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Assessing the environmental sustainability of grass silage and cattle slurry for biogas production

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A B S T R A C T

Grass silage and cattle slurry have been identified as potential significant resources for biogas production. While a higher proportion of grass silage enables a higher specific methane yield to be achieved, there are concerns that using high shares of grass silage may have negative environmental impacts. Previous studies which consider grass as a feedstock have focused on environmental sustainability in the context of greenhouse gas mitigation. However, there is a potential risk of burden shifting occurring if other environmental impacts, such as eutrophication and terrestrial acidification, are not taken into account. A consequential life cycle assessment was conducted to examine mono-digestion of cattle slurry and co-digestion with grass silage in different ratios on a volatile solids (VS) basis. The prior uses of the feedstocks were considered, along with the processes displaced by the biogas and digestate produced. The net environmental impact varied according to the proportion of silage and slurry digested. Higher environmental burdens were observed for mixes with a greater ratio of grass silage to slurry. The optimum environmental performance for the baseline scenario was observed at a VS ratio of 0.4:0.6 for silage and slurry, where there is a net reduction for all impact categories considered. The choice of marginal technologies that are displaced has a significant influence on the results, as have the assumptions about how the grass silage is sourced. This study provides greater insight into the environmental impacts of co-digesting an energy crop with animal manure in varying proportions.

1. Introduction

Addressing interrelated challenges for the climate and environment is one of the most urgent tasks faced by society. In 2019 the European Commission launched 'The Green New Deal', a set of policy initiatives with the overarching aim of making Europe climate neutral by 2050 (European Commission, 2019a). The Deal includes targets for clean energy, biodiversity, farming, and the circular economy. The 'Farm to Fork' Strategy includes targets to reduce nutrient losses by at least 50% and to reduce fertiliser use by 20% by 2030 (European Commission, 2020).

In anaerobic digestion (AD), organic material is converted to biogas in the absence of oxygen (Scarlat et al., 2018). AD is gaining attention as a technology that can provide solutions in several sectors such as renewable energy, waste management, nutrient recycling, and sustainable agriculture (Gustafsson and Anderberg, 2020). Biogas can be combusted in combined heat and power (CHP) units to generate heat and electricity (Hakawati et al., 2017). Alternatively, biogas can be purified to remove trace gases. The upgraded biomethane can then be injected into the natural gas network or used as a transport fuel (Scarlat et al., 2018). The material remaining after AD is known as digestate and can be used as a fertiliser (Scarlat et al., 2018). Biogas production in Europe doubled between 2008 and 2016, from 93 to 187 TWh (Gustafsson and Anderberg, 2020). This was mainly driven by favourable support schemes in place in several European Union Member States (Scarlat et al., 2018).

Different types of organic feedstock can be used for AD, such as farm manure, slurry, food-processing waste and farm crops (Himanshu et al., 2019). The use of crops like grasses and maize silage for biogas production increased significantly in several European countries in the past decade (IEA, 2018). This is largely due to the high methane yields that can be achieved from energy crops which increases the profitability of biogas production, along with favourable support schemes (Scarlat et al., 2018). However, several European countries, including Germany, Austria and Denmark, are

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Abbreviations

AD	anaerobic digestion
CAN	calcium ammonium nitrate
CHP	combined heat and power
CLCA	consequential life cycle assessment
DAP	diammonium phosphate
DM	dry matter
DMY	dry matter yield
ED	electoral division
EF	emission factor
GHG	greenhouse gas
GJ	gigajoule
iLUC	indirect land use change
K	potassium
kWe	kilowatt electrical
LCIA	life cycle impact assessment
LESS	low emission slurry spreading
MWe	megawatt electrical
N	nitrogen
N_2O	nitrous oxide
NBPT	N-(n-butyl) thiophosphoric triamide
NECP	National Energy and Climate Plan
NH_3	ammonia
NOx	nitrogen oxides
P	phosphorus
PJ	petajoule
SMY	specific methane yield
tkm	tonne kilometre
TWh	terawatt hour
VS	volatile solids

limiting the share of energy crops used for biogas production due to concerns around sustainability due to the impact of energy crops on land use change and food security (Scarlat et al., 2018).

The revised Renewable Energy Directive (2018/2001/EU) entered into force in December 2018 and defines a series of sustainability and greenhouse gas (GHG) emission criteria for biomass fuels (European Commission, 2019b). For gaseous biomass fuels, these criteria apply if the thermal input capacity is equal to or exceeding 2 MW (MW) (European Commission, 2019b). Member States may apply the sustainability and GHG emission saving criteria to installations with lower fuel capacity. The Directive also sets limits on feedstock which are high risk for indirect land use change (iLUC), such as palm oil (Dusser, 2019). Biomass fuels which are certified as low iLUC-risk are exempt from these limits (European Commission, 2019b). A strategy has been developed in Italy, called "Biogasdoneright", where farm-scale AD is adapted so it does not compete with traditional food and/or feed production through the use of sequential cropping (Valli et al., 2017).

The AD industry in Ireland is less developed compared to other European countries. In 2018 it was estimated that there were 38 AD plants in Ireland, including 8 landfill gas projects and 19 industrial facilities, including those for wastewater sludge treatment (IEA, 2018). In contrast, there were close to 1,000 AD plants in the United Kingdom by the end of 2016, including 279 plants utilising agricultural feedstocks (IEA, 2018). Gas Networks Ireland, operator of the gas network, have set a target of 11 TWh per annum for biomethane injected to the gas grid by 2030, which corresponds to 20% of current natural gas demand (Gas Networks Ireland, 2019). This is to be derived from grass, animal waste, crop residues and food waste (Gas Networks Ireland, 2019).

Grass may be specifically cultivated as an energy crop for AD (Pehme et al., 2017) or may alternatively be sourced as a by-product from landscape management from non-cultivated areas, including riverbanks, roadside verges and semi-natural grasslands (Nilsson et al., 2020; Bedoić et al., 2019; Boscaro et al., 2018). While residual grass from non-cultivated areas could mitigate the need for agricultural feedstock, this source of biomass has high recovery costs associated with harvesting, along with low energy returns (Boscaro et al., 2018). Another issue is the presence of inert materials such as plastics or cans, depending on the grass origin (Boscaro et al., 2018).

Research suggests that there is significant potential for biogas production from cultivated grass and cattle slurry in Ireland (O'Shea et al., 2017). Approximately 92% of agricultural land in Ireland is grassland (McEniry et al., 2013). The main use for grassland is as feed for livestock, where it is grazed for the majority of the year. It is also used in the production of silage for feed over winter months. Pasture-based farm enterprises dominate, with dairy and beef accounting for two-thirds of gross agricultural output in Ireland (SFSI, 2020). Excess grass silage, surplus to livestock requirements, has been identified as a potential source for biomethane production if management is optimised, particularly the application of fertiliser (McEniry et al., 2013).

Mono-digestion of grass silage has been shown to become unstable in long term operation (Wall et al., 2014). Co-digestion with cattle slurry can enhance the stability and longevity of biogas production. Cattle slurry has a low biogas conversion efficiency compared to other types of biomass such as energy crops (Esteves et al., 2019). Hence, co-digestion of cattle slurry and grass silage can improve biogas efficiency. Himanshu et al. (2019) examined how the characteristics of grass silage and cattle slurry feedstocks affect the cost of biogas production. They found that the proportion of grass silage should be maximised when co-digesting with slurry (3,270 tonnes silage and 2,533 tonnes slurry), as the total cost of methane production progressively decreased as the proportion of silage in the feedstock mixture increased. Higher proportions of grass silage leads to higher methane yields and increased plant profitability (Himanshu et al., 2019).

Given the potential for biogas from grass silage and cattle slurry, it is important to understand the potential consequences of changing the use of these feedstocks, intentionally or otherwise. This extends responsibility beyond the biogas system alone. A consequential life cycle assessment (CLCA) is a suitable approach to assess the environmental sustainability of biogas production from grass silage and cattle slurry in this regard (Weidema et al., 2018). CLCA is a modelling approach where "activities are included in the product system to the extent that they are expected to change as a consequence of change in demand for the functional unit" (UNEP/SETAC, 2011).

Previous CLCAs have examined the impact of mono-digestion of animal slurries (Pehme et al., 2017; De Vries et al., 2012; Hamelin et al., 2011), co-digestion with crops (Tsapekos et al., 2019; Pehme et al., 2017; Styles et al., 2016; Tonini et al., 2015) and with wastes/residues (Styles et al., 2016; Tonini et al., 2015). Those studies highlight the environmental benefits of mono-digestion of slurry compared with traditional manure management. They also emphasised the need to focus on feedstocks which do not compete with food or feed crops for land use.

Pehme et al. (2017) assessed the environmental impact of co-digestion of manure with natural and cultivated grass in Estonia.

Natural grass was unused grass from semi-natural grasslands. The cultivated grass was grown specifically for AD. Co-digestion with cultivated grass showed higher environmental impacts in the global warming, acidification and eutrophication impact categories compared with natural grass. Those authors concluded that it is necessary to improve the nitrogen balance of the supply chain of these scenarios, and to carefully consider the counterfactual use of the grass stream. In the study by Pehme et al. (2017), only one binary mix of manure and grass was assessed, alternative proportions for co-digestion were not considered.

O'Shea et al. (2017) estimated the potential biomethane resource from cattle slurry and grass silage in Ireland and used an optimisation model to determine profitable biomethane plant locations. Profitable plants produced 12 PJ of biomethane, which was 8.6% of the theoretical resource. Those authors briefly considered the potential GHG emissions savings associated with the plants in their model but acknowledged that a full life cycle assessment is needed to determine the sustainability of such a system.

While previous studies which consider grass as a feedstock have focused on environmental sustainability in the context of GHG mitigation (Nilsson et al., 2020, O'Shea et al., 2017), there is a potential risk of burden shifting occurring if other environmental impacts, such as eutrophication and terrestrial acidification, are not taken into account. The objective of this study is to assess the environmental impacts of digesting different proportions of grass silage and cattle slurry for biogas production, using CLCA. The intended application of this study is to provide an evidence-based resource for policymakers and researchers which can be used in combination with other complimentary models and analyses.

2. Methods

The methodology will now be described as follows; biomethane resource (Section 2.1), goal, scope and boundary definition (Section 2.2), life cycle inventory (Section 2.3), life cycle impact assessment (Section 2.4), scenario analysis (Section 2.5), sensitivity analysis (Section 2.6) and uncertainty analysis (Section 2.7).

2.1. Biomethane resource

County Tipperary, which is located in the "Golden Vale" (Fig. 1), has the second highest number of cattle and dairy cows in Ireland (Central Statistics Office, 2020). It has been identified as a region with significant potential for rural AD utilising grass silage and cattle slurry (O'Shea et al., 2017). An updated estimate of the biomethane resource for county Tipperary was determined using the methodology by O'Shea et al. (2017).

2.1.1. Calculation of cattle slurry resource

The smallest areas for which detailed livestock figures are available in Ireland are electoral divisions (EDs). There are 177 EDs in Tipperary. The number of bulls, dairy cows, other cows, and other cattle for each ED in Tipperary were obtained from the Census of Agriculture via the StatBank database of the (Central Statistics Office, 2020). Livestock numbers vary throughout the year as a result of production and culling. In order to avoid over, or under, estimation of livestock numbers, the average of the June and December livestock numbers were used. A scaling factor was applied to each category of cattle within the EDs using the method described by O'Shea et al. (2016).

The most recent Census of Agriculture was conducted in June 2010. There have been significant changes in livestock numbers nationally in the last 10 years. To provide up to date estimates, the total number of each cattle type in Tipperary county for every year from 2010 to 2019 was obtained from the CSO StatBank database (CSO, 2020). The average year-on-year increase or decrease in livestock numbers for each cattle type was determined and the number of each cattle type in each ED was adjusted accordingly.

The potential slurry resource was determined using the values for daily VS excretion for dairy cattle (3.09 kg dm/head/day) and other cattle (1.28 kg dm/head/day) from Ireland's National Inventory (EPA, 2019a). Total annual slurry production for each cattle type was calculated assuming animals are housed indoors for 16 weeks per year (EPA, 2019b) during which time slurry collection is feasible. The specific methane yield (SMY) was taken to be 143 L CH_4/kg VS (Wall et al., 2014).

2.1.2. Calculation of grass silage resource

The total area for pasture, grass silage and hay in Tipperary in 2010 was obtained from the Census of Agriculture (Central Statistics Office, 2020). In this assessment it is assumed that there is no change in area for each type of land use since 2010, as there is a lack of data for land use on a county level since the last Census of Agriculture was conducted. The potential grass silage resource was calculated using the approach described by McEniry et al. (2013). Grasslands were classified into three major soil groups as an indicator of productivity according to the 2018 National Farm Survey (Teagasc, 2019a). The average proportion of soil groups for all farm types was used. The proportion of the total silage area allocated to 1- and 2-cut silage systems was 79% and 21% of the grass silage area, respectively (McEniry et al., 2013).

The annual dry matter yield for grazed grass (pasture) can be calculated for a specified rate of N fertiliser application using the N response equation from Finneran et al. (2012):

$$DMY_{GG} = \left(-0.0444 \times N^2\right) + (38.419 \times N) + 6257.2$$

Where DMY_{GG} is the annual dry matter yield (kg DM/ha) of grazed grass and N is the annual rate of N fertiliser applied (kg/ha). In McEniry et al. (2013), DMY_{GG} is calculated using the average application rate of N fertiliser nationally (65 kg N/ha) and the maximum rate of N fertiliser application allowed under statutory limits (182 kg N/ha). Using the maximum rate of fertiliser application provides grass which is surplus to livestock requirements. However, application of N fertiliser at the maximum rate may be constrained by obligations under the Water Framework and Nitrates Directives as well as requirements to control ammonia and nitrous oxide emissions. Therefore, an intermediate rate of N fertiliser application (124 kg/ha) was used to calculate DMY_{GG} in this study. Average N application rates were maintained for silage and hay production as per McEniry et al. (2013).

The grass requirement for sheep and cattle was calculated using rates of consumption per head of livestock as per McEniry et al. (2013). The herbage utilisation rate for cattle was taken as 0.8 kg DM grass ingested per kg of grass grown, as per the Teagasc (Irish Agriculture and Food Development Authority) Sectoral Road Map (Teagasc, 2016a). Grass silage utilisation rates and the grass utilisation rate for sheep were taken from McEniry et al. (2013). The resource of grass in excess of livestock requirements was found by subtracting the total grass requirement of livestock from the total production of grass. The SMY of grass silage was taken to be 405 L CH_4/kg VS (Wall et al., 2014).

2.1.3. Mapping of cattle slurry and grass silage methane resource

Thematic maps of the potential biomethane resource associated with cattle slurry and surplus grass were generated using ArcMap v 10.7 (ESRI, 2018). These are shown in Fig. 2. The potential resource in catchments of 10 km radius was determined. To estimate the potential bioenergy supply from grass silage and cattle slurry, the

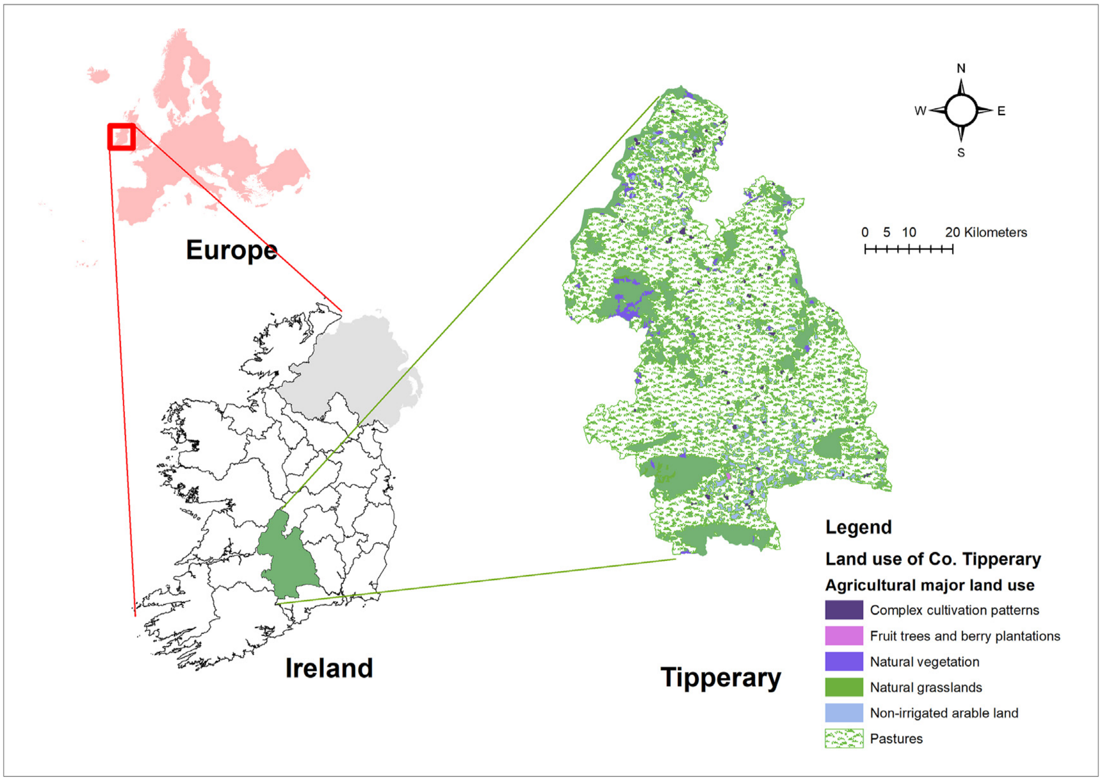
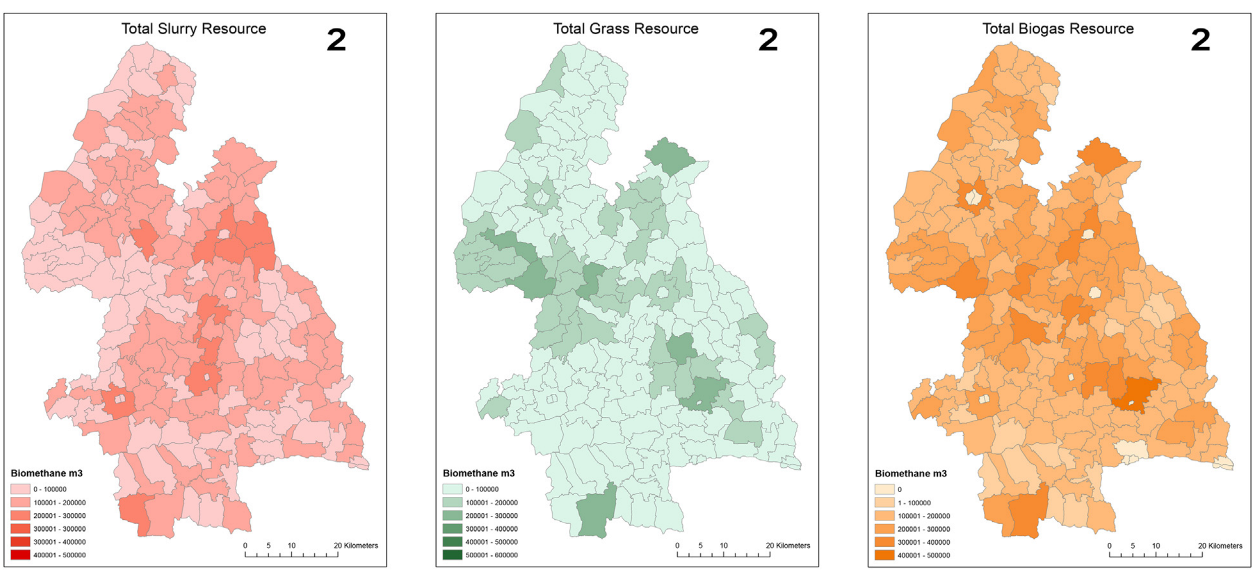


Fig. 1. Geographic scope of the study: County Tipperary, Ireland.



a) Total Slurry Resource for Tipperary b) Total Grass Resource for Tipperary c) Total Biogas Resource for Tipperary

Fig. 2. Biomethane resource (m^3 CH_4) per electoral division.

Sustainable Energy Authority of Ireland (SEAI) considers that 100% of the grass silage resource identified can be utilised in an "enhanced supply" scenario (SEAI, 2016a). For cattle slurry, 8% of the resource identified is utilised in the enhanced supply scenario. The estimate for the cattle slurry utilised is low due to the relatively small individual herd size in Ireland (SEAI, 2016a). For this assessment it is assumed 100% of the grass silage resource identified is utilised and 25% of the slurry resource. The higher estimate for slurry assumes that incentives enable more slurry from smaller farms to be utilised. The breakdown of the potential biomethane supply for Tipperary is shown in Table 1. The majority of EDs in the region have a biomethane resource which is less than 100,000 m^3. Based on the potential utilisable resource identified, an LCA was conducted based on small-scale co-digestion of grass and slurry, i.e. less than 1 MWe capacity (SEAI, 2016b).

Table 1
Potential biomethane resource in Tipperary. The CHP output corresponds to the upper range of biomethane estimated for each category.

Biomethane (m^3 CH_4)	No. of EDs	CHP output (kWe)
0-100,000	108	37.5
100,000–200,000	36	75
200,000–300,000	13	112.5
300,000–400,000	3	150
400,000–500,000	1	187.5
500,000–600,000	1	225

2.2. Goal, scope and boundary definition

The goal of this CLCA is to assess the environmental impact of diverting cattle slurry and grass silage to AD for the production of biogas. Based on the theoretical potential of feedstock determined in Section 2.1, the functional unit is based on the median biomethane output available in an ED and corresponds to a plant size of 150 kWe. The functional unit is defined as “one year of biogas plant operation”. The system boundaries are shown in Fig. 3. The incurred processes include operation of the biogas plant (digester and CHP operation, digestate storage and use) and the additional fertiliser required for the grass silage. The cattle slurry which was previously spread on land is also replaced by mineral fertiliser. The avoided processes are those which are displaced by the use of the feedstocks (land spreading of cattle slurry) and the co-products (biogas and digestate). Electricity produced via the CHP substitutes electricity from the national grid. The surplus grass silage in excess of livestock requirements is made available for biogas production by intensification on existing pasture, i.e. by the inclusion of additional N fertiliser. As digestate production is constrained, i.e. its volume cannot be changed in response to a change in demand for its output, it is not used for the production of feedstock in the system. However, the value of digestate to replace mineral fertiliser outside of the product system is taken into account (see section 2.3.2).

As the geographical scope of this study is County Tipperary, Ireland, foreground data is based on an Irish context. Background data is from the Ecoinvent database version 3.5 consequential system model (Wernet et al., 2016). The specific datasets are listed in the supplementary information. The data for the life cycle inventory has been made available in the repository Mendeley Data (see “Data availability”).

2.3. Life cycle inventory

This section describes the processes that constitute the CLCA model. The relevant methods which are used in the calculation of specific activity data, emissions and environmental burdens are outlined.

2.3.1. Description of biogas plant

The AD system in this study is based on the details provided by Himanshu et al. (2019) for a farm scale AD facility located at Teagasc, Grange, Dunsany, Co. Meath. It consists of a 1,500 m^3 single-stage digester that can co-digest grass silage and cattle slurry. It is assumed that the AD facility is located on a grassland farm with a cattle production enterprise that also requires grass. Animals are accommodated in slatted-floor housing for at least part of the year where slurry is collected in tanks beneath. Mono-digestion of slurry and co-digestion with grass silage in different ratios on a VS basis are assessed. The amount of feedstock required and biomethane produced is shown in Table 2. The digester is assumed to operate at a mesophilic temperature of 37 °C and the total digestion time is 75 days (Himanshu et al., 2019). In the baseline scenario for this assessment, biogas is used in a CHP. The CHP engine is taken to have an electric power output of 150 kW, with 30% electric efficiency and 50% heat efficiency (Gill and Fleming, 2009). A portion of the heat produced is used in the digestion process and the excess is dissipated. The fraction of electricity used for AD processes was taken as 0.10 (Styles et al., 2016). The net electricity produced is exported to the grid.

2.3.2. Identification of marginal technologies

CLCA uses market information to identify which activities are affected by a change in demand for the functional unit (Pehme et al., 2017). Each co-product (biogas and digestate) is assumed to substitute short term marginal technologies. These are existing

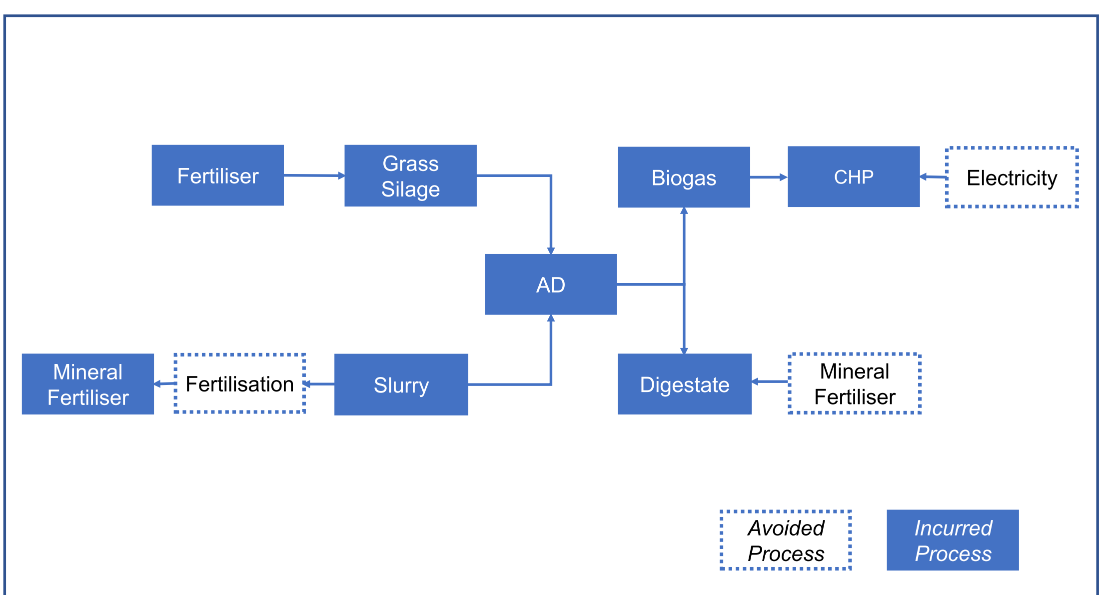


Fig. 3. System boundary for the major processes considered in this study. Incurred processes are shown in boxes with a solid outline and avoided processes are show in boxes with a dotted outline.

Table 2
Feedstock requirement and biomethane produced (Himanshu et al., 2019).

Silage:slurry on VS basis	0:1.0	0.2:0.8	0.4:0.6	0.6:0.4	0.8:0.20
Silage added (t $year^{-1}$)	0	527	1,196	2,071	3,270
Slurry added (t $year^{-1}$)	7,307	6,535	5,557	4,279	2,533
Organic Loading Rate (OLR)	0.91	1.02	1.16	1.33	1.58
Biomethane produced (m^3 $year^{-1}$)	92,814	122,970	161,206	211,339	280,038

technologies whose output changes due to small changes in demand in the market (Van Stappen et al., 2016). These will now be described.

The electricity that is supplied to the grid displaces electricity generated by other fuels. In this study, the marginal source of electricity is taken to be a mixture of different technologies, using different fuels (Lund et al., 2010; Mathiesen et al., 2009). The electricity mix that is displaced is based on Ireland's Draft National Energy and Climate Plan (NECP) (Government of Ireland, 2018). The mix is based on the NECP 2 scenario which assumes high oil prices and includes additional measures. The mix for the selected year (2024) consists of the following fuels for electricity generation; 45% natural gas, 22% coal, 2% peat, 23% wind and 8% biomass.

The digestate can substitute mineral fertiliser. Furthermore, the slurry previously used as an organic fertiliser and now fed to the digester also needs to be replaced by mineral fertiliser. According to the Teagasc Fertiliser Use Survey 2005–2015 (Dillon et al., 2018), Calcium Ammonium Nitrate (CAN) is the most commonly used fertiliser in Ireland for N nutrition and Diammonium Phosphate (DAP) is the most common for Phosphorus (P) and Potassium (K) nutrition.

2.3.3. Emissions

Avoided field emissions from the displaced slurry are included in the analysis, as well as the emissions incurred from the additional fertiliser that replaces slurry. An overview of the methods to calculate emissions for each process is given in Table 3. The fertiliser replacement value of slurry and digestate is determined using the MANNER-NPK software v1.01 (ADAS, 2013), assuming February application to grass on sandy loam soils, with broadcast spreading as the selected application method. Nitrous oxide (N_2O) emissions are based on the Intergovernmental Panel on Climate Change methodology (IPCC, 2006). A Tier 1 approach is used for cattle slurry and digestate, while a Tier 2 approach is used for mineral fertiliser as per Ireland's National Inventory (EPA, 2019a). Ammonia (NH_3) emissions are calculated using the European Environmental Agency methodology (2019); a Tier 1 approach is used for cattle slurry and digestate, while a Tier 2 approach is used for fertiliser.

Fugitive emissions are assumed to be 2.4% of methane produced (Scheutz and Fredenslund, 2019), which includes emissions from digestate storage. Digestate is stored in an open tank. N_2O and NH_3 emissions from digestate storage are calculated as per Styles et al. (2016) and the IPCC (2006). Emissions from the CHP plant are estimated using emission factors (EF) for Danish decentralised CHP plants by Nielsen et al. (2010), with a value of 1.6 g N_2O and 202 g nitrogen oxides (NOx) per GJ of biogas combusted. Data for construction of the AD plant and CHP unit are from the Ecoinvent database version 3.5 cut-off system model. The datasets are for a plant size of 500 m^3 and 160 kWe for the AD plant and CHP unit respectively. As the dataset for the AD plant is smaller than that in the present study (1,500 m^3), the environmental impacts are estimated by scaling up capacity using the approach described in Beausang et al. (2020).

2.4. Life cycle impact assessment

The software used in this assessment is openLCA v1.9.0 (GreenDelta, 2019). ReCiPe 2016 v.1.1 (Huijbregts et al., 2017) is the selected life cycle impact assessment (LCIA) method as it is the most comprehensive and recently updated impact assessment method available in openLCA v1.9.0. The selected characterisation factors are at the midpoint level, as they have a stronger relation to the environmental flows and a relatively low uncertainty (Huijbregts et al., 2017). The selected impact categories for this assessment are climate change, freshwater eutrophication and terrestrial acidification, which are relevant for agricultural and bioenergy systems (Czyrnek-Delêtre et al., 2017; Pehme et al., 2017). The impact factor values for the selected impact categories are listed in the supplementary information.

2.5. Scenario analysis

In the baseline scenario, surplus grass silage (in excess of livestock requirements) is made available for biogas production by additional N fertiliser, specifically CAN. The biogas is used in a CHP and is assumed to displace electricity which is based on the fuel mix for the year 2024. From the baseline scenario, key parameters were identified and tested in a scenario analysis to determine their contribution to the overall results. In each scenario, a specific parameter was tested, while all other parameters were the same as the baseline. These will now be described.

2.5.1. Protected urea

Switching from nitrate-based fertiliser to urea has been identified as a potential measure to mitigate N_2O emissions in Irish agriculture (Teagasc, 2019b). However, urea is vulnerable to NH_3 volatilisation (Krol et al., 2020). The urease inhibitor N-(n-butyl) thiophosphoric triamide (NBPT) has been shown to reduce NH_3 losses from surface-applied urea and increase yield and N uptake in temperate grasslands (Krol et al., 2020). In this scenario, protected urea (urea + NBPT) is assumed to be the source of N instead of CAN.

2.5.2. Biomethane upgrading

The predominant use for biogas in Ireland has been electricity and heat production. However, there is increasing interest in upgrading biogas to biomethane for injection to the gas grid or for use as a transport fuel. According to Gas Networks Ireland, biomethane may constitute 18% of the gas supply by 2030 (Gas Networks Ireland, 2019). In this scenario, 90% of the biogas produced is upgraded to biomethane for injection to the gas grid, while the remaining 10% is combusted onsite to provide heat and power for the biogas plant. The upgrading technology is assumed to be water scrubbing using the burdens of 770 MJ of electricity and 129 kg of water per tonne of CO_2 removed (Tsapekos et al., 2019). The avoided burdens from natural gas heating are from the Ecoinvent database version 3.5 consequential system model.

Table 3
Methods used to calculate activity data, emissions and environmental burdens.

Process	Method used to calculate emissions
Fertiliser application	
Direct N_2O emissions:	kg N_2O = fertiliser kg N x EF x (44/28) (IPCC, 2006)
	EF for CAN: kg N_2O–N/kg N = 0.0140 (EPA, 2019a)
	EF for protected urea: kg N_2O–N/kg N = 0.0040 (EPA, 2019a)
Indirect N_2O emissions (volatilisation)	kg N_2O = fertiliser kg N x 0.1 x0.01 x (44/28) (IPCC, 2006)
Indirect N_2O emissions (leaching)	kg N_2O = fertiliser kg N x 0.30 x 0.0075 x (44/28) (IPCC, 2006)
NH_3 emissions	kg NH_3 = fertiliser kg N x EF x (17/14) (EEA, 2019)
	EF for CAN: kg NH_3/kg N = 0.008 (EEA, 2019)
	EF for protected urea: kg NH_3/kg N = 0.1389 (Krol et al., 2020)
P leached	kg P = P application kg x 0.01 (Styles et al., 2016)
Fertiliser production	Ecoinvent v3.5 burdens for CAN and DAP expressed per kg N, P and K
Fertiliser transport	Ecoinvent v3.5 burdens per tkm
	Transoceanic ship: fertiliser t x 848 km
	Lorry 16–32 metric ton, Euro 5: fertiliser t x 185 km
Avoided slurry storage	
CH_4 emissions	kg CH_4 = Mg DM x 0.8 x 0.24[a] x 0.67 kg/m^3 CH_4 x 0.10[b] (Styles et al., 2016)
	Cattle slurry = 10% DM (Styles et al., 2016)
N_2O emissions	kg N_2O = Mg DM x 40.7[c] x 0.005[d] x (44/28) (Styles et al., 2016)
NH_3 emissions	Tier 2 mass-flow approach based on flow of total ammoniacal nitrogen (TAN) through the manure management system (EEA, 2019)
Avoided slurry application	
Direct N_2O emissions	kg N_2O = slurry kg N x 0.01 x (44/28) (IPCC, 2006)
Indirect N_2O emissions (volatilisation)	kg N_2O = slurry kg N x 0.20x 0.01 x (44/28) (IPCC, 2006)
Indirect N_2O emissions (leaching)	kg N_2O = slurry kg N x 0.30 x 0.0075 x (44/28) (IPCC, 2006)
NH_3 emissions	See above
P leached	kg P = P application kg x 0.01 (Styles et al., 2016)
Digestate storage	
NH_3 emissions	See above
Indirect N_2O emissions	kg NH_3 x 0.01 x (44/28) (Styles et al., 2016)
Digestate application	
Direct N_2O emissions	kg N_2O = digestate kg N x 0.01 x (44/28) (IPCC, 2006)
Indirect N_2O emissions (volatilisation)	kg N_2O = digestate kg N x 0.20 x 0.01 x (44/28) (IPCC, 2006)
Indirect N_2O emissions (leaching)	kg N_2O = digestate kg N x 0.30 x 0.0075 x (44/28) (IPCC, 2006)
NH_3 emissions	See above
P leached	P application kg x 0.01 (Styles et al., 2016)

[a] CH_4 producing capacity for manure type (dairy) (IPCC, 2006), [b] CH_4 conversion factor by system type (IPCC, 2006), [c] Total N, kg/Mg (Styles et al., 2016), [d] Storage system EF (IPCC, 2006).

2.5.3. Alternative feed

In the baseline scenario, grass silage in excess of livestock requirements is made available for biogas production by intensification on existing pasture by inclusion of additional N fertiliser. In the scenario analysis, alternative options for the provision of grass silage are considered for a silage:slurry ratio of 0.8:0.2 VS. According to Teagasc, a 3–4 kg concentrate feeding rate can be used to reduce daily silage feeding by 25% in a dairy herd (Teagasc, 2018). In this scenario a sample concentrate mix composed of 60% barley, 20% distiller's grains and 20% soya hulls is assessed (Teagasc, 2016b). This mix is described as a "high-energy ration suitable for all classes of stock and ad lib diets" (Teagasc, 2016b). Soya hulls were taken to be sourced from Brazil, which accounted for 33% of global production from 2016 to 2017 (Garcia et al., 2019). Data for transport of soya hulls were derived from Garcia et al. (2019). For this assessment it is assumed that distillers' grains and barley are sourced within Europe. Distillers' grains are a by-product of bioethanol production. France is the largest producer of bioethanol in the EU (USDA, 2019) and is also the biggest producer of barley in the EU (Eurostat, 2019). Transport burdens for distiller's grains and barley were derived assuming production in the Centre - Val de Loire region, which is the main area for cereal production in France (Eurostat, 2019).

2.5.4. Revised animal numbers

According to the 2018 National Farm Survey, cattle rearing farms in Ireland had an average family farm income of €8,311, which is the lowest in recent years (Teagasc, 2019a). In contrast, the average family farm income for dairy farms was €61,446 (Teagasc, 2019a). In this scenario, it is assumed that the slurry resource is provided by dairy farms (as in the baseline), but the grass silage resource is made available from a reduction in beef cattle numbers for a co-digestion silage:slurry ratio of 0.8:0.2 VS. The avoided processes from a reduction in animal numbers were modelled based on the system described by Sharma et al. (2018). These processes include enteric fermentation, manure management and application, excretion on pasture, concentrates and fertiliser incurred and are detailed in the supplementary information. The displaced beef is assumed to be substituted by a competing product on the market. According to Sharma et al. (2018) beef from Brazil is assumed to be the global substituted product.

2.6. Sensitivity analysis

Previous CLCA studies have shown that the marginal technologies assumed to be avoided or displaced by the AD plant are the most important factor in the results (Bacenetti et al., 2016). A sensitivity analysis was conducted on the marginal technologies displaced by biogas production. The fuel mix for displaced electricity generation in the baseline model is based on the NECP 2 scenario for the year 2024, which contains a significant amount of coal. In a sensitivity analysis, the fuel mix for displaced electricity generation is based on the NECP 2 scenario for the year 2030, which excludes coal and has a higher proportion of renewables: 53%

natural gas, 2% peat, 36% wind and 9% biomass (Government of Ireland, 2018).

The convention in CLCA is to assume a 1:1 substitution ratio between functionally equivalent product systems. This is a significant source of uncertainty, as in reality there are indirect and scale effects (Plevin et al., 2014). In the scenario analysis, biogas is upgraded to biomethane which is assumed to substitute natural gas on a 1:1 basis. In a sensitivity analysis, the ratio was assumed to be 1:0.5 biomethane to natural gas, based on the upper estimate by de Gorter and Drabik (2011).

2.7. Uncertainty analysis

Uncertainty analysis was performed using Monte Carlo Simulation implemented with openLCA 1.9.0. Data quality indicators in pedigree matrices were used to derive standard deviations which were attached to each parameter in the process inventories. The pedigree matrices include data reliability, completeness and temporal, geographical and technological correlations (Ciroth et al., 2013). Characteristic results were calculated for each scenario with 1,000 iterations with dependent sampling (Henriksson et al., 2014). The data quality indicators are included in the life cycle inventory dataset which is linked to this article (see "Data availability").

3. Results

The contribution of system processes to the LCIA results is shown in Fig. 4. The results for the selected impact categories are discussed below.

3.1. Global warming

For mono-digestion of slurry, there is a potential net reduction in emissions (−155,321 kg CO_2 eq). This is largely due to the avoided electricity that is substituted in the system, as well as the avoided storage and application of slurry (Fig. 4). The processes with the largest positive values include spreading of digestate and fugitive emissions from the plant, however these do not exceed the avoided emissions, so the overall result is negative. Conversely, for the co-digestion of silage:slurry at a ratio of 0.8:0.2 VS, there is a potential net increase in emissions (173,962 kg CO_2 eq). Similar to the result for mono-digestion of slurry, avoided electricity is the process with the largest negative value, however due to the emissions incurred from fertiliser production and application to provide additional grass silage for this mix, there is a net increase in emissions. The overall trend for global warming potential for the different mixes is shown in Fig. 5. As the proportion of silage increases, the potential net reduction in emissions decreases until there is a net increase in emissions at a silage:slurry ratio of 0.6:0.4 VS.

3.2. Freshwater eutrophication

There is a potential net reduction for the freshwater eutrophication impact category for both mono-digestion and co-digestion of all mixes considered. Fertiliser application and spreading of digestate had a net impact in this category, however avoided emissions from the storage of slurry and avoided electricity exceed the emissions incurred, leading to a net reduction in emissions (Fig. 4). As shown in Fig. 5, the potential net reduction in emissions decreases as the proportion of silage increases. This is because emissions savings from avoided slurry storage are reduced as the proportion of silage increases, along with higher burdens for the fertiliser incurred.

3.3. Terrestrial acidification

The overall trend of terrestrial acidification for mono-digestion and co-digestion mixes is shown in Fig. 5. For mono-digestion

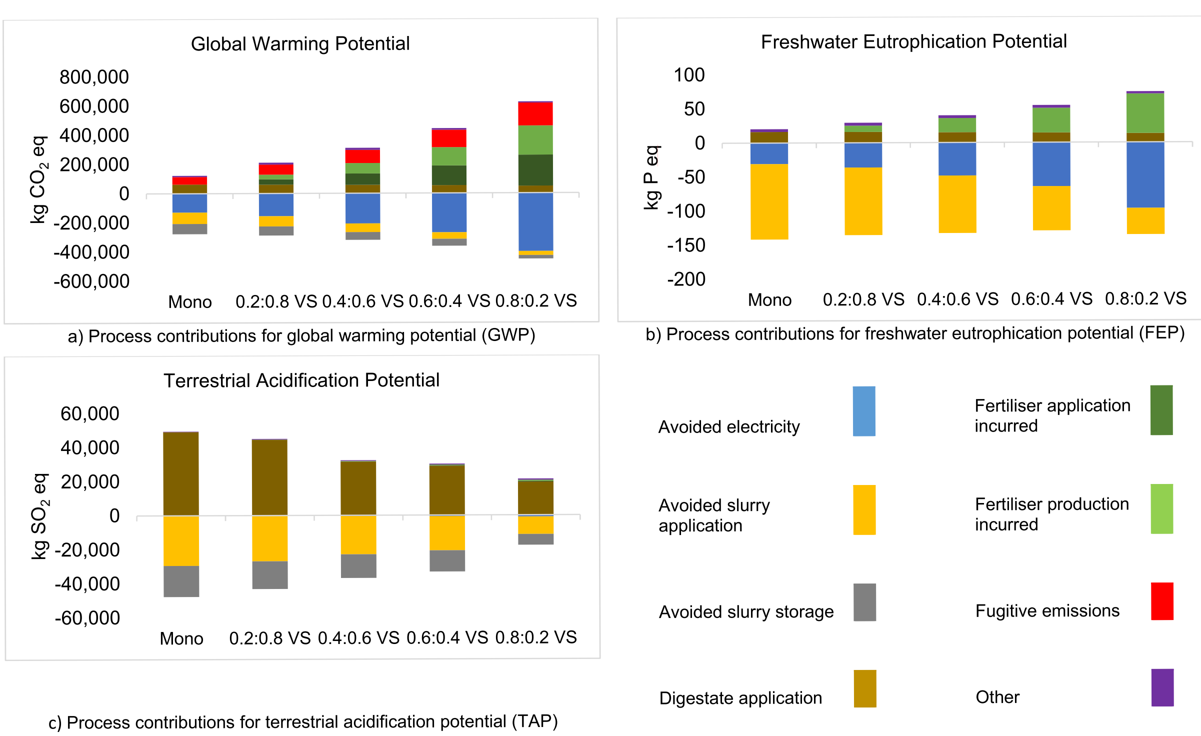
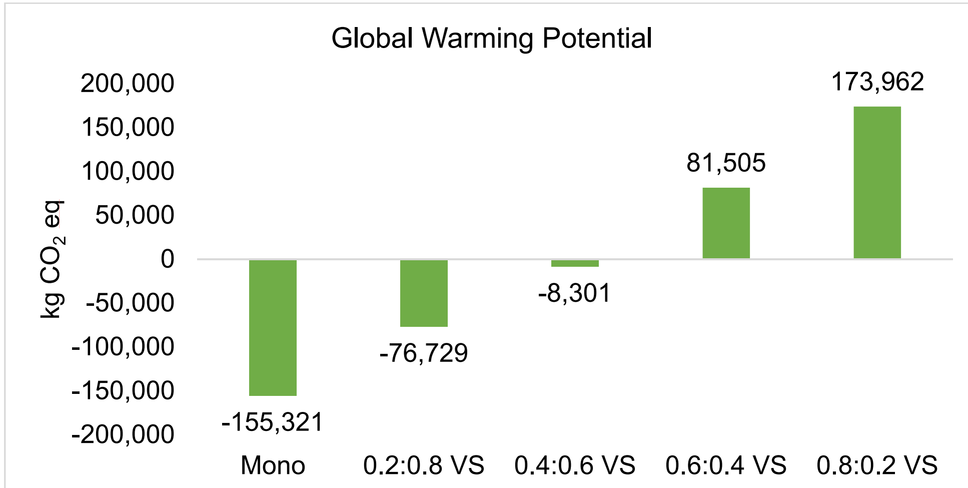
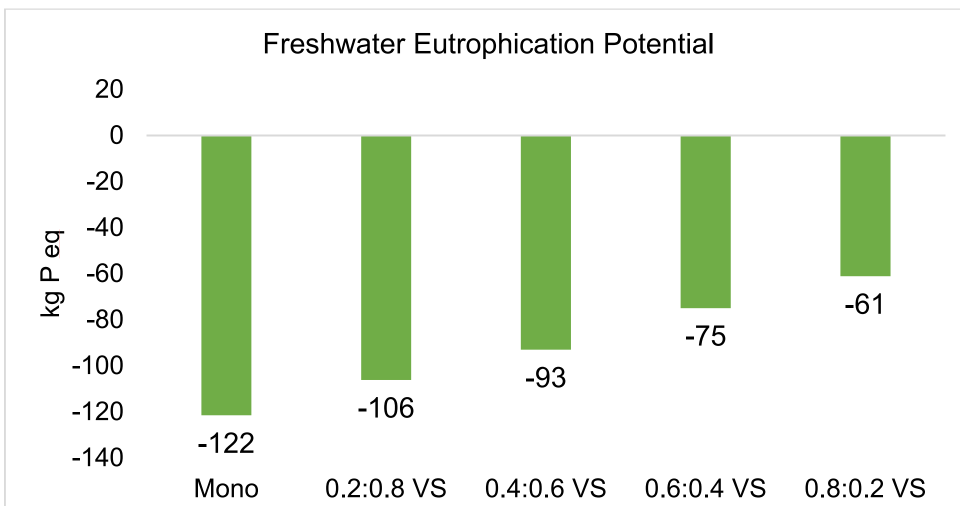


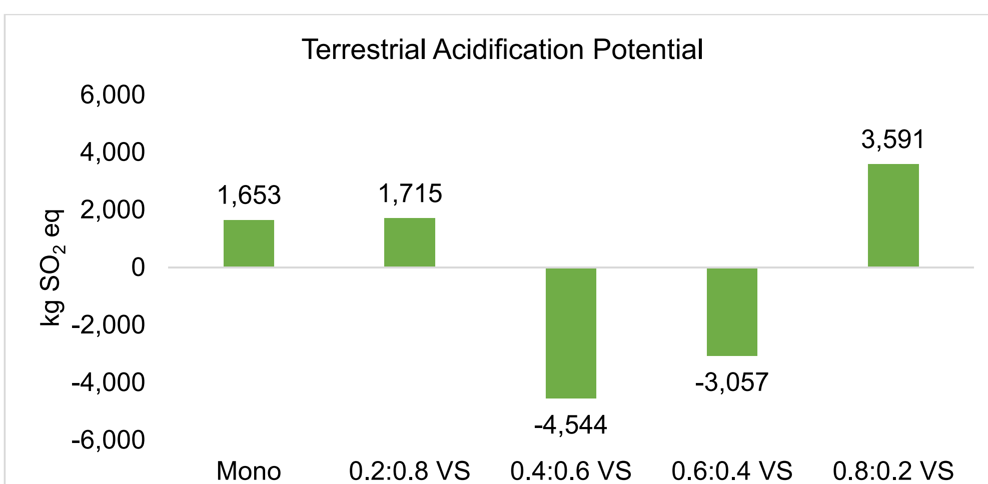
Fig. 4. Process contributions for the selected impact categories.



a) GWP results for mono-digestion and co-digestion mixes



b) FEP results for mono-digestion and co-digestion mixes



c) TAP results for mono-digestion and co-digestion mixes

Fig. 5. Impact category results for mono-digestion and co-digestion mixes.

and co-digestion of silage:slurry at a ratio of 0.2:0.8 VS, there is a potential net increase in emissions. This is due to the large contribution of ammonia emissions from the spreading of digestate (Fig. 4). As the proportion of silage increases, the contribution from digestate spreading decreases, and at silage:slurry ratios of 0.4:0.6 VS and 0.6:0.4 VS, the avoided emissions from slurry storage and application exceed the emissions from digestate spreading, leading to a potential net reduction for these two mixes. However, for the co-digestion of silage:slurry at 0.8:0.2 VS, there is again a net increase in emissions. This mix contains the lowest proportion of slurry, and the avoided emissions from slurry storage and application do not exceed the emissions from spreading of digestate.

3.4. Scenario analysis

The results for the scenario analysis will now be discussed.

3.4.1. Protected urea

For the global warming impact category, if protected urea is used instead of CAN, there is a potential net reduction in emissions for mono-digestion and all co-digestion mixes except silage:slurry at 0.8:0.2 VS, where there is a net increase in emissions (4,315 kg CO_2 eq). This is shown in Fig. 6. The result for freshwater eutrophication is similar to the baseline. Emissions savings decrease as the proportion of silage increases but a net reduction is seen for all mixes (Fig. 6). For the terrestrial acidification impact category, there is a net increase in emissions for all mixes except silage:slurry at 0.4:0.6 VS, where there is a net reduction (−2,560 kg SO_2 eq). Overall while the results for freshwater eutrophication are similar to the baseline in which CAN is the marginal fertiliser, the results for the global warming impact category show a reduced impact compared to the baseline, while the results in the terrestrial acidification category show a higher impact with the use of protected urea.

3.4.2. Biomethane upgrading

For this scenario, biogas is upgraded to biomethane and is assumed to substitute heating by natural gas. For global warming, there is a potential net reduction in emissions for mono-digestion and all co-digestion mixes except silage:slurry at 0.8:0.2 VS where there is a net increase in emissions (87,905 kg CO2 eq) as shown in Fig. 7. The results for the freshwater eutrophication category show a similar pattern (Fig. 7), there is a potential net reduction in emissions for mono-digestion and all co-digestion mixes except silage:slurry at 0.8:0.2 VS where there is a net increase in emissions (36 kg P eq). In this case, the emissions for incurred production of fertiliser exceed the emissions savings from avoided slurry application, leading to a net impact. For terrestrial acidification, the results show a similar trend to the baseline where biogas is combusted in a CHP (Fig. 7). There is a net increase in emissions for mono-digestion and co-digestion of silage:slurry at a ratio of 0.2:0.8 VS and 0.8:0.2 VS, as the emissions from spreading digestate exceed those avoided from slurry storage and application. At silage:slurry ratios of 0.4:0.6 VS and 0.6:0.4 VS, the avoided emissions from slurry storage and application exceed the emissions from digestate spreading, leading to a potential net reduction.

3.4.3. Alternatives for provision of grass silage

For co-digestion of silage:slurry at a ratio of 0.8:0.2 VS, alternative scenarios were considered for the provision of grass silage. In the first scenario, it was assumed that the amount of silage in livestock diets was reduced and replaced with concentrates. The results for this scenario are shown in Table 4. The results were higher compared to the baseline scenario, with a net impact for each of the three impact categories. This was due to the significant impact incurred from the production and transport of feedstuffs. In the second scenario, the slurry resource is provided by dairy farms (as in the baseline), but the grass silage resource is made available from a reduction in beef cattle numbers. The results for this scenario are shown in Table 4. Similar to the scenario for alternative feed, there is a net impact for all impact categories, but the burdens are significantly higher in this scenario. Most of the burden is incurred from the Brazilian beef that is assumed to be the global substituted product.

3.5. Sensitivity and uncertainty analyses

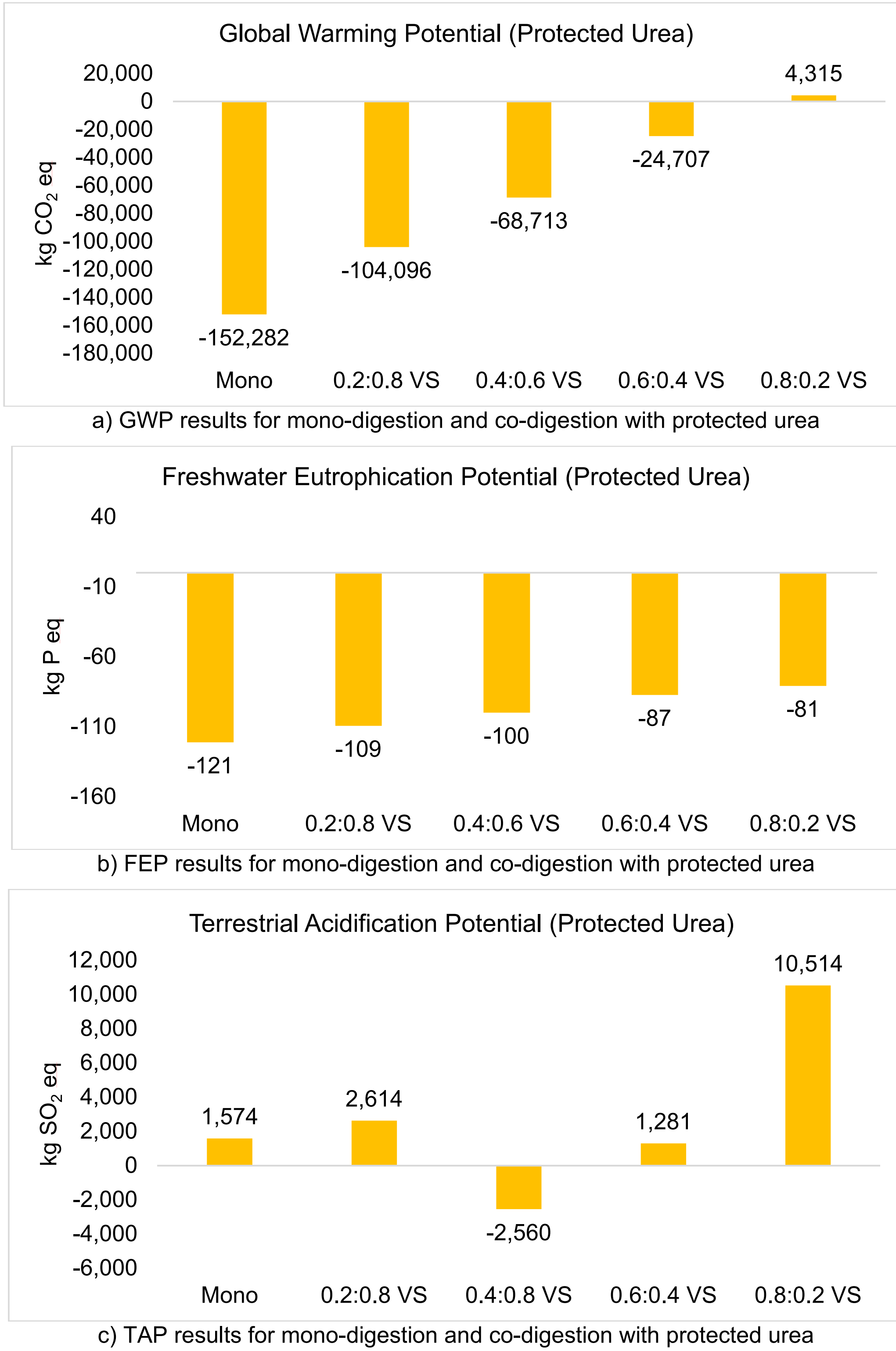
The results for the sensitivity analysis are shown in Table 5. When the NECP 2 electricity fuel mix for the year 2030 is displaced, the emissions savings are reduced for all three impact categories. For global warming potential there is a net impact at a VS ratio of 0.4:0.6 silage:slurry. This is also the case for a substitution ratio of 1:0.5 biomethane to natural gas. Monte Carlo simulation was used to examine uncertainty regarding the numerical data used, i.e. quantity uncertainty. This is visualised in Fig. 8. Reliability of data and temporal correlation were the indicators that contributed the most to quantity uncertainty, as the background data was often based on qualified estimates using datasets that were dated from six years or older.

4. Discussion

Mono-digestion of slurry generally has an improved environmental performance compared with traditional manure management in most impact categories (Beausang et al., 2020; Pehme et al., 2017). Previous studies have found a net reduction for global warming and freshwater eutrophication (Pehme et al., 2017; De Vries et al., 2012; Hamelin et al., 2011), which is in line with the results presented in this study. For terrestrial acidification, there is a net impact for mono-digestion of slurry in this study. This is in line with previous research (Pehme et al., 2017; De Vries et al., 2012; Hamelin et al., 2011). The result for terrestrial acidification is due to the impact of ammonia emissions from digestate. Digestate has been shown to have higher levels of ammonium compared with the organic substrate going into the AD process (Risberg et al., 2017).

Styles et al. (2016) suggest ammonia emissions could be addressed by covering digestate stores and injection application of digestate. Under the terms and conditions for the 2020 Nitrates Derogation, farms stocked above 170 kg organic N must use low emission slurry spreading (LESS) equipment for all slurry spread after April 15th, 2020 (Department of Agriculture, Food and the Marine, 2020), which includes trailing shoe and injection. LESS has been identified as a measure with significant potential to abate ammonia emissions and its uptake is expected to increase in the near future (Teagasc, 2020).

The impacts for co-digestion of slurry with substrates is highly dependent on the alternative use of the substrate in question. Generally, it is recommended that co-digestion with feedstocks that may otherwise be used for feed or food production should be avoided (Styles et al., 2016; Tonini et al., 2015). In the baseline scenario for this assessment, grass silage is made available for co-digestion without competing with feed for livestock by increasing productivity on existing pasture through additional N fertiliser. The results show that the optimum environmental performance for the baseline scenario can be found at a VS ratio of 0.4:0.6 for silage and slurry, where there is a net reduction for all impact categories considered. Alternative scenarios for the provision of grass silage, either by substituting with concentrates or reducing beef cattle numbers and substituting with a competing global product led to a higher environmental impact for the impact categories considered.



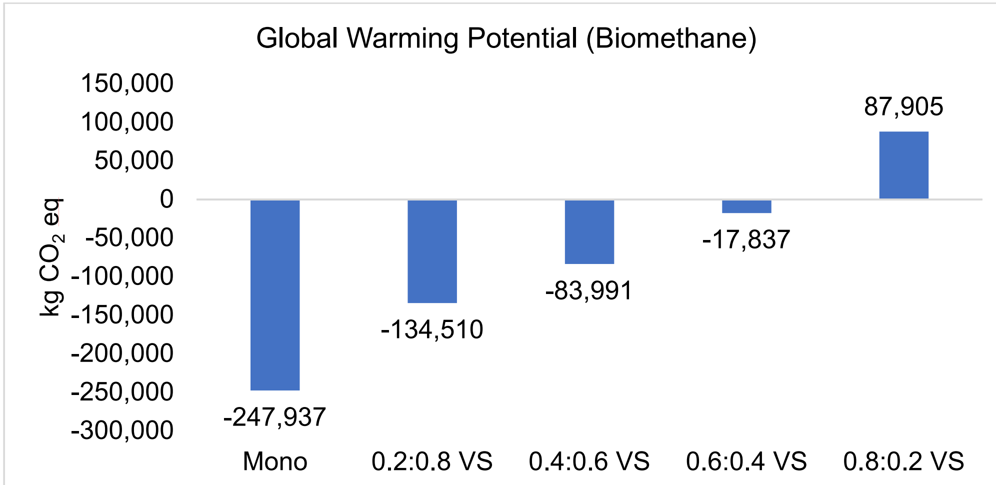
a) GWP results for mono-digestion and co-digestion with protected urea

b) FEP results for mono-digestion and co-digestion with protected urea

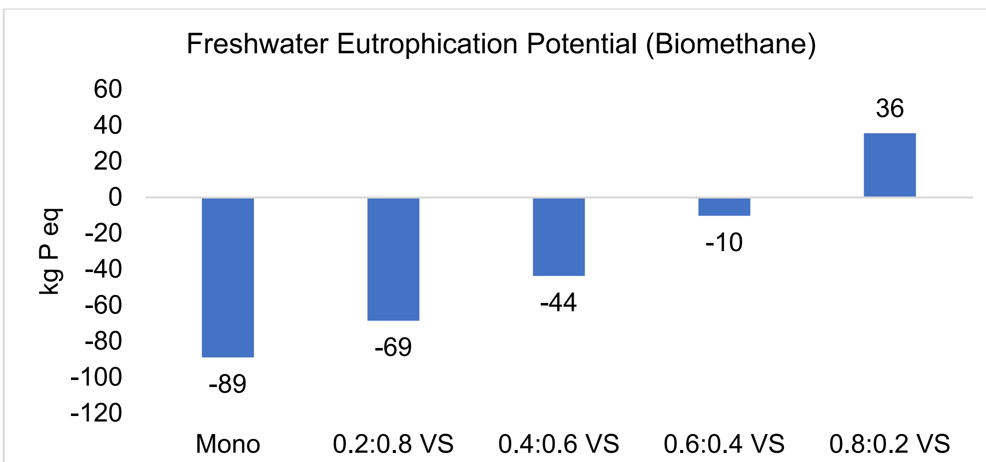
c) TAP results for mono-digestion and co-digestion with protected urea

Fig. 6. Impact category results for mono-digestion and co-digestion mixes using protected urea as the source of N fertiliser.

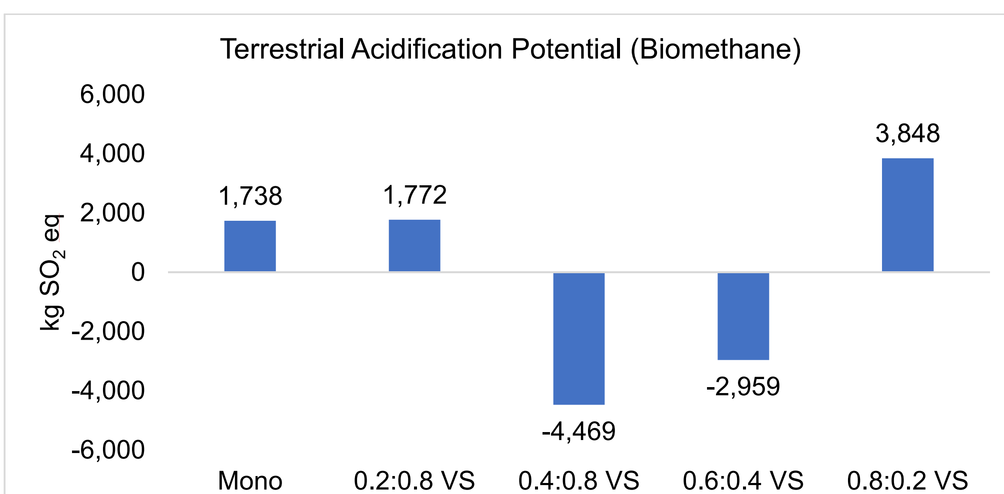
Co-digestion of slurry with higher proportions of grass silage (0.6:0.4 and 0.8:0.2) shows increased environmental burdens for global warming and terrestrial acidification. While the use of protected urea instead of CAN improves the environmental performance in the global warming category, there is a distinct trade-off with increased impacts for terrestrial acidification due to increased ammonia emissions. Agriculture is the dominant source of ammonia emissions in Ireland which arise from the decomposition of animal manures and the application of fertiliser (EPA, 2020). Ireland exceeded its emission ceiling for ammonia for three years in a row; 2016, 2017 and 2018, and emissions are projected to continue to increase unless additional measures are undertaken



a) GWP results for mono-digestion and co-digestion mixes with biomethane upgrading



b) FEP results for mono-digestion and co-digestion mixes with biomethane upgrading



c) TAP results for mono-digestion and co-digestion mixes with biomethane upgrading

Fig. 7. Impact category results for mono-digestion and co-digestion mixes with biomethane upgrading.

(EPA, 2020). While higher proportions of grass silage are favourable to decrease the cost of biogas production (Himanshu et al., 2019), the results of this study show there is potentially a higher risk of environmental impacts which must be considered.

For the majority of CLCAs, biogas is generally used for electricity generation. There is increasing interest in upgrading biogas to biomethane for use in the heat and transport sectors, which are more difficult to decarbonise (Rajendran et al., 2019). In the

Table 4
Results for alternative scenarios of 08:02 VS co-digestion.

Scenario	GWP (kg CO_2 eq)	FEP (kg P eq)	TAP (kg SO_2 eq)
08:02 VS (Baseline)	173,962	−61	3,591
Alt feed	251,851	72	5,160
Animal no's	1,567,850	719,635	754,588

scenario analysis, upgrading biogas to replace natural gas heating showed a better environmental performance for global warming compared to the baseline scenario. However, as shown in the sensitivity analysis this is dependent on the substitution ratio that is used. In the assessment by Styles et al. (2016), injection of biomethane to the gas grid also showed a better performance compared with electricity generation. Styles et al. (2016) also considered the use of upgraded biomethane for transport fuel, which showed even higher savings compared with grid injection. However, Tsapekos et al. (2019) found that CHP had a better environmental performance than biogas upgrading for transport fuel. Future assessments for biogas should take alternative downstream uses such as transport into account.

While a potential surplus of grass silage can be made available through additional N fertiliser, grass yield depends on several factors. Weather has a significant impact on yields and in the event of a drought, a surplus of grass for biogas production may not be available. For example, in 2018 exceptional spring rain led to the exhaustion of winter fodder in Ireland. This was followed with a drought in the summer which led to a significant reduction in the level of conserved grass (Met Éireann, 2018). This led to a fodder crisis with a significant feed deficit for the following winter and spring. Given the variability that can occur in the availability of grass from year to year, future research should consider how best to optimise digester performance in case of changes in the availability of feedstock. Research has shown that biogas yield is an important component in plant operating cost (Himanshu et al., 2019). Strategies to buffer digester performance should be identified in case a feedstock is not available.

In this study, surplus grass silage is made available for AD by intensification of existing pasture, i.e. from additional nitrogen fertiliser application. Accordingly, the assumptions regarding fertiliser use are particularly important. The application rates in McEniry et al. (2013) are based on average national fertiliser application rates in 2008. For grazed grass the application rate was 65 kg N/ha. More recent data on nitrogen application rates are available in the National Fertiliser Use Survey 2005–2015 (Dillon et al., 2018). The average national rate of nitrogen applied on grazed grass in 2015 was similar to the 2008 level, at 63 kg N/ha. However, the average rate for dairy farm systems was significantly higher than all other farm systems at 153 kg N/ha. In contrast, cattle and sheep systems have average rates of nitrogen application of 56 kg N/ha and 41 kg N/ha respectively. As a result, dairy systems may have less scope to provide surplus grass for AD compared to other farm systems if current rates of fertiliser application are close to statutory limits. Future research should consider which types of farm enterprises would be most suitable to provide surplus grass silage for AD.

5. Conclusion

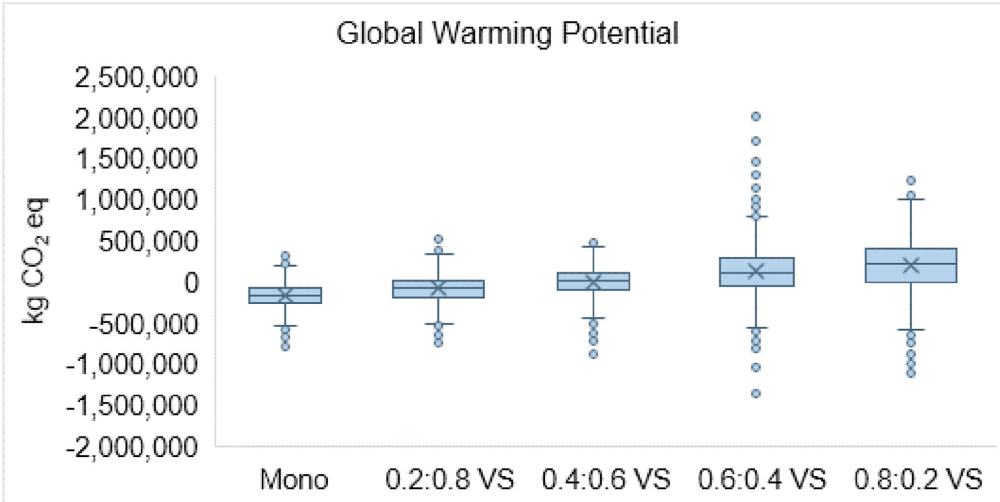
This study provides greater insights regarding the environmental sustainability of grass silage as a feedstock for co-digestion with cattle slurry. Co-digestion with the highest proportion of grass silage had a net environmental impact for all scenarios considered. The optimum environmental performance was observed at a VS ratio of 0.4:0.6 silage:slurry (1,196 and 5,557 tonnes respectively). This research suggests that co-digestion with lower proportions of grass silage may not lead to negative environmental impacts with intensification of grass production through additional fertiliser. These findings are dependent on the provision that the grass silage that is used does not displace feed for livestock. Policy support in Ireland should be directed to animal wastes to optimise the potential of AD to mitigate emissions from livestock production. The impact of ammonia emissions from digestate has a significant impact on the environmental performance of the system. Future research should consider mitigation strategies for digestate management to enhance environmental sustainability, particularly for mono-digestion of slurry and co-digestion with lower proportions of grass silage.

Data availability

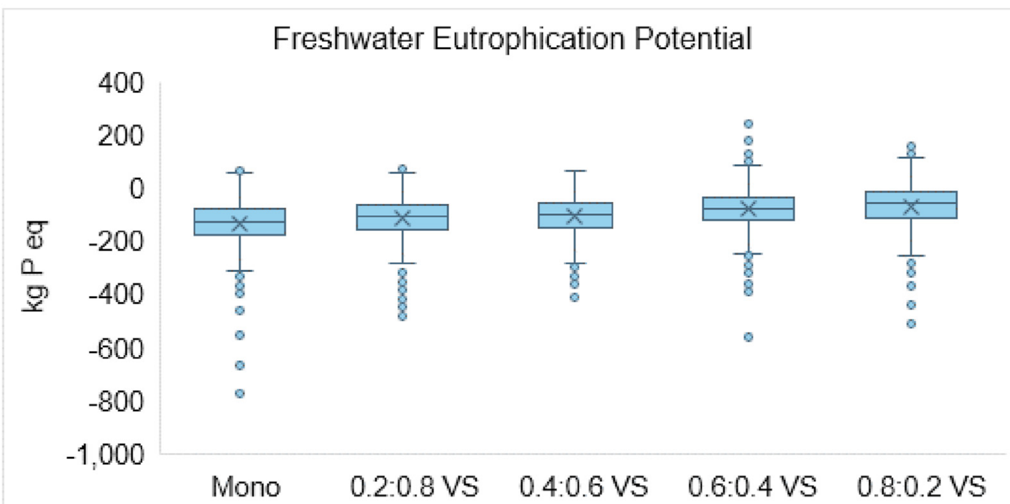
The life cycle inventory for this assessment along with the underlying data for the results presented have been made available in the repository Mendeley Data [https://doi.org/10.17632/wggm795njr.1].

Table 5
Results for sensitivity analysis.

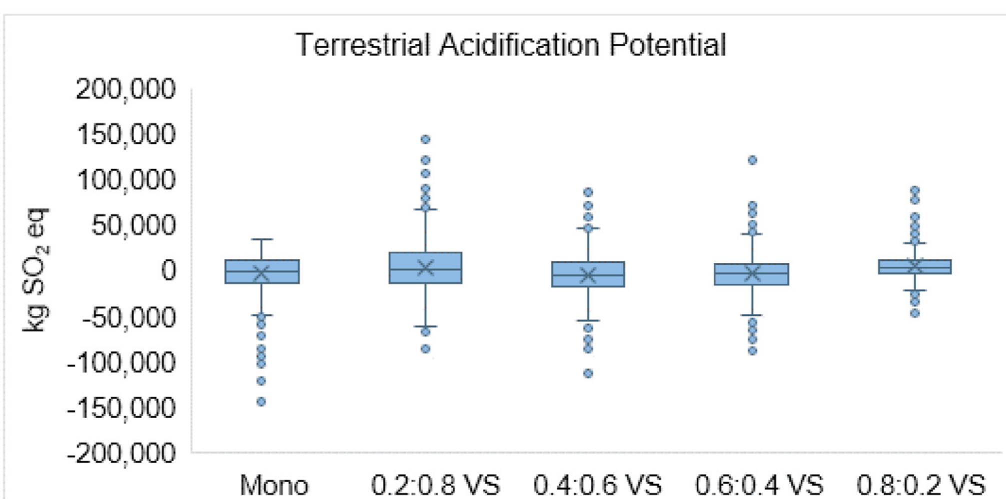
Scenario	Mono	0.2:0.8 VS	0.4:0.6 VS	0.6:0.4 VS	0.8:0.2 VS
Global Warming Potential					
Baseline	−155,321	−76,729	−8,301	81,505	173,962
NECP 2030	−111,039	−24,579	60,065	171,131	307,567
	−40%	−212%	114%	52%	43%
Biomethane 1:0.5	−84,394	−3,599	87,625	207,149	386,026
	−84%	−2032%	109%	61%	55%
Freshwater Eutrophication					
Baseline	−122	−106	−93	−75	−61
NECP 2030	−90	−70	−45	−12	33
	−34%	−53%	−107%	−526%	287%
Biomethane 1:0.5	−89	−67	−42	−8	38
	−37%	−58%	−121%	−802%	260%
Terrestrial Acidification					
Baseline	1,653	1,715	−4,544	−3,057	3,591
NECP 2030	1,962	2,079	−4,067	−2,431	4,524
	16%	18%	−12%	−26%	21%
Biomethane 1:0.5	1,781	1,828	−4,396	−2,862	3,976
	−10%	−14%	7%	15%	−14%



a) Results of Monte Carlo Simulation for GWP



b) Results of Monte Carlo Simulation for FEP



c) Results of Monte Carlo Simulation for FEP

Fig. 8. Results for Monte Carlo Simulation of selected impact categories.

CRediT authorship contribution statement

Ciara Beausang: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Visualization, Writing – original draft, Writing – review & editing. **Kevin McDonnell:** Supervision, Writing – review & editing. **Fionnuala Murphy:** Funding acquisition, Project administration, Supervision, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2021.126838.

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