Environmental balance of the UK biogas sector: An evaluation by consequential life cycle assessment

Styles, David; Dominguez, Eduardo Mesa; Chadwick, Dave

Science of the Total Environment

DOI:
10.1016/j.scitotenv.2016.03.236

Published: 01/08/2016

Peer reviewed version

Environmental balance of the UK biogas sector: an evaluation by consequential life cycle assessment

David Styles†, Eduardo Mesa Dominguez†, Dave Chadwick†*

**Corresponding author: d.styles@bangor.ac.uk +44 (0)1248 38 2502.*

†School of Environment, Natural Resources and Geography, Bangor, Gwynedd, UK, LL57 2UW.

KEYWORDS: anaerobic digestion, GHG, LCA, waste management, bioenergy, resource efficiency

ABSTRACT

Anaerobic digestion (AD) is expanding rapidly in the UK. Previous life cycle assessment (LCA) studies have highlighted the sensitivity of environmental outcomes to feedstock type, fugitive emissions, biomethane use, energy conversion efficiency and digestate management. We combined statistics on current and planned AD deployment with operational data from a survey of biogas plant operators to evaluate the environmental balance of the UK biogas sector for the years 2014 and 2017. Consequential LCA was applied to account for all major environmental credits and burdens incurred, including: (i) substitution of composting, incineration, sewer disposal, field decomposition and animal feeding of wastes; (ii) indirect land use change (ILUC) incurred by the cultivation of crops used for biogas production and to compensate for bakery and brewery wastes diverted from animal feed. In 2014, the UK biogas sector reduced greenhouse gas (GHG) emissions by 551-755 Gg CO_2e excluding ILUC, or 238-755 Gg CO_2e including ILUC uncertainty. Fossil energy depletion was reduced by 8.9-10.8 PJe, but eutrophication and acidification burdens were increased by 1.8-3.4 Gg PO_4e and 8.1-14.6 Gg SO_2e, respectively. Food waste and manure feedstocks dominate GHG abatement, largely through substitution of in-vessel composting and manure storage, whilst food waste and crop feedstocks dominate fossil energy credit, primarily through substitution of natural gas power generation. Biogas expansion is projected to increase environmental credits and loadings by a factor of 2.4 by 2017. If all AD bioelectricity replaced coal generation, or if 90% of biomethane replaced transport diesel or grid natural gas, GHG abatement would increase by 131%, 38% and 20%, respectively. Policies to encourage digestion of food waste and manures could maximize GHG abatement, avoiding the risk of carbon leakage associated with use of crops and wastes otherwise used to feed livestock. Covering digestate stores could largely mitigate net eutrophication and acidification burdens.

INTRODUCTION

Anaerobic digestion (AD) is an established technology to treat wet organic wastes that is increasingly being deployed as a renewable energy technology across Europe to convert a range of feedstocks into biogas and ultimately bioelectricity, bioheat or transport fuel. This has arisen largely in response to energy-related subsidies paid to AD operators, such as feed-in-tariffs (FITs) paid for bioelectricity and the Renewable Heat Incentive (RHI) paid for bioheat (OFGEM, 2016). Up until 2014, almost all biogas produced in the UK was used to fuel combined heat and power (CHP) engines, producing bioelectricity eligible for a FIT subsidy of up to £0.10/kWh in 2015 (OFGEM, 2015). In 2014, there were 32 small-scale plants (≤250kWe) with a cumulative installed capacity of 3.4MWe, 48 medium scale plants (>250kWe≤500kWe) with a cumulative installed capacity of 22.6MWe, and 72 large-scale plants (>500kWe) with a cumulative installed capacity of 136.0MWe (NNFCC, 2014). The overall efficiency of electricity generation can vary significantly across AD-CHP plants (Bacenetti et al., 2013), influencing their performance in terms of greenhouse gas (GHG) abatement and fossil fuel substitution. Similarly, the magnitude of eutrophication and acidification burdens arising largely from ammonia (NH_3) emissions during the storage and field application of digestate are highly dependent on the type digestate storage infrastructure and method of digestate application (Boulamanti et al., 2013: Styles et al., 2015a;b). However, up until now there has been little or no published information on the conversion efficiency of biogas feedstock into electricity, or on the methods of storing and spreading the residual digestate produced, for the UK biogas sector as a whole.

A review of 15 attributional life cycle assessment (LCA) studies (Hijazi et al., 2016) found that biogas energy has a lower GHG intensity than fossil reference energy, but can increase acidification and eutrophication burdens. Whiting and Azapagig (2014) reported that biogas produced from agricultural wastes resulted in significant GHG abatement when it substituted natural gas in CHP engines, but increased acidification and eutrophication burdens by 25 and 12 fold, respectively. Whilst high biogas yields from maize have been found to support generation of bioelectricity with a considerably lower GHG intensity than grid-electricity in Germany and the UK (Whiting and Azapagig, 2014; Claus et al., 2014), maize biogas was also found to increase environmental burdens across eight out of 11 impact categories compared with natural gas (Whiting and Azapagig, 2014). However, fugitive emissions of methane (CH_4) are highly variable (Adams et al., 2015), and may significantly reduce net GHG abatement achieved by crop-biogas. Furthermore, the aforementioned attributional LCA studies did not expand system boundaries to account for important indirect effects of AD system deployment, including the diversion of organic waste from alternative fates such as landfill or composting, and indirect land use change (ILUC) associated with the displacement of food crop production. Such effects could have an important influence on the environmental outcome of AD deployment at the national level, and may be captured by consequential LCA that accounts for indirect effects caused by market signals arising from an intervention (Weidema, 2001).

Styles et al. (2015a) applied consequential LCA to highlight the risk of carbon leakage from ILUC when animal feed is diverted to biogas plants on dairy farms, building on the recognised need to expand LCA boundaries in order to accurately evaluate the environmental efficiency of biogas and other renewable energy options (Rehl et al., 2012; Tufvesson et al., 2013; Tonini et al., 2016). Numerous recent studies have indicated that crop-biogas may actually increase GHG emissions when replacing fossil energy, if ILUC is incurred (Tonini et al., 2012; Hamelin et al., 2012; Styles et al., 2015b). LCA system boundaries have also been expanded to quantify significant GHG credits for biogas plants attributable to the avoidance of manure storage and waste management (e.g. composting), and fertilizer replacement by digestate (Borjesson and Berglund, 2007; Bacenetti et al., 2013; Boulamanti et al., 2013). For example, despite limited bioenergy yields (Lijó et al., 2014a), the digestion of pig slurry can achieve GHG abatement of 1.19 kg CO_2e per kWh of electricity generated when avoided slurry storage emissions are accounted for as a credit (Bancenetti et al., 2013).

Bywater (2011) characterized small-scale farm AD-CHP plants ≤ 250 kWe capacity in the UK, typically co-digesting animal manures and crops, with single-stage digestion and relatively short residence times, often storing digestate in pre-existing manure storage facilities. Larger AD plants usually digest waste and crop feedstocks, and are more likely to be optimized for electricity generation with e.g. two-stage digestion and longer residence times. Given recent findings on large differences in life cycle environmental performance for different types of feedstock and operating parameters, quantifying the overall environmental balance of AD deployment in the UK could provide useful insight for policy makers on the efficacy of policies relating to AD, in terms of climate change, energy security, and air and water pollution. Consequential LCA is the most relevant approach for such an evaluation because it accounts for direct and indirect environmental burdens and credits associated with the introduction of AD, considering process substitution (Weidema, 2001; Tufvesson et al., 2013; Tonini et al., 2016).

A detailed statistical overview of the UK biogas sector has recently been provided by NNFCC (2014a;b), including information on regional deployment, plant scale, feedstock requirements and land requirements, alongside projected biogas capacity development up until 2017. However, that overview did not include information on design and operational parameters critical to environmental performance. We compiled a uniquely detailed profile of the non-wastewater treatment plant (WWTP) biogas sector in the UK by combining plant size and feedstock data (NNFCC, 2014a;b) with operational data obtained from a survey of UK biogas plants. We then applied consequential LCA to evaluate the net environmental balance of the sector, and to explore alternative scenarios of deployment. This paper presents results from the biogas plant survey, and describes how they were used to estimate the environmental profile of the UK biogas sector through consequential LCA.

1. MATERIALS AND METHODS

2.1 Goal, scope and boundary definition

The aim of this study was to quantify the net environmental balance (burden ***change***) associated with recent and projected deployment of AD in the UK, considering incurred and avoided processes and emissions. The study involved the development and application of the *LCAD EcoScreen* tool, applying a consequential LCA approach to calculate the life cycle environmental balance of AD in relation to a reference flow of one Mg of dry matter (DM) feedstock input, for a range of characterized feedstocks, plant operating characteristics and counterfactual (substituted) material flows. Major processes accounted for are shown in Figure 1. The construction and manufacture of buildings and equipment were excluded from the scope, as is typical for bioenergy carbon footprints (EC, 2009).

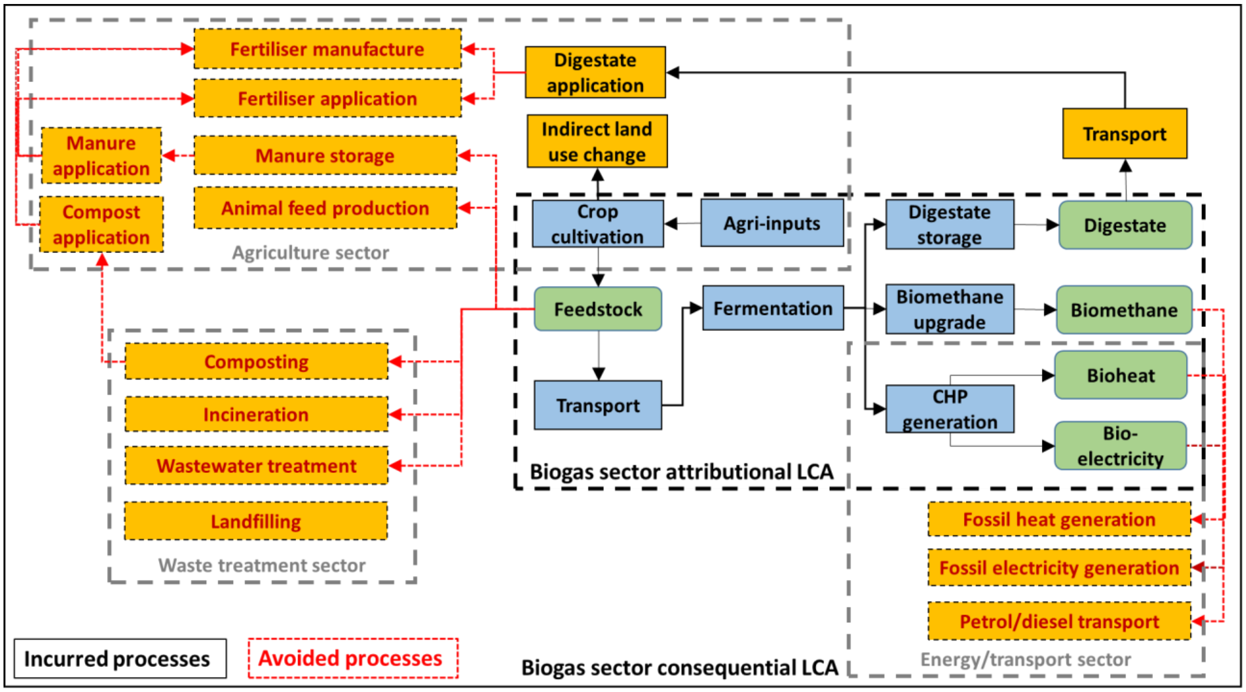


Figure 1. Simplified schematic representation of the major processes considered within the consequential LCA undertaken in this study, compared with an attributional LCA boundary.

Results were expressed in relation to four environmental impact categories based on CML (2010) characterization factors: global warming potential (GWP) expressed as CO_2e, eutrophication potential (EP) expressed as PO_4e, acidification potential (AP) expressed as SO_2e, and fossil resource depletion potential (FRDP) expressed as MJe. For example, 100-yr GWP factors for CH_4 and N_2O are 25 and 298, respectively. Scenario results were normalised against reported annual UK loadings for GWP (DECC, 2015), and against estimated UK loadings for EP, AP and FRDP after per capita extrapolation of EU loadings (Sleeswijk et al., 2008) based on EU 28 and UK population numbers of 510 million and 65 million, respectively (Eurostat, 2016).

2.2. Inventory compilation

LCA was undertaken for 1 Mg DM across 16 types of feedstock, and a total of 77 permutations of feedstock and plant operating parameters, based on national quantities of biogas feedstocks and plant operating characteristics described in the subsequent sections. Table 1 summarises the framework methodology employed, modified from Styles et al. (2015a;b). Inputs accounted for in detailed attributional LCA studies found to be of minor relevance to the four impact categories studied, such as water, lubricating oil and sodium hydroxide (e.g. Poeschl et al., 2012), were disregarded; the aim was to accurately capture wider environmental effects of the sector as a whole.

Table1. Methods applied within the *LCAD EcoScreen* tool to calculate activity data, emissions and environmental burdens in relation to a reference flow of one Mg feedstock dry matter

	Process	**Method and data to calculate primary emissions and burdens in relation to feedstock inputs**
Incurred processes	Cultivation	Burdens = Mg DM x crop cultivation burdens assuming mineral fertiliser application (Table 2) x 1.11 (10% silage loss).
	Indirect land use change	GWP and EP burdens = Mg DM / yield, Mg/ha x terrestrial C and N loss per ha at global agricultural frontier (Styles et al., 2015b), based on IPCC (2006) Tier 1 EFs.
	Transport	Burdens = Mg DM / DM % of wet weight (Table 3) x 5 km/50 km for crops/wastes x Ecoinvent v3.1 burdens per tkm for tractor-trailer/16-32 tonne truck (Table 2).
	Digester leakage	kg CH_4 = Mg DM x m^3/Mg CH_4 yield (Table 3) x 0.67 kg/m^3 x 1% digester loss (Adams et al., 2015).
	CHP combustion	kg CH_4 = Mg DM x m^3/Mg CH_4 yield (Table 3) x 0.67 kg/m^3 – 1% digester loss x 0.5% CHP slip. AP and EP burdens = MJ CH_4 x natural gas CHP burdens from Ecoinvent v3.1 (Table 2).
	Digestate storage	kg CH_4 = Mg DM x m^3/Mg CH_4 yield (Table 3) x 0.67 kg/m^3 x 1.5% for medium and large plants; CH_4 yield x 1.5%/4%/9% (closed/open tank/lagoon) for small plants. kg NH_3-N = Mg DM x total N, kg/Mg x % total N as NH_4-N (Table 3) x 2%/10%/52% for closed tank/open tank/lagoon (Misselbrook et al., 2012). Indirect N_2O-N = NH_3-N x 0.01 (IPCC, 2006).
	Digestate	Burdens = Mg DM / DM % of wet weight (Table 3) x 5 km (0 for manure digestate –

	transport	transported anyway) x burdens per tkm for tractor-trailer from Ecoinvent v3.1 (Table 2). Assumes 1 Mg digestate per 1 Mg feedstock wet weight.
	Digestate application	kg NH_3-N and kg NO_3-N = Mg DM x digestate NH_4-N (Table 3) – storage NH_3-N loss (above) x MANNER NPK EFs (Nicholson et al., 2013). kg N_2O-N = Mg DM x total N, kg/Mg (Table 3) – storage NH_3-N loss (above) x 0.01 + NH_3-N x 0.01 + NO_3-N x 0.0075 (IPCC, 2006). kg P leached = Mg DM x P content, kg/Mg (Table 3) x 0.01 (Withers, 2013). Fertiliser replacement credits = Mg DM x nutrient contents, kg/Mg (Table 3) – storage NH_3-N loss (above) x MANNER NPK availability factors (Nicholson et al., 2013) x fertilizer manufacture and application credits (described below).
Avoided processes (credits)	Avoided manure storage	Avoided kg CH_4 = Mg DM x 800 kg/Mg volatile solids x CH_4-producing capacity for manure type (IPCC, 2006) x 0.67 kg/m^3 CH_4 x CH_4 conversion factor by system type (IPCC, 2006). Avoided kg N_2O-N = Mg DM x total N, kg/Mg (Table 3) x storage system EFs (IPCC, 2006). Avoided kg NH_3-N = Mg DM x total N, kg/Mg (Table 3) x % total N as NH_4-N (Webb and Misselbrook, 2004) x storage system EFs (Misselbrook et al., 2012). Elaborated in SI 3.1.
	Avoided manure application	Avoided kg NH_3-N and kg NO_3-N = Mg DM x total N, kg/Mg (Table 3) x % total N as NH_4-N (Webb and Misselbrook, 2004) x MANNER NPK EFs (Nicholson et al., 2013). Avoided kg N_2O-N = Mg DM x total N, kg/Mg (Table 3) – storage NH_3-N loss (above) x 0.01 + NH_3-N x 0.01 + NO_3-N x 0.0075 (IPCC, 2006). Avoided kg P leached = Mg DM x P content, kg/Mg (Table 3) x 0.01 (Withers, 2013). Avoided fertiliser replacement credits = Mg DM x nutrient contents, kg/Mg (Table 3) – storage NH_3-N loss (above) x MANNER NPK availability factors (Nicholson et al., 2013) x fertilizer manufacture and application credits (described below).
	Avoided in-vessel composting	Avoided emissions, grid electricity use, fertiliser replacement and soil C sequestration as described in Styles et al. (2015b). Elaborated in SI 3.2. Net credit values in Table 2.
	Avoided landfilling	Avoided emissions, methane capture and grid electricity replacement credits as described in Styles et al. (2015a). Elaborated in SI 3.3. Net credit values in Table 2.
	Displaced animal feed	Additional incurred burdens = Mg feedstock DM x Ecoinvent v3.1 burdens for 1 Mg wheat-based concentrate feed (Table 2). ILUC effect from compensatory production calculated as described above for crop feedstock. Elaborated in SI 3.4.
	Avoided field residue decompositi on	Avoided emissions, soil C sequestration and fertiliser replacement credits described in SI 3.5. Net credit values in Table 2.
	Avoided incineration with energy recovery	Avoided burdens = 1 Mg DM animal processing waste x Ecoinvent v3.1 burdens for incineration, corrected for moisture content – credits for avoided natural gas electricity generation, elaborated in SI 3.6. Net credit values in Table 2.
	Avoided sewer disposal	Avoided burdens of wastewater treatment including AD approximated to environmental burdens of on-site AD – elaborated in SI 3.7. Net credit values in Table 2.
	Avoided marginal grid	Avoided burdens = Mg DM x m^3/Mg CH_4 yield (Table 3) x 0.67 kg/m^3 – 1% digester loss – 0.5% CHP slip x 50 MJ/kg LHV x CHP elec. efficiency (35% or 40% for small & medium or large scale AD) – 6% parasitic load x natural gas combined cycle

electricity generation	electricity generation burdens per MJ generated from Ecoinvent v3.1. Net credit values in Table 2.
Avoided oil/gas heating	Avoided burdens = Mg DM x m^3/Mg CH_4 yield (Table 3) x 0.67 kg/m^3 – 1% digester loss – 0.5% CHP slip x 50 MJ/kg LHV x 45% CHP heat efficiency – 33% parasitic heat use x 30%/27%/0% utilisation rate small/medium/large scale AD) x oil/gas heat burdens per MJ heat from Ecoinvent v3.1. Net credit values in Table 2.
Avoided NPK fertiliser manufacture	Avoided burdens = Mg DM x nutrient contents, kg/Mg (Table 3) – storage NH_3-N loss (above) x MANNER NPK availability factors (Nicholson et al., 2013) x Ecoinvent v3.1 burdens for ammonium-nitrate, triple superphosphate and potassium chloride expressed per kg N, P and K. Net credit values in Table 2.
Avoided NPK fertiliser application	Avoided kg NH_3-N = avoided fertilizer N application (above) x 0.017 (Misselbrook et al., 2012). Avoided kg N leached = avoided fertilizer N application (above) x 0.10 (Duffy et al., 2013). Avoided kg N_2O-N = avoided fertilizer N application (above) x 0.01 + NH_3-N x 0.01 + NO_3-N x 0.0075 (IPCC, 2006). Avoided kg P leached = avoided fertilizer P application (above) x 0.01 (Withers, 2013).

Burden data for key inputs and processes are summarized in Table 2. Crop cultivation burdens were based on Styles et al. (2015b) for UK cultivation of maize and grass, and from Ecoinvent v3.1 (Weidema et al., 2013) for fodder beet, and for rye as a proxy for “other cereal silage” (Table 2). The effect of possible land transformation caused by displacement of food production by cultivation of biogas crops, or by additional cultivation of wheat to compensate for animal feed diverted to AD, was evaluated for the purposes of uncertainty analyses. The ILUC factors proposed in Styles et al. (2015b) for land transformation at the global agricultural frontier were applied to an area equivalent to 100% of the area occupied by the biogas or compensatory wheat-feed crops. These factors equate to a GWP burden of 25 Mg yr^{-1} CO_2e and an EP burden of 9.9 kg yr^{-1} PO_4e, respectively, per hectare transformed, based on IPCC (2006) default LUC factors (representing a worst case effect). Methodology and embodied burdens are further elaborated in the following sections, and in SI Section 3.

Table 2. Environmental burden data applied to key inputs and processes, for global warming potential (GWP), eutrophication potential (EP) acidification potential (AP) and fossil resource depletion potential (FRDP)

Input/process	Reference unit	GWP kg CO_2e	EP kg PO_4e	AP kg SO_2e	FRDP MJe
Crop cultivation burdens					
Fodder beet	kg DM	0.33	0.0022	0.0030	3.35
Grass silage	kg DM	0.39	0.0028	0.0025	1.93
Maize silage	kg DM	0.19	0.0016	0.0012	0.92
Other cereal silage	kg DM	0.31	0.0031	0.0029	2.83
(Avoided) upstream burdens for inputs					
Ammonium nitrate-N	kg N	6.10	0.0068	0.024	55.7
Triple superphosphate	kg P	0.889	0.020	0.016	12.5
Potassium chloride K_2O	kg K	0.42	0.0007	0.0014	6.91
Diesel upstream	kg	0.69	0.0009	0.0062	51.6
(Avoided) field emissions					
Fertiliser N application	kg N	5.12	0.054	0.035	-
Manure/digestate N application*	kg N	5.05-6.92	0.034-0.230	0.155-0.525	-
P application	kg P	-	0.031	-	-
Tractor diesel combustion	kg diesel	3.06	0.0005	0.002	-
Tractor-trailer transport	tkm	0.161	0.00025	0.0011	2.14
16-32 tonne Euro IV truck transport	tkm	0.134	0.00011	0.0005	1.75
Biomethane combustion**					
CHP combustion	kWhth		0.00004	0.00045	
EURO V car	vkm		0.00003	0.0001	
Replaced fossil energy					
Marginal gas electricity generation	kWh_e	0.40	0.00004	0.0010	6.55
Coal electricity generation	kWh_e	1.02	0.0010	0.0015	16.9
Oil heating	kWh_{th}	0.33	0.00009	0.0006	4.50
Gas heating	kWh_{th}	0.25	0.00006	0.0004	3.89
EURO V Diesel car	Vkm	0.24	0.0002	0.0007	3.33
EURO V Petrol car	Vkm	0.28	0.0002	0.0008	3.83
Replaced waste management***					
Animal feed (avoided wheat cultivation)	kg DM biowaste	-0.58	-0.0071	-0.0041	-3.04
Field residue	kg DM biowaste	-0.16	0.0035	-0.0003	-0.51
Landfilling	kg DM biowaste	1.99	0.0006	0.0016	-6.01
In-vessel composting	kg DM biowaste	0.66	0.0032	0.0070	1.92
Incineration	kg DM biowaste	0.00	0.0009	0.0005	0.00

*based on MANNER-NPK model (Nicholson et al., 2013); **CHP and upgrade methane slip accounted for separately. ***these counterfactual waste management options, avoided by AD treatment of various waste fractions, are explained in SI section 3.

2.3. National AD feedstock profile

The *LCAD EcoScreen* tool was run for the annual quantities of feedstock required for all UK non-WWTP biogas plants operational as of October 2014, reported by NNFCC (2014a;b). The apportionment of reported feedstock quantities across 16 types, and five main categories of feedstock material (Table 3), is summarized in SI Section 2. In line with the CLCA objectives, only net additional transport incurred for each feedstock was used to calculate biogas sector transport burdens. Thus, no additional transport burdens were incurred for manure feedstocks compared with baseline manure management, nor for onsite processing of wet wastes. Crops and other waste feedstocks incurred transport burdens of 5 and 50 km, respectively. In all cases except for manures it was assumed that digestates incurred additional transport burdens of 5 km for field application.

Table 3. Quantities and characteristics of feedstocks digested in UK biogas plants, in declining order of magnitude, categorized according to five categories: food waste (FW), crop waste (CW), other waste (OW), crops (C) and manures (M)

Feedstock	Digest ate	References

Feedstock	Category	Transport distance (km)	% dry matter	Mg DM yr^{-1} digested **Error! Bookmark not defined.**	kg N Mg^{-1} DM	kg P Mg^{-1} DM	kg K Mg^{-1} DM	m^3 CH_4 Mg^{-1} DM	NH_4-N (% total N)	
Food waste	FW	50	26	462,410	27	11.8	4.1	369	80	WRAP (2010); Defra (2014)
Maize silage	C	5	30	158,130	14.1	6.2	3.8	332	37	FNR (2010)
Other industrial waste	OW	50	26	44,647	16.8	7.3	5.6	323	59	FNR (2010); WRAP (2009)
Other whole-crop cereal silage	C	5	33	43,486	18.6	8.1	8.2	312	37	FNR (2010)
Brewery waste	OW	0	22	41,949	34	14.8	5.1	297	59	Wellinger et al. (2013)
Cattle slurry	M	0	10	40,296	40.7	17.8	14.9	140	75	Defra (2010)
Crop waste	CW	5	17	37,027	26	11.4	6.6	240	59	Defra (2012); Deublein et al. (2008)
Grass silage	C	5	25	32,944	21.5	9.4	6.6	306	37	Styles et al. (2015a)
Animal processing waste	OW	0	15	26,338	20.4	8.9	19.4	199	59	FNR (2010); Wellinger et al. (2013)
Beet	C	5	23	20,206	19.4	8.5	6.9	313	37	Styles et al. (2015b)
Bakery waste	OW	50	61	18,755	37	16.2	1.7	304	59	Poeschl et al. (2012)
Pig slurry	M	0	4	8,809	99.3	43.4	37.3	283	80	FNR (2010); Defra (2010)
Poultry slurry	M	0	10	6,979	55.4	24.2	62.2	225	75	FNR (2010); Defra (2010)
Waste starch	OW	0	3.3	6,200	45	19.6	23.2	703	59	FNR (2010); Fang et al. (2011)
Poultry litter	M	0	40	5,943	50.2	21.9	34.6	225	75	FNR (2010); Defra (2010)
Cheese processing	OW	0	5	3,769	25	10.9	6.3	309	59	FNR (2010); Wellinger et al. (2013)

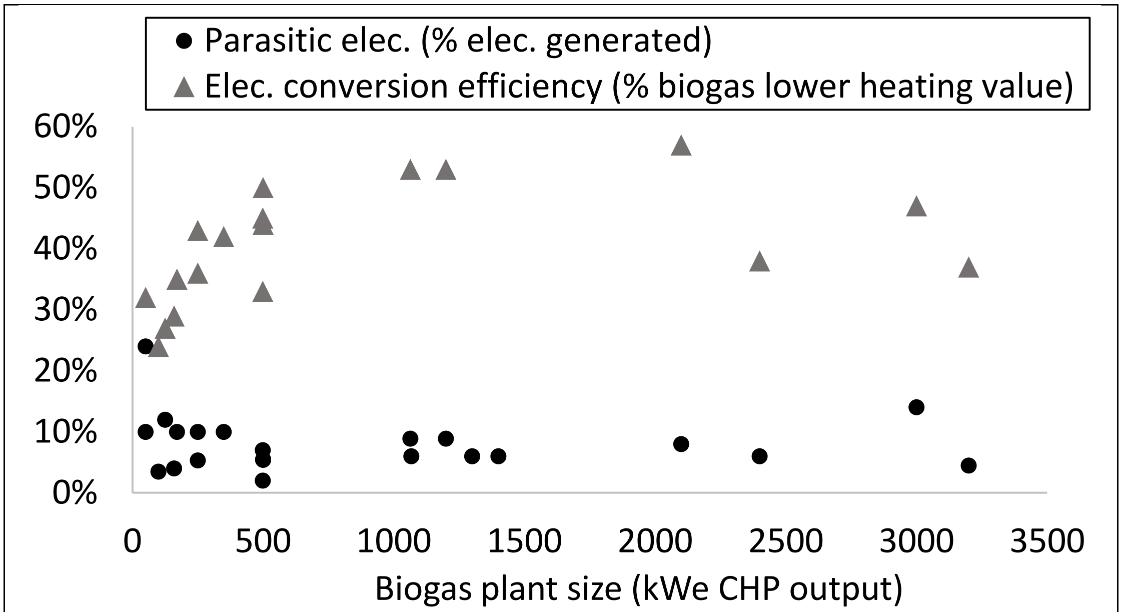
2.4. Plant operating assumptions

As of April 2014, 138 non-WWTP AD plants were operational in the UK (NNFCC, 2014a). An anonymous postal questionnaire was sent to the 78 AD plants for which postal addresses could be found online. The questionnaire contained 30 questions over five sections, covering: feedstock characteristics and quantities; heat and electricity generation and use; digestate storage and application details; operational issues such as maintenance requirements; operator perspectives on AD based on their experience. Twenty-four responses were received to the questionnaire survey relating to 26 biogas-CHP plants which were categorised according to size as per NNFCC (2014a): eight small plants (50-250 kWe); seven medium-sized plants (350-500 kWe); 11 large plants (≥1000 kWe). Not all respondents answered all questions, leaving a reduced number of data points for some parameters. Key results from the survey are summarised in Table SI1.1 and Figure 2.

Electricity conversion efficiency (η_{elec}) was estimated by dividing reported gross electricity generation (G_{elec}, MJ) by the lower heating value (LHV) of biomethane produced ($LHV_{biomethane}$, MJ), which was estimated based on feedstock inputs ($Q_{feedstock}$, Mg) for feedstocks i-n, using estimated default biomethane yields ($Y_{biomethane}$, $m^3\ Mg^{-1}$) reported in Table 3, a density of 0.67 kg m^{-3} and a LHV of 50 MJ kg^{-1} for methane:

$$\eta_{elec} = G_{elec} / [\Sigma(Q_{feedstock\ i\text{-}n} \times Y_{biomethane\ i\text{-}n}) \times 0.67 \times 50].$$

Values ranged from 0.27 to 0.57 (Figure 2), representing variation in actual net biomethane yields and CHP transformation efficiencies.



NB: Electricity conversion efficiency calculated as reported electricity output divided by LHV of default biomethane yields (Table 3) – high values indicate higher biomethane yields.

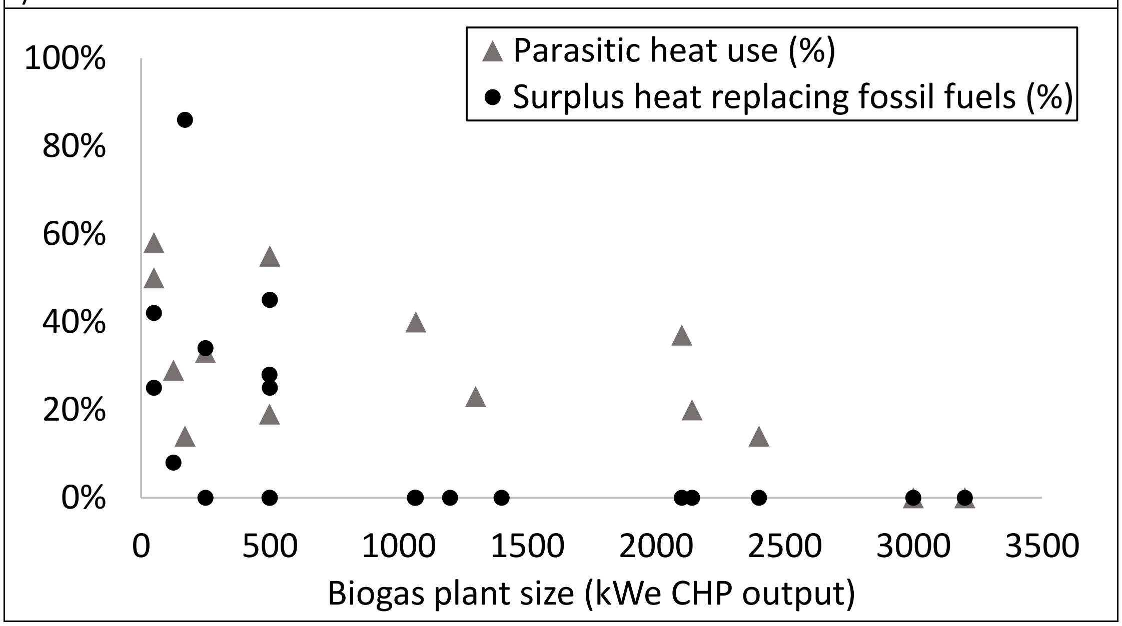


Figure 2. Key operational parameters plotted against AD-CHP electricity generating capacity (kWe)

Non-parametric Kruskal-Wallis analyses were undertaken in IBM SPSS Statistics 22 to test for differences in the distribution of values across AD plant size categories for the following parameters: estimated biomethane electricity conversion efficiency, parasitic electricity requirement, parasitic heat requirement, and the fraction of CHP heat output that replaces fossil-fuel heating. Statistically significant size effects were confirmed for electricity conversion efficiency ($p = 0.007$) and the fraction of CHP heat output replacing fossil fuel heating ($p = 0.015$) (Table SI1.1; Figure 2). Median values for each size category were applied in the national LCA evaluation for these parameters. Estimated biomethane-to-electricity conversion efficiencies (Figure 2; Table SI1.1) indicate that biomethane yields vary from default values presented in Table 3 depending on plant size. Based on median conversion efficiencies (Table SI1.1) and assuming constant CHP electrical efficiencies of 35% and 40% for small and medium/large scale biogas plants, respectively (Bacenetti et al., 2013), biomethane yields in Table 3 were multiplied by 0.91, 1.10 and 1.19 in order to calculate biomethane yields for small, medium and large plants, respectively, in the national LCA. Fractions of CHP electricity and heat output required to run the AD plant (parasitic loads), were not statistically different across size categories (Table SI1.1), so overall median values were used for all sizes of biogas-CHP plant considered in the national LCA evaluation.

2.5. Counterfactual fate of feedstock

Table 4 summarises counterfactual fates of all feedstocks, or in the case of crop feedstocks, the land required to cultivate the crops. Modelling of counterfactual fates was largely based on Styles et al. (2015a;b) and Ecoinvent v3.1 (Weidema et al., 2013), and is elaborated in SI Section 3. Table 1 and table 2 summarise methodology and results for key counterfactual (avoided) burdens. Important aspects are mentioned below.

Table 4. Feedstock quantities modelled according to specified permutations of counterfactual management and digestate storage options storage across three biogas-plant-size categories

	Small (≤250 kWe)			Medium (>250 ≤500 kWe)			Large (>500 kWe)		
Feedstock	**Quantity (Mg dry matter)**	**Counterfactual feedstock (or land) management**	**Digestate Storage**	**Quantity (Mg dry matter)**	**Counterfactual feedstock (or land) management**	**Digestate Storage**	**Quantity (Mg dry matter)**	**Counterfactual feedstock (or land) management**	**Digestate Storage**
Food waste				58,664	In-vessel composting	Lagoon	275,937	In-vessel composting	Lagoon
				9,550	In-vessel composting	Open tank	118,259	In-vessel composting	Sealed tank
Other industrial waste				5,664	In-vessel composting	Lagoon	26,642	In-vessel composting	Lagoon
				922	In-vessel composting	Open tank	11,418	In-vessel composting	Sealed tank
Bakery waste				2,379	Animal feed	Lagoon	11,192	Animal feed	Lagoon
				387	Animal feed	Open tank	4,796	Animal feed	Sealed tank
Brewery waste				5,322	Animal feed	Lagoon	25,033	Animal feed	Lagoon
				866	Animal feed	Open tank	10,728	Animal feed	Sealed tank
Crop waste	1,137	Field residues	Gas-tight	4,611	Field residues	Lagoon	21,690	Field residues	Lagoon
				751	Field residues	Open tank	9,296	Field residues	Sealed tank
Cattle manure	1,201	Crusted tank	Open tank	715	Crusted tank	Open tank	4,129	Crusted tank	Lagoon
	2,562	Open tank	Open tank	1,524	Open tank	Lagoon	8,808	Open tank	Lagoon
	4,243	Lagoon	Open tank	2,525	Lagoon	Lagoon	14,589	Lagoon	Lagoon
Maize silage	5,016	Food/feed production	Lagoon	19,425	Food/feed production	Lagoon	91,369	Food/feed production	Lagoon
				3,162	Food/feed production	Open tank	39,158	Food/feed production	Sealed tank
Other cereal silage	1,379	Food/feed production	Lagoon	5,342	Food/feed production	Lagoon	25,126	Food/feed production	Lagoon
				870	Food/feed production	Open tank	10,768	Food/feed production	Sealed tank
Grass silage				4,047	Food/feed production	Lagoon	19,035	Food/feed production	Lagoon
	1,045	Food/feed production	Lagoon	659	Food/feed production	Open tank	8,158	Food/feed production	Sealed tank
Poultry litter	838	Manure heap	Open tank	429	Manure heap	Lagoon	2,018	Manure heap	Lagoon
				70	Manure heap	Open tank	865	Manure heap	Sealed tank
Poultry slurry	678	Pit storage	Open	347	Pit storage	Lagoon	1,632	Pit storage	Lagoon

			tank	56	Pit storage	Open tank	700	Pit storage	Sealed tank
Fodder beet	641	Food/feed production	Lagoon	2,482	Food/feed production	Lagoon	11,675	Food/feed production	Lagoon
				404	Food/feed production	Open tank	5,004	Food/feed production	Sealed tank
Animal processing waste				3,341	Incineration	Lagoon	15,717	Incineration	Lagoon
				544	Incineration	Open tank	6,736	Incineration	Sealed tank
Waste starch				787	Wastewater treatment	Lagoon	3,700	Wastewater treatment	Lagoon
				128	Wastewater treatment	Open tank	1,586	Wastewater treatment	Sealed tank
Cheese processing waste				478	Wastewater treatment	Lagoon	2,249	Wastewater treatment	Lagoon
				78	Wastewater treatment	Open tank	964	Wastewater treatment	Sealed tank
Pig slurry	1,976	Pit storage	Gas-tight	1,011	Pit storage	Lagoon	4,756	Pit storage	Lagoon
				165	Pit storage	Open tank	2,038	Pit storage	Sealed tank

Emissions of CH_4, N_2O and NH_3 during manure storage were based on factors for specific types of storage system given in IPCC (2006) and Misselbrook et al. (2012) (Table 1). Cattle slurry is the dominant manure feedstock (Table 3). Data from the Defra farm practice survey indicate that 15% of dairy farms store liquid cattle slurry in a crusted tank, 32% in an open tank, and 53% in a lagoon (Defra, 2014a). Counterfactual management of cattle slurry was apportioned across these system types using the above percentages (Table 4). According to Defra (2014a), most "Pigs & Poultry" farms store their manure in a tank, so it was assumed that all pig and poultry slurry going to AD was diverted from "pit storage" and "liquid storage", respectively, as defined in IPCC (2006). It was assumed that all solid poultry litter going to AD was diverted from storage in manure heaps. Emissions and NPK fertilizer replacement for counterfactual manure spreading were calculated using the MANNER NPK model (Nicholson et al., 2013), assuming broadcast application in February and April to a spring crop, and June and September to an autumn crop, on a sandy-clay-loam soil.

Medium and large biogas plants digest a wide range of feedstock (Table SI1.1). In-vessel composting was considered to be the marginal type of waste management avoided by AD treatment of separately collected wet organic waste, such as food waste (Table 4). This reflects the dominant fate of separately collected organic waste, and represents a conservative long-term assumption given that a significant fraction of organic waste still goes to landfill in the UK (Mistry et al., 2011). Bakery and brewery wastes are highly valued as animal feeds; conservatively, it was assumed that these waste types were efficiently utilized to substitute wheat-based animal feed on an energy content basis before being diverted to AD (Table 4). Diversion to AD is thus associated with a possible ILUC effect, because additional wheat must be grown to compensate for the lost animal feed (Tonini et al., 2016), as described in SI 3.4. Waste starch and cheese processing wastes are often liquid, with low solids content, and therefore difficult to transport for alternative uses such as animal feed. These wastes were assumed to be disposed of as liquid effluent in the absence of on-site AD, to be treated in an off-site wastewater treatment plant (Table 4). Conservatively, it was assumed that off-site wastewater treatment involved AD of sewage sludge, leading to an identical environmental balance to on-site AD (from a modelling perspective, this leads to no net change in environmental burden whether wastewater is treated in an on-site or off-site AD plant). Owing to health and hygiene regulations surrounding the handling of animal processing waste, incineration was considered the most likely counterfactual fate for this waste stream (Table 4), modelled based on Ecoinvent v3.1 data (Table 2; SI 3.6). Finally, it was assumed that agricultural land used to cultivate biogas-crops would otherwise have been used to produce food or animal feed. Displaced food or feed production may thus incur ILUC. Waste streams going to medium- and large-scale biogas plants were modelled in quantities determined by the aforementioned statistics and assumptions (Table 4).

2.6. Digestate management

Digestate storage infrastructures for different biogas plant size categories were apportioned as per the distribution across surveyed plants (Table SI1.2). As shown in Table 1, digestate storage infrastructure strongly influences methane and ammonia emissions, although more complete digestion in medium and large-scale plants (indicated by higher electricity yields) constrains methane emissions from digestate storage to 1.5% of biomethane yield irrespective of storage type in these plants (Lijó et al., 2014b). For medium and large biogas plants, in order to avoid bias associated with apportioning particular feedstocks to particular digestate storage assumptions, each feedstock type was apportioned across the digestate storage practices in proportion to their reported prevalence for each size category (Table 4). Uncertainty around fugitive emissions from digestate storage is high (Adams et al., 2015), represented by an uncertainty range of ±25% in uncertainty analyses.

To constrain the number of model runs whilst avoiding bias for particular feedstocks, trailing hose application of digestate was assumed in all cases as a representative, average efficiency technique (Table SI1.1) – approximating to trailing shoe and dribble bar, and between splash plate and injection methods, in terms of NH_3 emissions and fertilizer-N-replacement that dominate the environmental profile of digestate application (Table 1). Emissions and NPK fertilizer replacement values were based on results for February and April application to a spring crop, and June and September application to an autumn crop, on sandy-clay-loam soils, modelled using MANNER NPK (Nicholson et al., 2013) parameterised with relevant digestate nutrient concentrations from Table 3.

2.7. Uncertainty and sensitivity analyses

Despite the use of detailed survey information to parameterise the LCA modelling, significant uncertainty remains over some biogas-plant factors, such as fugitive emissions, and especially over expanded boundary processes. Digestate storage emissions and fossil energy replacement burdens were varied by ±25% and ±10%, respectively. High uncertainty over avoided manure management and waste management burdens for individual feedstocks may average out to lower aggregate uncertainty across all feedstock input at the national scale; nonetheless, national aggregate values were varied ±25%. Uncertainty ranges were calculated excluding and including highly uncertain ILUC burdens for all crops and feedstocks diverted from animal feed.

Four alternative deployment scenarios were run to test for sensitivity to alternative marginal technologies, and the magnitude of environmental benefit associated with mitigation options. Relevant technologies and associated environmental burdens are summarized in Table 2.

Scenario I: The default marginal energy types replaced in the study are natural gas electricity generation via combined cycle turbine and oil heating (DECC, 2012). However, political drivers could lead to coal electricity generation becoming the marginal displaced technology (Finnveden et al., 2005). The possible effect of this was modelled using burdens for coal electricity generation in the UK taken from Ecoinvent v3.1 (Weidema et al., 2013).

Scenario II: Injection of biomethane into the natural gas grid was evaluated as an alternative to CHP in a simple scenario in which 90% of biogas produced nationally in medium and large scale biogas plants was upgraded and injected into the grid (10% combusted in onsite CHP to provide heat and power for the biogas plant). Environmental credits were calculated as avoided natural gas heating burdens from Ecoinvent v3.1 whilst incurred post-digestion burdens were calculated assuming 1.4% CH_4 slip during biomethane upgrading (Ravina & Genon, 2015) and NO_x emissions from biomethane combustion in boilers taken from Ecoinvent v3.1.

Scenario III: The use of upgraded biomethane as a transport fuel was also considered, based on vehicle-km direct burdens only for large Euro V diesel cars powered by biomethane, taken from Ecoinvent v3.1, and avoiding double-counting of upstream burdens for biomethane production calculated in the *LCAD EcoScreen* tool. Environmental credits were calculated as vehicle-km burdens for large Euro V cars powered by diesel, taken from Ecoinvent v3.1, and assuming 1 MJ of biomethane replaces 0.75 MJ diesel on a LHV basis (VTT, 2012).

Scenario IV: The environmental improvement associated with the retro-fitting of floating covers to all open digestate stores was explored, assuming an 80% reduction in NH_3 emissions (FNR, 2010).

2.8. Projected 2017 environmental balance

Results for the environmental balance of the UK biogas sector in 2014 were conservatively extrapolated to projected deployment in 2017, assuming that all existing plants and 40% of planned AD plants (NNFCC, 2014b) would be operational in that year. The environmental balance for each feedstock in 2014 was scaled up according to the ratio of tonnage for each feedstock in 2017 versus 2014 (Table SI2.1). This assumes a similar pattern of deployment across plant sizes in 2017 to 2014, and continued dominance of CHP for energy conversion.

3. RESULTS AND DISCUSSION

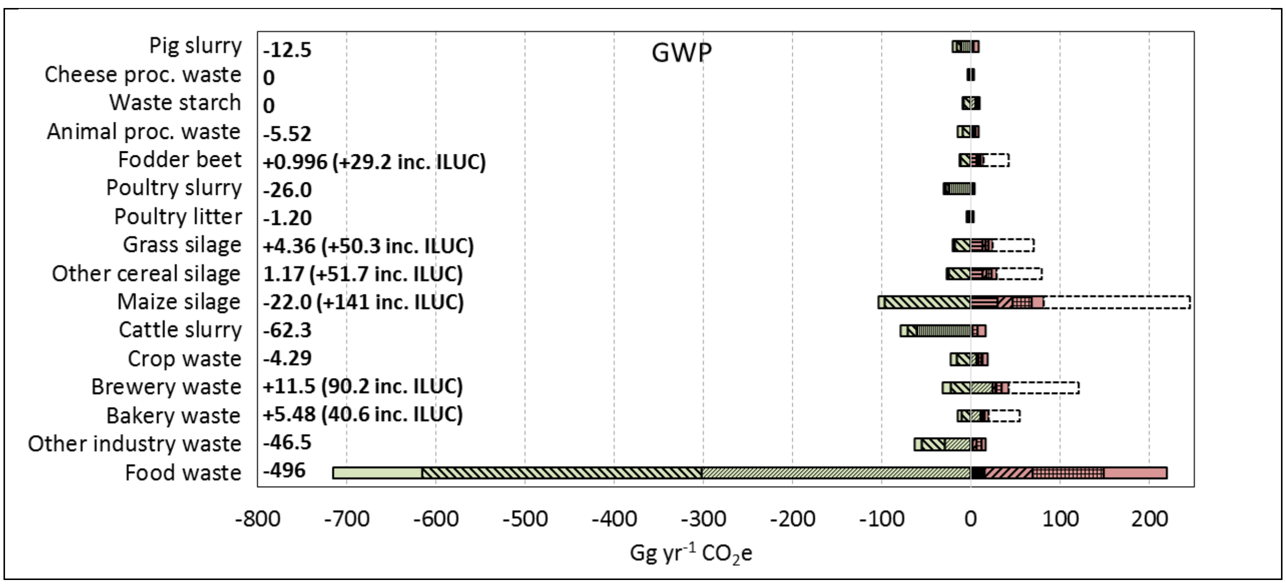
3.1 National balance of UK biogas sector in 2014

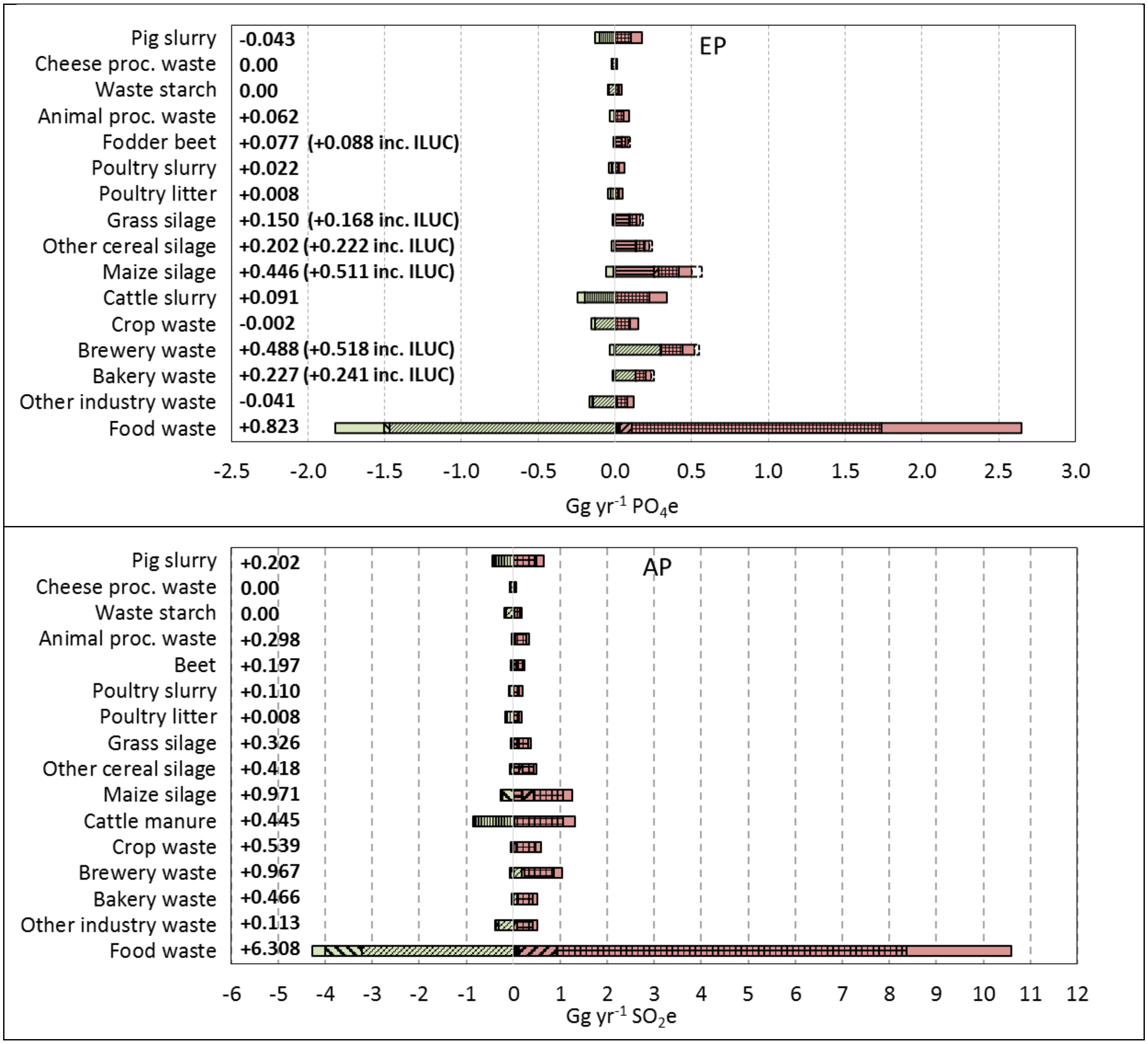
In 2014, deployment of AD in the UK reduced GHG emissions by 653 Gg CO_2e, or 251 Gg CO_2e if worst case ILUC from crop feedstock and diversion of residues from animal feed is accounted for, with a probable range of 238 to 755 Gg CO_2e after propagating major uncertainties (Table 5). The main GWP credits, in order of decreasing magnitude, were substitution of marginal grid electricity, avoided waste management, substitution of fertilisers and avoided manure management (Figure 3; Table SI4.1). Crops represented 27% of feedstock DM and contributed 2% towards GWP savings excluding ILUC, or cancelled out 42% of GWP savings if worst case ILUC was accounted for. Manures and food waste represented 6% and 48% of feedstock DM, and contributed 16% and 76% towards GWP savings from AD, respectively (Figure 4). The digestion of brewery waste and bakery waste increased GWP burdens by 11.5 and 5.5 Gg yr^{-1} CO_2e, respectively, assuming these wastes were diverted from optimum counterfactual use as animal feed, thus incurring compensatory cultivation of wheat feed. If this compensatory wheat cultivation in turn incurs ILUC, then net GWP burdens for these two waste fractions increase to 90.2 and 40.6 Gg yr^{-1}, respectively. The conservative assumption that liquid cheese processing waste and waste starch were diverted from treatment in centralized WWTP AD plants meant that these feedstocks had no net effect on the environmental balance of the non-WWTP AD sector. Aside from crop cultivation and possible ILUC, main GWP burdens arose from digestate storage, digestate application, and fugitive CH_4 leakage during fermentation (Figure 3; Table SI4.1).

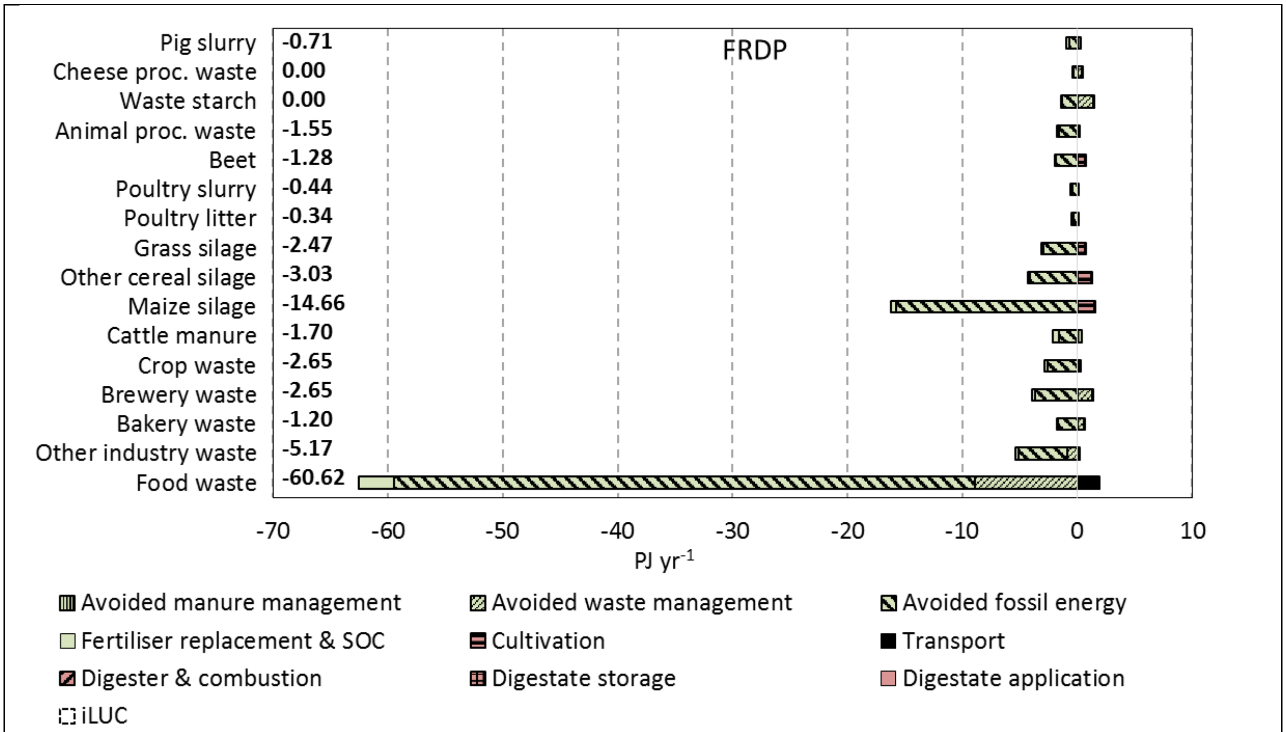
Anaerobic digestion increased national annual eutrophication and acidification burdens by 2.6 Gg PO_4e and 11.4 Gg SO_2e in 2014 excluding ILUC, with ranges of 1.8 to 3.4 Gg PO_4e and 8.1 to 14.6 Gg SO_2e after propagating major uncertainties (Table 5). Crops, food waste and "other waste" contributed approximately one third each towards eutrophication burdens, whilst food waste accounted for 55% of acidification burdens owing to its volume and high NH_3 emissions from NH_4-rich digestate (Figure 4). Notably, digestate storage was found to be a particular hotspot in our analyses, owing to the use of NH_4-driven NH_3 emission factors for different storage systems (Table 1; Misselbrook et al., 2012) which appear to be significantly higher than digestate storage emission factors reported by Amon et al. (2006) and applied in other studies (e.g. Lijó et al., 2014a;b). The higher NH_4-N content and pH of digestates compared with slurries would suggest that NH_3 emissions from digestate storage are likely to be higher than for slurry storage.

Despite their relatively small quantities, brewery and bakery waste made significant net contributions to the eutrophication burden of the AD sector, again owing to diversion from animal feed, incurring additional wheat cultivation. Dominant contributing processes were, in order of decreasing importance, digestate storage, digestate application, cultivation (for EP only), and CHP combustion (Figure 3; Table SI4.1 and Table SI4.2). Avoided manure management and avoided waste management represented the main acidification and eutrophication credits.

In 2014, AD reduced fossil resource depletion in the UK by 9.8 PJe, with a range of 8.9 to 10.8 PJ (Table 5). Food waste and crops contributed 62% and 22%, respectively, towards fossil resource savings. Burdens arising from waste transport, crop cultivation and avoided manure application (avoided fertilizer replacement) were small compared with the credits arising from substitution of marginal grid electricity (Figure 3; Table SI4.4). Digestate application (fertilizer replacement) and the avoidance of waste composting generated small but significant FRDP credits (Figure 3).




EP
Pig slurry -0.043
Cheese proc. waste 0.00
Waste starch 0.00
Animal proc. waste +0.062
Fodder beet +0.077 (+0.088 inc. ILUC)
Poultry slurry +0.022
Poultry litter +0.008
Grass silage +0.150 (+0.168 inc. ILUC)
Other cereal silage +0.202 (+0.222 inc. ILUC)
Maize silage +0.446 (+0.511 inc. ILUC)
Cattle slurry +0.091
Crop waste -0.002
Brewery waste +0.488 (+0.518 inc. ILUC)
Bakery waste +0.227 (+0.241 inc. ILUC)
Other industry waste -0.041
Food waste +0.823
-2.5 -2.0 -1.5 -1.0 -0.5 0.0 0.5 1.0 1.5 2.0 2.5 3.0
Gg yr-1 PO4e
AP
Pig slurry +0.202
Cheese proc. waste 0.00
Waste starch 0.00
Animal proc. waste +0.298
Beet +0.197
Poultry slurry +0.110
Poultry litter +0.008
Grass silage +0.326
Other cereal silage +0.418
Maize silage +0.971
Cattle manure +0.445
Crop waste +0.539
Brewery waste +0.967
Bakery waste +0.466
Other industry waste +0.113
Food waste +6.308
-6 -5 -4 -3 -2 -1 0 1 2 3 4 5 6 7 8 9 10 11 12
Gg yr-1 SO2e



NB: Values next to feedstock category names represent the net balance of annual burden change for that feedstock category.

Figure 3. Environmental burden *changes* arising from deployment of anaerobic digestion in the UK in the year 2014, by feedstock type and by (avoided) process (negative values = credits; positive values = burdens; net burdens written next to feedstock name).

3.2. Environmental balance across plant size categories

Figure 4 shows the breakdown of sectoral environmental credits and burdens, alongside feedstock input, across the three size categories of biogas plant considered, for the year 2014. Small, medium and large biogas plants accounted for 2%, 14% and 83%, respectively, of feedstock DM input (Table 4 and Table SI2.2). For medium and large biogas plants, the shares of sectoral environmental credits and burdens were similar to the shares of feedstock inputs. For small plants, GWP credits were considerably larger (up to 7% of national biogas GWP credit, including ILUC effect) and FRDP credit significantly smaller (1% of national biogas FRDP credit) than the 2% feedstock DM input to these plants. AP and EP burden contributions were also significantly smaller than the relative feedstock input. This reflects the environmental profile of the constrained mix of manure (56% DM), crop waste (5% DM) and purpose-grown-crop (39% DM) feedstocks fed in to small biogas plants (Table 4; Table SI1) – leading to large GWP credits for avoided manure storage, but smaller FRDP credits owing to the lower energy conversion efficiency of small scale plants.

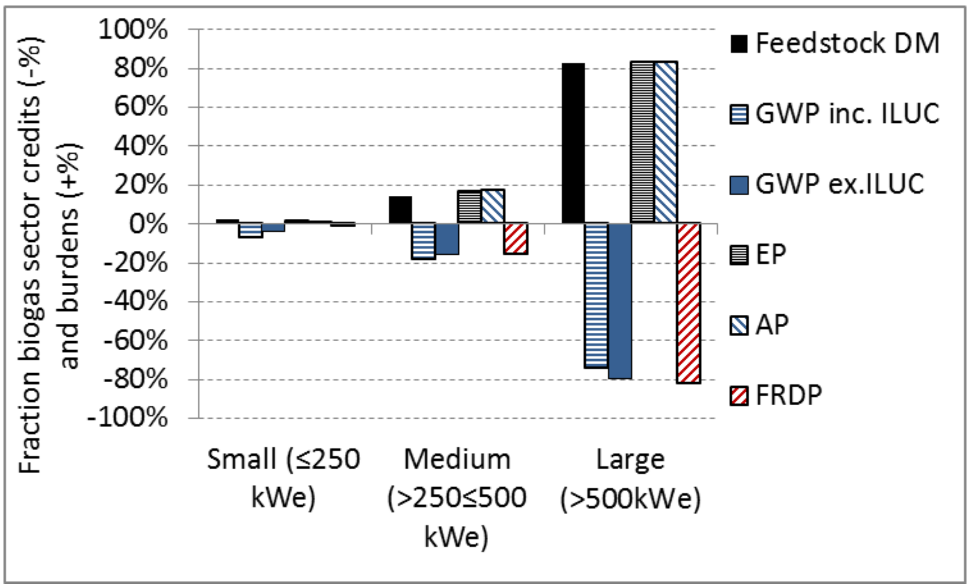
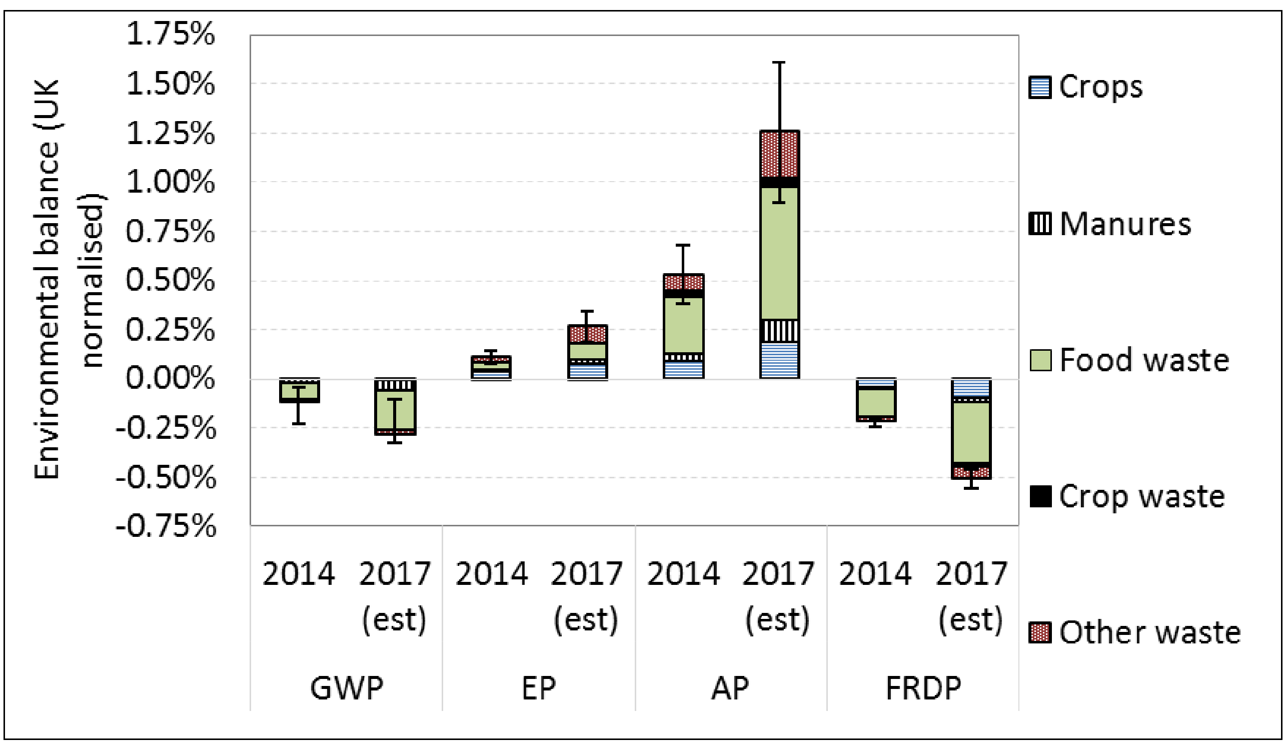


Figure 4. Distribution of sectoral total feedstock input and net environmental credits and burdens across different size categories of AD plant for the year 2014

3.3. Projected national balance of UK biogas sector

In 2014, the AD sector reduced UK total GWP and FRDP burdens by 0.1% and 0.2%, and increased EP and AP burdens by 0.1% and 0.5%, respectively (Figure 5). The net GWP reduction is close to zero at the top end of the uncertainty range, primarily owing to highly uncertain ILUC effects. The quantity of feedstock digested is projected to increase from 5,134 Gg fresh matter in 2014 to 12,118 Gg fresh matter in 2017, with a similar composition (Table SI2.1), leading to reductions of 0.3% and 0.5% in national GWP and FRDP burdens, and increases of 0.3% and 1.3% in national EP and AP burdens, respectively and excluding ILUC effects (Figure 5). Food waste will continue to dominate GWP and FRDP credits and AP burden increases (Figure 5), reflecting its relative volume, avoided composting burdens and NH_3 emissions from NH_4-rich digestate. However, whilst the projected 2017 environmental balance was extrapolated assuming that new biogas plants reflect the current profile of the sector, 34 out of the 415 "in development" biogas plants as of 2014 were biomethane-to-grid plants (NNFCC, 2014b). Therefore, according to grid injection performance reported in Table 5, the GHG and FRDP balance of the biogas sector in 2017 may be slightly better than indicated in Figure 5, though within error bar ranges.



Error bars represent propagated errors for uncertainty ranges shown in Table 5. 2017 values estimated based on NNFCC (2014b) projections.

Figure 5. Contribution of feedstock categories to the environmental balance of 2014 and projected (2017) AD deployment in the UK, normalized against UK environmental loadings

3.4. GHG abatement versus renewable energy generation

Projected AD deployment will lead to a net reduction in national annual GHG emissions of 1,595 Gg (1.59 Mt) CO_2e, but will still only treat approximately 38% and 5%, respectively, of national food wastes and manures in 2017. Thus, there remains considerable technical potential to achieve further GHG abatement through targeted expansion of AD. Treating all food waste generated and all handled (stored) manures in the UK through AD could lead to GHG abatement of 11,473 Gg (11.5 Mt) CO_2e per year, a 5% reduction in national GHG emissions, and could avoid 93 PJ of fossil energy consumption. The diversion of other waste streams, such as industrial wastes, towards AD could increase this theoretical potential further. The economic feasibility of digesting a large share of food waste and industrial wastes in large-scale AD plants is high, owing to gate fees, FiT income and economies of scale. However, whilst it may be economically feasible to install biogas plants on large pig, poultry and dairy farms, the 2015 FiT rate of £0.10 kWh^{-1} bioelectricity for biogas plants ≤ 250 kWe capacity is not sufficient to make small-scale manure AD economically viable (Mesa-Dominguez et al., 2015). Using the same CLCA methodology applied here, Mesa-Dominguez et al. (2015) estimated that digestion of all slurry from large UK dairy farms (> 400 milking cows) could lead to GHG abatement of up to 0.574 Gg yr^{-1} CO_2e, equivalent to a saving of 3.3 kg CO_2e for each kWh of bioelectricity generated. Even at a high FiT rate of £0.20 kWh^{-1}, this would translate into an abatement cost of £60 per Mg CO_2e. In comparison, large-scale maize-fed AD leads to net GHG abatement of just

0.16 kg CO_2e kWh^{-1} (excluding possible ILUC), at a cost of £539 Mg^{-1} CO_2e at the 2015 FiT rate of £0.087 kWh^{-1} for biogas plants >500 kWe capacity (OFGEM, 2015). Results presented in Figure 4 confirm that small-scale biogas plants fed with a high share of manure feedstock outperform the overall biogas sector in terms of GHG abatement, despite comparatively low energy yields. Furthermore, although large-scale maize AD-CHP performs comparatively well against other biogas feedstocks in terms of fossil energy substitution, *Miscanthus* heating pellets, solar PV panels and wind turbines substitute 2, 30 and 90 times more fossil energy per hectare of land utilized, at significantly lower cost (ADAS, 2014; Styles et al., 2015b; Styles, 2015). Thus, there is a case to replace or supplement current FiT and RHI subsidies that incentivize large-scale crop-fed AD plants with incentives that prioritise efficient GHG abatement through deployment of manure- and waste-fed AD. In the UK, a recent RHI review has proposed that the proportion of biogas originating from crop feedstock be limited to 50% for individual plants to receive RHI payments on biomethane injected into the grid (DECC, 2016).

3.5. Sensitivity analyses and mitigation options

The environmental balance of AD deployment in the UK is highly sensitive to the marginal types of waste management and electricity generation replaced. Conservative assumptions regarding the type of waste management option substituted by AD were applied in this study (Table 4), and may underestimate the environmental benefits of AD deployment in the UK. A 25% change in the aggregate environmental credit attributed to waste management substitution would change EP and GWP results by 13% and 11%, respectively (Table 5).

If policy intervention could ensure that AD bioelectricity substitutes electricity generated from coal, rather than natural gas, the environmental balance of national AD deployment would improve substantially; GWP and FRDP credits would more than double (Table 5). Compared with current use of biogas in CHP plants, upgrading biomethane produced in medium- and large-scale digesters to substitute grid natural gas or transport diesel would increase GHG abatement by 20% and 38%, respectively. Similar improvements would be achieved for FRDP, with minor reductions in EP and AP burdens. Meanwhile, introducing a regulatory requirement to cover digestate stores could effectively mitigate net eutrophication and acidification burdens arising from AD (Table 5).

Table 5. Sensitivity of the annual environmental balance of AD deployment to uncertainty in key parameters, and to alternative deployment scenarios, also expressed as percentage difference from results for 2014 deployment excluding land use change

		GWP Gg CO_2e/yr	**EP** Gg PO_4e/yr	**AP** Gg SO_2e/yr	**FRDP** PJe/yr
	Default CHP deployment ex ILUC	-653	2.6	11.4	-9.8
Uncertainty analyses	Default CHP deployment incl. ILUC	-251	2.8	11.4	-9.8
		-62%	**6%**	**0%**	**0%**
	Avoided manure management -25%	-629	2.7	11.7	-9.9
		-4%	**3%**	**3%**	**0%**
	Avoided manure management +25%	-678	2.5	11.0	-9.8
		4%	**-3%**	**-3%**	**0%**
	Avoided waste man -25%	-582	2.9	12.2	-9.7
		-11%	**13%**	**8%**	**-2%**
	Avoided waste man +25%	-724	2.3	10.5	-10.0
		11%	**-13%**	**-8%**	**2%**
	Digestate storage emissions -25%	-690	1.9	8.3	-9.8
		6%	**-26%**	**-27%**	**0%**
	Digestate storage emissions +25%	-616	3.3	14.5	-9.8
		-6%	**26%**	**27%**	**0%**
	Avoided fossil energy -10%	-595	2.6	11.5	-8.9
		-9%	**0%**	**1%**	**-10%**
	Avoided fossil energy +10%	-711	2.6	11.2	-10.8
		9%	**0%**	**-1%**	**10%**
	Lower bound ex. (inc.) ILUC	**-551 (-238)**	**1.83**	**8.12**	**-8.9**
	Upper bound	**-755**	**3.38**	**14.61**	**-10.8**
Scenario analyses	I. Replace coal electricity	-1508	2.3	10.6	-24.2
		131%	**-11%**	**-6%**	**146%**
	II. Grid injection (M&L plants)	-786	2.6	10.6	-12.2
		20%	**0%**	**-6%**	**24%**
	III. Transport (M&L plants)	-901	2.5	9.4	-12.8
		38%	**-6%**	**-17%**	**30%**

IV. Covered digestate stores	-687	0.5	1.6	-9.8
	5%	**-82%**	**-86%**	**0%**

3.5. Summary recommendations

The UK biogas sector delivers important waste management, renewable energy generation and nutrient recycling services, but requires the cultivation of crops and leads to significant emissions from digestate management. Accounting for these effects across multiple systems at a national level requires an expanded-boundary LCA approach, as concluded by Börjesson and Berglund (2007), Tonini et al. (2012) and Tufvesson et al. (2013). Consequential LCA supplements the detailed attributional LCA studies (Poeschl et al., 2012; Bacenetti et al., 2013; Boulamanti et al., 2013; Whiting and Azapagic, 2014; Lijó et al., 2014b) that provide invaluable evidence to benchmark and optimize the operational efficiency of particular systems. Although involving considerable uncertainty, the application of consequential LCA with conservative (worst case) assumptions and uncertainty ranges provides an evidence base for the role of AD in national policies on climate change, renewable energy generation and waste management, considering pertinent indirect effects and "unintended consequences".

The UK biogas sector delivers significant savings in GHG emissions and fossil resource depletion through replacement of: (i) grid-electricity; (ii) in-vessel composting of organic wastes; (iii) manure storage. The sector is projected to grow significantly, with FiT and RHI subsidies favouring more energy-efficient large-scale waste- and crop-fed biogas plants. However, there remains a large technical potential for further GHG abatement through expanded digestion of wastes and manures, whilst minimizing the digestion of crops that can lead to carbon leakage via indirect land use change. Our results also show that, where it is possible to use wastes to feed livestock (e.g. brewery and bakery wastes), this is a more environmentally efficient option than anaerobic digestion, supporting the conclusions of Tufvesson et al. (2013) and Tonini et al. (2016). Rather than basing FiT and RHI incentives on plant size, these subsidies could be based on feedstock type, with higher subsides for manures, and waste feedstocks that cannot be more efficiently utilized as animal feed, in order to derive maximum environmental benefit from AD. Landfill gate fees are partially responsible for driving expansion of AD; in the longer term, as landfilling of organic wastes is eliminated, differential fees (taxes) for waste management options could be based on their respective environmental efficiencies.

Eutrophication and acidification increases caused by the UK biogas sector are primarily attributable to somewhat uncertain NH_3 emissions from digestate management, and fertilizer application to biogas-crops. Covering all digestate stores could largely mitigate burdens caused by NH_3 emissions, and would be a simple regulatory control point that could be checked during plant planning and commissioning.

Finally, two thirds of the net heat output from AD-CHP plants is dumped; upgrading biomethane to substitute grid natural gas or transport diesel would considerably improve the overall environmental balance of AD deployment in the UK. There is some evidence that new RHI subsidies are beginning to encourage the development of biomethane-to-grid plants in the UK (NNFCC, 2014b), but incentives for transport biomethane are so far lacking. Given recent evidence of high NOx emissions from modern diesel engines, there remains a need to evaluate possible air quality and health effects of diesel substitution with biomethane, which could represent an area for cost-effective policy support (simultaneously tackling climate change, energy security and air pollution threats). Biomethane is particularly well suited to powering heavy goods vehicles and buses that are less easily adapted to electric propulsion.

4. CONCLUSIONS

Whilst detailed attributional LCA provides a precise environmental profile of direct AD system effects useful for management decisions, consequential LCA provides a less precise but more complete environmental profile of the AD sector that captures indirect environmental effects pertinent to policy making and regulation. Through the application of consequential LCA and the use of detailed feedstock input data and plant operating characteristics for the AD sector in the UK, this study highlighted the prevailing effects of counterfactual waste management and indirect land use change over biogas energy conversion efficiency in terms of GHG abatement. Accordingly, climate change policy priorities vis-a-vis AD should be to encourage digestion of food waste and manures, whilst restricting digestion of crop inputs and wastes that could be used as animal feed – even if this involves the deployment of smaller biogas plants with lower energy conversion efficiency for manures. It may be wise to focus on AD policy in terms of waste management, rather than renewable energy generation – there are far more efficient renewable energy options in terms of cost and land requirements. Nonetheless, upgrading the biogas produced for use as a transport fuel would considerably improve the environmental profile of the AD sector, compared with current dominant use for electricity generation and also compared with injection of upgraded biomethane into the gas grid. Biogas plant design, specifically the prevalence of open tanks and lagoons for digestate storage, is a dominant factor behind significant net eutrophication and acidification loadings from the UK biogas sector. Regulations requiring covered digestate stores, and injection application of digestate, could largely mitigate this problem.

AUTHOR INFORMATION

Corresponding Author

*David Styles. Email: d.styles@bangor.ac.uk; Tel.: (+44) (0)1248 38 2502.

ACKNOWLEDGMENT

The authors are grateful to the Knowledge Economy Skills Scholarships (KESS) programme, a pan-Wales higher level skills initiative led by Bangor University on behalf of the HE sector in Wales, part funded by the Welsh Government's European Social Fund (ESF) convergence programme for West Wales and the Valleys. The authors would also like to thank AD plant operators who returned questionnaire survey responses and hosted site visits.

Supporting Information. Supporting Information is included in a separate document, under the headings: 1. Survey Results; Methodology; 2. Feedstock Quantities; 3. Counterfactual Fates of Organic Waste Streams; 4. Contribution Analyses.

REFERENCES

Adams, P.W.R., Mezzullo, W.G., McManus, M.C., 2015. Biomass sustainability criteria: Greenhouse gas accounting issues for biogas and biomethane facilities, Energy Policy, 87, 95-109.

ADAS, Review of Land Use Climate Change: An assessment of the evidence base for climate change action in the agriculture, land and wider food chain sectors in wales. ADAS 2014, Leeds UK.

Amon B., Kryvoruchko V., Amon T. & Zechmeister-Boltenstern S., 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agriculture, Ecosystems & Environment, 112, 153-162.

Bacenetti, J., Negri, M., Fiala, M., González-García, S., 2013. Anaerobic digestion of different feedstocks: Impact on energetic and environmental balances of biogas process, Science of The Total Environment, 463–464, 541-551.

Börjesson, P.; Berglund, M., 2007. Environmental systems analysis of biogas systems—Part II: The environmental impact of replacing various reference systems. Biomass and Bioenergy, 31, 326-344.

Boulamanti, A.K.; Maglio, S.D.; Giuntoli, J.; Agostini, A., 2013. Influence of different practices on biogas sustainability. Biomass and Bioenergy, 53, 149–161.

Bywater, A., 2011. A Review of Anaerobic Digestion Plants on UK Farms - Barriers, Benefits and Case Studies. Royal Agricultural Society of England, Warwickshire.

Claus, S.; Taube, F.; Wienfort, B.; Svoboda, N.; Sieling, K.; Kage, H.; Senbayram, M.; Dittert, K.; Gericke, D.; Pacholski, A.; Herrmann, A., 2014. Life-cycle assessment of biogas production under the environmental conditions of northern Germany: greenhouse gas balance. Nitrogen workshop special issue papers, Journal of Agricultural Science, 152, 172–181.

CML, 2010. Characterisation Factors database available online from Institute of Environmental Sciences (CML). In Universiteit Leiden, Leiden.

DECC, 2012. Valuation of energy use and greenhouse gas (GHG) emissions. Department of Energy and Climate Change, UK Government 2012. Available at: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/68764/122-valuationenergyuseggemissions.pdf Last accessed July 2013.

DECC, 2015. Overview: GHG Inventory summary Factsheet. Department of Energy and Climate Change, London. Available at: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/442586/Overview_2013_without_Net_traded.pdf

DEFRA Comparative Life Cycle Assessment of Anaerobic Digestion; 2014. Available from http://sciencesearch.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=18631 (accessed 2 December, 2014).

DEFRA, 2010. Fertiliser Manual RB209. Defra, London.

DEFRA, 2012. Optimising inputs and ouputs from anaerobic digestion processes. Available from http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=14762 Last accessed 26.03.2016.

DEFRA, 2014a. Greenhouse gas mitigation practices – England Farm Practices Survey. Available at: https://www.gov.uk/government/collections/farm-practices-survey Last accessed 01.06.2015.

DEFRA, 2014b. Comparative Life Cycle Assessment of Anaerobic Digestion. Available from http://sciencesearch.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=18631 Last accessed 26.03.2016.

Deublein, D., Steinhauser, A., Eds. 2008. Biogas from waste and renewable resourecs: An introduction, WILEY-VCH Verlag GmbH & Co. KGaA, Weinheim.

EC, 2009. 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. OJEU: L 140/16.

Eurostat, 2016. Population and employment statistics page. Available at: http://appsso.eurostat.ec.europa.eu/nui/show.do Last accessed 15.01.2016

Fang, C.; Boe, K.; Angelidaki, I., Biogas production from potato-juice, a by-product from potato-starch processing, in upflow anaerobic sludge blanket (UASB) and expanded granular sludge bed (EGSB) reactors. Bioresource Technology 2011, 102, 5734–574.

Finnveden, G.; Johansson, J.; Lind, P.; Moberg, A.; 2005. Life cycle assessment of energy from solid waste—part 1: general methodology and results, Journal of Cleaner Production, 13, 213-229.

FNR, 2010. Guide to Biogas: from production to use. Fachagentur Nachwachsende Rohstoffe e.V. (FNR), Gülzow-Prüzen.

Hamelin, L.; Joergensen, U.; Petersen, B.M.; Olesen, J.E.; Wenzel, H., Modelling the carbon and nitrogen balances of direct land use changes from energy crops in Denmark: a consequential life cycle inventory. GCB Bioenergy 2012, 4, 889–907.

Hijazi, O., Munro, S., Zerhusen, B., Effenberger, M., 2016. Review of life cycle assessment for biogas production in Europe, Renewable and Sustainable Energy Reviews, 54, 1291-1300.

http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. IPCC 2006. Available at:

Lijó, L.; González-García, S.; Bacenetti, J.; Fiala, M.; Feijoo, G.; Lema, J.M.; Teresa Moreira, M.; 2014a. Life Cycle Assessment of electricity production in Italy from anaerobic co-digestion of pig slurry and energy crops, Renewable Energy, 68, 625-635.

Lijó, L.; González-García, S.; Bacenetti, J.; Fiala, M.; Feijoo, G.; Moreira, M.T., 2014b. Assuring the sustainable production of biogas from anaerobic mono-digestion, Journal of Cleaner Production, 72, 23-34.

Mesa-Dominguez, E.; Styles, D.; Zennaro, K.; Thompson, P., 2015. Evaluating cost-effective greenhouse gas abatement by small-scale anaerobic digestion, REA & Bangor University. Available at: http://www.biogas.org.uk/images/upload/news_116_REABangorUnismallscaleADreportfinal.pdf Last accessed 26.03.2016.

Misselbrook, T.H., Gilhespy, S.L., Cardenas, L.M., Eds. 2012. Inventory of Ammonia Emissions from UK Agriculture 2011. DEFRA, London.

Mistry, P, Procter, C., Narkeviciute, R., Webb, J., Wilson, L., Metcalfe, P., Solano-Rodriguez, B., Conchie, S., Kiff, B., 2011. Implementation of AD in E&W Balancing optimal outputs with minimal environmental impacts 2011 (AEAT/ENV/R/3162 April 2011).

Nicholson, F.A.; Bhogal, A.; Chadwick, D.; Gill, E.; Gooday, R.D.; Lord, E.; Misselbrook, T.; Rollett, A.J.; Sagoo, E.; Smith, K.A.; Thorman, R.E.; Williams, J.R.; Chambers, B.J.,

2013. An enhanced software tool to support better use of manure nutrients: MANNER-NPK. Soil Use and Management, 29, 473–484.

NNFCC, 2014a. Anaerobic digestion deployment in the United Kingdom. National Non Food Crop Centre, York.

NNFCC, 2014b. Anaerobic digestion deployment in the United Kingdom (updated). National Non Food Crop Centre, York.

OFGEM, 2014. Non-Domestic Renewable Heat Incentive (RHI). Available from: https://www.ofgem.gov.uk/environmental-programmes/non-domestic-renewable-heat-incentive-rhi

OFGEM, 2016. Feed-in Tariff Generation & Export Payment Rate Table for Non-Photovoltaic Installations - FIT Year 6 (1 April 2015 to 31 March 2016); 2015. Available from https://www.ofgem.gov.uk/sites/default/files/docs/2015/07/fit_non-pv_tariff_table_1_october_2015_0.pdf Last accessed 14.03.2016.

Poeschl, M., Ward, S., Owende, P., 2012. Environmental impacts of biogas deployment - Part I: Life Cycle Inventory for evaluation of production process emissions to air. Journal of Cleaner Production, 24, 168–183.

Ravina, M.; Genon, G., 2015. Global and local emissions of a biogas plant considering the production of biomethane as an alternative end-use solution. Journal of Cleaner Production, 102, 115-126.

Rehl, T.; Lansche, J.; Müller, J., 2012. Life cycle assessment of energy generation from biogas—Attributional vs. consequential approach. Renewable and Sustainable Energy Reviews, 16, 3766–3775.

Sleeswijk, W.; van Oers, A.L.; Guinée, J.; Struijs, J.; Huijbregts, M.; 2008. Normalisation in product Life Cycle assessment: An LCA of the Global and European Economic Systems in the year 2000. Science of The Total Environment, 390, 227–240.

Styles, D., 2015. Is bioenergy a sensible use of public money? Presentation delivered at "Talking Food vs Fuel", Hutchinsons Agronomist Conference 2015, Birmingham, February 2015.

Styles, D.; Gibbons, J.; Williams, A.P.; Dauber, J.; Urban, B.; Stichnothe, H.; Chadwick, D.; Jones, D.L., 2015b. Consequential life cycle assessment of biogas, biofuel and biomass energy options in an arable crop rotation. Global Change Biology Bioenergy, 7, 1305–1320

Styles, D.; Gibbons, J.; Williams, A.P.; Stichnothe, H.; Chadwick, D.R.; Healey, J.R., 2015a. Cattle feed or bioenergy? Consequential life cycle assessment of biogas feedstock options on dairy farms. GCB Bioenergy, 7, 1034-1049.

Tonini, D.; Hamelin, L.; Wenzel, H.; Astrup, T., 2012. Bioenergy production from perennial energy crops: a consequential LCA of 12 bioenergy scenarios including land use changes. Environmental Science & Technology, 46, 13521–13530.

Tonini, D., Hamelin, L., and Astrup, T.F., 2016. Environmental implications of the use of agro-industrial residues for biorefineries: application of a deterministic model for indirect land-use changes. GCB bioenergy, doi: 10.1111/gcbb.12290

Tufvesson, L.M.; Lantz, M.; Börjesson, P., 2013. Environmental performance of biogas produced from industrial residues including competition with animal feed – lifecycle calculations according to different methodologies and standards. Journal of Cleaner Production, 53, 214–223.

VVT (2012). Diesel and compressed natural gas combustion emissions from buses. Available at: http://lipasto.vtt.fi/yksikkopaastot/henkiloliikennee/tieliikennee/henkilo_tiee.htm Last accessed 28.09.2015).

Webb, J., Misselbrook, T.H., 2004. A mass-flow model of ammonia emissions from UK livestock production. Atmospheric Environment 38, 2163–2176.

Weidema, B.. 2001. Avoiding co-product allocation in life-cycle. Journal of Industrial Ecology, 4, 11–33.

Weidema, B.P.; Bauer, Ch.; Hischier, R.; Mutel, Ch.; Nemecek, T.; Reinhard, J.; Vadenbo, C.O.; Wernet, G, 2013. The ecoinvent database: Overview and methodology, Data quality guideline for the ecoinvent database version 3, www.ecoinvent.org

Wellinger, A., Murphy, J., Baxter, D., 2013. The biogas handbook. IEA, Paris. ISBN-13: 978 0 85709 498 8

Whiting, A., Azapagic, A., 2014. Life cycle environmental impacts of generating electricity and heat from biogas produced by anaerobic digestion, Energy, 70, 181-193.

WRAP, 2009. Anaerobic Digestate: Partial Financial Impact Assessment of the introduction of a Quality Protocol for the production and use of anaerobic digestate. WRAP, Oxford.

WRAP, 2010. Food Waste Chemical Analysis. Chemical characterisation of food wastes collected from Welsh Local Authorities for supporting decisions related to anaerobic digestion process design and operation. WRAP, Oxford.