Life cycle greenhouse gas emissions of microalgal fuel from thin-layer cascades

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Abstract
Thin-layer cascades (TLCs) enable algae cultivation at high cell densities, thus increasing biomass yields and facilitating the harvest process. This makes them a promising technology for industrial-scale algal fuel production. Using Life Cycle Assessment (LCA), we calculate the greenhouse gas (GHG) emissions of aviation fuel produced using algal biomass from TLCs. We find that the impact (81 g CO_2e per MJ) is lower than that of fuel from algal biomass cultivated in open race way ponds (94 g CO_2e). However, neither of the two cultivation systems achieve sufficient GHG savings for compliance with the Renewable Energy Directive II. Seawater desalination in particular dominates the TLC impact, indicating a trade-off between carbon and water footprint. In both cultivation systems, the mixing power and fertilizer consumption present further significant impacts. There is uncertainty in the correlation between mixing power and algal oil yield, which should be investigated by future experimental studies.

Keywords Life cycle assessment · Greenhouse gases · Microalgae · Fuel · Thin-layer cascade · Cultivation system

Introduction

In 2018, the global transport of passengers and goods was responsible for 8.0 Gt of CO_2 emissions (24% of total fuel-related emissions) [1]. Avoiding these emissions is the aim of several frameworks to which the international community has committed [2, 3]. Biofuels will play an important role in these efforts, despite the advent of batteries and fuel cells, especially in sectors like long-haul aviation, which require energy-dense fuels. To meet the demand of these sectors while avoiding land competition with the food and feed sector, low land-use change-risk biofuel-feedstocks are needed. Microalgae offer several advantages in this regard, i.e. the possibility to use marginal lands for their cultivation and their high theoretical biomass yield [4]. Yet, there are doubts whether algal fuels can achieve the greenhouse gas emission (GHG) reductions necessary to comply with existing regulations, such as the Renewable Energy Directive II in the EU (RED II) [5] and the Renewable Fuel Standard in the US (RFS) [6]. In a meta-analysis comprising 69 Life Cycle Assessments (LCAs) of renewable algal diesel, Tu et al. found that only 17 achieved sufficient GHG reductions for the RFS (50% GHG reduction compared to 2005 baseline diesel) [7]. The RED II mandates even higher GHG savings of 65% for biofuel producers starting operation after 1 January 2021 [5]. Needless to say, the algal fuel community is facing a challenge. Several studies have identified algae cultivation as the bottleneck towards economically- and environmentally feasible algal fuel production [8–11]. New cultivation technologies could reduce the impact, thereby enabling affordable and regulation-compliant algal fuel production. One such technology is proposed by Doucha et al. [12]: Comparing sloping thin-layer cascades (TLCs) to conventional open raceway ponds (ORPs), they find that the former require less power, water, and CO_2 input per unit biomass produced. Although TLCs date back to the 1950s, they have received relatively little attention to date [13, 14]. The existing literature is largely focused on engineering aspects, such as optimal operating conditions and determination of culture parameters [15–22]. Only few techno-economic assessments have been published [12, 23, 24]. To the author's knowledge, no LCA of algae cultivation in TLCs has been conducted so far – a gap we wish to close with this study.

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Method

Life Cycle Assessment (LCA) is used to calculate the Global Warming Potential (GWP 100) of 1 MJ of algal fuel (functional unit) produced using the TLC-algae fuel pathway shown in Fig. 1. The life cycle comprises the following stages: a biogas-fired combined heat and power (CHP) plant provides CO_2 to the algae cultivation facility where algae grow in autotrophic mode in TLCs, the biomass is harvested, converted into fuel, residual biomass is valorized energetically, fuel is transported and combusted. Modeling of each stage has been conducted in MS Excel based on engineering first principles and parameters have been tuned to values from literature and expert interviews. The resulting workbooks are available in the supporting information. Background models from the ecoinvent 3.6 APOS database are used to model upstream supply chains, such as electricity generation and fertilizer production [25]. The impact assessment has been conducted in Brightway2 using the ILCD 2.0 2018 midpoint, climate change total method [26]. Apart from fuel, the studied pathway produces electricity and heat. This multifunctionality is resolved by substitution with ecoinvent markets as indicated in the corresponding sections. The TLC pathway is compared to two other options, namely petroleum-based fuel and algal fuel produced using state-of-the-art ORPs for biomass cultivation. Both algae-based pathways are presented in the following.

Model

CO_2 source

Flue gas CO_2 from a combined heat and power plant (CHP) using biogas as fuel serves as the primary carbon source for autotrophic microalgae cultivation. CHP operation and upstream burdens are modeled based on ecoinvent activity *heat and power co-generation, biogas, gas engine, ES*. This activity has been edited by (a) eliminating all CO_2 emissions to the atmosphere and (b) implementing an energy penalty of 8% to account for CO_2 capture and compression [27]. After subtraction of the penalty, the CHP plant produces 5.8 MJ of useful heat and 4.1 MJ of electricity per kg CO_2. Useful heat is assumed to displace conventional heat production, here modeled by ecoinvent activity *heat and power co-generation, natural gas, combined cycle power plant, 400 MW electrical, ES*. Electricity output from the biogas CHP is assumed to displace the Spanish grid mix, *market for electricity, high voltage, ES*.

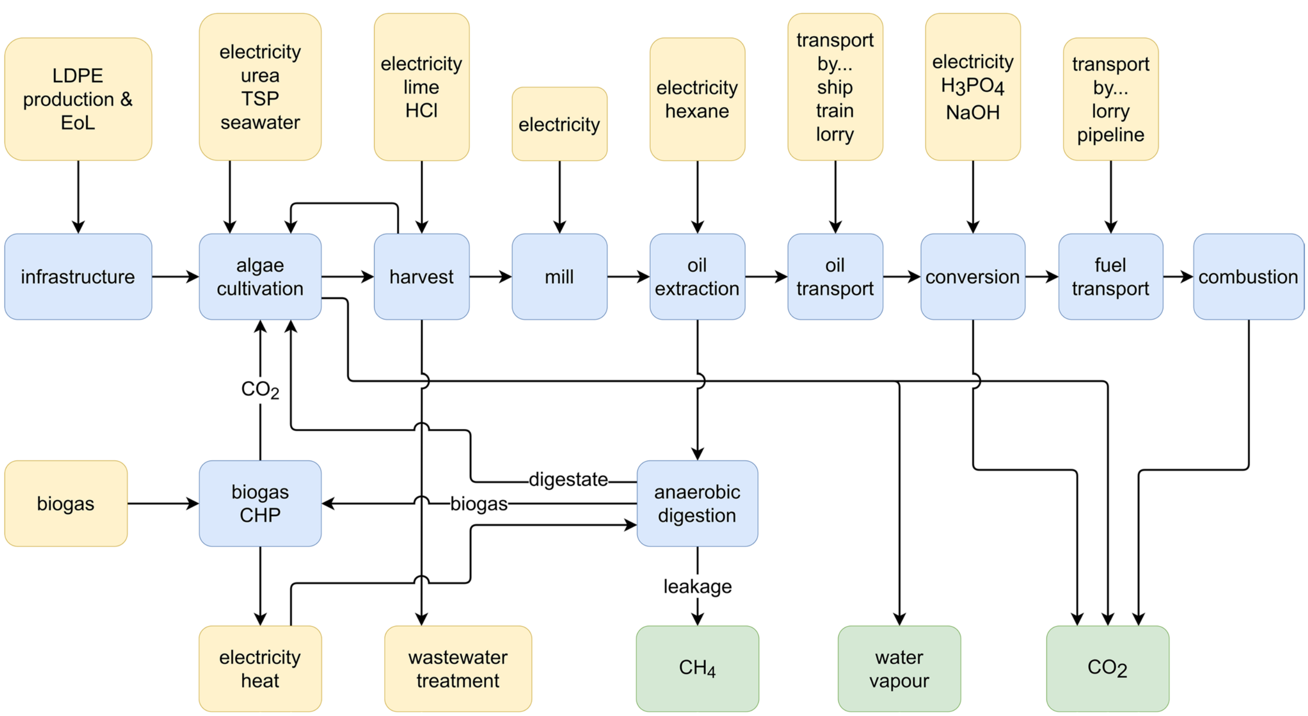

Fig. 1 Schema of the studied algal fuel production pathway. Boxes present process steps. Colors distinguish process types: foreground database (blue), background database (yellow), biosphere (green). *CHP*-combined heat and power, *LDPE*-low density polyethylene, *EoL*-end of life, *TSP*-triple superphosphate

Cultivation

Thin-layer cascades consist of inclined planes, here made from compacted sand, on which a thin fluid film containing the algae cells flows downstream, driven by gravity. Thanks to the film's thinness (here 6 mm), mutual shading between algae cells is reduced and the surface for gas exchange is greatly enhanced. This allows TLCs to maintain steady biomass growth rates up until cell concentrations of 40–50 g_{DW}/L [19, 21]. Seasonal productivity typically ranges between 22 and 25 $g_{DW}/(m^2d)$ [12]. For our study, we consider a hypothetical cultivation plant in a coastal area in Spain, which is operated during 8 months per year at an area-related productivity of 25 $g_{DW}/(m^2d)$ and a final cell oil content of 30 wt-%. At these operating conditions, the 100 ha facility yields 1 800 t of algae oil per year.

To maintain the flow on the inclined planes, a pump continuously circulates cultivation medium from the bottom reservoir back to the top of each plane. Excluding friction losses, the power demand per unit area is proportional to the plane inclination I, the fluid layer thickness h, the flow velocity u, the gravity constant $g = 9.81$ m/s^2, the medium density $\rho \approx 1\,000$ kg/m^3, and the inverse of the pump efficiency η_p [18]:

$$\frac{P}{A} = \frac{I\,h\,u\,g\,\rho}{\eta_p} \tag{1}$$

Using values from Doucha and Lívanský [18] for I (1.7%), h (6 mm), u (0.6 m/s) and assuming a pump efficiency of 80%, the specific power demand amounts to 0.75 W/m^2. The pump is operated only during day (12 h). At night, the cultivation medium is stored in a retention tank where it is mildly aerated. Doucha and Lívanský [12] approximate the power demand for aeration at 40% of the demand for pumping. The total power demand is then 12.6 Wh/(d m^2) or 0.50 Wh/g_{DW}. Note that this is a lower-bound estimate, as additional dissipation mechanisms such as pipe friction have been neglected.

To derive the nutrient demand, we follow the approach described by Geider and La Roche [28], yielding necessary C, N, and P input from the algae cell's macro-molecular composition. The algae cells are presumed to contain (by weight) 35% proteins, 30% lipids, 10% phospholipids, 20% carbohydrates, and 5% nucleic acids. Part of the nutrient demand is satisfied by anaerobic digestate and flue gas, which are by-products from the residue valorization unit (see section on residue valorization). These by-products reduce the C-demand by 30% and N- and P-demand by 48% each. The remaining demand is fulfilled by CO_2 from the biogas CHP, as well as by urea and triple superphosphate (TSP) from the market. It is further assumed that, due to technical limitations, 25% of the supplied CO_2 is lost to the atmosphere and 15% of the supplied N in urea is lost by other mechanisms. The net nutrient demand is then 1.9 kg CO_2, 0.17 kg urea, and 0.035 kg TSP per kg_{DW} biomass.

Lastly, water input is needed to compensate evaporation and technical losses. We assume that seawater is used to protect natural freshwater resources. For a Mediterranean climate, Guieysse et al. [29] estimate a net evaporation rate of 1.3 m^3/(m a). The evaporated seawater leaves behind salt, which accumulates in the cultivation medium. To maintain favorable cultivation conditions, this accumulation must be balanced by salt removal or by adding freshwater. A mass balance of both water and salt flows around the cultivation unit yields the following relationships:

$$\dot{m}_{\text{fresh}} = \dot{m}_{\text{evap}} + \dot{m}_{\text{harvest}}(1 - R)\left(1 - \frac{c_{\text{culture}}}{c_{\text{sea}}}\right) \tag{2}$$

$$\dot{m}_{\text{sea}} = \dot{m}_{\text{harvest}}(1 - R)\frac{c_{\text{culture}}}{c_{\text{sea}}} \tag{3}$$

where $\dot{m}_{\text{fresh}}$ is the freshwater demand, $\dot{m}_{\text{sea}}$ is the seawater demand, $\dot{m}_{\text{evap}}$ is the evaporation rate, $\dot{m}_{\text{harvest}}$ is the amount of water removed during harvest, R is the fraction of water returned to the cultivation unit after harvest, c_{culture} is the salt concentration in the cultivation unit and c_{sea} is the concentration of salt in seawater. Note that the flow rates $\dot{m}_i$ here are normalized by the biomass production rate, yielding units of gram water per gram biomass dry weight.

Looking at Eq. (2), we see that the second summand turns negative if the salt concentration in the cultivation unit is higher than in seawater ($c_{\text{culture}}/c_{\text{sea}} > 1$). This means that freshwater consumption can be reduced by reducing the water recycling rate (R) and increasing the culture salt concentration (c_{culture}). However, this reduction will come at the cost of an increased seawater demand. By rearrangement, one obtains an expression for the salt concentration at which no external freshwater source is necessary:

$$c^0_{\text{culture}} = c_{\text{sea}}\left(1 + \frac{\dot{m}_{\text{evap}}}{\dot{m}_{\text{harvest}}(1 - R)}\right) \tag{4}$$

In TLCs, the cell concentration at the time of harvest is relatively high (here 20 g_{DW}/L). This in turn means that the water flow rate is low ($\dot{m}_{\text{harvest}} = 50$ g/g_{DW}) compared to evaporation ($\dot{m}_{\text{evap}} = 220$ g/g_{DW}). According to Eq. (2), the salt concentration would have to be higher than 19% to eliminate freshwater use (for $c_{\text{sea}} = 3.5\%$ and $R = 0$). Such concentrations would require adapted algae strains, which are likely unsuitable for industrial-scale biofuel production. Consequently, an external freshwater source is necessary for algal fuel production with TLCs. This is in contrast to

ORPs, which feature lower biomass concentrations (here 0.5 g_{DW}/L), making harvesting an effective salt removal mechanism. For our analysis, we assume that the TLCs are nevertheless operated at a slightly elevated salt concentration of 5.3% for a moderate reduction in freshwater demand.

All water used in the cultivation stage is supplied via pipeline from the neighboring sea. Freshwater is provided by re-routing part of the seawater to a reverse osmosis plant. Energy demand of the pipeline is estimated using Eq. (5) and depends on the total flow rate $\dot{m} = 260$ g/g_{DW}, the gravity constant $g = 9.81$ m/s^2, the pump efficiency $\eta_p = 80\%$ and the head Δh. The latter will depend on the distance to the sea, the height difference between pipeline inlet and outlet, as well as friction losses along the way. Without a concrete design at hand, we assume an arbitrary head of 60 m. The power demand then amounts to 0.054 Wh/g_{DW}. Note that this estimate is one order of magnitude smaller than the mixing power demand (see Eq. (1)).

$$P = \frac{\dot{m}\ g\ \Delta h}{\eta_p} \tag{5}$$

For the ORP pathway, modeling is analogous apart from the following points: ORPs are deeper (here 30 cm) than TLCs (here 6 mm) and algae cells in deeper depths receive less sun light than cells at the surface. The lower average sun exposure leads to lower biomass yields on the order of 15 g_{DW}/(m^2 d). Consequently, a larger cultivation area is needed (175 ha) to achieve the same output per time unit. This in turn increases evaporation losses ($\dot{m}_{evap} = 360$ g/g_{DW}). Because ORPs achieve lower cell densities (0.5 g_{DW}/L) compared to TLCs, more water (and hence salt) is removed in the harvest process ($\dot{m}_{harvest} = 2\ 000$ g/g_{DW}). Recycling 64% of the harvested water ($R = 0.64$) at a slightly elevated salt concentration ${c^0}_{culture} = 5.3\%$ eliminates the need for an external freshwater source and yields a total seawater demand of 1 100 g/g_{DW}. The power demand for water transport via pipeline is then 0.22 Wh/g_{DW}. ORPs are typically mixed using paddle wheels whose energy demand depends on several factors, such as angular velocity and depth of the pond [30]. Here, a moderate mixing power requirement of 0.4 W/m^2 (= 0.48 Wh/g_{DW}) is assumed [30]. Nutrient demand per unit biomass is identical for both ORP and TLC cultivation scenarios, as identical biomass compositions and nutrient utilizations are assumed.

Concerning construction materials, we assume that both reactors are cost-effectively implemented by spreading LDPE pond liners over compacted sand. A pipeline is built to deliver saltwater from the nearby sea to the cultivation units. End of life (EoL) of pond liners and pipelines are accounted for using corresponding ecoinvent activities.

Harvest

The harvest procedure differs between ORP and TLC cultivation. In both scenarios, a final biomass concentration of 20 wt-% is desirable for downstream processing. In the TLC scenario, this can be achieved in one step, using a centrifuge (concentration factor 10). It is assumed that 95% of the biomass is recovered at an energy demand of 4 kWh/${m^3}_{feed}$ [7].

In the ORP scenario, due to the large amount of water entrained in the harvest stream, direct centrifugation is uneconomical. The biomass needs to be pre-concentrated by flocculation before centrifugation. There is a rich body of literature on flocculation methods such as pH-shift, electrocoagulation, addition of metal salts, natural and synthetic cationic polymers, and bioflocculants [31]. For marine algae in particular, the naturally high concentration of magnesium ions in the cultivation medium favors flocculation by pH-shift [32–38]. We assume that an addition of 200 mg/L of slaked lime ($Ca(OH)_2$) results in the recovery of 95% of the biomass (concentration factor 40). HCl is subsequently added at a stoichiometric ratio of 2:1 to remove precipitated magnesium hydroxide from the biomass [36, 39] and to neutralize the supernatant. The precipitate is then further dewatered in the centrifuge, operated at identical conditions to the TLC case. Note that the additional flocculation step in the ORP scenario leads to an overall higher loss of biomass. This in turn leads to a larger biomass demand per unit algal fuel.

Water removed during harvesting can be recycled back to the cultivation unit to reduce seawater demand. As noted before, salt entrained in the recyclate will affect the water-salt balance of the system. In the ORP scenario, 64% of the water can be recycled while keeping the balance intact. In the TLC scenario, all harvest water is discarded to minimize the freshwater demand. In both scenarios, water which is not recycled is sent to a wastewater treatment plant for decontamination before returning to the sea. Due to a lack of precise compositional data in the literature, this process is modeled using the general ecoinvent activity *treatment of wastewater, average, capacity 1.1E10l/year, CH*.

Oil extraction and transport

To extract oil from the algae cells, their cell walls are first mechanically disrupted in a ball mill. Power consumption for this process is estimated at 0.06 kWh/L based on the data sheet of an industrial manufacturer [40]. After cell disruption, liquid hexane is applied to separate the oil from the aqueous phase. An external heat source is necessary to regenerate the hexane. Based on Frank et al. [41], oil recovery ratio, heat demand, electricity demand, and specific

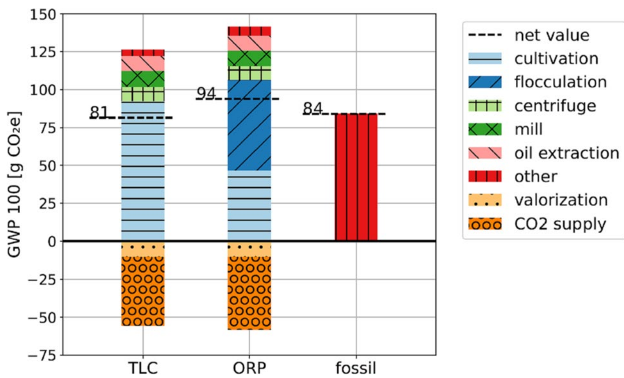

Fig. 2 Life cycle climate impact (GWP 100) of algal fuel from thin-layer cascades (TLC), open raceway ponds (ORP), and petroleum-based kerosene as the reference (fossil)

hexane loss are estimated at 95%, 6.1 MJ/kg_{oil}, 1.9 MJ/kg_{oil} and 5.2 g/kg_{oil}, respectively.

A single plant's oil output of 190 kg/h is small compared to the throughputs of conventional oil refineries. To leverage economies of scale, it is assumed that oil from multiple plants across Europe is collected and jointly processed in a dedicated bio-refinery in the Netherlands. The corresponding transport requirements are approximated as follows: 50 km via truck to a collection point, 100 km via train to a harbor, 3 500 km via ship to the refinery.

Conversion, fuel transport and use

In the bio-refinery, the algae oil is hydrotreated, yielding so-called HEFA fuel (hydrotreated esters and fatty acids, analogous to hydrotreated vegetable oil, HVO). The process model by Zschocke [42] is adopted: Phospholipids and other impurities are removed prior to conversion by application of phosphoric acid (0.62 g/kg_{oil}) and sodium hydroxide (1.9 g/kg_{oil}). The cleaned oil is then hydroprocessed, removing heteroatoms and saturating the carbon bonds. Hydrocracking increases the yield of middle distillates in the jet fuel range. Light fractions are consumed on-site to supply process energy and hydrogen for hydroprocessing. The product fractions (jet fuel, diesel, naphtha) total 1 MJ lower heating value by definition (functional unit). Direct CO_2 emissions during the conversion process amount to 0.50 kg/kg_{oil}.

From the refinery, the fuel fractions are transported to the end user. This process is modeled by 400 km pipeline transport followed by 50 km on truck. Final use is combustion in an engine, during which CO_2 is released to the atmosphere. Because this CO_2 is of biogenic origin and re-emitted within a short time frame (full loop: atmosphere → biomass → biogas → CHP flue gas → algae biomass → algal fuel → atmosphere), it is regarded as climate neutral in accordance with the ILCD guidelines [43].

Residue valorization

The aqueous residue left after algal oil extraction contains significant amounts of carbon and energy, which are recouped in a valorization unit. For energy recovery, the residual biomass is first converted into biogas via anaerobic digestion. A medium-to-low methane yield of 190 mL/g VS (volatile solids) is assumed to account for high salt concentrations in the feed [44]. The following assumptions are further made based on expert interviews: The biogas consists of methane (60 vol.-%) and CO_2 (40 vol.-%). Of the produced methane, 2% leak to the atmosphere. Electricity is needed for pumping and mixing in the digester (0.40 MJ/kg VS). Heat is needed to maintain mesophilic conditions (0.23 MJ/kg wet mass). We assume that digestate from the valorization unit contains bio-available N, P and C, which can supplement primary nutrients in the algae cultivation unit. Overall, 50% of N and P in the algae cake are salvaged, along with 34% of the carbon.

Biogas produced in the anaerobic digester is burnt in the CHP plant along with market biogas, producing heat, electricity, and CO_2. The secondary biogas (produced in the digester) supplies 0.24 MJ of useful heat and 0.20 MJ of electricity per unit fuel produced. As before, combustion CO_2 is captured and supplied to the cultivation plant. Credits are granted for the production of heat and electricity based on the ecoinvent activities *heat and power co-generation, natural gas, combined cycle power plant, 400 MW electrical, ES* and *market for electricity, high voltage, ES*, respectively.

Results and discussion

Figure 2 compares the GHG emissions (expressed per GWP 100) of the two algae pathways to that of conventional petroleum kerosene. The TLC pathway has the lowest GHG intensity at 81 g CO_2e per MJ lower heating value (LHV), followed by the fossil reference (84 g CO_2e) and the ORP pathway (94 g CO_2e). This result supports prior techno-economic analyses, which found that TLCs could be more resource-efficient than ORP technology [12, 24]. The TLC fuel achieves a GHG reduction of 4% compared to petroleum kerosene. This reduction is too small for the Renewable Energy Directive II, which demands 65% (for installations starting operation after 1 January 2021) [5]. Thus, both algal fuel pathways need further improvement before they can count towards the goals of the Paris Agreement [3]. The following sections highlight major emission sources in each pathway to direct future development efforts.

Cultivation and harvesting dominate the life cycle climate impact of both TLC- and ORP-algae-fuels. This finding is in line with previous LCAs on algae fuel production in ORPs [9–11, 45]. Within TLC cultivation, seawater desalination causes the largest impact (54 g CO_2e per MJ fuel LHV), mostly from energy used in the process. The impact can be reduced if renewable electricity is used in desalination or if freshwater is drawn from natural sources. In the latter case, the impact on local water reserves must be evaluated critically. Without reverse osmosis, the life cycle impact of TLC algal fuel can be as low as 27 g CO_2e, potentially fulfilling the RED II requirements.

The second largest GHG source in TLC cultivation is mixing (pumps for circulation during the day and air compressors at night, total 19 g CO_2e). Although the nominal power demand per unit area is higher for TLCs than ORPs, they achieve a proportionately higher biomass yield, resulting in a similar energy demand per unit biomass. Experimental power consumption values for large-scale TLCs are missing at the time of writing, which means our estimates are indicative. More generally, mixing influences biomass yields by controlling the light–dark-cycle imposed onto the algal cells. Reducing the power input will thus also reduce the biomass yield. Further research is necessary to quantify the relationship between both trends.

The third largest GHG contribution in TLC cultivation comes from urea supply (16 g CO_2e). Previous studies have highlighted the role of fertilizers in the GHG balance of algal fuel [8, 9, 11, 45]. Although the impact is dampened in our study by supplementing anaerobic digestate, it is still significant. Future studies should investigate, to what extent C, N and P in digestate are bio-available to microalgae. If the amount turns out to be insufficient, other nutrient sources (e.g. municipal or industrial wastewaters) should be investigated. Alternatively, the impact of artificial fertilizers can be reduced using green energy and hydrogen in their production.

In our ORP model, cultivation impacts are dominated by paddle wheel operation (19 g CO_2e), followed by urea consumption (17 g CO_2e) and seawater pumping (9 g CO_2e). Our estimate from paddle wheel operation is low compared to other studies such as [12] and [24]. Validation is complicated by a lack of data on large-scale ORP plants employing autotrophic algae growth under lipid-accumulating conditions. Again, mixing power and lipid/biomass yield are intertwined and further research is necessary to narrow down the range of meaningful assumptions. For urea production, the same comments as for the TLC apply. For seawater pumping, power demand follows Eq. 5 and is most easily reduced by locating the cultivation plant as close as possible to the sea. Furthermore, the transport demand can be reduced by avoiding water losses within the system. Strategies such as the one suggested by Collet et al. [11] (cover open cultivation systems by plastic-sheets on scaffoldings to recover evaporated water), seem promising in this regard.

Apart from cultivation, flocculation is a significant contributor to the ORP GHG balance. Its impact is shared equally between HCl consumption (29 g CO_2e) and wastewater treatment (29 g CO_2e). Note that the impact of lime consumption is negligible. HCl-use can be avoided if $Mg(OH)_2$ is allowed to remain in the precipitate. Future studies should investigate to which extent this can negatively affect downstream operations (the literature suggests that anaerobic digestion may be negatively affected by elevated alkaline earth metal concentrations in the feed [46]). Lastly, the impact of wastewater treatment is proportional to the amount of discarded water, as well as its pollution. Control over the former is limited by the water-salt balance described by Eq. 2 and 3. If the amount of wastewater is reduced by recycling more harvest water, an external freshwater source is necessary to prevent salt accumulation. The use of desalination in this regard may be prohibitive if no clean electricity is available (see TLC scenario). Concerning wastewater composition, we chose an 'average' scenario (according to ecoinvent definition) for our analysis, as no data on the composition of spent cultivation media was available. The actual impact of wastewater treatment will depend on the degree of pollution and the on local discharge regulations and will thus vary from location to location.

Conclusion

Our study presents for the first time a life cycle climate impact assessment of aviation fuel produced from algal biomass cultivated in thin-layer cascades (TLCs). Our results support the notion of prior techno-economic assessments, finding that TLCs could be more resource-efficient than established open raceway pond technology [12, 24]. Still, further improvements are necessary before fuel from TLC-algae can achieve the GHG savings mandated by the Renewable Energy Directive II [5]. Unlike ORPs, TLCs depend on an external freshwater source to prevent the accumulation of salt within the cultivation medium. Locations with abundant freshwater are thus preferable, as long as a low impact on local water reserves can be assured. Our results further indicate that power for water circulation presents a relevant contribution to the GHG footprint, although its real magnitude is rather uncertain. Future studies should investigate the correlation between power consumption and biomass yield in various climate conditions, to reduce the uncertainty. Lastly, our results support previous studies finding that artificial fertilizer consumption is relevant to the life cycle climate impact [8, 9, 11, 45]. Very high nutrient recovery rates from anaerobic digestate seem necessary to lower this impact. Future experiments should clarify to which degree such recovery rates can be achieved and–if not–which other, climate-friendly nutrient sources are available.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s00449-021-02612-9.

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Data availability The authors confirm that the data supporting the findings of this study are available within the article and its supplementary materials.

Code availability The following files are available free of charge in the supporting information: model_ORP.xlsx: ORP model. Model_TLC.xlsx: TLC model. Import_ORP.py: Python script to import ORP model into Brightway2. Import_TLC.py: Python script to import TLC model into Brightway2. Environment.yml: conda environment file (used to install Python dependencies, incl. Brightway2).

Declarations

Conflics of interest The authors declare no conflicts of interest.

Ethical approval This research involved no human participants and/or animals.

Consent to participate Not applicable.

Consent for publication Not applicable.

References

1. International Energy Agency (2019) Tracking transport. https://web.archive.org/web/20200316214719/https://www.iea.org/reports/tracking-transport-2019. Accessed 13 July 2021
2. United Nations Framework Convention on Climate Change (1997) Kyoto protocol. https://treaties.un.org/pages/ViewDetails.aspx?src=TREATY&mtdsg_no=XXVII-7-a&chapter=27&clang=_en. Accessed 13 July 2021
3. United Nations Framework Convention on Climate Change (2015) Paris agreement. https://treaties.un.org/pages/ViewDetails.aspx?src=TREATY&mtdsg_no=XXVII-7-d&chapter=27&clang=_en. Accessed 13 July 2021
4. Chisti Y (2007) Biodiesel from microalgae. Biotechnol Adv 25:294–306. https://doi.org/10.1016/j.biotechadv.2007.02.001
5. European Union (2018) Directive (EU) 2018/2001 of the European parliament and of the council of 11 December 2018 on the promotion of the use of energy from renewable sources. Off J Eur Union 5:82–209
6. United States Congress (2007) H.R.6 - Energy Independence and Security Act of 2007. https://www.congress.gov/bill/110th-congress/house-bill/6. Accessed 13 July 2021
7. Tu Q, Eckelman M, Zimmerman J (2017) Meta-analysis and harmonization of life cycle assessment studies for algae biofuels. Environ Sci Technol 51:9419–9432. https://doi.org/10.1021/acs.est.7b01049
8. Lardon L, Hélias A, Sialve B et al (2009) Life-cycle assessment of biodiesel production from microalgae. Environ Sci Technol 43:6475–6481. https://doi.org/10.1021/es900705j
9. Clarens AF, Resurreccion EP, White MA, Colosi LM (2010) Environmental life cycle comparison of algae to other bioenergy feedstocks. Environ Sci Technol 44:1813–1819. https://doi.org/10.1021/es902838n
10. Passell H, Dhaliwal H, Reno M et al (2013) Algae biodiesel life cycle assessment using current commercial data. J Environ Manage 129:103–111. https://doi.org/10.1016/j.jenvman.2013.06.055
11. Collet P, Lardon L, Hélias A et al (2014) Biodiesel from microalgae–life cycle assessment and recommendations for potential improvements. Renew Energy 71:525–533. https://doi.org/10.1016/j.renene.2014.06.009
12. Doucha J, Lívanský K (2014) High density outdoor microalgal culture. In: Bajpai R, Prokop A, Zappi M (eds) Algal biorefineries. Springer Netherlands, Dordrecht, pp 147–173
13. Masojídek J, Prášil O (2010) The development of microalgal biotechnology in the Czech Republic. J Ind Microbiol Biotechnol 37:1307–1317. https://doi.org/10.1007/s10295-010-0802-x
14. Grivalský T, Ranglová K, da Câmara Manoel JA et al (2019) Development of thin-layer cascades for microalgae cultivation: milestones (review). Folia Microbiol 64:603–614. https://doi.org/10.1007/s12223-019-00739-7
15. Doucha J, Lívanský K (1995) Novel outdoor thin-layer high density micro algal culture system: productivity and operational parameters. Algol Stud/Archiv für Hydrobiol, Suppl Vol 76:129–147. https://doi.org/10.1127/algol_stud/76/1995/129
16. Grobbelaar JU, Nedbal L, Tichy L, Setlik L (1995) Variation in some photosynthetic characteristics of microalgae cultured in outdoor thin-layered sloping reactors. J Appl Phycol 7:175–184. https://doi.org/10.1007/BF00693065
17. Doucha J, Straka F, Lívanský K (2005) Utilization of flue gas for cultivation of microalgae Chlorella sp.) in an outdoor open thin-layer photobioreactor. J Appl Phycol 17:403–412. https://doi.org/10.1007/s10811-005-8701-7
18. Doucha J, Lívanský K (2006) Productivity, CO_2/O_2 exchange and hydraulics in outdoor open high density microalgal (Chlorella sp.) photobioreactors operated in a Middle and Southern European climate. J Appl Phycol 18:811–826. https://doi.org/10.1007/s10811-006-9100-4
19. Doucha J, Lívanský K (2009) Outdoor open thin-layer microalgal photobioreactor: potential productivity. J Appl Phycol 21:111–117. https://doi.org/10.1007/s10811-008-9336-2
20. Apel AC, Weuster-Botz D (2015) Engineering solutions for open microalgae mass cultivation and realistic indoor simulation of outdoor environments. Bioprocess Biosyst Eng 38:995–1008. https://doi.org/10.1007/s00449-015-1363-1
21. Apel AC, Pfaffinger CE, Basedahl N et al (2017) Open thin-layer cascade reactors for saline microalgae production evaluated in a physically simulated Mediterranean summer climate. Algal Res 25:381–390. https://doi.org/10.1016/j.algal.2017.06.004

22. Schädler T, Caballero Cerbon D, de Oliveira L et al (2019) Production of lipids with Microchloropsis salina in open thin-layer cascade photobioreactors. Biores Technol 289:121682. https://doi.org/10.1016/j.biortech.2019.121682
23. del Mar Morales-Amaral M, Gómez-Serrano C, Acién FG et al (2015) Outdoor production of Scenedesmus sp. in thin-layer and raceway reactors using centrate from anaerobic digestion as the sole nutrient source. Algal Res 12:99–108. https://doi.org/10.1016/j.algal.2015.08.020
24. Acién Fernández FG, Fernández Sevilla JM, Molina Grima E (2019) Costs analysis of microalgae production. Biofuels from algae. Elsevier, Amsterdam, pp 551–566
25. Wernet G, Bauer C, Steubing B et al (2016) The ecoinvent database version 3 (part I): overview and methodology. Int J Life Cycle Assess 21:1218–1230. https://doi.org/10.1007/s11367-016-1087-8
26. Mutel C (2017) Brightway: an open source framework for Life Cycle Assessment. J Open Source Softw 2:236. https://doi.org/10.21105/joss.00236
27. Lively RP, Sharma P, McCool BA et al (2015) Anthropogenic CO2 as a feedstock for the production of algal-based biofuels. Biofuels Bioprod Biorefin 9:72–81. https://doi.org/10.1002/bbb.1505
28. Geider R, La Roche J (2002) Redfield revisited: variability of C:N:P in marine microalgae and its biochemical basis. Eur J Phycol 37:1–17. https://doi.org/10.1017/S0967026201003456
29. Guieysse B, Béchet Q, Shilton A (2013) Variability and uncertainty in water demand and water footprint assessments of fresh algae cultivation based on case studies from five climatic regions. Biores Technol 128:317–323. https://doi.org/10.1016/j.biortech.2012.10.096
30. Weissman JC, Goebel RP, Benemann JR (1988) Photobioreactor design: Mixing, carbon utilization, and oxygen accumulation. Biotechnol Bioeng 31:336–344. https://doi.org/10.1002/bit.260310409
31. Muylaert K, Bastiaens L, Vandamme D, Gouveia L (2017) Harvesting of microalgae: overview of process options and their strengths and drawbacks. Microalgae-based biofuels and bioproducts. Elsevier, Amsterdam, pp 113–132
32. Ayoub GM, Koopman B (1986) Algal separation by the lime-seawater process. J (W Pollut Control Federation) 58:924–931
33. Schlesinger A, Eisenstadt D, Bar-Gil A et al (2012) Inexpensive non-toxic flocculation of microalgae contradicts theories; overcoming a major hurdle to bulk algal production. Biotechnol Adv 30:1023–1030. https://doi.org/10.1016/j.biotechadv.2012.01.011
34. Wu Z, Zhu Y, Huang W et al (2012) Evaluation of flocculation induced by pH increase for harvesting microalgae and reuse of flocculated medium. Biores Technol 110:496–502. https://doi.org/10.1016/j.biortech.2012.01.101
35. Chen L, Wang C, Wang W, Wei J (2013) Optimal conditions of different flocculation methods for harvesting Scenedesmus sp. cultivated in an open-pond system. Biores Technol 133:9–15. https://doi.org/10.1016/j.biortech.2013.01.071
36. Vandamme D, Beuckels A, Markou G et al (2015) Reversible flocculation of microalgae using magnesium hydroxide. Bioenergy Res 8:716–725. https://doi.org/10.1007/s12155-014-9554-1
37. Lama S, Muylaert K, Karki TB et al (2016) Flocculation properties of several microalgae and a cyanobacterium species during ferric chloride, chitosan and alkaline flocculation. Biores Technol 220:464–470. https://doi.org/10.1016/j.biortech.2016.08.080
38. Pérez L, Salgueiro JL, Maceiras R et al (2017) An effective method for harvesting of marine microalgae: pH induced flocculation. Biomass Bioenerg 97:20–26. https://doi.org/10.1016/j.biombioe.2016.12.010
39. Chen Y, Cheng JJ, Creamer KS (2008) Inhibition of anaerobic digestion process: a review. Biores Technol 99:4044–4064. https://doi.org/10.1016/j.biortech.2007.01.057
40. Willy A. Bachofen AG Maschinenfabrik (2012) Brochure: DYNO-MILL KD Horizontale Rührwerkskugelmühlen. https://www.wab-group.com/de/nassmahltechnik/produkte/ruehrwerkskugelmuehle/dyno-mill-kd/. Accessed 13 July 2021
41. Frank ED, Han J, Palou-Rivera I et al (2011) Life-cycle analysis of algal lipid fuels with the GREET model. https://greet.es.anl.gov/files/algal-lipid-fuels. Accessed 13 July 2021
42. Zschocke A (2014) Abschlussbericht zu dem Vorhaben Projekt BurnFAIR: Arbeitspakete 1.1 bis 1.4. https://www.dbfz.de/fileadmin/user_upload/Referenzen/Berichte/Abschlussbericht_BurnFAIR.pdf. Accessed 13 July 2021
43. Fazio S, Castellani V, Sala S, et al (2018) Supporting information to the characterisation factors of recommended EF life cycle impact assessment methods: new methods and differences with ILCD, EUR 28888 EN, Publications Office of the European Union, Luxembourg. https://doi.org/10.2760/671368. ISBN 978-92-79-76742-5, JRC109369
44. Ward A, Ball A, Lewis D (2015) Halophytic microalgae as a feedstock for anaerobic digestion. Algal Res 7:16–23. https://doi.org/10.1016/j.algal.2014.11.008
45. Collet P, Hélias A, Lardon L et al (2011) Life-cycle assessment of microalgae culture coupled to biogas production. Biores Technol 102:207–214. https://doi.org/10.1016/j.biortech.2010.06.154
46. Ward AJ, Lewis DM, Green FB (2014) Anaerobic digestion of algae biomass: a review. Algal Res 5:204–214. https://doi.org/10.1016/j.algal.2014.02.001