Environmental hotspots of lactic acid production systems

Ögmundarson, Ólafur; Sukumara, Sumesh; Laurent, Alexis; Fantke, Peter

Published in:
GCB Bioenergy

Link to article, DOI:
10.1111/gcbb.12652

Publication date:
2020

Document Version
Publisher's PDF, also known as Version of record

Link back to DTU Orbit

Citation (APA):
Ögmundarson, Ó., Sukumara, S., Laurent, A., & Fantke, P. (2020). Environmental hotspots of lactic acid production systems. GCB Bioenergy, 12(1), 19-38. https://doi.org/10.1111/gcbb.12652

General rights
Copyright and moral rights for the publications made accessible in the public portal are retained by the authors and/or other copyright owners and it is a condition of accessing publications that users recognise and abide by the legal requirements associated with these rights.

- Users may download and print one copy of any publication from the public portal for the purpose of private study or research.
- You may not further distribute the material or use it for any profit-making activity or commercial gain
- You may freely distribute the URL identifying the publication in the public portal

If you believe that this document breaches copyright please contact us providing details, and we will remove access to the work immediately and investigate your claim.
Environmental hotspots of lactic acid production systems

Ólafur Ögmundarson1,2 | Sumesh Sukumara2 | Alexis Laurent1 | Peter Fantke1

1Quantitative Sustainability Assessment, Department of Technology, Management and Economics, Technical University of Denmark, Kgs. Lyngby, Denmark
2The Novo Nordisk Foundation Center for Biosustainability, Technical University of Denmark, Kgs. Lyngby, Denmark

Correspondence
Peter Fantke, Quantitative Sustainability Assessment, Department of Technology, Management and Economics, Technical University of Denmark, Produktionstorvet 424, 2800 Kgs. Lyngby, Denmark.
Email: pefan@dtu.dk

Abstract
Using selected bio-based feedstocks as alternative to fossil resources for producing biochemicals and derived materials is increasingly considered an important goal of a viable bioeconomy worldwide. However, to ensure that using bio-based feedstocks is aligned with the global sustainability agenda, impacts along the entire life cycle of biochemical production systems need to be evaluated. This will help to identify those processes and technologies, which should be targeted for optimizing overall environmental sustainability performance. To address this need, we quantify environmental impacts of biochemical production using distinct bio-based feedstocks, and discuss the potential for reducing impact hotspots via process optimization. Lactic acid (LA) was used as an example biochemical derived from corn, corn stover, and macroalgae (Laminaria sp.) as feedstocks of different technological maturity. We used environmental life cycle assessment (LCA), a standardized methodology, considering the full life cycle of the analyzed biochemical production systems and a broad range of environmental impact indicators. Across production systems, feedstock production and biorefinery processes dominate life cycle impact profiles, with choice in energy mix and biomass processing as main influencing aspects. Results show that uncertainty decreases with increasing technological maturity. When using Laminaria sp. (least mature among selected feedstocks), impacts are mainly driven by energy utilities (up to 86%) due to biomass drying. This suggests to focus on optimizing or avoiding this process for significantly increasing environmental sustainability of Laminaria sp.-based LA production. Our results demonstrate that applying LCA is useful for identifying environmental impact hotspots at an earlier stage of technological development across biochemical production systems. With that, our approach contributes to improving the environmental sustainability of future biochemical production as part of moving toward a viable bioeconomy worldwide.

KEYWORDS
biochemicals, corn, corn stover, environmental sustainability, hotspots, lactic acid, Laminaria sp., life cycle assessment, uncertainty
1 | INTRODUCTION

Increased utilization of renewable resources for biochemical production helps to potentially reducing greenhouse gas (GHG) emissions by substituting petrochemicals with bio-based chemicals. Since 2001, subsidies, campaigns, mandates, and bans have encouraged the emergence of biochemical-based materials on the global market (OECD, 2014). Producing bioplastics is the main biochemical application (Society of Chemical Industry, 2016), although bioplastics production currently only contributes with ~1% to total global plastics production. The market share of bioplastics, however, is foreseen to increase rapidly, and in January 2018, the European Commission published a new European Strategy for Plastics in a Circular Economy (EC, 2018) including a “Vision for Europe’s new plastic Economy” until 2030. Main focus of this strategy is on tackling environmental challenges related to conventional plastics along all life cycle stages from resources extraction to end-of-life (EoL) handling, and different ways of producing more environmentally sustainable plastics are proposed (EC, 2018). Worldwide production of bio-based building block chemicals and bio-based polymers is, hence, foreseen to increase, by about 26% and 10%, respectively, until 2022 (Aeschelmann et al., 2016).

Moving from fossil-based to bio-based chemicals, however, also comes with challenges. These range from high production costs and difficulties in matching performance properties of fossil-based chemicals, to questionable improvements in environmental sustainability profiles of biochemicals and derived products (Hottie, Bilec, & Landis, 2013; Ögmundarson, Herrgard, Forster, Hauschild, & Fantke, 2019; Zhu, Romain, & Williams, 2016). Yet, there are also positive examples, where the production of bio-based lactic acid (LA) has shown to perform environmentally better than functionally equivalent fossil-based products (Vink & Davies, 2015). This is, for example, the case when system boundaries are drawn at factory gates. However, cultural and geographical differences in waste handling can further influence conclusions, depending on—among other aspects—whether waste is incinerated with or without energy recovery, landfilled, or composted. It furthermore remains unclear, where the main environmental impacts occur along the life cycle of each selected feedstock production system with its specific technological maturity level (Ögmundarson et al., 2019).

As a consequence, existing claims emphasizing a favorable environmental sustainability performance of biochemicals and derived products are often unjustified and are in many cases exclusively based on reduced GHG emissions in specific production processes (Hottle et al., 2013). Failing to assess all life cycle stages can, in fact, lead to large environmental impacts (i.e., so-called hotspots) that remain unconsidered. Focusing only on selected life cycle stages and processes can additionally introduce a shifting of the burden, for example, from reducing GHG emissions during feedstock processing to potentially larger impacts during waste handling (Laurent, Bakas, et al., 2014; Laurent, Clavreul, et al., 2014). To avoid such “burden shifting” when assessing and optimizing the environmental performance of biochemical production systems, it was earlier proposed to include at least land-use and water use, eutrophication, and ecotoxicity impacts, in addition to commonly assessing global warming (Ögmundarson et al., 2019).

Currently, biochemicals and related biopolymers are mostly derived from agricultural crops (first generation feedstock; Yang, Choi, Park, & Kim, 2015) with a high technology readiness level (TRL; Taylor et al., 2015) of 8 (first-of-a-kind commercial system) or 9 (full commercial application; see Table 1). Biochemicals that are derived mostly from non-agricultural commodities, such as lignocellulosic biomass and wood (second generation feedstock; Yang et al., 2015), have yet to reach high commercialization levels. This is currently mainly limited by high costs associated with the conversion of these feedstocks into fermentable sugars (Bardhan, Gupta, Gorman, & Haider, 2015). Due to technological challenges associated with utilizing second generation feedstocks, for example, during the pretreatment phase of the biomass (Biddy et al., 2016; Chen & Qiu, 2010) and related economic challenges of biochemical production from second generation biomass (Chandel, Garlapati, Singh, Antunes, & da Silva, 2018; Lynd et al., 2017), alternative feedstock sources are increasingly receiving attention. These feedstocks include engineered crops, algae, and urban residues, such as household waste (third generation feedstock; Yang et al., 2015).

In support of optimizing the environmental sustainability performance of biochemical production, it is therefore essential that we understand, where along the life cycle of biochemicals environmental impact hotspots occur for each selected feedstock, and how any changes in inputs and outputs of the production system can influence such hotspots. In such effort, environmental life cycle assessment (LCA) can be a useful evaluation tool. LCA is an ISO-standardized and widely applied methodology for assessing the environmental performance of different product systems, and is thus well suited for quantifying impact hotspots along the entire life cycle of biochemical production systems using different feedstocks (Zhu et al., 2016).

Existing LCA studies performed on biochemicals show important trends and limitations. When biochemicals and derived products are compared to their functionally equivalent fossil-based products, LCA results indicate that with respect to certain impact categories, biochemicals often perform better (e.g., global warming: Hanes, Cruze, Goel, & Bakshi, 2015; Madival, Auras, Singh, & Narayan, 2009;
Patel et al., 2018; Vink & Davies, 2015; human toxicity: Landis, 2010; Papong et al., 2014; acidification: Hanes et al., 2015; Landis, 2010), whereas they may perform worse than their fossil-based counterparts in other impact categories (e.g., particulate matter exposure: Landis, 2010; Madival et al., 2009; Smidt et al., 2015; ecotoxicity: Landis, 2010; Smidt et al., 2015; van der Harst, Potting, & Kroeze, 2014; eutrophication: Breedveld et al., 2014; Gironi & Piemonte, 2011; Papong et al., 2014; Urban & Bakshi, 2009; land-use: Daful, Haigh, Vaskan, & Görgens, 2016; Patel et al., 2006). However, several studies focus exclusively on assessing global warming impacts and do not include other relevant environmental impacts, thus overlooking burden shifting from one environmental problem to another. In addition, there is a large variation in the coverage of life cycle stages across these studies, leading to potentially unaddressed burden shifting between life cycle stages. These trends and limitations suggest a strong need for the development of a more comprehensive overview of the differences in environmental performance profiles across feedstocks, life cycle stages, and impact categories, to identify sustainability related optimization potentials of biochemical production from conventional and emerging bio-based feedstocks (Ögmundarson et al., 2019).

To address this need, we aim in the present study (a) to consistently define biochemical production systems across selected bio-based feedstocks, with a focus on LA as an important building block chemical (European ioplastics, 2017; Wee, Kim, & Ryu, 2006); (b) to characterize the environmental performance of the selected LA production systems using a full LCA approach; and (c) to discuss related environmental impact hotspots and their potential drivers for each production system. We finally show how hotspot results can help to better inform technology system design, identify optimization potentials of future processes, and operationalize decision support. Although impact hotspots cannot be directly compared across the considered feedstock systems due to their different TRLs, our results provide an important starting point to focus future improvement efforts within each of the different biochemical production systems.

2 | MATERIALS AND METHODS

Our assessment follows the formal guidelines provided in the International Reference Life Cycle Data System (ILCD) Handbook for LCA practice (EC, 2010), which is compliant
with the ISO14044:2006 standard (ISO, 2006b), and the EN 16760:2015 standard specified for assessing bio-based products (CEN, 2015).

2.1 | Scope definition: Lactic acid life cycle system description

We consider the entire life cycle of different LA production systems, that is, from cradle to grave. This includes (a) the harvesting of renewable biomass (i.e., corn, corn stover, and *Laminaria sp.*); (b) the biorefinery process; (c) the polymerization of LA to polylactic acid (PLA), considered to be one of the most important LA products (European Bioplastics, 2017; OECD, 2014), and (d) EoL scenarios, despite these being sometimes considered unfeasible (Vink, Davies, & Kolstad, 2010). The functional unit, which reflects a system’s reference function and forms the basis of its environmental performance assessment (ISO, 2006a), is defined as “the utilization of 1 kg of LA, with 99.9% purity, for household product packaging application in the United States” (US). Using the monomer as functional unit emphasizes the different biorefinery setups based on the considered feedstocks. Since we assume the same polymerization process across monomers produced from the different feedstocks, our functional unit reflects the focus on the distinct monomer production systems. Household product packaging application thereby refers to food packaging, which follows the waste streams of household waste. A variety of production systems is considered, each with its specific feedstock to provide the LA specified in the functional unit (see Figure 1).

The default geographical location for feedstock harvesting, chemical production, and EoL is chosen to be the US (from cradle to grave), because nowadays this is where most LA and PLA are being produced. We assume that future LA production systems using *Laminaria sp.* (third generation feedstock) will most likely be developed there as well, and test this assumption in a scenario analysis using China (CN) and Iceland (ICE) as alternative geographical locations. When scaling up the production of macroalgae-based biochemicals, a combination of mowing and farming is most likely required. The scenario analysis of different geographical locations will highlight differences in energy and heat production between the three countries and the potential effects of these differences on environmental performance results. For simplicity, we assume that the biorefinery is in all scenarios located in direct proximity to the respective feedstock source.

Figure 1 illustrates which processes are included in our LCA (i.e., within the defined system boundaries) of LA production. For all three assessed feedstock generations, the same life cycle stages are assessed. Differences in modeling of the biorefinery stage are due to distinct compositions of the assessed biomasses. For the first generation biomass (corn), pretreatment is not required; however, for...
the second (corn stover) and third (Laminaria sp.) generation biomasses, both chemical pretreatment and separation are required to enable fermentation (Davis et al., 2013; Humbird et al., 2011; Tomás-Grasa, Ögmundarson, Gavala, & Sukumara, n.d.).

Inputs (e.g., use of water, energy utilities, and chemicals) and outputs (waste streams and environmental emissions) are assessed, as well as avoided production of energy (through energy conversion from unfermented solids), and avoided production of virgin materials (through the recycling of plastics from household waste), via system expansion, a methodological choice to catch the multifunctionality of the products. Among emerging feedstocks, brown algae is the single largest macroalgae resource and is considered to be one of the most likely contenders for the production of biofuels and chemicals, being particularly attractive for fermentation-derived products due to their high content of carbohydrates and near or full absence of lignin (whereas at least some red algae contain lignin), and being available in all three studied locations (FAO, 2019; Jung, Lim, Kim, & Park, 2013). For our present study, we have chosen to focus on three different Laminaria sp., because they are found in all three different geographical locations assessed in this study, namely Laminaria digitata, Laminaria saccharina, and Laminaria japonica, and their average chemical composition is presented in ESI.3.

Table 2. Main data sources for the life cycle assessment of lactic acid derived from selected bio-based feedstocks (for additional details, see Sections S.1–S.4 in Supporting Information)

| Life cycle stage       | Corn feedstock | Corn stover feedstock | Laminaria sp. feedstock |
|------------------------|----------------|------------------------|-------------------------|
|                        | Foreground system | Background system | Foreground system | Background system | Foreground system | Background system |
| Biomass production and harvesting | Ecoinvent 3a | Ecoinvent 3, including iLUC4 | Fertilizer value of corn stoverb | Ecoinvent 3a | Onshore collection of Laminaria sp.c | Ecoinvent 3a |
| Biorefinery           | Techno-economic assessment (see ES and ES) | Ecoinvent 3a | Techno-economic assessment (see ES and ES) | Ecoinvent 3a | Techno-economic assessment (see ES and ES) | Ecoinvent 3a |
| Polymerization        | IHS Markit databased | Ecoinvent 3a | IHS Markit database | Ecoinvent 3a | IHS Markit database | Ecoinvent 3a |
| End-of-life           | Country-specific handling of household wastee | Ecoinvent 3a | Country-specific handling of household waste | Ecoinvent 3a | Country-specific handling of household waste | Ecoinvent 3a |

Wernet et al. (2016).
Karlen, Kovar, and Birrell (2015).
Asco Harvester (2018).
IHS Markit: Leading Source of Critical Information (2018).
OECD (2018).

The systems reflect US conditions (referred to as the “base case”) throughout the life cycle of LA production and across assessed feedstocks (see Table 2). Due to current technological limitations, not all carbons can be fermented, such as alginate. They nevertheless constitute a large proportion of the carbon content of the biomass. It is therefore interesting to assess what the environmental benefits would be, if the alginate was fermentable. Fermenting alginate is hence included as additional scenario in our study across geographical scenarios.

2.2 | System modeling and data collection

The assessed biorefinery scenarios for producing LA from the three different feedstock generations are differentiated by pretreatment practices of the biomass (see Figure 1), where chemical pretreatment and separation are included in the second and third generation processes. Physical pretreatment, fermentation, and purification steps are assumed common for all three feedstock generations. For the third generation biomass, intensive air-drying is chosen as a drying method due to current scalability issues of other methods, like freeze-drying and mild-air drying (Adams, Bleathman, Thomas, & Gallagher, 2017). Drying was included as a pretreatment method for three different reasons. First, results from small-scale experiments at The Novo Nordisk Foundation Center for Biosustainability utilizing Laminaria sp. for LA production showed that to minimize risk of external contamination during fermentation, the best way was to dry the biomass. Hence, drying was assumed to be an effective pretreatment method. Second, given that drying along with the entire biorefinery process is very energy-intensive, considering geographical differences in the availability of energy sources for biorefineries is crucial. Therefore, we assessed three distinct locations, demonstrating that biomass drying is not favorable without easy access to renewable energy sources. Third, transportation of biomass is required,
and in case of *Laminaria* sp., the high water content needs to be reduced to avoid the transportation of large amounts of water.

To secure technological representativeness of the biorefinery process (conversion of the feedstock to monomers), we used Aspen Plus® v.8.8 to simulate both the processes in a techno-economic assessment and also create the relevant inventories. For the production of LA from corn, this process is modeled based on data available from process design of commercial LA production (Carlson & Peters, 1997). The same approach was followed for modeling LA production from corn stover (Humbird et al., 2011). For LA production from *Laminaria* sp., the fermentation was modeled like the fermentation process for corn because of lack of existing commercial processes for producing commodity chemicals from *Laminaria* sp. (Tomás-Grasa et al., n.d.).

Following the guidelines of the ILCD Handbook, an attributional model with use of system expansion and average life cycle inventory (LCI) data was selected (EC, 2010; Wernet et al., 2016). By applying Aspen Plus®, we can populate data that are otherwise not available, due to a lack of industrial data for the production of commodity chemicals (Dunn, Adom, Sather, Han, & Snyder, 2015). Furthermore, when assessing future uses of emerging bio-based feedstocks, such as *Laminaria* sp., Aspen Plus® populates the necessary inventory based on user-defined assumptions. The calculated mass flows of the biorefinery from Aspen were then used to populate the mass flows for the annual LA production of 110,000 metric tons. These calculations were subsequently used to identify needs regarding amounts of biomass per feedstock process for either harvesting or cultivation.

For the polymerization process, the IHS Markit© database (IHS Markit: Leading Source of Critical Information, 2018) was used to identify mass flows, while for EoL processes, the average distribution of waste in 2014 in the US was chosen (United States Environmental Protection Agency [EPA], 2018). For all background systems, pre-existing processes from Ecoinvent v3 were applied (Wernet et al., 2016) and refined for each country whenever country-specific background information was available (see Table 2 and Sections S.1–S.3 in Supporting Information, for additional inventory details).

### 2.3 Life cycle impact assessment

The life cycle impact assessment was performed with SimaPro v.8.5. All impact categories included in the ReCiPe...
2016 methodology (hierarchist perspective, v.1.02) were assessed at both midpoint and damage (i.e., endpoint) levels. Midpoint impact indicators are defined somewhere along the cause–effect chain, while endpoint level impacts are translated into damages that can be aggregated into different areas of protection (Figure 2). Translating the various midpoint indicators into damages is a fundamental step for getting an overview of the magnitude of possible trade-offs across impact categories.

At midpoint level, considered impact categories include global warming, stratospheric ozone depletion, ionizing radiation, ozone formation, human toxicity, fine particulate matter impacts, tropospheric ozone formation, acidification, eutrophication, ecotoxicity, land-use, resource depletion, and water consumption. At damage level, impacts are translated into damages on human health, ecosystem quality, and natural resources (Huijbregts et al., 2017).

Furthermore, as part of our base case for LA production from corn and because of its particular relevance for crop-based feedstocks, we have also included the biophysical indirect land-use change (iLUC) attributional model (Schmidt, Weidema, & Brandão, 2015). The iLUC model contributes to LCA results through specific LCI flows. This includes intensification of already-farmed land (to meet increased demand for crops), which contributes to relevant environmental impacts via increased nitrogen fertilizer use, and transformation from secondary forests to croplands contributing to CO$_2$ emissions (Schmidt et al., 2015; see Figure 2). Assessing iLUC is thus modeled as inputs from the technological system as potential net primary production (NPP0) kg carbon per kg of corn (Schmidt et al., 2015).

Impact hotspots in our analysis are defined as most significant impacts (e.g., after weighting) in terms of life cycle process and impact magnitude. Changes in hotspots across impact categories cannot be directly compared at midpoint level, as they are expressed in different units. In order to compare hotspots across impact categories, results are aggregated into damages affecting the three considered areas of protection, where all contributing impact categories are expressed in a common unit per area of protection.

### 2.4 Scenario definitions

We defined four scenarios to evaluate the sensitivity of the results to changes in the system modeling and settings, as summarized in Table 3. When using Laminaria sp. for biorefinery production, only certain parts of the available carbohydrates are usable for fermentation purposes, namely cellulose, laminarin, and mannitol. There is currently no technology to ferment alginate, which accounts for about 30% of the carbohydrates composition of Laminaria sp. (Schiener, Black, Stanley, & Green, 2015). To better understand what the environmental gains could be, if alginate was fermentable, a first sensitivity scenario was defined to account for this aspect. In a second sensitivity scenario, we explored how the geographical location and local energy mixes influence the environmental impacts related to energy utilities. In a third scenario, we assumed onshore collection of Laminaria sp. with a seaweed mower as a way of acquiring the Laminaria sp. biomass, thereby evaluating the effects of a cultivation technique of the same biomass on impact results. Finally, we analyzed in a fourth sensitivity scenario, if transportation and EoL processes in a different geographical area, Germany, was tested, where recycling percentages are higher than in the US (results presented in ES).

### 2.5 Uncertainty assessment

To address uncertainty of our assessment results, we conducted a Monte Carlo analysis. For all life cycle stages of the three selected feedstock generations (inputs and outputs), we estimated the uncertainty using the Pedigree matrix approach (Owsianik, Ryberg, Renz, Hitzl, &
Hauschild, 2016), with the exception that the maize grain process was adapted from the ecoinvent database, which already had geometric standard deviations assigned to the respective flows. Moreover, uncertainty of input data was assessed based on predefined data quality criteria (reliability, completeness, temporal correlation, geographical correlation, further technological correlation, and basic uncertainty factor), while uncertainty factors were calculated using squared geometric standard deviations for foreground processes. For background processes, pre-assigned geometric standard deviations from ecoinvent were used. For the Monte Carlo simulations, 10,000 iterations were performed.

3 | RESULTS

3.1 | Relevance of life cycle stages across selected feedstock generations

The presentation of results in this section is in line with the general definition of life cycle stages in this study, namely (a) harvesting of renewable biomass; (b) the biorefinery process; (c) the polymerization; and (d) EoL scenarios. However, within each life cycle stage (e.g., biorefinery stage), different processes contribute differently to impact distributions per feedstock generation.

When producing LA from corn, the production of biomass and the biorefinery process are the two main contributing life cycle stages dominating impacts related to global warming, stratospheric ozone depletion, freshwater eutrophication, marine eutrophication, human carcinogenic toxicity, land-use, and water consumption (Table 4a). This is consistent with earlier findings, where these two life cycle stages are found to be the main drivers of overall environmental performance for LA (Gironi & Piemonte, 2011; Landis, 2010; Madival et al., 2009) and succinic acid (Breedveld et al., 2014; Smidt et al., 2015).

Indirect land-use change contributes mostly to the impacts of biomass production, for example, causing a 35% increase in marine eutrophication impacts (for details regarding the calculation of iLUC and its contributions to other impacts see ESI.6), which demonstrates the importance of assessing iLUC for first generation feedstocks. To the best of our knowledge, this is the first time that iLUC of LA production from corn has been evaluated. When examining the different hotspots of LA production systems from corn (Table 4a), impacts associated with global warming, ozone formation, fine particulate matter impacts, terrestrial acidification, terrestrial ecotoxicity, mineral resource scarcity, and fossil resources scarcity are all driven by high contributions from the biorefinery stage. The process with the highest contribution to biorefinery-related impacts is the use of triethanolamine in the purification of LA production, followed by the treatment of refinery sludge (see LCI for first generation process in ESI.1). The purification process has also been identified earlier as substantial contributor to impacts of production systems for succinic acid (Morales et al., 2016) and life cycle (Helmes et al., 2018).

When producing LA from corn stover, the biorefinery process becomes the dominant life cycle stage across impact categories (Table 4b). High energy utility demand is the mainly contributing process, with triethanolamine use in the purification of the LA and handling of refinery sludge as additional significant contributors. Another reason for the high freshwater eutrophication impacts is the need for increased fertilizer use, in order to substitute the nutrients that the corn stover would otherwise have provided to the corn fields in cases where it is not used as feedstock (see LCI for second generation process in ESI.2). This highlights the biorefining process as an environmental hotspot for focus in future optimization efforts, while earlier studies assessing the environmental impacts of LA production from corn stover did not specify any dominating life cycle stage (Adom & Dunn, 2017).

With regard to LA production from Laminaria sp. (Table 4c), high utility inputs for drying in the biomass process and intensive energy usage in the biorefinery both result in environmental hotspots that are of similar magnitude. This is not the case, however, for the impact categories of marine eutrophication, freshwater ecotoxicity, marine ecotoxicity, and human non-carcinogenic toxicity, because of the difference in the composition of the energy mix used in the LCA model (see ESI.3). Process inputs, such as the use of a neutralizing agent, phosphate fertilizer and calcium carbonate, in addition to energy utilities in the biorefinery, are all inputs into the process flow that cause environmental hotspots. While our study is to the best of our knowledge the first assessing life cycle environmental impacts of LA production from Laminaria sp., previous studies have already highlighted these energy-intensive processes as important contributors to environmental impact profiles (Milledge & Harvey, 2016; Seghetta, Hou, Bastianoni, Bjerre, & Thomsen, 2016). In Table 4d, we present the results for the scenario without biomass drying, indicating potential environmental gains when drying and related energy use can be avoided.

A general trend in our results across selected feedstock generations is that utilities are the dominating contributors (energy inputs), including energy for drying of biomass, steam and electricity use, ultimately accounting for up to 86% of total environmental impacts. Energy utility inputs are also the largest contributor to most other impact categories assessed. This is especially the case for LA production from Laminaria sp., which is the most energy-intensive across the three considered feedstock generations. This is partly because this production system is currently the least technologically optimized, and because it may be the least energy efficient production system when considering the respective TRL. This is also reflected in the uncertainty ranges presented...
## TABLE 4  
Environmental impact results for lactic acid production from corn (a), corn stover (b), and *Laminaria* sp. (c), including 95% confidence interval ranges (in parentheses) and hotspots (expressed as % contribution of life cycle stages, BM, biomass production; BR, biorefinery; PM, polymerization; and EoL, end-of-life treatment). Negative environmental impacts are related to crediting use of unfermented solids for energy production in the biorefinery stage and crediting for avoided production of virgin material when recycling waste materials.

| Impact categories                                      | Unit       | Total results (2.5th–97.5th%) | Life cycle stages |
|--------------------------------------------------------|------------|-------------------------------|------------------|
|                                                        |            |                               | BM (%) | BR (%) | PM (%) | EoL (%) |
| (a) Lactic acid from corn (TRL 8–9)                    |            |                               |        |        |        |        |
| Global warming                                         | kg CO₂ eq  | 4.2 (1.3−4.8)                 | 47.5   | 59.1   | 1.8    | −8.5  |
| Stratospheric ozone depletion                           | kg CFC11 eq| 2 × 10⁻³ (8.4 × 10⁻⁰⁸−1.6 × 10⁻⁰⁵) | 95.8   | 3.4    | 0.1    | 0.6   |
| Ionizing radiation                                     | kBq Co-60 eq| −0.22 (−1.2 to 0.015)         | 14.9    | −121.6 | 3.6    | 3.1   |
| Ozone formation, human health                           | kg NO₂ eq  | 7.1 × 10⁻³ (4.4 × 10⁻⁹−8.9 × 10⁻⁹) | 37.7   | 75.8   | 2.4    | −15.8 |
| Fine particulate matter exposure                        | kg PM₂.⁵ eq| 1.8 × 10⁻² (5.8 × 10⁻³−3.6 × 10⁻²) | 18.2   | 82.8   | 1.0    | −1.9  |
| Ozone formation, terrestrial ecosystems                 | kg NO₂ eq  | 7.3 × 10⁻³ (4.5 × 10⁻³−9.1 × 10⁻³) | 37.6   | 77.3   | 2.7    | −17.5 |
| Terrestrial acidification                               | kg SO₂ eq  | 0.077 (0.037−0.13)            | 20.3   | 81.2   | 0.4    | −1.8  |
| Freshwater eutrophication                              | kg P eq    | 5.2 × 10⁻⁵ (−3.1 × 10⁻⁵ to 1.7 × 10⁻⁵) | 978.6  | −1,057.4 | 120.0 | 58.8  |
| Marine eutrophication                                  | kg N eq    | 5.6 × 10⁻³ (1.8 × 10⁻⁵−3.5 × 10⁻⁵) | 84.8   | 9.3    | 0.1    | 5.9   |
| Terrestrial ecotoxicity                                 | kg 1,4-DCB | 12 (7.1–19)                   | 20.0   | 72.8   | 10.8   | −3.6  |
| Freshwater ecotoxicity                                  | kg 1,4-DCB | 0.12 (−0.055 to 0.35)         | 25.1   | −6.1   | 3.6    | 77.4  |
| Marine ecotoxicity                                      | kg 1,4-DCB | 0.17 (−0.064 to 0.49)         | 19.8   | 2.9    | 4.1    | 73.2  |
| Human carcinogenic toxicity                             | kg 1,4-DCB | 0.048 (−0.19 to 0.24)         | 73.7   | 34.4   | 10.0   | −18.1 |
| Human non-carcinogenic toxicity                         | kg 1,4-DCB | 4.3 (−0.041 to 13)            | 7.6    | 37.7   | 4.9    | 49.8  |
| Land use                                                | m² a crop eq| 1.4 (0.99–1.9)                | 93.5   | 6.5    | 0.1    | 0.0   |
| Mineral resource scarcity                               | kg Cu eq   | 0.012 (7.8 × 10⁻⁰³−1.8 × 10⁻⁰³) | 33.6   | 59.0   | 11.8   | −4.4  |
| Fossil resource scarcity                                | kg oil eq  | 0.65 (6.4 × 10⁻⁰³−1.1)         | 36.6   | 137.8  | 2.7    | −77.1 |
| Water consumption                                       | m³         | 0.43 (2.0 to 2.8)             | 75.1   | 26.2   | 0.6    | −1.9  |
| (b) Lactic acid from corn stover (TRL 4–5)              |            |                               |        |        |        |        |
| Global warming                                         | kg CO₂ eq  | 7.9 (6.0–9.2)                 | 36.7   | 66.8   | 1.0    | −4.5  |
| Stratospheric ozone depletion                           | kg CFC11 eq| 3.3 × 10⁻⁶ (2.6 × 10⁻⁶−4.0 × 10⁻⁵) | 40.7   | 54.7   | 0.9    | 3.7   |
| Ionizing radiation                                     | kBq Co-60 eq| 0.3 (0.038–1.3)              | 0.0    | 95.2   | 2.6    | 2.2   |
| Ozone formation, human health                           | kg NO₂ eq  | 1.2 × 10⁻² (9.7 × 10⁻³−1.4 × 10⁻²) | 47.7   | 60.4   | 1.4    | −9.5  |
| Fine particulate matter exposure                        | kg PM₂.⁵ eq| 0.024 (0.015–0.035)           | 22.0    | 78.7   | 0.7    | −1.4  |
| Ozone formation, terrestrial ecosystems                 | kg NO₂ eq  | 1.2 × 10⁻² (9.9 × 10⁻³−1.5 × 10⁻²) | 47.4   | 61.5   | 1.6    | −10.5 |
| Terrestrial acidification                               | kg SO₂ eq  | 0.064 (0.038–0.11)            | 16.2   | 85.5   | 0.4    | −2.2  |
| Freshwater eutrophication                              | kg P eq    | 3.5 × 10⁻³ (1.3 × 10⁻³−6.6 × 10⁻³) | 35.3   | 62.0   | 1.8    | 0.9   |

(Continues)
| Impact categories                      | Unit         | Total results (2.5th-97.5th%) | Life cycle stages |
|----------------------------------------|--------------|--------------------------------|-------------------|
|                                        |              |                                | BM (%)           |
| Marine eutrophication                  | kg N eq      | $1.3 \times 10^{-1} (8.7 \times 10^{-4} - 1.9 \times 10^{-3})$ | 12.1             |
| Terrestrial ecotoxicity                | kg 1.4-DCB   | 22 (14–37)                     | 42.1             |
| Freshwater ecotoxicity                 | kg 1.4-DCB   | 0.33 (0.16–0.69)               | 25.8             |
| Marine ecotoxicity                     | kg 1.4-DCB   | 0.46 (0.22–0.95)               | 26.3             |
| Human carcinogenic toxicity            | kg 1.4-DCB   | 0.29 (0.095–0.82)              | 38.3             |
| Human non-carcinogenic toxicity        | kg 1.4-DCB   | 9.5 (3.7–23)                   | 27.0             |
| Land use                               | m² a crop eq | 0.17 (0.17–0.33)               | 0.0              |
| Mineral resource scarcity              | kg Cu eq     | 0.011 (0.011–0.031)            | 0.0              |
| Fossil resource scarcity               | kg oil eq    | 1.8 (1.2–2.2)                  | 39.5             |
| Water consumption                      | m³           | 0.15 (–3.6 to 3.2)             | 13.5             |

(c) Lactic acid from macroalgae with drying (TRL 2–3)

| Impact categories                      | Unit         | Total results (2.5th-97.5th%) | Life cycle stages |
|----------------------------------------|--------------|--------------------------------|-------------------|
|                                        |              |                                | BM (%)           |
| Global warming                         | kg CO₂ eq    | 11 (7.14–15.2)                 | 50.9             |
| Stratospheric ozone depletion          | kg CFC11 eq  | $5.8 \times 10^{-6} (3.6 \times 10^{-6} – 1.0 \times 10^{-5})$ | 51.4             |
| Ionizing radiation                     | kBq Co-60 eq | 0.27 (–0.13 to 2.7)            | 45.4             |
| Ozone formation, human health          | kg NO₂ eq    | 0.015 (0.011–0.022)            | 54.5             |
| Fine particulate matter exposure       | kg PM₂.₅ eq | 0.02 (8.6 × 10⁻⁷–3.2 × 10⁻⁵)   | 50.1             |
| Ozone formation, terrestrial ecosystems | kg NO₂ eq    | 0.016 (0.011–0.022)            | 54.9             |
| Terrestrial acidification              | kg SO₂ eq    | 0.045 (0.032–0.061)            | 51.1             |
| Freshwater eutrophication              | kg P eq      | $3.0 \times 10^{-3} (9.0 \times 10^{-5} – 1.0 \times 10^{-2})$ | 43.6             |
| Marine eutrophication                  | kg N eq      | $1.1 \times 10^{-3} (7.3 \times 10^{-4} – 2.0 \times 10^{-3})$ | 9.2              |
| Terrestrial ecotoxicity                | kg 1.4-DCB   | 29 (17–73)                     | 41.9             |
| Freshwater ecotoxicity                 | kg 1.4-DCB   | 0.3 (0.078–0.74)               | 28.3             |
| Marine ecotoxicity                     | kg 1.4-DCB   | 0.42 (0.13–1)                  | 28.8             |
| Human carcinogenic toxicity            | kg 1.4-DCB   | 0.26 (0.096–1.2)               | 55.6             |
| Human non-carcinogenic toxicity        | kg 1.4-DCB   | 7.1 (2.9–22)                   | 24.7             |
| Land use                               | m² a crop eq | 1.1 (0.74–1.4)                 | 86.4             |
| Mineral resource scarcity              | kg Cu eq     | 0.022 (0.013–0.047)            | 51.7             |
| Fossil resource scarcity               | kg oil eq    | 3.1 (1.9–4.6)                  | 61.6             |
| Water consumption                      | m³           | 0.46 (–2.64 to 3.2)            | 43.0             |

(Continues)
TABLE 4  (Continued)

| Impact categories                                      | Unit       | Total results (2.5th–97.5th%) | Life cycle stages |
|---------------------------------------------------------|------------|--------------------------------|-------------------|
|                                                         |            |                                | BM (%)           | BR (%) | PM (%) | EoL (%) |
| (d) Lactic acid from macroalgae without drying (TRL 2–3)|            |                                 |                  |        |        |         |
| Global warming                                          | kg CO₂ eq  | 5.7 (0.22–11.4)                 | 2.6              | 102.4  | 1.4    | −6.4    |
| Stratospheric ozone depletion                           | kg CFC11 eq| 3.3 × 10⁻⁶ (9.9 × 10⁻⁷–6.4 × 10⁻⁶)| 2.8              | 92.2   | 1.0    | 4.0     |
| Ionizing radiation                                      | kBq Co-60 eq| 0.28 (−1.12 to 2.44)            | 1.2              | 93.7   | 2.7    | 2.3     |
| Ozone formation, human health                           | kg NO₂ eq  | 7.3 × 10⁻³ (1.2 × 10⁻³–1.4 × 10⁻²)| 2.0              | 111.3  | 2.3    | −15.6   |
| Fine particulate matter exposure                        | kg PM₂.⁵ eq| 0.01 (−7.8 × 10⁻³–2.8 × 10⁻³)   | 1.0              | 100.7  | 1.8    | −3.4    |
| Ozone formation, terrestrial ecosystems                  | kg NO₀ eq  | 7.4 × 10⁻³ (1.1 × 10⁻³–1.4 × 10⁻²)| 2.1              | 112.7  | 2.7    | −17.4   |
| Terrestrial acidification                               | kg SO₂ eq  | 0.023 (6.9 × 10⁻³–4.1 × 10⁻²)   | 1.3              | 103.6  | 1.2    | −6.2    |
| Freshwater eutrophication                               | kg P eq    | 2 × 10⁻³ (−3.8 × 10⁻³–9.5 × 10⁻³)| 0.2              | 95.2   | 3.1    | 1.5     |
| Marine eutrophication                                   | kg N eq    | 1.1 × 10⁻³ (4.8 × 10⁻⁴–2 × 10⁻³) | 0.0              | 69.3   | 0.3    | 30.4    |
| Terrestrial ecotoxicity                                 | kg 1,4-DCB | 19 (9–40)                       | 0.1              | 95.3   | 6.9    | −2.3    |
| Freshwater ecotoxicity                                  | kg 1,4-DCB | 0.24 (−0.11 to 0.69)            | 0.1              | 58.9   | 1.8    | 39.2    |
| Marine ecotoxicity                                      | kg 1,4-DCB | 0.33 (−0.13 to 0.93)            | 0.2              | 60.2   | 2.1    | 37.5    |
| Human carcinogenic toxicity                              | kg 1,4-DCB | 0.16 (−0.14 to 0.78)            | 0.5              | 102.0  | 3.0    | −5.5    |
| Human non-carcinogenic toxicity                         | kg 1,4-DCB | 6.3 (0.3–18.1)                  | 0.2              | 62.0   | 3.4    | 34.4    |
| Land use                                                | m² a crop eq| 0.15 (0.07–0.25)                | 0.3              | 99.5   | 0.6    | −0.4    |
| Mineral resource scarcity                               | kg Cu eq   | 1.2 × 10⁻² (5 × 10⁻³–2.1 × 10⁻²)| 0.6              | 91.5   | 12.5   | −4.7    |
| Fossil resource scarcity                                | kg oil eq  | 1.5 (−0.2 to 3.1)               | 9.3              | 127.4  | 1.3    | −38.0   |
| Water consumption                                       | m³         | 0.26 (−2.8 to 3.2)              | 0.0              | 102.1  | 0.9    | −3.0    |
in Table 4, for all impact categories assessed, with highest uncertainty related to the Laminaria sp. scenario. This high utility demand in the third generation feedstock process can be somewhat explained by the high liquid concentration of the Laminaria sp., requiring an energy-intensive drying process before fermenting the biomass (see ESI.3).

Polymerization does not appear as a hotspot in the overall impacts over the cradle to grave study for any of the considered feedstocks. Exceptions are freshwater and marine ecotoxicity, and human non-cancer toxicity impacts in the corn feedstock scenario. This is due to the proportionally high number of electricity and chemical inputs, as reported by the IHS Markit database, which was chosen as a data source widely used by industry. However, we note that the reliability of this data source has never been assessed, which was factored into uncertainty results from the Monte Carlo analysis (see ESI.5). Polymerization has also been reported in previous studies as potential environmental hotspot in LA production, mainly related to impacts associated with global warming and acidification from high energy consumption (Papong et al., 2014).

Processes in the EoL stage, which are generally independent of the chosen feedstock, have an overall positive effect on environmental impact profiles of LA and derived products. This is mainly related to environmental benefits originating from the proportion of plastics that can be recycled, and associated avoidance of producing virgin plastics. This leads to a potential reduction of up to 92% in fossil resource-scarcity impacts per kg LA derived from corn. Not all impact indicators are, however, positively affected by the EoL stage. For example, there is a more than 70% increase in freshwater and marine ecotoxicity impacts related to emissions from waste management, municipal waste incineration, and sanitary landfilling (see Table 4a). If the EoL stage is excluded, neither the positive nor negative impacts can be considered in related decisions for optimizing biochemical production systems. This demonstrates that it is essential to include the EoL stage when assessing the overall environmental performance of biochemicals, which is also indicated elsewhere (Laurent, Bakas, et al., 2014; Laurent, Clavreul, et al., 2014).

### 3.2 Relevance of different impact categories

With respect to individual impact categories, special attention should be given to global warming, land and water use, eutrophication, ecotoxicity, and indirect land-use change for the system using first generation biomass (Ögmundarson et al., 2019). When comparing our study with earlier studies of similar scope and system boundaries (Ingrao et al., 2015; Vink & Davies, 2015), our results vary by a factor of 0.6–2.5 for global warming impacts of LA/PLA production from corn. For land-use and freshwater eutrophication, our results are about one order of magnitude lower than results from some existing studies (Groot & Borén, 2010; Madival et al., 2009), while being in the same range as results from other studies (Papong et al., 2014). For marine eutrophication, our results are about 30% lower than what has been presented elsewhere (Landis, 2010), while compared with results presented earlier for water use impacts (Vink, Glassner, Kolstad, Wooley, & O’Connor, 2007) and ecotoxicity impacts (Landis, 2010), our results are up to one order of magnitude higher. These differences can partly be explained by including iLUC impacts in our study, affecting various other impact categories to different extents. More specifically, intensification of corn production and transformation to arable land due to increased demand for corn products leads to an increase in global warming impacts (Groot & Borén, 2010; Madival et al., 2009; Papong et al., 2014; see Table 5). Additional differences are related to using highly specific energy mixes with, for example, a high renewable energy usage in other studies (Vink & Davies, 2015), while using country-specific energy mixes that does not take into account local differences for energy inputs (Adom & Dunn, 2017; Vink & Davies, 2015).

Compared to our results, previously published studies assessing LA/PLA production from second generation biomass generally show global warming impacts that are about one order of magnitude lower (Adom & Dunn, 2017; Daful et al., 2016). This is mainly explained by the fact that we assign fertilizer value per kg corn stover needed to fulfill the functional unit as part of the system expansion (i.e.,

### Table 5 Global warming impacts

(وة express in kg CO₂ equivalents emitted to air) of cradle to grave lactic acid production from three feedstock generations (for details, see ESI.10)

| Lactic acid from | Global warming | Global warming without iLUC | Biogenic carbon storage | Net biogenic carbon emissions |
|------------------|----------------|-----------------------------|-------------------------|-----------------------------|
| corn             | kg CO₂ eq.     | 4.19                        | 3.67                    | 2.55                        | 1.64                        |
| corn stover      | kg CO₂ eq.     | 7.90                        | —                       | 2.91                        | 4.99                        |
| Laminaria sp.    | kg CO₂ eq.     | 11.0                        | —                       | 4.51                        | 6.52                        |

Abbreviation: iLUC, indirect land-use change.
deriving benefits from the avoidance of synthetic fertilizer production). In addition, our study considers the entire product life cycle as compared to assessing the production stage only, which is seen in most previous studies. These discrepancies additionally emphasize that assessing the full life cycle is essential for identifying and comparing all relevant impact hotspots and trade-offs between life cycle-stages and impact categories.

Using *Laminaria* sp. as feedstock in biorefineries is not yet a common practice, despite its potential as biomass, which is readily accessible along most coastlines. Seghetta, Hou, et al. (2016) studied the life cycle impacts of cultivated *Laminaria* sp. utilization for energy and feed production. Their results reported for global warming impacts are somewhat higher than results in the present study. This is mainly due to differences in modeled infrastructure and frequency of transportation to and from *Laminaria* sp. cultivation areas. Energy used for drying of the *Laminaria* sp. biomass before fermentation is in our scenario the main driver of global warming impact results.

It is important to note that especially in technologically immature production systems, the level of uncertainty has a significant influence on the interpretation of the above comparisons. Large parts of the uncertainty are thereby associated with assumptions made regarding process design of emerging feedstocks and how heat integration is incorporated (Adom & Dunn, 2017). Impacts are moreover highly influenced by the way feedstock is grown (corn), collected (corn stover), or mowed (*Laminaria* sp.), especially with regard to impacts related to terrestrial acidification and marine eutrophication, which are highly influenced by fertilizer or pesticide use, all leading to potential shifts in environmental hotspots between the considered feedstock processes.

### 3.3 Potential for optimizing environmental performance

To enable optimization of biochemical production systems with respect to reducing environmental impact hotspots and the role of data availability and assumptions at different TRLs, we first provide an overview of characteristics found when assessing LA derived from different bio-based feedstocks (Table 6). Since data availability and system understanding increase with increasing TRL, both scale-up challenges and uncertainty in assessment results increase with each new feedstock generation, being lowest for corn, via corn stover to *Laminaria* sp. This has also been shown in similar studies, where uncertainty was reported to be a function of technological maturity when assessing the environmental sustainability of algal-based biorefineries (Thomassen, Dael, Lemmens, & Passel, 2017).

Figure 3 compares damage level life cycle impact results across the different feedstock processes, aggregated by area of protection and including Monte Carlo uncertainty ranges for each of the feedstock generations (see ESI.5 detailing calculations of the geometric standard deviations used in the uncertainty analysis). Overall, LA production from corn (highest TRL among considered feedstocks) shows the best overall environmental performance, especially for human health (Figure 3a) and natural resources (Figure 3c). This implies a low potential for further reducing environmental impacts in comparison to other feedstocks (of lower technological maturity) in our study. LA production from *Laminaria* sp., in contrast, shows

| TABLE 6 | Characteristics of assessing environmental life cycle impacts of lactic acid production using three feedstock generations with different technological readiness levels (TRL) |
|---|---|---|
| Single product production process | Conceptual biochemical production | Conceptual biochemical production |
| Scaling up | Commercial biorefinery size assumed and scaling up done by applying techno-economic assessment for each of the biorefinery stages of the feedstock processes | Commercial biorefinery size assumed and scaling up done by applying techno-economic assessment for each of the biorefinery stages of the feedstock processes |
| Level of detail and data availability for each of the life cycle stages | Feedstock cultivation: high | Feedstock cultivation: high |
| Biorefinery: high | Feedstock collection: medium | Feedstock collection: medium |
| Polymerization: medium | Biorefinery: high | Biorefinery: high |
| End-of-life: medium | Polymerization: medium | Polymerization: medium |
| End-of-life: medium | End-of-life: medium | End-of-life: medium |
| Process optimization potential | Low | Medium: process optimization (lignin recovery) |
| Assessment results uncertainty (see ES) | Low (good data and system descriptions available at TRL 8–9) | Medium (good data and system description available for selected processes, and poor data available for other processes at TRL 4–5) |
| | | High (poor data and system description available at TRL 2–3) |
**FIGURE 3** Cumulative scores across considered impacts (including indirect land-use change influencing various impact categories) that contribute to areas of protection (a: human health; b: ecosystem quality; c: natural resources) per functional unit (FU) across selected bio-based feedstock generations for lactic acid production. Uncertainty ranges are indicated as error bars, and excluding the biomass drying process from the *Laminaria* sp. life cycle is shown as additional scenario (*Laminaria* sp. w/o drying). DALY, disability-adjusted life years

**FIGURE 4** Changes in environmental hotspots per area of protection (AoP) when assessing future scenarios of fermenting alginate for lactic acid production from *Laminaria* sp. across three countries (a–c), when changing geographical location of cradle to grave assessments from the United States, US, to China, CN (d), and from the US to Iceland, ICE (e), as well as when excluding impacts associated with indirect land-use change, iLUC (f)
highest impacts and uncertainties across areas of protection, and therefore has the highest potential for process optimization in terms of improving environmental performance. We note that uncertainty ranges across selected feedstock generations are overlapping to the extent that with the current overall uncertainty, it is difficult to identify the most relevant processes for further system optimization. However, we observe that the environmental performance of producing LA from Laminaria sp. can be dramatically improved relative to the use of corn or corn stover as feedstock, if the drying process is drastically optimized or removed completely. Without the drying of Laminaria sp., environmental impacts of the Laminaria sp. scenario are similar or even lower than impacts associated with the other two feedstock generations. This indicates great potential for Laminaria sp. to become more competitive in terms of improving environmental performance, when optimizing the biomass drying process. To support this finding, additional efforts are required with focus on investigating the influence of different drying technologies or the influence of using non-dried Laminaria sp. in biorefineries on environmental performance of LA. An earlier study already demonstrates that the drying of Laminaria sp. indeed causes comparatively high impacts of energy usage (Seghetta, Hou, et al., 2016).

In addition to testing Laminaria sp. drying, we tested the influence of different background data inputs for the Laminaria sp. scenario on environmental performance, with results presented in Figure 4. Assuming that alginate could also be used as input for fermentation would yield a considerable reduction in environmental impacts across considered countries (Figure 4a–c). The highest impact reduction contributing to reduced human health damages is associated with global warming (15%) and particulate matter impacts (19%), both related to the reduction in biomass needed per kg product produced, resulting in less energy needed for the drying of Laminaria sp. biomass. This reduced demand for biomass also drives the decrease in damages on ecosystem quality, with the highest reduction associated with global warming impacts on terrestrial ecosystems (18%) and land-use impacts (9%). Regarding natural resources, fossil resource scarcity is reduced by 36% from decreased biomass demand. Furthermore, increased feedstock yield is observed when assuming that fermentation of alginate in CN also induces benefits by reducing energy-related impacts. This applies, for example, to global warming impacts on terrestrial ecosystems and to terrestrial acidification (Figure 4b). Despite the energy mix in ICE being the most environmentally friendly across considered locations, we note that the mowing of Laminaria sp. biomass is mainly driven by energy generation from fossil fuels. Yet, fermenting alginate can reduce damages on natural resources by more than 50%, with similar trends also seen for reducing damages on human health and ecosystem quality (Figure 4c).

Since energy inputs have the single most dominating impact on the production system of LA from Laminaria sp., we evaluated if the results would change with a different composition of the considered energy mix. When changing the location of LA production from the US to CN, a drastic increase in damages on human health (108%) and on ecosystem quality (95%; Figure 4d) is observed, due to differences in the energy mix composition and related inputs. Damages on natural resources are reduced by 30% in the same scenario for CN, and by 76.5% when moving production to ICE (Figure 4e). This is because the country-specific ecoinvent processes chosen for modeling the background of the heat-mix production composition in the US cause higher impacts related to fossil resources scarcity as compared to background processes for the other considered locations (See ESI.7 for details). For the CN scenario, this might come as a surprise, but is explained by the fact that due to the lack of a country-specific Chinese heat-mix production process in the ecoinvent database, we assumed average world processes in this scenario. However, for both human health and ecosystem quality damages, an opposite trend is observed when moving LA production from the US to CN versus ICE. For ICE, where 87% of the heat and electricity production comes from renewable resources (Statistics Iceland, 2018), results in a reduction in damages on human health (31%) and ecosystem quality (33%), as compared to an increase in damages on human health (130%) and ecosystem quality (99%) for CN. The US and CN both rely to a large extent on fossil resources for their energy mix (IEA, 2017; EIA, 2018), where burning fossil fuels emits large quantities of particulate matter, explaining the related substantial reduction presented in Figure 4e for ICE. Reduction in damages on ecosystem quality somewhat differs when compared to the US and CN. These results emphasize that it is important to consider the location of production systems, as related energy mixes may strongly influence environmental performance results—a conclusion also reported in previous studies (Espinosa, Laurent, & Krebs, 2015; Yates & Barlow, 2013).

Finally, most existing LCA studies on biochemical production do not consider impacts from indirect land-use change. When excluding iLUC impacts from our analysis, we see a reduction of up to 10% across areas of protection for the corn feedstock scenario (Figure 4f). This reduction could be even higher when choosing locations other than those considered in the present study, due to possible differences in land-use competition. Thus, ignoring iLUC in LCA may lead to an underestimation of environmental impacts whenever land-use competition is relevant. This is supported by the few available studies in which iLUC is included in the impact assessment (Hjuler & Hansen, 2018).

4 | DISCUSSION

We have demonstrated how LCA can effectively be applied for evaluating and optimizing the environmental performance
of LA production systems using selected bio-based feedstock systems with different levels of technological maturity. Our results emphasize how each selected feedstock system and TRL comes with a distinct set of environmental hotspots. Hence, each of these feedstock systems and TRLs require different focus points for process optimization in support of striving for increased environmental sustainability of biochemical production without the aim of identifying the most environmentally friendly feedstock. We note that while our results allow for comparing LA production systems in terms of their environmental performance, this comparison does not imply a fair evaluation across systems. This is mainly due to differences in assumptions and data available across TRLs. For a fair comparison across systems, optimization potentials across TRLs need to be further assessed and scale-up factors developed. However, our study allows to pinpoint important hotspots within each TRL for the selected feedstocks to focus technology advances during maturation of each system.

We recommend to specifically focus on biomass production and refinery processes, which dominate overall impact profiles across selected feedstock systems. Moreover, enabling the potential fermentation of alginate will drastically reduce environmental impacts. When additionally using bacteria in the biorefinery process that ferment Laminaria sp. without the need to first dry the feedstock would significantly reduce global warming and several other impacts, rendering LA production from Laminaria sp. the potentially best-in-class option among the selected bio-based feedstock systems. Intensive air-drying is potentially not an industrially viable process, except in countries like ICE where there is easy access to excessive heat and steam. Still, some kind of pretreatment of the Laminaria sp. is needed to slow down the degradation which needs to be avoided to preserve the nutrient values and proteins of the already sensitive biomass. Simultaneously drying of the Laminaria sp. can reduce risk of feedstock contamination. In addition, microorganisms feed more successfully on dry than on wet macroalgae biomass, which is currently a serious challenge for using non-dried macroalgae. When assessing different pretreatment options, like freeze-drying or mild air-drying, the results may lead to different performance profiles, and hence, it is important to specify the drying process. This is also the case for chemical separation or ensiling of the Laminaria sp. (van Hal, Huijgen, & López-Contreras, 2014; Herrmann et al., 2015; Hou, From, Angelidaki, Huijgen, & Bjerre, 2017; Seghetta et al., 2017; Stévant, Rebours, & Chapman, 2017).

Although our findings are not generalizable toward other than the selected feedstocks and toward other biochemicals, we highlight important aspects that should generally be considered. As next step, it would be desirable to compare our results for LA ultimately also with fossil-based chemicals, but LA is not produced from fossil sources, and therefore, a functionally equivalent fossil-based chemicals would need to be defined. In addition, such comparison is already available in literature, as demonstrated by Hottle et al. (2013) and Ögmundarson et al. (2019).

Having demonstrated that the local energy mix greatly influences impact results, we strongly recommend taking production location into account for any future assessments. Finally, various yet distinct impact categories dominate overall impact profiles across selected feedstock systems, mostly those associated with feedstock production and biorefining but also those influenced by EoL scenarios. Hence, we emphasize that there is a need to consider a broader set of impact categories, including indirect land-use change, and to cover the entire life cycle, in order to successfully identify and reduce relevant trade-offs and avoid any possible burden shifting across impact categories and across life cycle-stages. Furthermore, when moving to farming and/or intensified mowing of macroalgae, related impacts on marine biodiversity need to be included to consider possible trade-offs, but methods to include such aspects in LCA are currently not yet mature.

Overall, applying LCA in a comprehensive manner gives the bio-based industry the opportunity to actively incorporate and prioritize environmental sustainability in its decision-making process at an early stage of development of biochemicals. LCA helps pinpoint relevant environmental hotspots and related optimization potentials along the entire biochemical life cycle, before making large scale-up investments. LCA also allows for the integration and alignment of environmental aspects with techno-economic aspects, enabling overall optimization of biochemical production systems (Kern et al., 2017; Patel, Zhang, & Kumar, 2016) and our results are currently being evaluated by researchers at the Novo Nordisk Foundation Center for Biosustainability to consider environmental aspects in the optimization of macroalgae biorefinery processes. For assessing feedstock systems with low technological maturity, we recommend focusing future research on reducing uncertainty of inventory data, which requires access to industry data, and improving assumptions in system models with respect to the upscaling of early-stage technologies.

Finally, in support of achieving overarching global goals for sustainable development, we ultimately also need to benchmark life cycle impacts associated with chemical production from both fossil- and bio-based feedstocks against nature’s own capacity to recover from chemical emissions (Fantke & Illner, 2019). For LA production, our study provides a possible starting point for evaluating environmental sustainability along entire life cycles in support of assessing progress toward achieving sustainability goals when moving toward a bio-based industry, and our findings can easily be extended and applied to other biochemicals and feedstocks. Implementing these processes in industry can help improving the sustainability performance of biochemical production, which is a declared goal of the biochemical industry.
ACKNOWLEDGEMENTS

This work was supported by the EU FP7 project Biorefinery 2G (grant 613771). We thank M. Owsianiak and M. Ryberg for constructive comments, J. Schmidt for support in the iLUC calculations, and A. Ö. Kristjánsdóttir for making inventory data for Laminaria sp. scenarios available on the Asco Harvester website.

ORCID

Ölgud Ögmundarson https://orcid.org/0000-0003-3171-2388
Sumesh Sukumara https://orcid.org/0000-0002-7924-458X
Alexis Laurent https://orcid.org/0000-0003-0445-7983
Peter Fantke https://orcid.org/0000-0001-7148-6982

REFERENCES

Adams, J. M. M., Bleathman, G., Thomas, D., & Gallagher, J. A. (2017). The effect of mechanical pre-processing and different drying methodologies on bioethanol production using the brown macroalgae Laminaria digitata (Hudson) JV Lamouroux. Journal of Applied Phycology, 29(5), 2463–2469. https://doi.org/10.1007/s10811-016-1039-5

Adom, F. K., & Dunn, J. B. (2017). Life cycle analysis of corn-stover-derived polymer-grade l-lactic acid and ethyl lactate: Greenhouse gas emissions and fossil energy consumption. Biofuels, Bioproducts and Biorefining, 11(2), 258–268. https://doi.org/10.1002/1bb.1734

Aeschelmann, F., Carus, M., Baltus, W., Carrez, D., de Guzman, D., Käb, H., ...Ravenstijn, J. (2016). Bio-based building blocks and polymers. Global capacities and trends—2017–2022. Huerth, Germany: nova-Institut GmbH.

Asco Harvester. (2018). Macroalgae data. Retrieved from http://www.ascoharvester.is/

Bardhan, S. K., Gupta, S., Gorman, M. E., & Haider, M. A. (2015). Biorenewable chemicals: Feedstocks, technologies and the conflict with food production. Renewable and Sustainable Energy Reviews, 51, 506–520. https://doi.org/10.1016/j.rser.2015.06.013

Biddy, M. J., Davis, R., Humbird, D., Tao, L., Dowe, N., Guarnieri, M. T., ... Beckham, G. T. (2016). The techno-economic basis for coproduct manufacturing to enable hydrocarbon fuel production from lignocellulosic biomass. ACS Sustainable Chemistry & Engineering, 4(6), 3196–3211. https://doi.org/10.1021/acs.suschemeng.b600243

Breedveld, L., Fontana, S., Misericocchi, C., Vannini, L., Papaprydes, C. D., Vouyiouka, S., Huizen, D. (2014). LCA of vegetarian burger packed in biobased polybutylene succinate. In R. Schenck & D. Huizen (Eds.), Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2014), San Francisco, California, USA, 8–10 October, 2014. (pp. 157–166). Vashon, WA: American Center for Life Cycle Assessment.

Carlson, T. L., & Peters, E. M. (1997). Low PH lactic acid fermentation (Patent). Minneapolis, MN: United States Patent. Retrieved from https://patentimages.storage.googleapis.com/a1/c7/e4/2d3b01d6b3f7b/US6475759.pdf

Chandel, A. K., Garlapati, V. K., Singh, A. K., Antunes, F. A. F., & da Silva, S. S. (2018). The path forward for lignocellulose biorefineries: Bottlenecks, Solutions, and perspective on commercialization. Bioresource Technology, 264, 370–381. https://doi.org/10.1016/j.biortech.2018.06.004

Chen, H., & Qiu, W. (2010). Key technologies for bioethanol production from lignocellulose. Biotechnology Advances, 28(5), 556–562. https://doi.org/10.1016/j.biotechadv.2010.05.005

Daful, A. G., Haigh, K., Vaskan, P., & Görgens, J. F. (2016). Environmental impact assessment of lignocellulosic lactic acid production: Integrated with existing sugar mills. Food and Bioproducts Processing, 99, 58–70. https://doi.org/10.1016/j.fbp.2016.04.005

Davis, R., Kinchin, C., Markham, J., Tan, E. C. D., Laurens, L. M. L., Sexton, D., ... Lukas, J. (2013). Process design and economics for the conversion of algal biomass to biofuels: Algal biomass fractionation to lipid- and carbohydrate-derived fuel products. Golden, CO: National Renewable Energy Laboratory.

Dunn, J. B., Adom, F., Sather, N., Han, J., & Snyder, S. (2015). Life-cycle analysis of bioproducts and their conventional counterparts in GREET™. Argonne, IL: Argonne National Laboratory.

Espinosa, N., Laurent, A., & Krebs, F. C. (2015). Ecodesign of organic photovoltaic modules from Danish and Chinese perspectives. Energy & Environmental Science, 8(9), 2537–2550. https://doi.org/10.1039/C5EE01763G

European Bioplastics. (2017). Bioplastics: Facts and figure. Online publication. Berlin, Germany. Retrieved from https://www.europaean-bioplastics.org/news/publications/

European Commission [EC]. (2010). The International reference life cycle data system (ILCD) handbook – General guide for life cycle assessment – Detailed guidance (1st ed.). Brussels, Belgium. https://doi.org/10.2788/38479

European Commission [EC]. (2018). A European strategy for plastics in a circular economy. COM(2018) 28 final (Vol. 16). Brussels, Belgium: European Commission.

European Committee for Standardization [CEN]. (2015). European standard EN 16760:2015—Bio-based products – Life cycle assessment. Retrieved from https://standards.cen.eu/dyn/www/f?p=204:110::FSP_PROJEC_T:39303&cs=1379CA21646C9EFCDB785629FC62F8F3

Evans, L. K., & Edwards, M. S. (2011). Bioaccumulation of copper and zinc by the giant kelp Macrocystis pyrifera. Algae, 26(3), 265–275. https://doi.org/10.4490/algae.2011.26.3.265

Fantke, P., & Illner, N. (2019). Goods that are good enough: Introducing an absolute sustainability perspective for managing chemicals in consumer products. Current Opinion in Green and Sustainable Chemistry, 15, 91–97. https://doi.org/10.1016/j.cogsc.2018.12.001

Food and Agriculture Organization of the United Nations [FAO]. (2019). Fisheries and aquaculture software. FishStatJ – Software for fishery and aquaculture statistical time series. Retrieved from http://www.fao.org/fishery/

Gironi, F., & Piemonte, V. (2011). Life cycle assessment of polyactic acid and polyethylene terephthalate bottles for drinking water. Environmental Progress & Sustainable Energy, 30(3), 459–468. https://doi.org/10.1002/ep.10490

Groot, W. J., & Borén, T. (2010). Life cycle assessment of the manufacture of lactide and PLA biopolymers from sugarcane in Thailand. The International Journal of Life Cycle Assessment, 15(9), 970–984. https://doi.org/10.1007/s11367-010-0225-y

Hanes, R. J., Cruze, N. B., Goel, P. K., & Bakshi, B. R. (2015). Allocation games: Addressing the ill-posed nature of allocation in...
greenhouse gas (GHG) balance of succinic acid-based plastic end products made from lignocellulosic biomass. Biofuels, Bioproducts and Biorefining, 6(3), 246–256. https://doi.org/10.1002/bbb.1849

Patel, M., Cranke, M., Dornberg, V., Herrmann, B., Roes, L., Häusing, B., ... Recchia, E. (2006). Medium and long-term opportunities and risk of the biotechnological production of bulk chemicals from renewable resources – The potential of white biotechnology. Utrecht, the Netherlands: The BREW Project.

Patel, M., Zhang, X., & Kumar, A. (2016). Techno-economic and life cycle assessment on lignocellulosic biomass thermochemical conversion technologies: A review. Renewable and Sustainable Energy Reviews, 53, 1486–1499. https://doi.org/10.1016/j.rser.2015.09.070

Schiener, P., Black, K. D., Stanley, M. S., & Green, D. H. (2015). The seasonal variation in the chemical composition of the kelp species Laminaria digitata, Laminaria hyperborea, Saccharina latissima and Alaria esculenta. Journal of Applied Phycology, 27(1), 363–373. https://doi.org/10.1007/s10811-014-0327-1

Schmidt, J. H., Weidema, B. P., & Brandão, M. (2015). A framework for modelling indirect land-use changes in life cycle assessment. Journal of Cleaner Production, 99, 230–238. https://doi.org/10.1016/j.jclepro.2015.03.013

Seghetti, M., Hou, X., Bastianoni, S., Bjerre, A.-B., & Thomesen, M. (2016). Life cycle assessment of macroalgal biorefinery for the production of ethanol, proteins and fertilizers – A step towards a regenerative bioeconomy. Journal of Cleaner Production, 137, 1158–1169. https://doi.org/10.1016/j.jclepro.2016.07.195

Seghetti, M., Romeo, D., D’Este, M., Alvarado-Morales, M., Angelidaki, I., Bastianoni, S., & Thomesen, M. (2017). Seaweed as innovative feedstock for energy and feed – Evaluating the impacts through a life cycle assessment. Journal of Cleaner Production, 150, 1–15. https://doi.org/10.1016/j.jclepro.2017.02.022

Seghetti, M., Tørring, D., Bruhn, A., & Thomesen, M. (2016). Bioextraction potential of seaweed in Denmark — An instrument for circular nutrient management. Science of the Total Environment, 563–564, 513–529. https://doi.org/10.1016/j.scitotenv.2016.04.010

Sen Nag, O. (2019). World leaders in corn (maize) production, by country – WorldAtlas.com. Retrieved from https://www.worldatlas.com/articles/world-leaders-in-corn-maize-production-by-country.html

Smidt, M., den Hollander, J., Bosch, H., Xiang, Y., van der Graaf, M., Lambin, A., & Duda, J.-P. (2015). Life cycle assessment of biobased and fossil-based succinic acid. In J. Dewulf, S. De Meester, & R. A. Lambin, A., & Duda, J.-P. (2015). Life cycle assessment of macroalgal biorefinery for the 3rd generation biomass: A techno-economic assessment on lactic acid production. Biofuels, Bioproducts and Biorefining, in review.

The Organization for Economic Cooperation and Development [OECD]. (2014). Biobased chemicals and bioplastics. Paris, France: OECD Publishing. https://doi.org/10.1787/5jxwrjfx0dfj-en

The Organization for Economic Cooperation and Development [OECD]. (2018). The Organization for Economic Cooperation and Development statistics. Retrieved from http://www.oecd.org/

Thomassen, G., Van Dael, M., Lemmens, B., & Van Passel, S. (2017). A review of the sustainability of algal-based biorefineries: Towards an integrated assessment framework. Renewable and Sustainable Energy Reviews, 68, 876–887. https://doi.org/10.1016/j.rser.2016.02.015

Tomás-Grasa, E., Ögmundarson, Ö., Gavala, H. N., & Sukumara, S. (n.d.). Commodity chemical production from 3rd generation biomass: A techno-economic assessment on lactic acid production. Biofuels, Bioproducts and Biorefining, in review.

United States Energy Information Administration [EIA]. (2018). U.S. primary energy consumption by source and sector, 2017. Washington, DC: EIA.

United States Environmental Protection Agency [EPA]. (2018). Plastics: Material-specific data. Retrieved from https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/plastics-material-specific-data#PlasticsTableandGraph

Urban, R. A., & Bakshi, B. R. (2009). 1,3-propanediol from fossils versus biomass: A life cycle evaluation of emissions and ecological resources. Industrial & Engineering Chemistry Research, 48(17), 8068–8082. https://doi.org/10.1021/ie801612p

van der Harst, E., Potting, J., & Kroese, C. (2014). Multiple data sets and modelling choices in a comparative LCA of disposable beverage cups. The Science of the Total Environment, 494–495(50), 129–143. https://doi.org/10.1016/j.scitotenv.2014.06.084

van Hal, J. W., Huijgen, W. J. J., & López-Contreras, A. M. (2014). Opportunities and challenges for seaweed in the biobased economy. Trends in Biotechnology, 32(5), 231–233. https://doi.org/10.1016/j.tibtech.2014.02.007

Vink, E. T. H., & Davies, S. (2015). Life cycle inventory and impact assessment data for 2014 Ingeo™ polylactide production. Industrial Biotechnology, 11(3), 167–180. https://doi.org/10.1089/lin.2015.0003

Vink, E. T. H., Davies, S., & Kolstad, J. J. (2010). The eco-profile for current Ingeo® polylactide production. Industrial Biotechnology, 6(4), 212–224. https://doi.org/10.1089/ind.2010.6.212

Vink, E. T. H., Glassner, D. A., Kolstad, J. J., Wooley, R. J., & O’Connor, R. P. (2007). The eco-profiles for current and near-future NatureWorks® polylactide (PLA) production. Industrial Biotechnology, 3(1), 58–81. https://doi.org/10.1089/ind.2007.3.058

Wee, Y., Kim, J., & Ryu, H. (2006). Biotechnological production of lactic acid and its recent applications. Retrieved from https://doi.org/citeulike-article-id:7853424

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., & Weidema, B. (2016). The ecoinvent database version 3 (part I): Overview and methodology. International Journal of Life Cycle Assessment, 21(9), 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

Yang, X., Choi, H. S., Park, C., & Kim, S. W. (2015). Current states and prospects of organic waste utilization for biorefineries. Renewable and Sustainable Energy Reviews, 49, 335–349. https://doi.org/10.1016/j.rser.2015.04.114

Yates, M. R., & Barlow, C. Y. (2013). Life cycle assessments of biodegradable, commercial biopolymers—A critical review. Resources, Conservation and Recycling, 78(5), 54–66. https://doi.org/10.1016/j.resconrec.2013.06.010
Zhu, Y., Romain, C., & Williams, C. K. (2016). Sustainable polymers from renewable resources. *Nature, 540*(7633), 354–362. https://doi.org/10.1038/nature21001

**SUPPORTING INFORMATION**

Additional supporting information may be found online in the Supporting Information section at the end of the article.