Give them credit—the greenhouse gas performance of regional biogas systems

Sinéad O'Keeffe1 | Uwe Franko2 | Katja Oehmichen3 | Jaqueline Daniel-Gromke4 | Daniela Thrän1,3

1Department of Bioenergy, Helmholtz Centre for Environmental Research (UFZ), Leipzig, Germany
2Department of Soil System Science, Helmholtz Centre for Environmental Research (UFZ), Halle/Saale, Germany
3Bioenergy Systems Department, Deutsches Biomasseforschungszentrum (DBFZ), Leipzig, Germany
4Biochemical Conversion Department, Deutsches Biomasseforschungszentrum (DBFZ), Leipzig, Germany

Correspondence
Sinéad O’Keeffe, Department of Bioenergy, Helmholtz Centre for Environmental Research (UFZ), Leipzig, Germany. Email: sinead.o-keeffe@ufz.de

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Abstract
Anaerobic digestion to produce biogas is an important decentralised renewable energy technology. Production varies extensively between different countries and within countries, as biogas production is heavily dependent on local and regional feedstocks. In Germany, distinct regional differences can be observed. Therefore, understanding the kinds of biogas systems operating within a region is crucial to determine their greenhouse gas (GHG) mitigation potential and carbon neutrality. This is the first study to conduct an integrated life cycle assessment of biogas configurations in the landscape (biogas plants and their biomass catchments) for an entire region. RELCA a ‘REgional Life Cycle inventory Assessment’ approach was used to model the GHG mitigation potential of 425 biogas plants in the region of Central Germany (CG), aggregated to nine biogas clusters, based on feedstock mix (e.g., animal manures and energy crops) and installed capacity. GHG emission profiles were generated to compare and to identify the role of GHG credits and size of installed capacity on the mitigation performance of the regional biogas clusters. We found that smaller scaled slurry dominant clusters had significantly better GHG mitigation performance (−0.1 to −0.2 kg CO2eq kWhel−1), than larger energy crop dominant (ECdom) clusters (0.04–0.16 kg CO2eq kWhel−1), due to lower cultivation emissions and larger credits for avoided slurry storage. Thus, for the CG region larger ECdom clusters should be targeted first, to support GHG mitigation improvements to the overall future electricity supplied by the regional biogas systems. With the addition of GHG credits, the CG region is producing biogas with GHG savings (−0.15 kg CO2eq kWhel−1, interquartile range: 0.095 kg CO2eq kWhel−1). This infers that biogas production, as a waste management strategy for animal manures, could have important ramifications for future policy setting and national inventory accounting.

KEYWORDS
biogas, greenhouse gas, life cycle, livestock, regional, spatial, waste management
1 | INTRODUCTION

Anaerobic digestion, to produce biogas, is one of the most important decentralised renewable energy technologies which is undergoing rapid expansion in order to mitigate against climate change and to ensure a secure energy supply (World Energy Council, 2016). Almost half of the global biogas production occurs in Europe, followed by Asia (32%) then the Americas (17%), with less than 2% of the production occurring in Africa and Oceania continents. The leading countries of biogas production are China, the USA and Germany and it is foreseen that biogas will play an important role in the future global energy supply (WBA, 2017; World Energy Council, 2016). How biogas is produced in different countries can vary extensively, within Europe alone different structures of biogas production exist. In Germany and Italy, most of the biogas is produced in small- to medium-scaled plants, while in England the approach is to use a limited number of larger scaled plants (AEBIOM, 2017). Therefore, to assess the environmental benefits or burdens of biogas production, the specific contexts of these biogas systems need to be taken into account.

Germany for example, as one of the world leaders in biogas production, to achieve its greenhouse gas (GHG) mitigation targets of 40% reduction from 1990 levels by 2020, set in motion the ‘Energiewende’ (Energy transformation with renewables). Biogas energy has been an important contributor to this ‘Energiewende’ (FNR, 2013), with nearly 8,500 biogas plants, contributing approximately 17% of the German renewable power supplied in 2016 (Daniel-Gromke et al., 2018). The majority of these biogas plants use agriculturally sourced feedstock, such as livestock manures and energy crops. This has been mainly driven by substrate-related bonuses provided under the legislative framework of the German Renewable Energy Sources Act (EEG) (BMEL, 2011; Daniel-Gromke et al., 2018).

While the EEG bonuses have had a strong role in determining what feedstock is potentially used, biogas plants are generally planned and built according to available feedstocks (Appel, Ostermeyer-Wiethaup, & Balmann, 2016). This has resulted in distinct regional differences within Germany (Daniel-Gromke et al., 2018; Scheftelowitz, Becker, & Thrän, 2018). Therefore, the number of possible biogas configurations, that is, the combination of feedstocks, sizes and technologies, across Germany is very large (Daniel-Gromke et al., 2018; FNR, 2009). This means that even the use of national averages could lead to obscuring significant regional environmental effects (BMEL, 2011). Understanding what kind of biogas systems are operating within a region and the influencing regional factors is crucial to determine the GHG mitigation potential and carbon neutrality of these renewable energy systems (Chaplin-Kramer et al., 2017; O’Keeffe, Majer, Bezama, & Thrän, 2016a).

The preferred method for assessing the GHG mitigation potential of bioenergy systems has been life cycle assessment (Cherubini & Strømman, 2011; COM (2005) 670; /28/ EC). To date, there have been several life cycle assessments of biogas systems focusing on different aspects, such as, feedstock, technical efficiency, system boundaries and credits (Bachmaier, Effenberger, & Gronauer, 2010; Lanske & Müller, 2012; Lijó et al., 2017; Meyer-Aurich et al., 2012; Scholz, Meyer-Aurich, & Kirschke, 2011; Styles et al., 2014; Vázquez-Rowe, Marvuglia, Rege, & Benetto, 2014). However, very few have investigated the biogas production within a region and those that did, have done so with limited inclusion of regional variability (Bachmaier et al., 2010; Dressler, Loewen, & Nelles, 2012; Lijó et al., 2017). Recently, regionally contextualised life cycle concepts were promoted to include greater spatial details in the life cycle assessments of regional bioenergy production (O’Keeffe et al., 2016a). Furthermore RELCA, a ‘REgional Life Cycle inventory Assessment’ approach was developed (O’Keeffe, Wochele-Marx, & Thrän, 2016b) to carry out within regional assessments (e.g. distribution of GHG emissions within a region). RELCA, combines conventional geographical modelling and catchment delineation to assess the potential environmental implications of bioenergy configurations (bioenergy plants and their biomass catchments) within a focus region. For this paper, it was used to model the GHG mitigation potential of 425 biogas plants and their catchments operating in a focus region in Germany. The aim of this paper was to use the RELCA approach to: (a) generate the GHG emission profiles of biogas systems unique to the region and to compare their mitigation performance as a function of energetic output (kWhₑ) and; (b) to identify the role of GHG credits and installed capacity size in influencing the mitigation performance of the regional biogas systems. This paper provides the systematic overview of the full LCA.

2 | MATERIALS AND METHODS

2.1 | Life cycle method

A schematic of the RELCA framework and the associated modelling steps are provided in Supporting Information Data S1.1. A ‘within regional’ life cycle scope was implemented (O’Keeffe et al., 2016a). The regional scope is defined as one scale lower than a country and denotes the foreground activities relating to the bioenergy systems being assessed (Supporting Information Data S1.1). The RELCA approach applied in this study was retrospective and complied with the ISO LCA standards (EC-JRC, 2010; ISO 14044, 2006), as well as the GHG accounting method of the IPCC (2006a). GHG balances were conducted from field-to-CHP (Combined Heat and Power) plant for all biogas plants modelled (DBFZ, 2011). An attributional life cycle approach, with system expansion with substitution was
implemented. The system boundaries are outlined in Figure 1. The functional unit was the energetic output, (net) kilowatt hour of electricity provided to the grid.

For the individual biogas plants (Figure 1), the boundaries started at the biogas plant, where the upstream emissions related to the production of livestock and hence animal manures were not allocated to biogas production as animal manures were considered as a waste product of the livestock industry (IPCC, 2006b, 2006c; Meyer-Aurich et al., 2012). However, the avoided emissions from the conventional storage of these manures were included in the life cycle assessment, because the improved management of manures is directly linked to the anaerobic digestion and biogas supply (ISO, 2006). This also means that only the land directly supplying the biogas plants was included in this analysis. The regional foreground is set to the eastern German region of Central Germany (CG), which consists of three federal states, or ‘Bundesländer’; Saxony, Saxony-Anhalt and Thuringia.

2.2 Conversion modelling

2.2.1 Background data sources

The DBFZ has been monitoring and collecting data about German biogas production for over a decade (Daniel-Gromke et al., 2018; DBFZ, 2011). In collaboration with the DBFZ, data relating to biogas production was analysed for the region of Central Germany. The first year with the most complete data available (land use and biogas technologies) was chosen for this study. The operational base year for biogas production was assumed to be from the point of harvest in autumn 2010, through to the autumn 2011. During this time period, there was an estimated 570 biogas plants in operation in the CG region. This also includes: agricultural plants (e.g. using energy crops and slurry), plants using municipal organic wastes (e.g. domestic food waste) and plants using industrial wastes (e.g. distillery wastes).

For biogas production, there is a large variability in the types of different feedstocks, feedstock mixtures and technologies used. Therefore, there is a need to simplify the complexity of such biogas systems and develop representative model plants, or ‘typical plants’ using key parameters (FNR, 2006). As a first step to reduce the scope of our study, we focused this study on those biogas plants that were using agricultural feedstocks only.

2.2.2 Data mining

The DBFZ biogas database was screened and in a second step biogas plants were organised according to the feedstock mix.

**FIGURE 1** Visual representation of the RELCA modelling conducted for regional biogas production. For all biogas plants modelled in the Central Germany region (n = 425), the flows included in the calculations are shown, for both the biogas and reference system.
In this step those plants using chicken wastes (e.g. poultry manures) or organic municipal solid wastes were excluded from the analysis, as these were thought not to be representative of the ‘typical regional’ biogas plant found in CG. This resulted in a complete dataset for 166 biogas plants, which were plants that had both quantitative (tonnes of feedstock per year) and qualitative data (description of feedstock mix) associated with them. A further 107 biogas plants were found to have qualitative data only and another 163 plants had the parameter of installed capacity (kW) only, making a total of 436 agricultural biogas plants. However, with further screening for spatial sensitivities, some plants were found to be in close proximity to each other (i.e. found in same GIS grid cell, 250 m × 250 m). These plants were then aggregated and determined as one plant location, with the summed installed capacity allocated to the associated location. This resulted in a total of 425 plant locations being included in the analysis, accounting for approximately 75% of the total biogas plants reported for the region in 2010 and approximately 71% of the total installed biogas capacity in the region.

For reducing complexity, it was decided to cluster the biogas plants based on two characteristics relevant for the German Renewable Energy Sources Act (EEG) these were: (a) size classes, and (b) dominant feedstock classes, that is, either animal manures or maize silage (Supporting Information Data S1.2). The final cluster descriptions are outlined in Table 1. The regional distribution of the clusters showed that they were broadly spread out across the region, with no great spatial clustering. However, it must be noted that a greater number of energy crop-based clusters were located in the northern part of the CG region in the federal state of Saxony-Anhalt (Supporting Information Data S1.3). All other clusters were distributed evenly across the region.

### 2.2.3 Development of model plants for CG

Model biogas plants for each cluster were then derived using the complete dataset (166) and Equations 1–4. Feedstock types \( F_i \) were categorised as: Animal Manure (AM), Animal Slurry (As), Maize Silage (MS), Cereal grains (Cer) and grass silage (GS). In order to determine the required amount of each feedstock type needed \( F_{ij} \) (in t/a) associated with cluster \( j \) (CL1–9, Table 2) was divided by the total summed installed capacity \( IC_j \) (in kW) of cluster \( j \) (Equation 1). The methane production potential for each feedstock type \( i \) in the substrate mix (Table 2) was determined (Supporting Information Data S1.4) and the total summed methane potential \( CH_4,j \) of the feedstock mix for cluster \( j \) (Table 2) was then calculated. From this, the potential gross electricity production \( GE_j \) for each cluster could be calculated: (a) assuming the energetic value of methane to be 9.97 kWh (FNR, 2014), (b) using the electrical efficiency of the specific cluster \( j \) and (c) assuming an average methane loss of approximately 5% (Equation 2) (Reinelt, Liebetrau, & Nelles, 2016) (see Supporting Information data for more detailed description S1.4).

Based on (DBFZ) expert knowledge and data available from the database (i.e. age of plant, technology configurations), the various operational efficiencies and conditions were determined for each model biogas plant associated with a particular cluster (see Supporting Information Data S1.4).

The methane production potential \( CH_4,j \) was calculated as follows:

\[
CH_4,j = \sum F_{ij} \times IC_j
\]  

(1)

The gross electricity production potential \( GE_j \) was calculated as follows:

\[
GE_j = CH_4,j \times 9.97 \times n_{el,j} \times 0.95
\]  

(2)

The net electricity production \( NE_j \) was calculated as follows:

\[
NE_j = GE_j \times 0.9
\]  

(3)

The potential gross electricity production \( F_{t,BGP} \) for each cluster could be calculated as follows:

\[
F_{t,BGP} = (IC_{BGP} \times F_{ij,BGP})
\]  

(4)

For simplicity, it was assumed that all biogas plants used approximately 10% of their own electricity for operational purposes and did not use electricity from the German grid for running the biogas plant. It was also assumed they supplied 90% of their net electricity \( (NE_j \text{ in kWh}_a) \) to the grid (Equation 3). Once the representative feedstock mixes were determined for

| Cluster\(^{a}\) | Size kW\(_{el}\) | Feedstock\(^{b}\) | Number\(^{c}\) |
|-----------|-------------|----------------|-------------|
| 1\(^{d}\)  | <150        | Mans           | 27          |
| 2\(^{d}\)  | 150–500     | Mans           | 126         |
| 3\(^{d}\)  | 150–500     | EC             | 19          |
| 4\(^{d}\)  | 501–1,000   | Mans           | 65          |
| 5\(^{d}\)  | 501–1,000   | EC             | 19          |
| 6\(^{d}\)  | >1,000      | EC             | 10          |
| 7\(^{d}\)  | 150–500     | N/A            | 76          |
| 8\(^{d}\)  | 501–1,000   | N/A            | 69          |
| 9\(^{d}\)  | >1,000      | N/A            | 14          |

\(^{a}\)Clusters (CL1–6 had a subset with adequate data to generate model biogas plants, whereas CL7–9 had no data available with regard to dominant feedstock and therefore, model plants were developed for these plants based on an analysis of mean feedstocks associated with the same installed electrical capacity category that is, CL7 is the average of CL2 and CL3, CL8 is the average of CL4 and CL5 and CL9 was assumed to be the same as CL6. \(^{b}\)These are the predominant feedstock associated with this cluster (i.e. contributes a greater weight to the feedstock mix). Mans = Animal manures (a mixture of slurry (9%DM) and Manure (25%)). EC = Energy crops: Maize silage; Cereal grains: Rye, Barley, Triticale; N/A = not applicable because they had no feedstock data associated with them. \(^{c}\)The number of data points or biogas plants associated with each cluster. 

\(^{d}\)Clusters refer to the energy crop dominant clusters (ECdom). 

\(^{e}\)Super script refers to the slurry dominant clusters (SLdom). 

\(^{f}\)Superscript refers to Energy crop dominant clusters (ECdom).
each cluster, these were then used to generate the appropriate feedstock mix for each individual biogas plant (BGP) based on their individual installed electrical capacities (ICBGP in kW) and the cluster j to which they were assigned (Equation 4). The feedstock demand for each feedstock type of each biogas plant (F_i,BGP in t/a) was then used to generate the catchment areas for each biogas plant found within the CG region (n = 425).

2.2.4  Mass flows of biogas plant-production of digestate

The mass flows for each of the 425 biogas plants were calculated, based on the biogas (b_i) and methane production potentials of the different substrates (Equation 5), and the other associated operational parameters, such as biogas density (ρ_i) (Supporting Information Data S1.4).

For calculation of the digestate output from each plant, it was assumed (Equation 6) that the total amount of organic dry matter fed into the digester (ODM_pin in t/a) minus the weight of the biogas produced (B_p in t/a), determined the potential organic dry matter (ODM_pout) leaving the fermenter and thus, found in the digestate (Equation 6). Therefore, the digestate dry matter (DigDM) could then be estimated (Equation 7) from the sum of the ODM_pout of a biogas plant and the residual dry matter (R) (i.e. dry matter not degraded – referring to ash content of the feedstocks). It was assumed, for simplicity, that the water content remained constant between input and output flows.

\[ B_p = \sum (b_i \times \rho_i) + \ldots (b_{i+n} \times \rho_{i+n}) \]  
\[ ODM_{pout} = ODM_{pin} - B_p \]  
\[ DigDM = ODM_{pout} + R \]  

The destructive rates of the ODM were estimated to be between 80% and 89% for the different clusters, which were found to be comparable to the literature (FNR, 2009). Based on the literature and expert discussions, it was assumed that during the digestion process the ammoniacal nitrogen undergoes an absolute increase of approximately 20% (FNR, 2009; Möller & Müller, 2012; Möller, Schulz, & Müller, 2010; Möller, Stinner, Deuker, & Leithold, 2008), but there is however, a 10% loss of Total N (IFEU, 2008). It was also assumed that the phosphorous or potassium input and output remained constant and did not change (Bachmann, Uptmoor, & Eichler-Löbermann, 2016). The relevant aspects of digestate are discussed further below (section 2.3.2).

2.2.5  Conversion emissions

The GHG emissions were based on the measurements of Liebetrau et al. (2010) and are outlined in Supporting Information Data S1.4. Furthermore, as the biogas plants were thought to be energetically self-sufficient, no electrical energy was necessary from the national grid, therefore, no associated upstream emissions were calculated. Emissions relating to the infrastructures of the clusters were not included as it was assumed, based on the literature, that the impacts would have a comparatively small contribution to the overall emission balance (Bachmaier et al., 2010; Rehl, Lansche, & Müller, 2012).

2.2.6  Crop allocation modelling (CRAM) and catchment modelling (CAMod) steps

The CRAM approach of Wochele, Priess, Thrän, and O’Keeffe (2014) was implemented. The output geodataset provided the regional distribution of the different biogas feedstocks (MS, Cer, GS) using 6.25 ha grid cells (250 m x 250 m). For each grid cell, important regional geographical variables (e.g. climate, soil types and agricultural suitability) were also provided, as well as yields, determined at the county level (German statistical office, 2010a). This was then used to model management and emissions associated with the cultivation of the selected energy crops (Supporting Information Data S1.5).

Catchments were modelled according to O’Keeffe et al. (2016b), with the demand for each cultivated feedstock type
(i.e. MS, Cer, GS) used to generate the catchment area for each biogas plant. The catchments grew in size until the demands of all feedstock types were satisfied for each of the biogas plants \((n = 425)\) in one simulation run. However, if the grid cell from an allocated crop was closer to one biogas plant over another, that grid cell was allocated to the closest biogas plant, in order to avoid catchment area overlap (Tobler, 1970). Additionally, the catchment supplying the fodder demand of the associated livestock was modelled, with arable land devoted to maize and grass silage, assumed to be dedicated to livestock production first and then to biogas production. However, only the direct flows relevant for biogas production were considered in the life cycle assessment presented here (Figure 1).

2.3 | Biomass modelling

2.3.1 | Management flows

Once the catchments had been delineated, the cultivation management could then be estimated for each constituent grid cell of the associated biogas catchment. Similar to the methods outlined in O’Keeffe et al. (2016b), all calculation steps carried out for biomass cultivation were estimated for each cropping grid cell using MATLAB, (2017).

All flows relating to biomass cultivation were for one harvest only, until just after the point of harvesting for the biogas plant. The various management steps are outlined in Table 3, for the Nitrogen (N) fertiliser application rates calculations and tables are provided in Supporting Information Data S1.6.

### Table 3

Crop management practices assumed for the CG region, foreground regional flows\(^a\) (all units are kg ha\(^{-1}\) a\(^{-1}\), unless otherwise stated)

| Management          | Maize silage | Grass silage | Cereals Rye | Barley | Triticale |
|---------------------|--------------|--------------|-------------|--------|-----------|
| Sowing rate\(^c\)   | 0.038        | 0.003        | 0.087       | 0.19   | 0.19      |
| Fertilisers\(^d\)   |              |              |             |        |           |
| P Demand            | 19.65–34.73 (26.16) | 1.34–43.95 (20.4) | 14.32–22.79 (14.11) | 19.39–30.24 (24.84) | 13.48–26.22 (17.39) |
| K Demand            | 105–186 (140) | 17.3–251 (110) | 20.45–32.55 (24.44) | 27.7–43.2 (35.48) | 52–102 (74.79) |
| Crop protection\(^e\) |              |              |             |        |           |
| Herbicide           | 2.13         | 0.68         | 0.3         | 0.3    | 0.3       |
| Insecticide         | 0.012        |              |             |        |           |
| Fungicide           | 0.84         | 0.45         |             |        | 0.05      |
| Growth regulator    |              |              |             |        | 0.05      |
|                     |              |              |             |        |           |

Notes. \(^a\)Upstream production flows are outlined in Supporting Information Data S1.9. \(^b\)Grass silage related to two broad categories of grasslands: grass leys (intensive grassland on arable land) and pastures (extensive grassland), herbicide was assumed to be applied to grass leys only. \(^c\)Sowing rates are in t/ha seeds. For grassland this refers to leys only. \(^d\)Fertilisers – Nutrient applied P = phosphorus provided by P\(_2\)O\(_5\) in fertiliser; K = potassium provided by K\(_2\)O in fertiliser. P & K rates/demand were estimated based on assumed off-take from yields (LLG, 2001; TLL, 2007a, 2009, 2015). The rates of P and K modelled (mean in brackets) are provided here. For lime applications the following assumptions were made: CaO = assuming it takes 1.785 kg of CaCO\(_3\) to neutralise the same area as 1 kg of CaO (2016) (used to convert to Ecoinvent units). A blanket application of 3 t/ha for arable land and 2 t/ha for grassland were assumed (2010). Due to space limitations, detailed data relating to the N fertiliser regime can be found in Supporting Information Data S1.6. \(^e\)Data on crop protection products and recommended dosages was gathered for the region (Supporting Information Data S1.9). Once a final list of plant protection products were identified, the active ingredients of the crop protection products were determined (2014). The active ingredients associated with a fungicide, herbicide and pesticide products were then cross checked with the national survey data of (Rollberg, 2013).

2.3.2 | Rates of organic amendments

Rates of organic amendments (i.e. digestate from a biogas plant or slurry in the reference system) were based in accordance with the various national and regional agricultural authorities (KTBL, 2012; LLG, 2001, 2011; TLL, 2007b; VDLUFA, 2000). The amount of organic amendments applied had to be adjusted for the different crops based on the potential amount of nutrients being supplied by the specific organic amendment associated with each biogas plant. Three major modelling constraints were implemented. The first was to keep the applied total organic nitrogen below the legally specified limit of 170 kg N ha\(^{-1}\) a\(^{-1}\). The second relates to the composition of the organic amendments and the most appropriate quantity to avoid excessive over application of P and K (Supporting Information Data S1.6). The third specified that the amount of digestate applied across all crops should not exceed the amount available to be spread (i.e. how much digestate available). If initial rates resulted in a greater application of N or a surplus of P, K (above the specified limit) or exceeded the amount available, then it was assumed that a lower volume of slurry was applied. The reduction in application rates continued uniformly across the catchment until all constraints were satisfied across all biogas catchments. Digestate and slurry were assumed to be applied using a trailing hose (KTBL, 2010).

The fertiliser characteristics for the digestate and for the reference system are outlined in the Supporting Information Data S1.6. The resulting difference between organic...
amendments was applied and crop NPK demand was then used to estimate the chemical fertilizer required, for each grid cell of the biogas catchment. The estimated Non-Regional management flows, as defined by O’Keeffe et al. (2016b) are outlined in Supporting Information Data S1.9.

2.3.3 | Field operations
The fuel consumption relating to field operations was also estimated using the online KTBL tool (KTBL, 2012). Regression functions were generated based on distance from the biogas plant and three different soil categories, light, medium and heavy (based on % clay content). In the case of fertiliser application, regressions were based on weight of fertiliser to be applied. From these regressions, the amount of diesel required could then be estimated for each field operation (Supporting Information Data S1.7). Emissions associated with field operations were also calculated using Ecoinvent v3.3 (Wernet et al., 2016) (Supporting Information Data S1.7).

2.3.4 | Transport
It was assumed that all arable crops (MS, Cer) were brought to the farmstead using a tractor in combination with two trailers (KTBL, 2010, 2012). For the harvesting of grass feedstock, a loader waggon was assumed. For detailed calculations, see Supporting Information Data S1.8.

2.3.5 | Associated GHG cultivation emissions
A Tier 2 approach was implemented (IPCC, 2006c; O’Keeffe, Majer, Drache, Francho, & Thrain, 2017) to estimate the GHG emissions associated with cultivation. The nitrous oxide emissions associated with the production of each of the different feedstocks were estimated for each of the constituent grid cells of the associated biogas catchments, according to the German national guidelines (Thünen-Institute, 2011) and using the soil-based emission factor of Brocks, Jungkunst, and Bareth (2014).

2.3.6 | Non regional – indirect cultivation emissions
Non regional, upstream emissions for fertiliser production, plant protection products and diesel fuels, were estimated according to O’Keeffe et al. (2017) and are outlined in Supporting Information Data S1.9. All upstream emissions were attributed to each constituent grid cell of a biogas catchment found within the associated Federal states using MATLAB 2017 based scripts (see O’Keeffe et al., 2016b for more details). The global warming potential (GWP) characterisation factors used were according to the IPCC (2014) recommendations for a 100 year period. These were: GWP of 1 kg CO_{2eq} for CO_2, N_2O was assumed to have a GWP of 265 kg CO_{2eq} and CH_4 was assumed to have a GWP of 28 kg CO_{2eq}.

2.4 | Calculation of credits
2.4.1 | Reference system credits
Biogas production in CG is unique, with respect to other biogas producing regions in Germany, due to the greater proportion of livestock manures (slurry at 10% DM and solid manures at 25% DM) being used as a feedstock (Supporting Information Data S1.10). Based on the available data (Daniel-Gromke et al., 2018; DBFZ, 2011), greater than 70% of the biogas plants in CG were using livestock manures from cattle production systems, with a further 3% using a mixture of manures from pig and cattle, as well as 14% using pig slurry as the main feedstock. Therefore, in an attempt to simplify the modelling approach required for the reference system and hence credits, it was decided as a first step to focus only on a potential ‘cow based’ reference system.

Due to the economic sensitivity of such data, it is not easy to obtain data on the exact farm profiles corresponding to a particular biogas plant. Therefore, we estimated the potential associated livestock reference system (Figure 1) based on the operational parameters obtained for the different biogas plants and livestock statistics for the region (German Statistical Office, 2010b). Only those flows necessary for comparison with the biogas plant were considered (Supporting Information Data S1.10). These were: (1) a GHG credit for the potential avoided GHG emissions, which would have been lost without the use of a biogas digester (i.e. due to storage of manures – slurry credit), and (2) the partial N and C flows of the associated livestock system which were relevant for calculating: (a) the fertilizer replacement potential of both the animal manures (FRV_Mana) and (b) the associated digestate replacing it (FRV_dig), as well as, (c) the potential difference in carbon being replaced back to the soil.

2.4.2 | GHG emissions avoided – ‘slurry storage credit’
The avoided emissions due to storage of livestock manures were calculated according to national guidelines (IPCC, 2006b; Thünen-Institute, 2011), using a Tier 2 approach. Livestock statistics for each county (Landkreis) k were used to determine the number of cows (L), divided into a total of nine cow types (t) (Supporting Information Data S1.10). The feed intake for the different cow types, t was derived based on regional reports and literature values.
(Dämmgen, Meyer, Rösemann, Haenel, & Hutchings, 2013; DLG, 2006; LfL, 2015, 2016; TLL, 2006). This was used to calculate the potential volatile solids (VS), based on gross energy intake (Supporting Information Data S1.10.2). Assuming that the dry matter excreted is equal to the sum of volatile solids excreted and ash (Thünen-Institute, 2011), it was also possible to estimate for each cow type (t) the potential amount of excrement produced annually, as well as the different emission factors (EFCH4,λ) associated with each cow type. Livestock excrement (Ex) is defined here as what is directly excreted by the cow. Furthermore, the weighted excrement production factor (CEx,λ) could be calculated. This was calculated by multiplying the amount of total excrement produced per cow type t in a county k (Ex,t,k) by their total number in a county (Lt,k) and taking it as a fraction of total excrement potentially produced in a county k (TotEx,k) (Equation 8).

\[
CEx_{t,k} = \frac{L_{t,k} \times Ex_{t,k}}{TotEx_k}
\] (8)

Using the CEx,t,k factor for a particular cow type, combined with the total amount of animal excrement demand of a particular biogas plant (i.e. how much used per year), in county k (FEx,k), the share (SEx,t,k) or amount (in t/a) of excrement potentially contributed by each cow type could be calculated, based on their weighted share (Equation 9). Thus, knowing how much excrement each cow type produces (Ex), the number of each cow type t, potentially associated with each biogas plant located in a particular county (L,BGP,t) could then be calculated (Equation 10). Hence, we could calculate, based on the different L,BGP,t and their different emission factors (EFCH4,λ), the potential CH4 emissions savings (CH4 in m3/a) due to avoidance of open storage of manures, as the manures will be used instead for the production of biogas (Equation 11). These avoided emissions are used to estimate the slurry credit (SLcred).

\[
S_{Ex,t,k} = CEx_{t,k} \times F_{Ex,k}
\] (9)

\[
L_{BGP,t} = \frac{S_{Ex,t,k}}{Ex_t}
\] (10)

\[
SL_{cred} = L_{BGP,t} \times EF_{CH4,t,BGP}
\] (11)

### 2.4.3 | Nitrogen (N)-based flows – fertiliser credit

The amount of livestock N excreted was also estimated for two relevant flows for the biogas system: (a) the amount of N2O and NH3 (i.e. indirect N2O lost) and hence influential factor on the fertiliser replacement value of the animal manures (FRV_{Mans}) and, (b) the amount of potential N going into the biogas plant, which in turn will affect the FRV of the digestate (FRV_{dig}). The fertiliser replacement value refers to how much chemical fertiliser (N, P, K in kg/a) can be replaced through the use of organic amendments (Lalor et al., 2011). For detailed calculations, see Supporting Information Data S1.10.

### 2.4.4 | Soil organic carbon (SOC) credits

No land transformations regarding the conversion of grassland into arable land were identified from the available land use statistics for the CG region (Supporting Information Data S1.11). Therefore, it was assumed that no CO2 emissions associated with land use changes were released. However, changes in carbon fluxes from cropland remaining cropland (IPCC, 2006a) were considered and the approach outlined by Petersen, Knudsen, Hermansen, and Halberg (2013) was used, to account for the potential effect of soil carbon changes for 1 year on the atmospheric CO2. It was assumed that from the annual addition of carbon, approximately 10% of the C added to the soil organic matter will be sequestered in a 100 year perspective and this factor was implemented here, as in previous studies (Knudsen, Meyer-Aurich, Olesen, Chirinda, & Hermansen, 2014; Mogensen, Kristensen, Nguyen, Knudsen, & Hermansen, 2014).

The first change in carbon fluxes calculated related to the substitution of animal slurry with digestate from the biogas plant (Supporting Information Data S1.10.6). The second change in the carbon flux, related to the reduction of carbon input to soil organic matter, due to the increase in cultivated area dedicated to Maize silage. The changes in carbon fluxes were accounted for using a more simplified approach of Witing et al. (2018) and in accordance with the approach of the carbon turnover models in CANDY (Franko, Oelschlägel, & Schenk, 1995) and CCB (Franko, Kolbe, Thiel, & Ließ, 2011). The final loss or gain of useful carbon was then multiplied by the Petersen et al. (2013) factor, which was then used as the carbon credit in the final GHG balances (Supporting Information Data S1.10.6).

### 2.4.5 | Heat credit

For each cluster, data were available to determine an external heat use factor (Supporting Information Data S1.12). The heat used by an external users was then used to generate the heat credit, assuming the heat from the biogas plants are replacing 70% of heat from natural gas and 30% of heat from oil fuel (Oehmichen & Thrän, 2017). Thus, a credit of 312.12 g CO2eq is assigned to each kilowatt hour of residual heat sourced from the CHP for external use.
**2.4.6 | Statistical analysis**

Emission profiles were generated for each biogas cluster using MATLAB 2017 Statistics and Machine Learning Toolbox™. These emission profiles refer to the distribution of emissions determined for the various biogas plants analysed within a cluster. Therefore, deviations from the median values are due to plant size and plant locations. Significant differences between different regional clusters were tested in R studio (Team R, 2017), using a Kruskal–Wallis test and a post hoc test, Wilcoxon pairwise comparison, with a Holm correction, as the data were found to be nonparametric. All boxplots and bar charts in this paper were produced using MATLAB (2017).

**2.4.7 | Sensitivity analysis**

Two sensitivity analyses were conducted. The first was calculated to account for the uncertainty relating to electricity demand for processing energy. In the base case, the biogas plants were assumed to supply their own electricity; however, it could be the case that electricity is also being used from the grid to run the various processes of the biogas plants. Therefore, for the sensitivity analysis, the process energy demand was assumed again to be 10% of produced electricity. The emission factor for each kilowatt hour of electricity used was 0.58 kg CO₂eq kWhₑ⁻¹ (Oehmichen & Thrän, 2017; Thrän & Pfeiffer, 2015). A sensitivity analysis was also conducted with regard to whether the digestate storage was open or closed. The factors from Liebetrau et al., 2010 were used (Supporting Information Data S1.4).

**3 | RESULTS**

**3.1 | Overall results**

The RELCA simulations show that for biogas production in the CG region, the GHG emissions range from −0.23 to +0.21 kg CO₂eq kWhₑ⁻¹ (Figure 2b). Figure 2b–d show emission profiles associated with each biogas cluster, the deviations coming largely from the different sizes and locations of the associated biogas plants. Slurry dominant clusters (SLdom) had better GHG profiles, showing significant mitigation potential, with median values ranging from −0.10 to −0.20 kg CO₂eq kWhₑ⁻¹, in comparison to energy crop dominant (ECdom) clusters, ranging from 0.04 to 0.16 kg CO₂eq kWhₑ⁻¹ (Figure 2b). This is mainly as the latter, with a greater demand for energy crops and lower proportion of slurry in the feedstock mix, have in turn significantly higher associated cultivation emissions and lower slurry credits (Figure 2a,d).

For the calculations conducted here, we have used regionally specific biogas plants and conditions (i.e. soils, management) for the assessment. Therefore, comparing our results
directly with the literature is difficult, due to the wide diversity of approaches, assumptions and scopes used to assess biogas systems. This difficulty has already been acknowledged (Patterson, Esteves, Dinsdale, & Guwy, 2011; Rehl et al., 2012). However, it is still important to put our results into context with some form of benchmark. Thus, if we simply compare with the ranges found across the literature for slurry and energy crop-based biogas plants, regardless of the particular context (e.g. size, feedstock ratios or types of credits) the range goes from \(-0.91\) to \(0.207\) kg CO\(_{2eq}\) kWh\(_{el}\)\(^{-1}\) (Bachmaier et al., 2010; Dressler et al., 2012; IFEU, 2008) making our results for the CG region relatively conservative in comparison.

Comparing the medians of each cluster with the GHG emission estimated for electricity supplied by the German electricity grid indicated that in this study the biogas clusters in CG had relatively better GHG performance, resulting in GHG savings of \(0.38–0.74\) kg CO\(_{2eq}\) kWh\(_{el}\)\(^{-1}\).

Additionally, it must also be noted that the results for CL7 and CL8 were determined to have GHG emission profiles similar to CL2 and CL4 respectively. However, they were statistically different from CL3 and CL5. This is interesting, because CL7 and CL8 are based on the overall averaged data of the size categories 150–500 kW (i.e. CL2 and CL3) and 501–1,000 kW (i.e. CL4 and CL5) respectively. Thus, showing the importance of determining representative model plants for a region and that even within a region, the use of averages may obscure important results.

### 3.2 Cultivation

For cultivation emissions (Figure 2d), the trends of the clusters, from the lowest to the highest GHG cultivation emissions were as follows: \(CL2 < CL1 < CL7 < CL8 < CL4 < CL9 < CL6 < CL3 < CL5\), with the underlying factor of location playing a large role in the ordering. As with other LCA of biogas (Bachmaier et al., 2010; Dressler et al., 2012; Lansche & Müller, 2012), cultivation contributed the greatest emissions to the overall balances. For the CG region it was found to contribute 52%–67% of the total GHG emissions, for the SL\(_{dom}\) and EC\(_{dom}\) clusters respectively. Nitrogen-related emissions contributed, for all cases, greater than 80% of the total cultivation emissions. Soil N\(_2\)O contributed 72%–78% of the cultivation emissions, with fertilizer N contributing 9%–14% of emissions. In total, nitrogen-related emissions accounted for 46%–59% of the total emissions.

### 3.3 Conversion

Conversion contributed 33%–47% of the total emissions, with slurry dominant clusters having, naturally, a greater contribution of emissions coming from conversion (>40%), as the contribution from cultivation tended to be relatively lower. For the conversion step, the GHG emissions corresponded to scale, CL1 with the smallest installed capacity, having the lowest conversion emissions and CL9, the largest. This difference is due to the extra processing steps required for managing the more solid energy crop feedstock. Approximately 60% of conversion emissions were found to come from the CHP plant, due to gas slips, while approximately 30% of conversion emissions are related to the digestate storage.

It must also be mentioned that although the digestate storage was assumed to be closed, gas-tight conditions were not assumed and therefore, some leakage was assumed (Liebetrau et al., 2013). Furthermore, due to the lack of data, in some cases it was difficult to determine if the digestate storage of a cluster was open or closed (Supporting Information Data S1.4). Therefore, in the base case modelling (Figure 2), we modelled all the clusters with closed storage. However, depending on sizes and general year of installations, smaller and older plants tend to be associated with open digestate storage. If this is the case, this means that the GHG balances will increase by about 0.154 kg CO\(_{2eq}\) kWh\(_{el}\)\(^{-1}\), reducing the mitigation potential of CL1 from \(-0.193\) to \(-0.038\) kg CO\(_{2eq}\) kWh\(_{el}\)\(^{-1}\). For energy crop dominant plants with open digestate storage, taking CL3 as an example, the emissions would increase dramatically from 0.160 to 0.314 kg CO\(_{2eq}\) kWh\(_{el}\)\(^{-1}\). While these GHG balances are still better than a fossil-based energy system, they are still relatively high emissions for a system required to support GHG savings. Therefore, policies supporting biogas production should ensure that digestate is suitably managed, for example, with the use of gas-tight storage, or longer retention times in the digester (i.e. greater digestion of substrates reducing residual gas in digestate). This could result in significant savings, as shown here in some cases cutting some emission balances by half.

It was also assumed that the biogas plants supply their own process energy, which is also an uncertainty in our analysis. Therefore, if we assume that the electricity required for conversion processes is taken from the grid, this results in decreasing the GHG mitigation performance of the clusters by between \(0.041\) kg CO\(_{2eq}\) kWh\(_{el}\)\(^{-1}\) and \(0.082\) kg CO\(_{2eq}\) kWh\(_{el}\)\(^{-1}\). The larger plants with greater process energy demands (e.g. more pumps to operate) will have a greater reduction. This is interesting, as even though the functional unit of the systems increases (i.e. dominator goes up), resulting in a reduction of emissions across the other process steps, the overall increase in the conversion step, pushes the balances up, and this leads to a reduction in mitigation. Therefore, while perhaps buying electricity from the grid is more lucrative in economic terms, it is not in GHG terms. This should also be a point of consideration for policies supporting biogas production, particularly in countries or regions where the electricity supplied from the grid is heavily dependent on fossil fuels.
3.4 | Transport and storage

Transport and storage contributed relatively little to the GHG performance of the clusters, approximately 1%–2% of the total emissions. In general the median distance travelled for feedstock was the shortest for CL1, at 1.05 km. All other clusters with installed capacity less than 1 MWel had median distance ranging from 3.09 to 4.42 km. However, for the CL 6 and CL 9, which are greater than 1 MWel, the median distance was found to be approximately 10.30 km.

3.5 | Direct and indirect regional emissions

Assuming that biogas plants are energetically self-sufficient, the total direct emissions (i.e. generated in the regional foreground) were found to contribute 82%–88% of the emissions. The indirect GHG emissions (i.e. non regional) contributed 12%–18%. The indirect emissions were higher (15%–18%) for the ECdom clusters. This can be related to the higher use of chemical fertilizer being used for these clusters. For the sensitivity analysis, less than one per cent change was found between the indirect and direct ranges in the case of external electricity used from the grid. With an approximately five per cent change observed for the sensitivity effect of open digestate storage, naturally leading to an increase in direct emissions, increasing them to up to 93% of the total emissions.

3.6 | Credits

The large influence of credits on the overall results, as a function of energetic output (per kWhel), can clearly be seen in Figure 2a,b. Indeed, the strong influence which credits can have on lowering the GHG balances of biogas systems has been noted in other studies (Bachmaier, Effenberger, Gronauer, & Boxberger, 2012; Lansche & Müller, 2012; Oehmichen & Thrän, 2017).

3.6.1 | Avoided emissions from slurry storage (SLcredits)

The need to include and account for greater detail in calculating slurry credits is gaining more traction in the literature (Lansche & Müller, 2012; LWK, 2011; Oehmichen & Thrän, 2017). The slurry credits were found to range from 0.023 to 0.39 kg CO2eq kWhel−1. In comparison to the literature values, which range from approximately 0.091 to 0.2 kg CO2eq kWhel−1 (Bachmaier et al., 2010; Lansche & Müller, 2012), our values were relatively higher. This is mainly due to the regional specification, which was influenced by the livestock composition and the higher use of animal excrements. Not every region in Germany may have such large associated GHG savings. Indeed, this will vary from region to region and from country to country.

3.6.2 | Fertilizer saving credit

While the approach here is limited, assuming a credit for cattle slurry only and using general statistics rather than actual farm data, the results are still valid to show the need for including a more detailed reference system for estimating SL credits, when biogas plants are using livestock excrements. This is still an area which needs greater research and will become even more relevant with the updated calculation approaches determined for the EU Renewable Energy Directive II (COM (2016) 767/F2).

3.6.3 | Carbon credits

The carbon credits ranged from −0.02 to +0.008 kg CO2eq kWhel−1, resulting again in the ECdom clusters performing better than the SLdom clusters (Figure 2a), due to...
the greater volume of available organic amendments (i.e. manures vs. digestate). Although the carbon content of manures was estimated to be higher (i.e. as biogas production results in a loss of carbon out of the system), the more useful carbon content of the digestate was estimated to be much higher (see Witing et al., 2018). However, the carbon credits had only a slight effect on the GHG balances. In spite of this it is still important to include it, as it can provide an indication of effects to soil quality and to date very few studies include the use of a soil carbon credits within an LCA approach (Claus et al., 2014). While our approach here is relatively simple, this study shows the benefits of including aspects of soil carbon flows within the life cycle accounting of biogas assessments. From the modelling results here and indeed what has been recently shown in the literature (Witing et al., 2018), biogas may have the potential for improving the SOC in the CG region, however more detailed field-based research would be needed to determine this conclusively.

3.6.4 | Heat credits

The heat credits for the CG clusters were found to range from $-0.027$ to $-0.22$ kg CO$_2$eq kWh$_{el}^{-1}$, almost as lucrative in GHG savings as the SL credits. CL1 had the smallest heat credit, with CL5 having the largest associated heat credit. Similar to other studies, the difference between the heat credits depends on the share of heat used (Bachmaier et al., 2010; Oehmichen & Thrän, 2017). In the CG region, the larger energy crop plants appeared to have a higher share of external heat use (see Supporting Information S1.12). In the case of CG, for the larger plants the heat credit in some cases helped to mitigate against somewhat larger cultivation emissions associated with these clusters, CL5 is a good example of this. However, it must also be noted here that the data associated with the smaller cluster were relatively uncertain and thus, it could be substantially higher (only 8% had heat use data available). Altogether, it shows that the use of surplus heat can provide substantial GHG savings and this is particularly relevant for the larger EC$_{dom}$ clusters.

3.7 | Effect of size

It is generally assumed that the bigger the bioenergy plant the better the GHG mitigation performance, this is mostly as, all things being equal the denominator is much bigger (scale effect). However, the results presented here for biogas show the opposite. This is because the effect of size is confounded

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**FIGURE 3** Provides an overview of greenhouse gas emission profiles (y axis kg CO$_2$eq kWh$_{el}^{-1}$) with respect to size classes (installed kW-x axis): (a) shows size classes aggregated for clusters 1–6 according to energetic output-no credits (b) shows size classes divided into slurry dominant (blue, CL1, CL2, CL4) and energy crop dominant (grey, CL 3, CL5, CL6) according to energetic output (kg CO$_2$eq kWh$_{el}^{-1}$), no credits. The comparative sizes for SL$_{dom}$ clusters were: <150 kW, 150–500 kW, 501–1,000 kW. For E$_{dom}$ clusters it was 150–500 kW, 501–1,000 kW, >1,000 kW) (c, d) show the same as (a) and (b) respectively for energetic output (kg CO$_2$eq kWh$_{el}^{-1}$), with credits included. Clusters with the same letters are not significantly different from one another. Clusters denoted with ‘***’ are significant from every other cluster ($p < 0.001$). Extreme outliers were removed
with feedstock. Due to the low energy density of livestock manures, they often need to be digested with crops which have higher energy content (Auburger, Petig, & Bahrs, 2017; FNR, 2013; Meyer-Aurich, Lochmann, Klauss, & Prochnow, 2016). Larger plants therefore, usually have a greater share of energy crop in their feedstock mix. This in turn means they will have correspondingly higher associated cultivation emissions (Table 1; Figure 2d). To date, very few studies have investigated the combined effect of size and feedstock on the GHG balances of biogas systems, most focus predominantly on the effect of feedstock alone (Bachmaier et al., 2010; Delzeit & Kellner, 2013).

The emission profiles for the CG region aggregated to size categories only and excluding credits are shown in Figure 3a. No significant difference between the two smaller clusters, which have relatively similar feedstock mixes, can be seen. However, clusters grouped within the size category 500–1000 kWel have significantly higher emissions, with plants greater than 1,000 kWel being associated with the most significantly highest emissions. On first glance, it appears there is an effect of size. However, when the sizes were compared within either SL_dom or EC_dom cluster groupings (Figure 3b), no significant differences were found between the sizes for the EC_dom clusters. For the slurry-based clusters, however, the largest size category of 500–1,000 kW, was found to have significantly higher GHG balances. This is mainly due to the greater share of energy crops in its feedstock mix, a difference of approximately 5%–10%.

With the addition of credits (Figure 3c), the smaller plants can be seen to significantly outperform the larger plants and the effect of scale appears to become even stronger. However, when we look at Figure 3d, again we can see the effect of feedstock and the related credits for the different clusters and increase in size does not necessarily mean an increase in GHG emission.

For example, SL_dom cluster CL2, has a better performance than CL1, which is smaller (locational effects play a role in this difference). Likewise, EC_dom cluster, CL5, has a better performance than CL3 which is smaller (heat credits play a role, as CL5 was found to supply almost 50% more of its surplus heat, whereas CL3 used only 27%). Therefore, while we see here that size is an important factor in decreasing GHG mitigation performance of biogas plants, the effect of size is confounded in the feedstock being used and the amount of surplus heat being supplied to external users.

### 3.8 Mitigation performance of the regional biogas system

So far smaller SL_dom clusters were shown to have a better GHG mitigation performance than the larger scale EC_dom clusters. However, it begs the question, how much electricity do each of these clusters contribute to the grid for our operational year and how does this relate to GHG savings?

Overall about 70% of the total modelled electricity supplied was provided by 85% of the plants (n = 425) modelled (SL_dom clusters). These plants also contributed 86% of the overall GHG savings, which are mostly due to the allocation of slurry credits (Table 4). However, it should also be noted that the plants in the SL_dom clusters still require energy crops to support the low energy density animal manures. This results in energy crops contributing between 47% and 61% of the energy content of the substrates found in the SL_dom clusters (CL1, CL2, CL4, CL7, CL8). It also results in approximately 67% of the direct land supplying the biogas plants to be dedicated to SL_dom clusters.

While the proportion of plants in the EC.dom clusters operating in the CG region is relatively low at 13%, they contributed about 30% of the electricity production of the plants analysed (Table 4). They contributed 14% of the overall GHG

### TABLE 4 Overview of contributions from all agricultural-based biogas clusters in the CG region

| Cluster | %Plants² | %Electricity³ | %Total emissions⁴ | %Savings electricity⁵ | %Savings (credits)⁶ | %Total savings⁷ | %Land⁸ |
|---------|---------|--------------|-----------------|-----------------------|---------------------|----------------|--------|
| 1       | 6%      | 1%           | 1%              | 8%                    | 2%                  | 2%             | 1%     |
| 2       | 30%     | 20%          | 17%             | 34%                   | 27%                 | 27%            | 16%    |
| 3       | 4%      | 3%           | 4%              | 3%                    | 1%                  | 1%             | 4%     |
| 4       | 15%     | 18%          | 17%             | 16%                   | 20%                 | 20%            | 18%    |
| 5       | 4%      | 6%           | 7%              | 3%                    | 4%                  | 4%             | 7%     |
| 6       | 2%      | 9%           | 11%             | 1%                    | 4%                  | 4%             | 12%    |
| 7       | 18%     | 13%          | 12%             | 20%                   | 16%                 | 16%            | 11%    |
| 8       | 16%     | 20%          | 19%             | 16%                   | 22%                 | 22%            | 18%    |
| 9       | 3%      | 10%          | 12%             | 2%                    | 4%                  | 4%             | 13%    |

Notes: Values have been rounded up and therefore in some cases may differ slightly from 100%.

²Total land use estimated for the 425 plants was 94,514 ha. ³Total installed electrical capacity modelled for the 425 plants was 208 MW_el, representing approximately 7.35% of the total German installed electrical capacity (2011). ⁴Total summed emissions of the biogas clusters prior to the allocation of credits. ⁵Savings in GHG emissions in comparison to the German electricity supply. ⁶Savings in relation to allocated credits. ⁷Total savings, added savings from credits and savings in comparison to German grid.
savings, similar to SL\textsubscript{dom} clusters, in that the proportion of plants is similar to the proportion of GHG savings. For EC\textsubscript{dom} clusters, savings are mostly related to heat credits. This also shows that fewer plants can contribute more electricity, due to the energy-dense crops being used (i.e. 85\textperthousand\textperthousand–96\textperthousand\textperthousand of the feedstock energy content). Additionally, it can be seen that to produce approximately 30\textperthousand\textperthousand of the electricity supply also requires about 33\textperthousand\textperthousand of the total direct land area being supplied to the biogas plants.

If we compare directly the smaller SL\textsubscript{dom} clusters (CL1, CL2) against the largest EC\textsubscript{dom} clusters (CL6, CL9), we see that the SL clusters consisted of approximately 36\textperthousand\textperthousand of all plants installed, provided 21\textperthousand\textperthousand of the electricity to the grid with 29\textperthousand\textperthousand GHG savings and 17\textperthousand\textperthousand of the total direct land used. The EC\textsubscript{dom} clusters, on the other had consisted of 5\textperthousand\textperthousand of all plants installed, contributed 19\textperthousand\textperthousand of the electricity to the grid, had a GHG saving of 8\textperthousand\textperthousand and used 25\textperthousand\textperthousand of the land associated with biogas production. With this we can see that while EC\textsubscript{dom} plants are lucrative in supplying electricity to the grid, they are not so lucrative in GHG savings, predominantly related to their higher cultivation emissions and relatively lower slurry credits. While this is not an entirely surprising result, it does perhaps lead to the consideration that the percentage GHG savings of a regional biogas cluster should be the same or indeed more than its energetic contribution and if it is not, then these are the regional clusters which need to be targeted first for modifications. In this way, improvements are provided to the overall electricity by the regional biogas systems.

4 | DISCUSSION, CONCLUSIONS AND RECOMMENDATIONS

The aim of this paper was to generate the GHG emission profiles of biogas systems or clusters unique to a region in Germany, in order to compare their mitigation performance, as well as to identify the role of GHG credits and size in influencing the mitigation performance of these regional biogas systems. The approach and results presented in this paper, are the first (at time of writing), to attempt a more integrative assessment of biogas configurations in the landscape for an entire region and thus, avoid the obscurity of national averages.

The regional biogas supply was divided into biogas clusters, to help us identify possible regional trends with regard to substrates, credits and size. From this, it was determined that per kilowatt hour delivered to the grid, SL\textsubscript{dom} clusters had a better GHG mitigation performance, than the EC\textsubscript{dom} clusters. This was particularly apparent at the smaller scales. However, it must be noted that the GHG savings of these smaller plants could change if they are storing their digestate in open storage tanks, reducing their mitigation potential by 26\textperthousand\textperthousand and in some cases ending with an even worse GHG performance than the EC\textsubscript{dom} clusters. Therefore, the allocation of subsidies for SL\textsubscript{dom} biogas plants should be done ensuring that biogas plants installed are adapted to site conditions (e.g. livestock numbers, quality of animal manures), while still meeting technical requirements which can adequately reduce GHG emissions (e.g. sufficient residence time to reduce residual gas emissions or gas-tight digestate storage with gas detection). However, what we also see is that while size is an important factor in decreasing GHG mitigation performance of biogas plants, this effect of size is confounded in the feedstock being used. This in turn means that there are more underlying regional factors, than size alone, which are responsible for the GHG performance of these biogas systems.

It was also shown that while EC\textsubscript{dom} clusters are lucrative in supplying energy to the grid, they are not so lucrative in GHG savings, predominantly related to their higher cultivation emissions and relatively lower slurry credits. This is important to point out, as a balance needs to be struck between security of supply (i.e. electricity delivery to the grid) and GHG savings. For Germany, as the potential amount of agricultural biogas plants have reached their peak, any further increases in capacity means alteration of the current biogas systems installed. Therefore, investigating the contribution from agricultural biogas clusters or systems within regions, with regard to their supply of electricity to the grid (i.e. security of supply), GHG saving and land use will become more prevalent in the coming years. A starting point will be the ability to identify which types of regional biogas clusters may need modifications to achieve greater GHG mitigation potentials in the future regional biogas supply. In addition, such an examination of the regional production can help to show where potential trade-offs may need to be examined in more detail, such as, effects of location. From the analysis conducted in this study, it appears that larger EC\textsubscript{dom} clusters should be targeted first, to support such GHG mitigation improvements to the overall future electricity supplied by the regional biogas systems.

Furthermore, from the analysis conducted here, the CG region is producing biogas which appears to be not only carbon neutral, but appears to result in GHG savings, with a regional median emission of \(-0.156 \text{ kg CO}_2\text{eq kWh}^{-1}\) (interquartile range: 0.0095 \text{ kg CO}_2\text{eq kWh}^{-1})). Showing the important role which credits can also have on the potential GHG mitigation performance of a biogas system. This infers that the allocation of credits under the context setting, biogas as a waste management strategy for livestock manures, could have very important ramifications for future policy setting, in particular for the EEG in Germany and the Renewable Energy directive II (Com (2016) 767/F2) in Europe. It also leads to the question, under which national GHG accounting method should such credits be allocated in order to avoid double counting, under the agricultural sector or under the...
energy sector? This is brought into question mainly as many biogas systems are integrated within existing agricultural systems.

Finally, the approach presented in this paper has shown the value of conducting an integrated regional life cycle of biogas systems and although focusing on a particular region in Germany, it can be applied to other regions around the world. Indeed, such approaches are recommended for biogas production, as it is predominantly decentralised and heavily dependent on local and regional feedstocks (O’Keeffe et al., 2016a; World Energy Council, 2016). Therefore, understanding what kind of biogas systems are operating within a region and the influencing regional factors is crucial to not only determine the GHG mitigation potential and carbon neutrality of these renewable energy systems, but also to improve them in an effective and sustainable way.

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**ORCID**

Sinéad O’Keeffe © https://orcid.org/0000-0001-7043-5665

**REFERENCES**

AEBIOM. (2017). European Biomass Association. Statistical Report 2017. European Bioenergy Outlook. Key Findings. Retrieved from http://www.aebiom.org/statistical-report-2017/statistical-report-2017-17-10-17/ (accessed 1 May 2018).

Appel, F., Ostermeyer-Wiethaup, A., & Balman, A. (2016). Effects of the german renewable energy act on structural change in agriculture – The case of biogas. *Utilities Policy, 41*, 172–182. https://doi.org/10.1016/j.jup.2016.02.013

Auburger, S., Petig, E., & Bahrs, E. (2017). Assessment of grassland as biogas feedstock in terms of production costs and greenhouse gas emissions in exemplary federal states of Germany. *Biomass and Bioenergy, 101*, 44–52. https://doi.org/10.1016/j.biombioe.2017.03.008

Bachmaier, J., Effenberger, M., & Gronauer, A. (2010). Greenhouse gas balance and resource demand of biogas plants in agriculture. *Engineering in Life Sciences, 10*, 560–569. https://doi.org/10.1002/elsc.2010000073

Bachmaier, H., Effenberger, M., Gronauer, A., & Boxberger, J. (2012). Changes in greenhouse gas balance and resource demand of biogas plants in southern Germany after a period of three years. *Waste Management & Research, 31*, 368–375.

Bachmann, S., Uptmor, R., & Eichler-Löbermann, B. (2016). Phosphorus distribution and availability in untreated and mechanically separated biogas digestates. *Scientia Agricola, 73*, 9–17. https://doi.org/10.1590/0103-9016-2015-0069

Baywa. (2014). Kalkdünger. Retrieved from http://www.baywa.de/pflanzenbau-obst/ackerbau_gruenland/duengemittel/kalkduenger/ (accessed 1 February 2014).

BMEL. (2011). Scientific Advisory Board on Agricultural Policy at the Federal Ministry of Food, Agriculture and Consumer Protection. Promotion of biogas production through the Renewable Energy Sources Act (EEG), http://www.bmel.de/SharedDocs/Downloads/EN/Ministry/Biogas-EEG.pdf?__blob=publicationFile (accessed 10 March 2017).

Broms, S., Jungkunst, H.F., & Bareth, G. (2014). A regionally disaggregated inventory of nitrous oxide emissions from agricultural soils in Germany-A GIS based empirical approach. *ERDKUNDE, 68*, 125–144. https://doi.org/10.3112/erdkunde.2014.02.04

BVL. (2013). Bundesamt für Verbraucherschutz und Lebensmittelsicherheit online database on plant protection products. https://portal.bvl.bund.de/psm/jsp/ (accessed 1 January 2014).

Chaplin-Kramer, R., Sim, S., Hamel, P., Bryant, B., Noe, R., Mueller, C., ... Daily, G. (2017). Life cycle assessment needs predictive spatial modelling for biodiversity and ecosystem services. *Nature Communications, 8*, 15065. https://doi.org/10.1038/ncomms15065

Cherubini, F., & Strømman, A. H. (2011). Life cycle assessment of bioenergy systems: state of the art and future challenges. *Bioresource Technology, 102*, 437–451. https://doi.org/10.1016/j.biortech.2010.08.010

Clauß, S., Taube, F., Wienforth, B., Svoboda, N., Sieling, K., Kage, H., ... Herrmann, A. (2014). Life-cycle assessment of biogas production under the environmental conditions of northern Germany: greenhouse gas balance. *The Journal of Agricultural Science, 152*, 172–181. https://doi.org/10.1017/S0021859613000683

Com (2005) 670 Communication from the Commission of 21 December 2005, Thematic Strategy on the sustainable use of natural resources. Com (2016) 767/F2 European Commission. Proposal for a Directive of the European Parliament and of the council on the promotion of the use of energy from renewable sources. Retrieved from https://ec.europa.eu/transparency/regexp/rep/1/2016/EN/COM-2016-767-F2-EN-MAIN-PART-1.PDF (accessed 1 May 2017).

Dümmgen, U., Meyer, U., Rösemann, C., Haenel, H. D., & Hutchings, N. J. (2013). Methane emissions from enteric fermentation as well as nitrogen and volatile solids excretions of German calves – A national approach. *Landbauforsch Applied Agricultural Forestry Research, 63*, 37–46.

Daniel-Gromke, J., Rensberg, N., Denysenko, V., Stinner, W., Schmalfuß, T., Scheifelowitz, M., ... Liebetrau, J. (2018). Current developments in production and utilization of biogas and biomethane in Germany. *Chemie Ingenieur Technik, 90*, 17–35. https://doi.org/10.1002/cite.201700077

DBFZ. (2011). Deutsche Biomasse Forschung Zentrum. Monitoring zur Wirkung des Erneuerbare-Energien-Gesetz (EEG) auf die Entwicklung der Stromerzeugung aus Biomasse. Retrieved from https://www.dbfz.de/fileadmin/eeg_monitoring/berichte/08_Monitoring_ZB_Maerz_2011.pdf

DBFZ. (2012). Deutsches Biomasseforschungszentrum. Report Nr. 12. Monitoring zur Wirkung des Erneuerbare Energien-Gesetz (EEG) auf die Entwicklung der Stromerzeugung aus Biomasse. Retrieved
Delzeit, R., & Britz, W. (2012). An economic assessment of biogas production and land use under the German Renewable Energy Source Act. Kiel Working Paper No. 1767. Retrieved from https://www.ifeu.de/oekobilanzen/pdf/THG_Bilanzen_Bio_Erdgas.pdf (accessed 15 April 2013).

IPCC. (2006a). Intergovernmental Panel of Climate Change. IPCC Guidelines for National Greenhouse Gas Inventories. Chapter 5 Cropland. Agriculture, forestry and other land use, vol. 4.

IPCC. (2006b). Intergovernmental Panel of Climate Change. IPCC Guidelines for National Greenhouse Gas Inventories. Chapter 10: Emissions from livestock and manure management, vol. 4.

IPCC. (2006c). Intergovernmental Panel of Climate Change. IPCC Guidelines for National Greenhouse Gas Inventories, Chapter 11, N2O emissions from managed soils, and CO2 emissions from lime and urea application, vol. 4.

IPCC. (2014). Intergovernmental Panel on Climate Change, 2014, Climate Change 2014: Synthesis Report Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R. K. Pachauri & L.A. Meyer (Eds.)]. Geneva: IPCC.

O'KEEFFE ET AL.

Delzeit, R., & Kellner, U. (2013). The impact of plant size and location on profitability of biogas plants in Germany under consideration of processing digestates. *Biomass and Bioenergy*, 52, 43–53. https://doi.org/10.1016/j.biombioe.2013.02.029

Directive 2009/28/EC of the European Parliament and of the council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. OJL 140/16.

DLG. (2006). Deutsche Landwirtschafts-Gesellschaft. Schätzung der Futteraufnahme bei der Milchkuh. Retrieved from http://www.dlg.org/fachinfos-rinderender.html (accessed 25 June 2017).

Dressler, D., Loewen, A., & Nelles, M. (2012). Life cycle assessment of the supply and use of bioenergy: Impact of regional factors on biogas production. *The International Journal of Life Cycle Assessment*, 17, 1104–1115. https://doi.org/10.1007/s11367-012-0424-9

EC-IRC. (2010). General guide for life cycle assessments-detailed guidance document for life cycle assessment (LCA). ILCD handbook-International Reference Life Cycle Data System, European Union. Retrieved from http://bookshop.europa.eu/en/international-reference-life-cycle-data-system-ilcd-handbook-pblBN24708/ (accessed 1 January 2012).

FNR. (2006). Fachagentur Nachwachsende Rohstoffe (Agency of Renewable Resources), Handreichung Biogasgewinnung und -nutzung.

FNR. (2009). Fachagentur Nachwachsende Rohstoffe e.V. (FNR). Biogas-Messprogramm II - 61 Biogasanlagen im Vergleich. Retrieved from http://www.fnr-server.de/ftp/pdf/literatur/pdf_f_385-messprogramm_ii.html (accessed 1 February 2016).

FNR. (2013). Fachagentur Nachwachsende Rohstoffe e.V. Biogas, an introduction. Retrieved from https://mediathek.fnr.de/media/downloadable/files/samples/bt/birosch.biogas-2013-en-web-pdf.pdf (accessed 1 June 2016).

FNR. (2014) Fachagentur Nachwachsende Rohstoffe e.V. Bioenergy in Germany: Facts and Figures. Retrieved from http://mediathek.fnr.de/media/downloadable/files/samples/b/a/basisdaten_9x16_2013_engl_web.pdf (Accessed 1 March 2014).

Franko, U., Kolbe, H., Thiel, E., & Ließ, E. (2011). Multi-site validation of a soil organic matter model for arable fields based on generally available input data. *Geoderma*, 166, 119–134. https://doi.org/10.1016/j.geoderma.2011.07.019

Franko, U., Oelschlägel, B., & Schenk, S. (1995). Simulation of temperature-, water- and nitrogen dynamics using the model CANDY. *Ecological Modelling*, 81, 213–222. https://doi.org/10.1016/0304-3800(94)00172-E

German Federal Statistical Office. (2010a). *Landwirtschaftliche Bodennutzung und pflanzliche Erzeugung*. Wiesbaden: Statistisches Bundesamt.

German Federal Statistical Office. (2010b). *Landwirtschaftliche Betriebe mit Viehhaltung und Zahl der Tiere Landwirtschaftszählung – Haupterhebung*. Wiesbaden: Statistisches Bundesamt.

IFEU. (2008). Basisdaten zu THG-Bilanzen für Biogas-Prozessketten und Erstellung neuer THG-Bilanzen. Retrieved from http://www.ifeu.de/oekobilanzen/pdf/THG_Bilanzen_Bio_Erdgas.pdf (accessed 15 April 2013).
Lijó, L., Lorenzo-Toja, Y., González-García, S., Bacenetti, J., Negri, M., & Moreira, M. T. (2017). Eco-efficiency assessment of farm-scaled biogas plants. Bioresource Technology, 237, 146–155. https://doi.org/10.1016/j.biortech.2017.01.055

LLG. (2001). Landesanstalt für Landwirtschaft, Forsten und Gartenbau, Sachsen-Anhalt. Grundlagen der Düngedarmermittlung für eine gute fachliche Praxis beim Düngen. Retrieved from https://llg.sachsen-anhalt.de/fileadmin/Bibliothek/PoImtk_Pund_Verwaltung/MLU/LLFG/Dokumente/GL_Duengedarmermittlung (accessed February 2013).

LLG. (2011). Landesanstalt für Landwirtschaft, Forsten und Gartenbau, Sachsen-Anhalt. Hinweise zur umweltgerechten Düngung von Körner-, Silo- und Energiemais. Retrieved from http://www.llg.sachsen-anhalt.de/fileadmin/Bibliothek/PoImtk_Pund_Verwaltung/MLU/LLFG/Dokumente/11_mais_dueng.pdf (accessed 1 September 2017).

MATLAB. (2017). Matlab and statistics toolbox release. Natick, MA: The MathWorks Inc.

Meyer-Aurich, A., Lochmann, Y., Klauss, H., & Prochnow, A. (2016). Comparative advantage of maize- and grass-silage based feedstock for biogas production with respect to greenhouse gas mitigation. Sustainability, 8, 617. https://doi.org/10.3390/su8070617

Meyer-Aurich, A., Schattauer, A., Hellebrand, H. J., Klauss, H., Plöchl, M., & Berg, W. (2012). Impact of uncertainties on greenhouse gas mitigation potential of biogas production from agricultural resources. Renewable Energy, 37, 277–284. https://doi.org/10.1016/j.renene.2011.06.030

Mogensen, L., Kristensen, T., Nguyen, T. L. T., Knudsen, M. T., & Hermansen, J. E. (2014). Method for calculating carbon footprint of cattle feeds – Including contribution from soil carbon changes and use of cattle manure. Journal of Cleaner Production, 73, 40–51. https://doi.org/10.1016/j.jclepro.2014.02.023

Möller, K., & Müller, T. (2012). Effects of anaerobic digestion on digestate nutrient availability and crop growth: A review. Engineering in Life Sciences, 12, 242–257. https://doi.org/10.1002/elsc.201100085

Möller, K., Schulz, R., & Müller, T. (2010). Substrate inputs, nutrient flows and nitrogen loss of two centralized biogas plants in southern Germany. Nutrient Cycling in Agroecosystems, 87, 307–325. https://doi.org/10.1007/s10705-009-9340-1

Möller, K., Stinner, W., Deuker, A., & Leithold, G. (2008). Effects of different manuring systems with and without biogas digestion on nitrogen cycle and crop yield in mixed organic dairy farming systems. Nutrient Cycling in Agroecosystems, 82, 209–232. https://doi.org/10.1007/s10705-008-9196-9

Oehmichen, K., & Thrán, D. (2017). Fostering renewable energy provision from manure in Germany – Where to implement GHG emission reduction incentives. Energy Policy, 110, 471–477. https://doi.org/10.1016/j.enpol.2017.08.014

O’Keeffe, S., Majer, S., Bezama, A., & Thrán, D. (2016a). When considering no man is an island—assessing bioenergy systems in a regional and LCA context: A review. The International Journal of Life Cycle Assessment, 21, 885–902. https://doi.org/10.1007/s11367-016-1057-1

O’Keeffe, S., Majer, S., Drache, C., Franko, U., & Thrán, D. (2017). Modelling biodiesel production within a regional context – A comparison with RED Benchmark. Renewable Energy, 108, 355–370. https://doi.org/10.1016/j.renene.2017.02.024

O’Keeffe, S., Wochele-Marx, S., & Thrán, D. (2016b). RELCA: a REnGional Life Cycle inventory for Assessing bioenergy systems within a region. Energy, Sustainability and Society, 6, 1–19.

Patterson, T., Esteves, S., Dinsdale, R., & Guwy, A. (2011). Life cycle assessment of biogas infrastructure options on a regional scale. Bioresource Technology, 102, 7313–7323. https://doi.org/10.1016/j.biortech.2011.04.063

Petersen, B. M., Knudsen, M. T., Hermansen, J. E., & Halberg, N. (2013). An approach to include soil carbon changes in life cycle assessments. Journal of Cleaner Production, 52, 217–224. https://doi.org/10.1016/j.jclepro.2013.03.007

Rehl, T., Lanske, J., & Müller, J. (2012). Life cycle assessment of energy generation from biogas – Attributional vs. consequential approach. Renewable and Sustainable Energy Reviews, 16, 3766–3775. https://doi.org/10.1016/j.rser.2012.02.072

Reinelt, T., Liebetrau, J., & Nelles, M. (2016). Analysis of operational methane emissions from pressure relief valves from biogas storages of biogas plants. Bioresource Technology, 217, 257–264. https://doi.org/10.1016/j.biortech.2016.02.073

Rößberg, D. (2013). Erhebungen zur Anwendung von Pflanzenschutzmitteln in der Praxis im Jahr 2011. Journal für Kulturpflanzen, 65, 141–151.

Scheftelowitz, M., Becker, R., & Thrán, D. (2018). Improved power provision from biomass: a retrospective on the impacts of German energy policy. Biomass and Bioenergy, 111, 1–12. https://doi.org/10.1016/j.biombioe.2018.01.010

Scholz, L., Meyer-Aurich, A., & Kirschke, D. (2011). Greenhouse gas mitigation potential and mitigation costs of biogas production in Brandenburg, Germany. AgBioForum, 14, 133–141.

Styles, D., Gibbons, J., Williams, A. P., Stichnothe, H., Chadwick, D. R., & Healey, J. R. (2014). Cattle feed or bioenergy? Consequential life cycle assessment of biogas feedstock options on dairy farms. GCB Bioenergy, 7, 1034–1049.

Team R. (2017). RStudio: Integrated development for R. Boston, MA: RStudio, Inc. Retrieved from http://www.rstudio.com/

Thrán, D., & Pfeiffer, D. (2015) (Eds.) Method handbook – Material flow-oriented assessment of greenhouse gas effects. In Series of the funding programme “Biomass energy use”, Vol. 04, Leipzig: ISSN 2364-897X

Thünen-Institute. (2011). Calculations of gaseous emission and particulate emissions from German Agriculture 1990 - 2009. Special Issue 342. Retrieved from https://www.thuenen.de/media/publikationen/landbauforschung-sonderhefte/lbf_sh342.pdf

TLL. (2006) Thüringen Landesanstalt für Landwirtschaft. Leitlinie zur effizienten und umweltverträglichen Mutterkuhhaltung. Available at http://www.tll.de/ainfo/pdf/muku0206.pdf (accessed 25 June 2017).

TLL. (2007a). Leitlinie zur effizienten und umweltverträglichen Erzeugung von Silomais für die Fütterung und Nutzung als Gärsustruktur in Biogasanlagen. Retrieved from http://www.tll.de/ainfo/pdf/simal0807.pdf (accessed 1 September 2016).

TLL. (2007b). Thüringer Ministerium für Landwirtschaft, Forsten, Umwelt und Naturschutz. Landwirtschaft und Landschaftspflege in Thüringen. Düngung in Thüringen 2007 nach „Guter fachlicher Praxis“. Retrieved from https://www.thuenen.de/media/publikationen/landbauforschung-sonderhefte/lbf_sh342.pdf

TLL. (2009). Leitlinie zur effizienten und umweltverträglichen Erzeugung von Winterroggen. Retrieved from http://www.tll.de/ainfo/pdf/tll_wro.pdf (accessed 1 August 2016).
TLL. (2015). Leitlinie zur effizienten und umweltverträglichen Erzeugung von Wintergerste. Retrieved from http://www.tll.de/ainfo/pdf/ll_wg.pdf (accessed 1 September 2016).

Tobler, W. R. (1970). A computer movie simulating urban growth in the Detroit region. *Economic Geography, 46,* 234–240. https://doi.org/10.2307/143141

Vázquez-Rowe, I., Marvuglia, A., Rege, S., & Benetto, E. (2014). Applying consequential LCA to support energy policy: Land use change effects of bioenergy production. *Science of the Total Environment, 472,* 78–89. https://doi.org/10.1016/j.scitotenv.2013.10.097

VDLUFA. (2000). Verband Deutscher Landwirtschaftlicher Untersuchungs- und Forschungsanstalten. Bestimmung des Kalkbedarfs von Acker- und Grünlandböden.

WBA. (2017). World Bioenergy Association. Global Bioenergy Statistics 2017. Retrieved from https://worldbioenergy.org/uploads/WBA%20GBS%202017_hq.pdf (accessed 1 April 2018).

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

Witing, F., Prays, N., O’Keeffe, S., Gründling, R., Gebel, M., Kurzer, H.-J., … Franko, U. (2018). Biogas production and changes in soil carbon input – A regional analysis. *Geoderma, 320,* 105–114. https://doi.org/10.1016/j.geoderma.2018.01.030

Wochele, S., Priess, J., Thrän, D., & O’Keeffe, S. (2014). Crop allocation model “CRAM” – An approach for dealing with biomass supply from arable land as part of a life cycle inventory. In C. Hoffmann, D. Baxter, K. Maniatis, A. Grassi & P. Helm (Eds.), *EU BC&E Proceedings 2014* (pp. 36–40). Florence: ETA-Florence Renewable Energies.

World Energy Council. (2016). World Energy Resources. Bioenergy 2016. Retrieved from https://www.worldenergy.org/wp-content/uploads/2017/03/WEResources_Bioenergy_2016.pdf (accessed 1 April 2018).

**SUPPORTING INFORMATION**

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

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