Life cycle assessment of ethanol production from miscanthus: A comparison of production pathways at two European sites

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Abstract
Lignocellulosic ethanol represents a renewable alternative to petrol. Miscanthus, a perennial plant that grows on marginal land, is characterized by efficient use of resources and is considered a promising source of lignocellulosic biomass. A life cycle assessment (LCA) was performed to determine the environmental impacts of ethanol production from miscanthus grown on marginal land in Great Britain (Aberystwyth) and an average-yield site in Germany (Stuttgart; functional unit: 1 GJ). As the conversion process has substantial influence on the overall environmental performance, the comparison examined three pretreatment options for miscanthus. Overall, results indicate lower impacts for the production in Stuttgart in comparison with the corresponding pathways in Aberystwyth across the analysed categories. Disparities between the sites were mainly attributed to differences in biomass yield. When comparing the conversion options, liquid hot water treatment resulted in the lowest impacts, followed by dilute sulphuric acid. Dilute sodium hydroxide pretreatment represented the least favourable option. Site-dependent variation in biomass composition and degradability did not have substantial influence on the environmental performance of the analysed pathways. Additionally, implications of replacing petrol with miscanthus ethanol were examined. Ethanol derived from miscanthus resulted in lower impacts with respect to greenhouse gas emissions, fossil resource depletion, natural land transformation and ozone depletion. However, for other categories, including toxicity, eutrophication and agricultural land occupation, net scores were substantially higher than for the fossil reference. Nevertheless, the results indicate that miscanthus ethanol produced via dilute acid and liquid hot water treatment at the site in Stuttgart has the potential to comply with the requirements of the European Renewable energy directive for greenhouse gas emission reduction. For ethanol production at the marginal site, carbon sequestration needs to be considered in order to meet the requirements for greenhouse gas mitigation.

KEYWORDS
biofuel, environmental performance, life cycle assessment, lignocellulosic ethanol, marginal land, miscanthus, perennial crop, pretreatment
1 | INTRODUCTION

Lignocellulosic biomass is an abundant, renewable resource and is considered a promising option to contribute to the biomass demand of a growing bioeconomy (Kumar, 2014; Lewandowski, 2016). Over the last decade, the potential to produce transportation fuels from lignocellulosic feedstocks has been accentuated (van der Weijde et al., 2013). This trend is supported by the advancement of conversion technologies and coincides with ambiguity in sustainability considerations of sugar- or starch-based biofuels (McKechnie, Pourbafrani, Saville, & MacLean, 2015). Controversies related to these first generation feedstocks include concerns about food competition, land use change or intensive agricultural inputs and have directed the research focus towards lignocellulose-based solutions (Falano, Jeswani, & Azapagic, 2014). At present, first commercialization ventures of lignocellulosic ethanol production in Europe can be observed (Clariant International Ltd, 2017a, 2017b) and thus potential impacts have to be analysed and considered.

Lignocellulosic feedstocks are mainly composed of cellulose, hemicellulose and lignin (Carroll & Somerville, 2009). These cross-link to form a network, which contributes to a plant’s recalcitrance to (bio-)chemical and physical treatments (Achinas & Euverink, 2016). Cellulose is a polysaccharide of linearly interlinked glucose molecules. These glucose chains cross-link by means of hydrogen bonds and Van der Waals forces to form compact crystalline structures that are insoluble in water and resistant to degradation (Marriott, Gómez, & McQueen, 2015). Hemicellulose is a polysaccharide consisting of a diverse set of hexoses and pentoses, with xylose usually being the main component. It cross-links with cellulose microfibrils via hydrogen bonds and with lignin to form lignin carbohydrate complexes that contribute to cell wall recalcitrance (Chundawat, Beckham, Himmel, & Dale, 2011). Lignin is a complex heteropolymer consisting of three aromatic monomers (Bonawitz & Chapple, 2010). It provides structural strength to plant tissues and protects the polysaccharides (Mathew, Parameshwaran, Sukumaran, & Pandey, 2016; Yu et al., 2014). The relative abundance and interaction of the lignocellulosic components define a plant’s cell wall recalcitrance.

In the production of lignocellulosic ethanol via biochemical pathways, cellulose and hemicellulose serve as sugar source for the fermentation reactions. For an efficient conversion, several process steps are required: Pretreatment, (enzymatic) hydrolysis, fermentation and product stream purification. Pretreatment is required to overcome the lignocellulose recalcitrance and to increase accessibility of the polysaccharides to the degrading enzymes (Torres, van der Weijde, Dolstra, Visser, & Trindade, 2013). Various methods exist, deploying (combinations of) physical, physicochemical, chemical and biological approaches (Cao, Sun, Liu, Yin, & Wu, 2012; da Costa Sousa, Chundawat, Balan, & Dale, 2009; Murnen et al., 2007; Prasad, Sotenko, Blenkinsopp, & Coles, 2016). These processes and techniques differ in efficiency and material/energy demands and have various advantages and drawbacks with respect to production of inhibitory by-products and further process steps (Achinas & Euverink, 2016; Sims, Taylor, Jack, & Mabee, 2008). For instance, the efficiencies of subsequent hydrolysis processes strongly depend on the previous treatment. During hydrolysis, cellulose and (partially) hemicellulose are hydrolysed to obtain soluble monomeric sugar molecules for the further conversion to ethanol. Today, these processes are most commonly performed by means of enzymatic reactions (Achinas & Euverink, 2016).

However, enzymatic activity can be negatively affected by highly recalcitrant lignocellulosic structures, as well as adsorption to and inhibition by lignin (Agor, Cicek, Sparling, Berlin, & Levin, 2011; Cao et al., 2012; Klein-Marcuschamer, Oleskowicz-Popiel, Simmons, & Blanch, 2010). Accordingly, an efficient pretreatment process alters the lignocellulosic structure and decreases the adherence of lignin to cellulose and enzymes, in order to facilitate optimal sugar release (Klein-Marcuschamer et al., 2010; Padmanabhan et al., 2016). Free sugar monomers can then be metabolized and degraded to ethanol by means of microorganisms such as yeasts (e.g., Saccharomyces cerevisiae) or bacteria (e.g., Zymomonas mobilis). As only sugars are utilized for ethanol production, the third lignocellulosic main component, lignin, can be used as a by-product. For instance, it can be combusted for heat and power generation, along with other residues of the refinery process (Humbird et al., 2011).

Lignocellulosic feedstock may be supplied from diverse sources, including agricultural residues and dedicated biomass crops. For a sustainable provision of sufficient biomass on an industrial level, several prerequisites have to be considered. Most importantly, feedstocks are required to provide high yields within low-input management schemes (only small amounts of mineral nutrients and pesticides applied; van der Weijde et al., 2013). Among others, miscanthus is considered a crop that potentially fulfils these requirements. It is a perennial rhizomatous C4 grass originating from East Asia (Cadoux, Riche, Yates, & Machet, 2012).

Plants can grow up to four metres and yield high amounts of biomass in a range of conditions (Brosse, Dufour, Meng, Sun, & Ragauskas, 2012; Lewandowski et al., 2016; van der Weijde et al., 2013). Due to an efficient nutrient relocation mechanism and an extensive root system, miscanthus is characterized by efficient resource use and the potential to sequester atmospheric carbon (Clifton-Brown, Breuer, & Jones, 2007; Lewandowski & Schmidt, 2006).
An additional benefit is miscanthus’ potential to grow on marginal land. Marginal areas are unsuitable for the (economically) benign cultivation of traditional crops due to a range of potential factors such as moisture, salinity, temperature, soil depth or even field shape (Clifton et al., 2016; Quinn et al., 2015). Moreover, it has been reported that miscanthus can be grown on heavy metal-contaminated or saline soils (Nsanganwimana, Pourrut, Mench, & Douay, 2014; Pidlisnyuk, Stefanovska, Lewis, Erickson, & Davis, 2014; Stavridou, Hastings, Webster, & Robson, 2017; Sun, Yamada, & Takano, 2014). Considering these aspects, miscanthus cultivation could contribute to a sustainable biomass provision, reducing land use competition and improving the profitability of agricultural land of lower quality (Allison, Morris, Clifton-Brown, Lister, & Donnison, 2011; Clifton-Brown et al., 2016; Hu, Wu, Persson, Peng, & Feng, 2017). However, the intensive cultivation of miscanthus requires further optimization of the plant. With more than 20 species known, the genus holds large potential for further optimization with respect to environment- or application-related adaptions. In order to advance the market potential of lignocellulosic ethanol, current breeding efforts focus on the creation of new hybrids aiming to increase biomass yields and optimize lignocellulose degradability (Purdy, Maddison, Cunniff, Donnison, & Clifton-Brown, 2015; van der Weijde et al., 2016).

The biochemical conversion of lignocellulosic biomass to ethanol for fuel provision has been studied intensively from an environmental perspective. A wide range of feedstocks and conversion processes have been analysed by means of life cycle assessments (LCA). LCA is a method for assessing the environmental impacts of products throughout their life cycle including the raw material acquisition, production, transportation as well as end use and disposal. Morales, Quintero, Conejeros, and Aroca (2015) reviewed 60 LCA studies of lignocellulosic ethanol production from diverse feedstocks. They demonstrated that most studies focus on global warming potential and revealed a clear reduction potential when lignocellulosic instead of fossil feedstocks are used. This was shown for agricultural residues, such as corn stover or wheat straw, as well as for dedicated energy crops. For instance, switchgrass, a perennial grass comparable to miscanthus, offers a reduction potential of 53%–93% of greenhouse gas (GHG) emissions in comparison to petrol. Furthermore, various studies observed lower impacts on ozone layer depletion and heavy metal emission-related toxicities. However, for other impact categories, such as acidification, eutrophication and ecotoxicity trends are less clear and depend strongly on feedstock types. Despite the fact, that a wide variety of feedstocks have been analysed, only a few have focused on the environmental performance of miscanthus ethanol. Some exceptions, considering the biochemical conversion, include studies from the USA and Europe (Cronin et al., 2016; Dwivedi et al., 2015; Falano et al., 2014; Meyer, Wagner, & Lewandowski, 2017; Scown et al., 2012). Although it is deemed a promising option for a sustainable provision of biomass, the cultivation of miscanthus on marginal lands for biofuel production has only been considered in a single study. This study focused on the Mediterranean region and did not consider the conversion pathway in detail (Schmidt, Fernando, Monti, & Rettenmaier, 2015), although the environmental performance of lignocellulosic ethanol is strongly influenced by the design of the conversion process. A large variety of options exist, mainly differing in the pretreatment step and influencing conversion conditions, yields, energy requirement, material demand and by-product generation (Falano et al., 2014; McKechnie et al., 2015).

Hence, this study aims to analyse the environmental performance of ethanol production from miscanthus at a marginal site and to contrast the results with the performance at an average-yield site. To do so, an environmental LCA of lignocellulosic ethanol from miscanthus is performed. The cradle-to-grave analysis follows the ISO norms 14040 and 14044 (ISO, 2006a, 2006b). It enables the identification of main drivers for different environmental impacts and the evaluation of variation due to site-specific factors (incl. biomass composition and by-product generation). In addition, the study analyses the GHG emission reduction potential in comparison with petrol. In order to consider differences in the conversion process, the analysed systems extend to the refinery and three relevant pretreatment techniques were compared.

The combination of two European miscanthus cultivation scenarios—a marginal site in the UK and an average site in Germany—and three refinery models provided six production pathways for the analysis. With this approach, the study contributes to the identification of favourable pathways for the production of lignocellulosic ethanol and gives insights into the potential role of marginal areas for the sustainable production of biofuels.

2 | MATERIALS AND METHODS

2.1 | Scope

The studied system is composed of two main parts: the agricultural production and the subsequent conversion in a refinery (Figure 1). The boundaries include miscanthus cultivation and the production and transport of required inputs such as fertilizers, pesticides, propagation material, management and harvesting operations over a 20-year cultivation period. Harvest was assumed to take place from the third year on. The model of the feedstock production was adapted to two locations in Europe, Aberystwyth in the
UK (ABER) and Stuttgart in Germany (STR). The sites differ mainly in yields as well as in emissions to air and water. In Aberystwyth, feedstock production takes place on a marginal site, which is characterized by shallow soils, a slow establishment phase and low yields. Furthermore, the British site is characterized by higher precipitation rates and comparatively low soil clay content (Hastings et al., 2017; Wagner, Kiesel, Hastings, Iqbal, & Lewandowski, 2017; van der Weijde, Dolstra, Visser, & Trindade, 2017). The system boundaries include transport of biomass from the field to the refinery and all processes of the conversion in the plant (pretreatment, enzymatic hydrolysis, fermentation and distillation/separation). The refinery is assumed to have a yearly biomass intake of 250,000 t dry matter (DM), which represents the capacity of recently started commercial lignocellulose refinery projects in Romania and Slovakia (both with a yearly production capacity of 50,000 t ethanol (Clariant International Ltd, 2017b, Clariant International Ltd, 2017a). Co-products of the refinery stage include heat and electricity which are produced by combustion of process residues and reused within the system. If a surplus of electricity is produced, a system expansion approach is applied and electricity exports are assumed to replace local grid electricity. In addition to the refinery processes, the system includes the product distribution to the service station and the use phase.

In Europe, bioethanol is usually used as a low-level blend with petrol. Nevertheless, combustion of pure ethanol in a medium European passenger car was considered as this allows a transparent comparison of the environmental performance.

The functional unit for the system is defined as 1 gigajoule (GJ). For the life cycle impact assessment, the characterization model ReCiPe is applied (Goedkoop et al., 2013). Selected impact categories include those comparable to the ones described by Morales et al. (2015), as these represent prevailing categories in the assessment of lignocellulosic ethanol. This includes the midpoint indicators climate change (CC), fossil resource depletion (FD), freshwater (FET), marine (MET), and terrestrial ecotoxicity (TET), human toxicity (HT), freshwater (FE) and marine eutrophication (ME), terrestrial acidification (TA), ozone depletion (OD) and photochemical oxidant formation (POF). In addition, natural land transformation (NLT) and agricultural land occupation (ALO) were assessed, as both categories were found to be of relevance for miscanthus cultivation and utilization (Wagner & Lewandowski, 2017). The relative importance of impact results is assessed by means of a normalization step (ISO, 2006b). The applied European midpoint level normalization factors were based on the ReCiPe methodology (Goedkoop et al., 2013).

For reasons of simplification, an assumption was made: Biogenic carbon dioxide (CO2) emissions from the combustion of fermentation residues or ethanol, as well as biogenic CO2 emissions from ethanol fermentation, were not accounted for as the released carbon is in balance with the carbon sequestered in the biomass (Gerbrandt et al., 2016).
2.2 | Life cycle inventory

2.2.1 | Miscanthus cultivation

Agronomic operations for the cultivation of miscanthus include soil preparation, establishment, fertilization, harvest and recultivation of the field after the cultivation period. Data on yield, amounts of pesticides, and propagation material were obtained from multi-annual field trials in Aberystwyth and Stuttgart and were based on a cultivation period of 20 years (field trial description in Lewandowski et al., 2016). Data for yields and inputs were summed up over the whole cultivation period including the establishment phase. Input quantities were calculated per year (Table 1). A similar procedure was applied for agricultural operations, including harrowing, (chisel) ploughing, planting, mulching, spraying of fertilizers, mowing and chipping. For all processes besides chipping, standard inventory data from the ecoinvent v3.4 database were used (Wernet et al., 2016). For harvesting, the standard ecoinvent process for chopping of maize was adapted according to Hastings et al. (2017).

Nitrous oxide (N$_2$O) emissions and soil carbon dynamics represent major sources of variability in life cycle considerations of lignocellulosic ethanol (Gerbrandt et al., 2016). It has been shown that during long-term cultivation of miscanthus CO$_2$ is sequestered in the soil (McCalmont et al., 2017). However, the underlying dynamics are complex and thus sequestration was not considered in the present life cycle inventory. Potential impacts are discussed in the final section. Direct N$_2$O and nitric oxide (NO) emissions from nitrogen fertilizer were estimated according to Bouwman, Boumans, and Batjes (2002), while indirect N$_2$O emissions from mineral fertilizer and N$_2$O emissions from harvesting residues were modelled according to IPCC (2006). Ammonia emissions and nitrate (NO$_3$) losses were calculated according to EEA (2001) and the NO$_3$ model described in Faist Emmenegger, Reinhard, and Zah (2009). Phosphate and phosphorus emissions to surface and groundwater, as well as heavy metal emissions to soils were estimated based on Nemecek and Kägi (2007). For pesticide-related emissions, it was assumed that the entire mass of pesticides is released to agricultural soil (Nemecek, Dubois, Huguenin-Elie, & Gaillard, 2011).

The analysed miscanthus accession is OPM 6, a hybrid derived from crosses between M. sacchariflorus and M. sinensis. This accession has performed well across various European environments and provides high biomass and carbohydrate yields (van der Weijde, Dolstra, et al., 2017). Harvested biomass from both sites was analysed for main lignocellulose components as described in van der Weijde, Kiesel, et al. (2017). As biomass quality in the third cultivation year is representative for well-established plantations (van der Weijde, Dolstra, et al., 2017), results for the third harvest were selected for modelling of the entire cultivation period (Table 2). At both locations, biomass was assumed to be harvested, chipped by a maize chopper and transported to the refinery by truck. For transport distances, the area required to collect sufficient biomass was estimated based on the annual demand of the refinery, yield per hectare, fraction of arable land in the corresponding regions (Wales and Baden-Württemberg), and the fraction of arable land on which miscanthus is grown (assumed). The plant is assumed to be located in the centre of the area, and the radius of a circle with the corresponding area was considered as transport distance from the field to the refinery (Table 3). Postharvest losses were assumed to be up to 5%.

**Table 1** Fertilizer and agrochemical inputs for miscanthus cultivation (Lewandowski et al., 2016)

|                        | Aberystwyth | Stuttgart |
|------------------------|-------------|-----------|
| Propagation material   | 15,000.00   |           |
| (plugs/ha)             |             |           |
| N, as CAN (kg/ha * a)  | 60.00       |           |
| K$_2$O, as K$_2$SO$_4$ (kg/ha * a) | 120.00   |           |
| P$_2$O$_5$, as Ca(H$_2$PO$_4$)$_2$ (kg/ha * a) | 30.00 |           |
| Pesticides (kg/ha * a) | 0.28        |           |
| Yield (kg DM/ha * a)   | 9,745.00    | 15,316.00 |

**Table 2** Biomass composition of miscanthus (OPM6) at the two analysed sites (van der Weijde, Kiesel, et al., 2017)

|       | Aberystwyth | Stuttgart |
|-------|-------------|-----------|
| Cellulose (g/kg DM) | 486.44      | 516.71    |
| Hemicellulose (g/kg DM) | 278.85      | 273.56    |
| Lignin (g/kg DM)      | 95.27       | 100.36    |
sugar recovery efficiencies after pretreatment and hydrolysis were used for further modelling (Table 4). The selection was based on the objective of choosing conditions characterized by low residence time, reasonable chemical input and high solid loading.

Assumptions regarding the heat energy requirements for the conversion processes were based on a process simulation by Kumar and Murthy (2011). The reported values for dilute acid (DA), dilute alkali (DAH) and liquid hot water (LHW) were adjusted by means of an estimation model to reflect the pretreatment temperatures given in the references (Table 4; Mafe, Davies, Hancock, & Du, 2015). It was assumed that all treatments were performed at a solid loading of 20%, and that this assumption had no effects on the reported sugar recovery. Electricity requirements, which also include energy for pre-processing of the biomass, were adopted without further adjustment.

For the dilute sulphuric acid treatment (DA), a pretreatment mixture containing 0.73 wt% sulphuric acid was considered (Guo, Zhang, Ha, Jin, & Morgenroth, 2012). The use of diluted acid affects the biomass primarily by hydrolysis of the hemicellulose fraction. As a result, monosaccharides for fermentation are obtained and cellulose is more easily accessible for enzymatic saccharification (Pedersen & Meyer, 2010). After the acid-catalysed reaction, neutralization agents such as sodium hydroxide or ammonium are required. In this study, application of lime was assumed, which results in the formation of gypsum. Gypsum could be considered a by-product. However, due to its impurity, this is unlikely and landfilling was assumed (Humbird et al., 2011). In the dilute alkali pathways (DAH), a concentration of 2.5 wt% was assumed for the pretreatment mixture (Haque et al., 2013). During the alkaline treatment, cross-links between hemicelluloses and lignin are cleaved resulting in the solubilization of lignin. The efficient removal of lignin renders cellulose and hemicellulose susceptible to enzymatic hydrolysis and enables high hydrolysis yields (Silverstein, Chen, Sharma-Shivappa, Boyette, & Osborne, 2007). After the pretreatment, the slurry was assumed to be neutralized by addition of sulphuric acid (Kumar & Murthy, 2011). In liquid hot water treatment (LHW), the water itself (and acetic acid released from hemicellulose) acts as a catalyst and solubilizes hemicellulose. Neither addition of further reactants nor a neutralization step is required (Agbor et al., 2011).

Irrespective of the applied pretreatment, treated substrates are hydrolysed by cellulas. For hydrolysis, a generic enzyme loading of 0.02 mg protein per gram of cellulose in the hydrolysate was assumed (Humbird et al., 2011; Kumar & Murthy, 2011). The inventory for enzyme provision was based on a recent publication by Gilpin and Andrae (2017). Fermentation efficiencies of 95% for glucose and 70% for xylose were adopted from Kumar and Murthy (2011). Yeast application at a dosage of 2.5 kg/t DM was assumed, and impacts were calculated considering key inputs of yeast production as reported by Dunn, Mueller, Wang, and Han (2012).

The resulting ethanol yields were calculated considering efficiencies of sugar recovery after pretreatment and hydrolysis as given in the references (Table 4), as well as the fermentation efficiencies (Kumar & Murthy, 2011), and the specific biomass compositions for Aberystwyth and Stuttgart (Table 5).

After fermentation, the product is distilled and purified by means of two distillation columns and separation via a molecular sieve (Kumar & Murthy, 2011). The bottom stream of the first distillation column is separated into a solid and a liquid fraction. The liquids are partially evaporated and the evaporation condensate, along with the liquid fraction of the first separation, is treated in anaerobic digestion. Methane generation was estimated on the basis of the chemical oxygen demand (COD; 0.239 kg/kg COD; Kumar & Murthy, 2011), which was calculated based on component-specific degradation efficiencies (Barta, Reczey, & Zacchi, 2010). In order to ensure sufficient nutrient supply

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**Table 3** Data for the estimation of transport distances for raw biomass

|                        | Aberystwyth | Stuttgart |
|------------------------|-------------|-----------|
| Annual biomass demand (kg) | 250,000,000 | 250,000,000 |
| Yield (kg DM/ha * a)     | 9,745       | 15,316    |
| Fraction arable land    | 11.89%      | 22.93%    |
| Arable land available for Miscanthus cultivation | 10% | 10% |
| Collection area (km²)   | 21,582      | 7,120     |
| Radius (km)             | 82.88       | 47.61     |

Source: aWelsh Government (2018). bStatistisches Landesamt Baden-Württemberg (2017).

**Table 4** Selected conditions for pretreatment of miscanthus.

| Pretreatment | Conditions | Sugar recovery based on raw material | Reference       |
|--------------|------------|--------------------------------------|-----------------|
| Dilute alkali| 105°C; 2.5 wt% NaOH | Glucose: 76.42% Xylose: 35.27% | Haque et al. (2013) |
| Dilute acid  | 150°C; 0.73 wt% H₂SO₄ | Glucose: 69.35% Xylose: 57.00% | Guo et al. (2012) |
| Liquid hot water | 200°C | Glucose: 78.38% Xylose: 49.84% | Khullar (2012) |
during anaerobic digestion, supplementation of ammonia (4.5 g/kg COD) and phosphoric acid (2.0 g/kg COD) was considered (Barta et al., 2010). Wastewater from anaerobic digestion contains residual organic matter and is treated by aerobic digestion (Barta et al., 2010; Kumar & Murthy, 2011). The process was modelled by means of a standard ecoinvent process for the treatment of wastewater from a biomass refinery (Wernet et al., 2016). For DAH pathways, it was assumed that cleaned water is additionally sent to a reverse osmosis system in order to remove sodium sulphate formed during the neutralization of sodium hydroxide after the pretreatment. Biogas, sludge from wastewater treatment, as well as solids and evaporate from the distillation bottom stream are combusted for heat and power generation (heating value calculation for the residues was based on data provided in Humbird et al. (2011)). A boiler efficiency of 70% was assumed and the generated steam covered the process requirements in the first instance (Kumar & Murthy, 2011). Excess energy was converted to electricity with an assumed conversion efficiency of 30% and was primarily consumed within the refinery (Kumar & Murthy, 2011). Surplus production is exported to the corresponding national grid, substituting the local electricity mix according to the ecoinvent database (Wernet et al., 2016).

As no specific data for emissions from the combustion of miscanthus fermentation residues were available, emission factors were sourced from an ecoinvent dataset reporting on the combustion of bagasse (Wernet et al., 2016). The remaining ash is rich in nutrients and could be reused as a soil additive (Risse & Gaskin, 2013; Wagner & Lewandowski, 2017). Consequently, the present study assumed disposal in a landfill.

### 2.2.3 Use phase and comparison with the petrol system

Impacts of the distribution of ethanol to service stations were modelled according to an ecoinvent standard process and include transport by ship, train and truck (Wernet et al., 2016). Emissions from the use phase of ethanol were estimated according to the EMEP/EEA air pollutant emission inventory guidebook and included carbon monoxide (CO), nonmethane volatile organic compounds (NMVOC), mono nitrogen oxides (NOx), particulate matter, N2O, NH3, lead and fossil CO2 from the combustion of lubricants (EEA, 2017). As comprehensive data on combustion emissions of pure ethanol were not available, the closest alternative, E85, was selected. For comparison of the ethanol pathways and the fossil reference, specific emissions from the use phase were assumed in accordance with the EMEP/EEA guidebook (EEA, 2017), considering characteristics of emissions and engine efficiencies for both fuels in medium passenger cars. For the fossil reference system, standard ecoinvent inventory data was assumed for petrol production and provision (Wernet et al., 2016).

### 2.2.4 Construction-and transport-related impacts

Impact data for plant construction include data for major construction inputs (concrete, steel and energy) and are derived from the ecoinvent database (Wernet et al., 2016). Inputs of the agricultural and the refinery stage were

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**Table 5** Major inputs and outputs of the conversion pathways. Ethanol yield is corrected for a purification efficiency of 98.76% (according to Kumar & Murthy, 2011)

|                  | Dilute alkali | Dilute acid | Liquid hot water |
|------------------|---------------|-------------|------------------|
| **Inputs**       |               |             |                  |
| Water requirements (kg/t DM) | 1552.7        | 1506.6      | 1,490.8          |
| Sulphuric acid (kg/t DM)        | 98.0          | 29.2        | –                |
| Sodium hydroxide (kg/t DM)      | 80.0          | –           | –                |
| Calcium hydroxide (kg/t DM)     | –             | 22.1        | –                |
| Energy (MJ/t DM)                | 4,498.3       | 4,579.5     | 5,090.9          |
| Electricity (kWh/t DM)          | 141.5         | 133.0       | 133.7            |
| Enzymes (kg/t DM)               | 10.1          | 9.5         | 9.7              |
| **Outputs**                  |               |             |                  |
| Ethanol (kg/t DM)              | 248.0         | 236.4       | 251.8            |
| Electricity surplus (kWh/t DM) | 119.5         | 105.9       | 110.5            |

*Note. ABER: Aberystwyth; STR: Stuttgart.*
assumed to be transported by truck (EURO 5 emission stage) over a distance of 150 km (Wernet et al., 2016).

3 | RESULTS

Aggregated results for the environmental performance of the analysed pathways are displayed in Table 6. The individual contribution of the cultivation and refinery phase, as well as impacts from biomass transport and ethanol distribution and use are presented in Figures 2–4. Negative scores represent environmental savings through the generation of surplus electricity from the combustion of lignin and other residues.

3.1 | Contribution analysis of the impact results

3.1.1 | Climate change and fossil depletion

Climate change (CC) impacts range from 29.0 and 30.6 kg per functional unit for LHW and DA in Stuttgart to 61.0 kg CO₂ eq for DAH in Aberystwyth (Table 6). Miscanthus cultivation and harvest account for 16.5–18.0 kg CO₂ eq in Stuttgart and 24.0–26.3 kg CO₂ eq in Aberystwyth (Figure 2a). Main drivers are emissions from fertilizer production and fertilizer-induced N₂O emissions. With respect to the pretreatment pathways, LHW and DA result in lower emissions than DAH. For LHW, conversion-related emissions (including enzymes) amount to approximately 16 kg CO₂ eq, whereas values are significantly higher for DA and DAH at 21 and 41 kg CO₂ eq, respectively. This difference is mainly due to the applied reactants including sulphuric acid, lime and sodium hydroxide. The latter accounts for approximately a third of the overall net score of the alkaline pathways.

A major share of emissions in all pathways is caused by enzyme-related impacts. Between 12.2 and 15.1 kg CO₂ eq are introduced by upstream burdens which are mainly due to the use of large quantities of heating and cooling energy during enzyme provision (supplied by natural gas). As greenhouse gas emissions are strongly correlated to fossil fuel use, results for the impact category fossil resource depletion (FD) show similar patterns as observed for GHG emissions (Figure 2b). However, for FD, the overall contribution of the agricultural stage is less pronounced. In both categories, credits from surplus electricity partially offset burdens from the process. Savings are smaller in Aberystwyth than in Stuttgart, as credits are inversely related to the carbon intensity of the local grid. In addition, lignin content of OPM 6 is lower in Aberystwyth, resulting in a reduced heating value of the fermentation residues. Irrespective of the site, DAH provides larger credits than LHW and DA. This is due to the pretreatment-specific degradation efficiencies and energy requirements that both are lower for DAH compared to the other treatments.

3.1.2 | Freshwater and marine eutrophication

Eutrophication is related to the release of nutrients, such as nitrogen and phosphorus to the environment (Goedkoop et al., 2013). For the analysed pathways, net overall

| Impact category                  | Unit  | DA STR | DA ABER | DAH STR | DAH ABER | LHW STR | LHW ABER |
|----------------------------------|-------|--------|---------|---------|----------|---------|----------|
| Agricultural land occupation     | m² a  | 111.26 | 176.85  | 114.65  | 182.63   | 104.41  | 166.38   |
| Climate Change                   | kg CO₂ eq | 30.59  | 40.48   | 50.00   | 61.01    | 28.96   | 37.84    |
| Fossil resource depletion        | kg oil eq | 6.56   | 8.02    | 13.65   | 15.44    | 6.06    | 7.36     |
| Freshwater ecotoxicity           | kg 1,4-DB eq | 2.51   | 3.24    | 6.61    | 7.58     | 2.39    | 2.98     |
| Freshwater eutrophication        | kg P eq | 0.1E-02 | 1.8E-02 | 1.3E-02 | 3.1E-02 | 0.5E-02 | 1.7E-02 |
| Human toxicity                   | kg 1,4-DB eq | 151.63 | 196.92  | 184.28  | 233.68   | 144.08  | 184.94   |
| Marine ecotoxicity               | kg 1,4-DB eq | 1.67   | 2.01    | 4.99    | 5.52     | 1.60    | 1.84     |
| Marine eutrophication            | kg N eq | 0.28   | 0.59    | 0.30    | 0.61     | 0.27    | 0.55     |
| Natural land transformation      | m²     | 1.8E-02 | 2.1E-02 | 2.3E-02 | 2.6E-02 | 1.6E-02 | 1.9E-02 |
| Ozone depletion                  | kg CFC-11 eq | 3.5E-06 | 3.2E-06 | 2.0E-05 | 2.0E-05 | 3.0E-06 | 2.6E-06 |
| Photochemical oxidant formation  | kg NMVOC | 0.28   | 0.30    | 0.35    | 0.38     | 0.25    | 0.29     |
| Terrestrial acidification        | kg SO₂ eq | 0.33   | 0.37    | 0.54    | 0.58     | 0.29    | 0.33     |
| Terrestrial ecotoxicity          | kg 1,4-DB eq | 4.5E-02 | 5.2E-02 | 5.0E-02 | 5.9E-02 | 4.2E-02 | 4.9E-02 |

Note. ABER: Aberystwyth; DA: Dilute sulphuric acid; DAH: Dilute sodium hydroxide; LHW: Liquid hot water; STR: Stuttgart.
freshwater eutrophication (FE) potential ranges from 0.001 to 0.031 kg P eq (Figure 2c). Major burdens for DAH pathways are due to sodium hydroxide production, which accounts for more than a third of the gross score (excluding credits). For all pathways, substantial burdens are derived from enzyme production and fertilizer-related emissions. The latter represent the major contributor to the agricultural stage. With regard to credits from surplus electricity, substantial differences are observed between the two locations. In Stuttgart, credits almost offset the impacts of the DA and LHW pathways (and a considerable amount of DAH), resulting in net results close to zero for DA and LHW.

For marine eutrophication (ME), credits from electricity production are marginal compared to the major impacts of the cultivation stage at both locations (Figure 2d). Irrespective of the pathway, between 74% and 87% of the overall impacts are related to feedstock production. In Aberystwyth, impact scores for marine eutrophication are found to be about twice as high as in Stuttgart. The difference is more pronounced than the yield variation between the locations. This is due to the quantity of nitrate losses, which depends on site-specific characteristics. Only minor differences are found between the figures for the distinct pretreatment pathways at the same location and can be attributed to differences in the pathway-specific biomass requirements.
3.1.3 | Terrestrial acidification

Terrestrial acidification (TA) accounts for emissions with the potential to change soil acidity (Goedkoop et al., 2013). In the present study, combustion of fermentation residues and the use phase of ethanol contribute between 20% and 39% to the overall burdens (Figure 2e, combustion of ethanol is included in Distribution & Use). Impacts from ethanol combustion are mainly due to emissions of nitrogen oxides and ammonia, as no sulphur emissions from ethanol combustion were assumed. Provision of sulfuric acid accounts for about 10% of the TA potential in the DA pathways. In DAH pathways, sodium hydroxide and sulphuric acid account for more than 40% of the net score. Enzyme-related impacts are mainly due to upstream burdens from glucose production (from maize starch). Emissions during the agricultural stage are predominantly related to fertilization, and the magnitude of the impacts reflects the variations in yields and biomass requirements of the individual pathways.

3.1.4 | Human toxicity and ecotoxicity (freshwater, marine and terrestrial)

The persistence and effect of chemicals in the environment, as well as on humans (human toxicity) is described by the impact categories human toxicity (HT) and ecotoxicity (freshwater [FET]/terrestrial [TET]/marine [MET]) (Goedkoop et al., 2013). In all production pathways, the overall scores of HT are dominated by the agricultural phase and...
the disposal of ash via landfilling (Figure 3a, included in Process, other than enzymes). Impacts of the cultivation phase result mainly from fertilization-related emissions of phosphorus.

For FET and MET, impacts related to the cultivation stage are less prominent. In both categories, benefits from electricity generation are marginal contributors to the overall score for the Stuttgart-based pathways. Here, credits partially offset impacts, resulting in lower net scores for the German pathways. In general, results for both categories show a high degree of similarity, with poorer overall scores for pathways in Aberystwyth (Figure 3b,c).

Similarly, by-product credits are insignificant for TET. Irrespective of site and treatment, enzyme provision (mainly upstream burdens from maize starch and glucose production) is responsible for the largest burdens. Cultivation-related impacts result mainly from fertilizer application (Figure 3d).

### 3.1.5 Photochemical oxidant formation

The reaction of nitrogen oxides (NOx) and volatile organic compounds results in the formation of photochemical oxidants (POF). Reported as NMVOC, oxidants are considered a rough measure of air pollution (Goedkoop et al., 2013). For the analysed pathways, a major share (44%–64%) of the overall scores is due to residue and ethanol combustion. In addition, fugitive ethanol emissions during transportation and storage (0.5%, according to Wernet et al., 2016) account for about 5% of the net scores (Figure 3e; combustion of ethanol is included in Distribution & Use).

### 3.1.6 Ozone depletion

The emission of chlorinated or brominated substances contributes to the destruction of the Earth’s stratospheric ozone layer (Goedkoop et al., 2013). Contributions from the agricultural stage are mainly due to the production of fertilizers. The provision of enzymes and chemicals causes additional burdens throughout the analysed pathways (Figure 4a). However, due to extensive upstream burdens, sodium hydroxide represents the largest contributor, accounting for up to 60% of the total impacts of the DAH pathways. As a result, total scores of DAH pathways are almost three times higher than for the other treatments.

### 3.1.7 Agricultural land occupation and natural land transformation

Land use indicators, such as agricultural land occupation (ALO) and natural land transformation (NLT), account for...
the potential damage to ecosystems due to occupation and transformation (Goedkoop et al., 2013). Impacts of agricultural land occupation are predominantly derived from the cultivation phase (Figure 4b). With respect to geographical differences between the production pathways, variations in results are caused by yield differences in miscanthus cultivation between the two sites. In the same geographical boundary, variation is primarily caused by the pathway-specific feedstock requirements. In comparison with ALO, sources of impacts in natural land transformation are more diverse (Figure 4c). In all analysed pathways, enzyme production is a major contributor, which is due to upstream impacts from glucose production and energy provision from natural gas. In addition, landfilling of ash contributes substantial impacts to the overall results of all pathways (Figure 4c; landfilling of ash is included in Processes, other than enzymes).

3.2 Normalization

Normalized impacts for the analysed pathways are presented in Figure 5 and can be divided into three groups according to their magnitude. The first group includes FET, MET and HT, representing the categories with the highest relevance. The second group encompasses the eutrophication (FE and ME), as well as the land use categories (NLT and ALO). The remaining categories (CC, TA and others) constitute the third group. This ranking is observed for all analysed pathways with only few exceptions. For instance, the relative magnitude of FE results is substantially lower for LHW and DA pathways in Stuttgart than for the corresponding pathways in Aberystwyth. This is due to the site-specific credits from electricity exports that are substantially higher in Germany. Overall, the results indicate that the production of miscanthus ethanol is accompanied by significant impacts in the toxicity categories. For HT in particular, consistently high impacts are observed for all analysed pathways. Overall, these considerations underline the importance of including and focusing on more than just GHG emissions in the sustainability analysis of miscanthus ethanol.

3.3 Comparison with petrol

In comparison with the fossil reference petrol, the analysed ethanol pathways are favourable in three out of 13 analysed impact categories (Figure 6 and Figure 7). These include CC, FD and NLT. In addition, DA and LHW pathways at both locations result in lower ozone depletion potential than the fossil reference. In eight categories, including ALO, FET, HT, MET, ME, POF, TA and TET, petrol has substantially lower impacts than the analysed miscanthus pathways. In the remaining category, FE, a single pathway, DA in Stuttgart outperforms the fossil reference, which is due to credits derived from electricity export. Overall, it is apparent that the replacement of petrol with miscanthus ethanol is associated with environmental advantages and disadvantages. Although results of the different pathways show clear variation, the overall pattern is typical for bioenergy: Reduction in CC and FD are accompanied by detrimental effects in most of the other impact categories. The largest discrepancies in the environmental performance of miscanthus ethanol and petrol are observed for ALO, HT and ME, mainly due to impacts from residue combustion and fertilizer application in the biofuel pathways.

4 DISCUSSION

The following section examines the results with regard to uncertainties in the inventory and considers potential implications. First, the contributions of feedstock cultivation and biomass variability to the environmental impacts are
discussed. Second, the conversion process is addressed. Finally, the potential of miscanthus ethanol to mitigate GHG emissions is considered and corresponding trade-offs with other environmental aspects are investigated.

4.1 | Miscanthus cultivation

Irrespective of location and pathway, feedstock cultivation is a substantial contributor to the overall environmental impacts. For ALO, ME and to a lesser degree FE and HT, agricultural activities account for the largest burdens. For most impact categories, the regional variation reflects the yield differences between the marginal site in Aberystwyth and the more fertile site in Stuttgart (9.7 t DM/ha vs. 15.3 t DM/ha). Thus, as previously reported (Gerbrandt et al., 2016; Meyer et al., 2017; Schmidt et al., 2015), biomass yield is one of the major factors determining the overall performance of ethanol production from miscanthus. Biomass yield potential therefore needs to be taken into consideration when deciding whether miscanthus production on a specific site is recommendable from an environmental point of view (also see Wagner et al., 2019). For newly developed miscanthus accessions, such as OPM 6, long-term experience with regard to yield stability over cultivation periods of 20 years and across locations is scarce and represents a source of uncertainty in the model.

Major impacts during feedstock cultivation result from fertilizer production and fertilization-related emissions. For instance, the most relevant impact categories, including HT, FET and MET, depend mainly on fertilizer dosage. In addition, fertilizer-induced N₂O emissions contribute significantly to the global warming potential of the biomass

FIGURE 6 Comparison of the environmental performance of bioethanol production pathways and the fossil reference system (petrol). The option scoring highest in each category equals 100%. Dilute sulphuric acid (DA), dilute sodium hydroxide (DAH), liquid hot water (LHW) indicate the pretreatment pathway. Stuttgart (STR) and Aberystwyth (ABER) represent the location of feedstock cultivation. Impact categories include agricultural land occupation (ALO), climate change (CC), fossil resource depletion (FD), freshwater ecotoxicity (FET), freshwater eutrophication (FE), human toxicity (HT), marine ecotoxicity (MET)

FIGURE 7 Comparison of the environmental performance of bioethanol production pathways and the fossil reference system (petrol). The option scoring highest in each category equals 100%. Dilute sulphuric acid (DA), dilute sodium hydroxide (DAH), liquid hot water (LHW) indicate the pretreatment pathway. Stuttgart (STR) and Aberystwyth (ABER) represent the location of feedstock cultivation. Impact categories include marine eutrophication (ME), natural land transformation (NLT), ozone depletion (OD), photochemical oxidant formation (POF), terrestrial acidification (TA) and terrestrial ecotoxicity (TET)
production stage. The present model assumed generic amounts of fertilizers for both sites, although in practice inputs are adjusted to local conditions and are often lower than assumed here. In addition, the IPCC N₂O emission factor may overestimate the actual emissions, as has been shown for N₂O emission measurements in rape seed (Ruser et al., 2017). Accordingly, crop and site-specific assessments are required.

Impact categories related to nitrate emissions tend to have higher results for the feedstock cultivation in Aberystwyth than in Stuttgart. In particular, for marine eutrophication, impacts are higher at the site in the UK than for the biomass production in Germany. This not only results from lower biomass yields but also from conditions in Aberystwyth, which are conducive to nitrate leaching, including higher precipitation rates and lower soil clay content. As a standard model for the calculation of nitrate loss was applied (Faist Emmenegger et al., 2009), the actual emissions may be overestimated for miscanthus. Nitrate leaching tends to be lower under established perennial crops than for the cultivation of annuals. The presence of a permanent root system ensures a continuous nutrient uptake throughout the year. In addition, nitrogen mineralization is reduced, as no tilling is required during the cultivation period of miscanthus (Jørgensen, 2005).

The present study did not consider soil carbon changes, although miscanthus holds potential for carbon sequestration if the cultivation takes place on arable land. Estimates for soil carbon increases range between 0.7 and 2.2 t C/(ha * a) (equivalent to 2.6–8.0 t CO₂/(ha * a)) (McCalmont et al., 2017; Milner et al., 2016). Even the assumption of a lower sequestration rate of 0.7 t C/(ha * a) would result in substantial GWP credits for the production system. For production in Stuttgart and Aberystwyth, savings of 24.4–26.6 kg CO₂/GJ and 40.2–43.8 kg CO₂/GJ, respectively, could be expected. Considering the difference in the sequestration potentials, it becomes apparent that CO₂ uptake could partially offset higher GHG impacts of the miscanthus cultivation on marginal land. However, these effects depend on various factors, such as previous land use, climatic conditions and organic carbon content of the soil (McCalmont et al., 2017; Zan, Fyles, Girouard, & Samson, 2001). Overall, we conclude that, for the miscanthus production step, the choice of high-yielding genotypes with low requirements for nitrogen fertilization substantially improves the environmental performance of feedstock supply.

### 4.2 | Biomass characteristics

Ethanol yield is a key factor in lignocellulosic ethanol production (Gerbrandt et al., 2016). The potential yield depends on the cellulose and hemicellulose content of the biomass, the effectiveness of pretreatment and hydrolysis as well as on the fermentation efficiency. It should be noted that the assumed efficiencies of pretreatment, hydrolysis and fermentation were drawn from literature. It should also be noted that the considered values are clearly lower than commonly assumed in environmental life cycle or techno-economic analyses (Wilosso et al., 2012). Accordingly, the ethanol output of the production pathways could be substantially higher, thereby lowering impacts from the agricultural and refinery stage. However, lower recovery efficiencies are also balanced out to a certain extent in the overall impacts, as more organic matter is available for combustion and electricity generation.

Moreover, these efficiencies do not represent specific values for the considered miscanthus composition. Lignocellulosic conversion efficiencies underlie a high degree of uncertainty and variability due to interactions of biomass composition and degradability (van der Weijde et al., 2016; van der Weijde, Dolstra, et al., 2017). Torres et al. (2016) indicated that improved lignocellulose degradability benefits the overall environmental performance of ethanol production when higher yields and lower thermochemical requirements are to be expected. In miscanthus, lignin content has been shown to be negatively correlated to lignocellulose degradability (van der Weijde et al., 2016), whereas increased hemicellulose contents regularly contribute to improved digestibility (Xu et al., 2012).

Considering the given biomass quality in the present study, improved digestibility could be expected for OPM 6 in Aberystwyth compared to Stuttgart. Potential effects were explored in a basic scenario analysis assuming reduced thermochemical requirements during the pretreatment (ΔT = −20°C) and increased cellulose conversion. Overall, improved lignocellulose degradability and the corresponding reduction in pretreatment requirements indicate potential to reduce impacts in comparison with the base case. However, the environmental savings due to improved digestibility have only minor impact on the overall results of the present study. Accordingly, we conclude that site-dependent variation in biomass composition and the corresponding variability in lignocellulose degradability are negligible for the comparison of the environmental performance of ethanol production in Aberystwyth and Stuttgart as analysed in the present study.

### 4.3 | Conversion

Conversion processes (including enzymes, chemicals, and energy inputs) are main contributors to the total scores for most impact categories. Site-related variations assessed through modelling here are rather small and are mainly a consequence of differences in energy provision for enzyme production on the local scale, as well as of distinct
feedstock requirements due to the site-characteristic biomass composition. Major differences between the effects of pretreatment options modelled here were due to the application of chemicals. Sodium hydroxide is the most influential, as it introduces substantial upstream burdens and reduction in consumption could result in significant environmental savings for the DAH production pathways. Lower concentrations of sodium hydroxide have been shown to be effective as well. However, the corresponding processes regularly require higher temperatures or extended residence times (e.g., Lewandowska et al., 2016; Świątek, Lewandowska, Świątek, Bednarski, & Brzozowski, 2014). In addition, reutilization of the pretreatment solution is a promising option and has been successfully demonstrated for miscanthus resulting in substantial savings of sodium hydroxide (Han, Kim, Cho, Choi, & Chung, 2016).

Regardless of the treatment pathway, enzyme production is a major contributor to the conversion-related impacts for most categories. This is in line with other studies, which found life cycle results of lignocellulosic ethanol sensitive to impacts from enzyme provision (Gerbrandt et al., 2016; McKechnie et al., 2015; Wang, Littlewood, & Murphy, 2013). Related impacts are subject to uncertainty, as reported burdens for enzyme provision vary widely. For instance, figures for CO$_2$ equivalents range from 2.5 to 22.0 kg CO$_2$ eq/kg enzyme (Agostinho, Bertaglia, Almeida, & Giannetti, 2015; MacLean & Spatari, 2009). In the present study, a recent inventory published by Gilpin and Andrae (2017) was used. With respect to the comprehensive nature of the inventory and GHG emissions of approximately 10 kg CO$_2$ eq/kg enzyme, the data are considered to offer reasonable estimates. However, uncertainty remains high, which is also due to varying assumptions in literature with respect to the enzyme loading (Gerbrandt et al., 2016). As lignin adheres to enzymes and renders them ineffective, the amount of lignin in enzymatic hydrolysis can be a decisive factor (X. Zhao, Zhang, & Liu, 2012). A substantial lignin removal, as commonly given in alkaline pretreatments, could allow lower enzyme dosage and thereby contribute to substantial reductions in enzyme-related impacts for DAH (Cha et al., 2014; Chang & Holtzapple, 2000). In addition, impacts related to enzyme production could be reduced by replacing the carbon source (cornstarch glucose in the present study) with sugar cane molasses or lignocellulosic materials.

4.4 | Energy considerations

Lignin contributes a major part to the overall heating value of the fermentation residues. The reported lignin contents in the present study are substantially lower than commonly assumed in techno-economic or life-cycle analyses of miscanthus (Boakye-Boaten, Kurkalova, Xiu, & Shahbazi, 2017; Scown et al., 2012). This is due to the fact that these numbers represent measurements of acid detergent lignin (ADL), which are significantly lower than the corresponding values for Klason lignin (van der Weijde et al., 2016). For switchgrass, it has been reported that Klason lignin represents the heating value more accurately than ADL (Jung, Varel, Weimer, & Ralph, 1999). Accordingly, the present model might underestimate the heating value of the fermentation residues and credits from by-product generation alike.

Credits from by-product generation are strongly influenced by the characteristics of the local electricity generation. A comparison of impacts for the production of electricity in Germany, UK and the European Union presents large discrepancies in the various impact categories between local grids (Wernet et al., 2016). For instance, in Germany impacts for freshwater and marine ecotoxicity, as well as for freshwater eutrophication are about five times higher than for the British electricity mix. As credits are inversely related to the magnitude of the impacts, these considerations largely influence the overall results with respect to the regional variation and can be a decisive factor in the assessment of these categories.

4.5 | Emissions from combustion of fermentation residues and ethanol

Impacts from the combustion of fermentation residues and the use phase of ethanol contribute substantially to POF. Emissions from residue combustion were modelled based on factors for the combustion of sugarcane bagasse and, consequently, are subject to uncertainty due to the differences in composition. Similarly, impacts on POF from the use phase of ethanol might be overestimated, as the current study assumed emissions representing the combustion of an 85%-ethanol blend. Commonly, carbon monoxide formation is lower for pure ethanol due to a more efficient combustion in the engine (Pagliuso, 2010). Nevertheless, impacts from the use phase of ethanol are less relevant than the emissions from the combustion of fermentation residues and have only limited impact on the overall results.

4.6 | Potential for GHG mitigation and overall environmental performance of miscanthus ethanol

According to the presented results, miscanthus ethanol provides a suitable option for the mitigation of GHG emissions. This applies to pathways characterized by feedstock cultivation on both average and marginal sites. With respect to the different treatments, DAH offers the smallest savings, whereas DA and LHW have substantially higher reduction potentials. The present study indicates a lower
saving potential than those reported by previous publications on miscanthus ethanol (Falano et al., 2014; Scown et al., 2012; Wang, Han, Dunn, Cai, & Elgowainy, 2012). Reasons include the consideration of diverse by-products, for example chemicals such as acetic and lactic acid (Falano et al., 2014) or the assumption of substantial electricity exports. In particular, assessments based on the National Renewable Energy Laboratory’s report on an integrated lignocellulosic ethanol refinery (Humbird et al., 2011), commonly assume a fixed rate for electricity exports (0.090 MJ/MJ ethanol; Scown et al., 2012). In the present study, smaller electricity surpluses were computed, which partially explains the observed differences.

In order to promote the use of energy from renewable resources, the European Union has established the Renewable Energy Directive (RED). The legislation includes GHG emission reduction targets for the provision and use of sustainable biofuels. Potential savings for miscanthus ethanol as modelled in the present study are shown in Table 7, assuming a reference value of 83.8 kg CO₂ eq/GJ for the petrol system (Directive/28/EC/2009: Annex V). Using this reference, only DA and LHW in Stuttgart would meet the 60% target. DA and LHW in Aberystwyth would fulfil the 50% target. If, as proposed in the calculation methodology (Directive/28/EC/2009 2009), a low soil carbon sequestration rate (0.7 t C/(ha * a)) is assumed, all analysed pathways would meet the requirements (Table 7).

Overall, the study has shown that miscanthus ethanol holds potential for the reduction of GHG emissions in the transportation sector and that this applies for feedstock cultivation at both average-yielding sites and marginal land. The magnitude of savings depends strongly on site-specific factors such as biomass yield and soil carbon changes. For the marginal site in particular, carbon sequestration is a major parameter for the eligibility under the European legislation. In addition, results across all impact categories reveal clear differences between the various conversion pathways. DAH proved to be the least promising option due to substantial burdens from sodium hydroxide counter-balancing the advantages of lower pretreatment requirements. In general, liquid hot water treatment has come with the lowest impacts across the analysed categories. Although mitigation of GHG emissions is the main rationale for the promotion of biofuels, it should be noted that ethanol from miscanthus has a significantly poorer performance than petrol in eight of 13 analysed impact categories, irrespective of the pretreatment and conversion pathway chosen. This is mainly due to inputs such as chemicals or cellulytic enzymes, that are necessary for the conversion of lignocellulosic feedstocks via biochemical pathways. By analysing the results of the toxicity and eutrophication categories, it becomes apparent that climate change mitigation through the use of miscanthus ethanol is accompanied by potentially negative impacts on human health and ecosystems.

### Table 7: Potential greenhouse gas savings of miscanthus ethanol in comparison with a fossil fuel comparator, as proposed in the Renewable Energy Directive (RED) (Directive/28/EC/2009 2009). Two cases are presented, examining implications of carbon sequestration accounting

|                  | Dilute acid (%) | Dilute alkali (%) | Liquid hot water (%) |
|------------------|----------------|------------------|---------------------|
|                  | DA STR | DA ABER | DAH STR | DAH ABER | LHW STR | LHW ABER |
| Savings according to RED baseline (83.8 kg CO₂ eq) | 64     | 52     | 40      | 27      | 65      | 55       |
| Savings according to RED (incl. 0.7t C/(ha * a)) | 95     | 103    | 72      | 79      | 95      | 103      |

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