answered and a more detailed guideline on global land use impact assessment is required.

Additionally, accounting for the impact of using sea surface (e.g., seaweed farming) in LCA is even a more difficult task. The study of Taelman et al. (2014) provides an LCA method (CEENE 2014) as a first attempt to quantify the impact of marine areal occupation. Seaweed production systems have to a certain extent an environmental impact related to the use of sea surface due to the fact that the infrastructure used at sea may interfere with the availability of sunlight for NPP production. Therefore, the complexity of the marine environment should be fully taken into account. More research and experimental data are required on light scattering, turbulent conditions, the way seaweeds hang in the water column and the light permeability of seaweed blades. The need for more data as well as the necessity for combining fields of expertise around the complex marine environment should lead to a better estimation of the impact of sea surface occupation. Therefore, a multidisciplinary approach is required and joint initiatives of research institutions, policy and industry are essential.

Apart from the challenges related to a better quantification of land or sea use when cultivating algae, there is a need for more LCA case-studies on algae production for applications other than biofuels. Especially for Europe and on the shorter term, there is a strong indication that algae should be cultivated for higher value applications (e.g., as a cosmetic ingredient). Moreover, data is missing on direct

emissions from algae cultivation systems. In the case of wastewater treatment, direct emissions from open ponds can have a significant contribution to the environmental footprint of the whole system. Therefore, emission-related measurements should be made for a better quantification of such aspects in LCA.

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