Comparing environmental consequences of anaerobic mono- and co-digestion of pig manure to produce bio-energy – a life cycle perspective

J.W. De Vries[a,1], T.M.W.J. Vinken[a], L. Hamelin[b], I.J.M. De Boer[c]

[a] Wageningen UR Livestock Research, Wageningen Universtity and Research Centre, P.O. Box 135, 6700 AC, Wageningen, the Netherlands.

[b] Institute of Chemical Engineering, Biotechnology and Environmental Technology (KBM), University of Southern Denmark, Campusvej 55, 5230 Odense M., Denmark.

[c] Animal Production Systems Group, Wageningen University, P.O. Box 338, 6700 AH, Wageningen, the Netherlands.

Abstract

The aim of this work was to assess the environmental consequences of anaerobic mono- and co-digestion of pig manure to produce bio-energy, from a life cycle perspective. This included assessing environmental impacts and land use change emissions (LUC) required to replace used co-substrates for anaerobic digestion. Environmental impact categories considered were climate change, terrestrial acidification, marine and freshwater eutrophication, particulate matter formation, land use, and fossil fuel depletion. Six scenarios were evaluated: mono-digestion of manure, co-digestion with: maize silage, maize silage and glycerin, beet tails, wheat yeast concentrate (WYC), and

[1] Corresponding Author: Tel.: +31 (0)320-238044; Fax: +31 (0)320-238094, E-mail address: jerke.devries@wur.nl (J.W. De Vries).

roadside grass. Mono-digestion reduced most impacts, but represented a limited source for bio-energy. Co-digestion with maize silage, beet tails, and WYC (competing with animal feed), and glycerin increased bio-energy production (up to 568%), but at expense of increasing climate change (through LUC), marine eutrophication, and land use. Co-digestion with wastes or residues like roadside grass gave the best environmental performance.

Keywords: consequential LCA, pig slurry, renewable energy, indirect land use change, greenhouse gases

1. Introduction

The demand for renewable energy is rising because of increasing social awareness of consequences related to non-renewable energy use, e.g. fossil fuel depletion, energy security, and climate change (CC). Renewable energy production in the European Union, for example, is targeted to reach 20% of total energy production by 2020 (EU, 2009). This transition requires insight into environmental consequences of producing renewable energy, including CC, fossil fuel depletion, and land use changes. Life cycle assessment (LCA) is an internationally accepted method to gain insight into the environmental consequences of a product or system (ISO-14040, 2006).

Bio-energy is a form of renewable energy and is produced from biomass. Biomass can be converted by anaerobic digestion (AD) into biogas, composed of methane (CH_4), carbon dioxide (CO_2) and some trace gases (e.g., hydrogen gas), which can then be used to produce bio-energy in the form of electricity, heat, or transport fuel (De Vries et al., 2012; Hamelin et al., 2011). The remaining product after AD, i.e. digestate, can be recycled as organic fertilizer for crop cultivation to substitute mineral fertilizer (Börjesson & Berglund, 2007). Main substrates for AD include agricultural biomass in the form of animal manures and energy crops (e.g. maize), organic residues from the processing industry (e.g. glycerin, beet tails, and gut and intestines from slaughtering houses), and other residues such as, roadside grass or forest residues (Cherubini & Strømman, 2011).

Environmental LCA studies of AD of pig and cattle manure (raw or separated fraction) and energy crops, such as maize and rye grass focused on bio-energy production, greenhouse gas (GHG) emission reduction potentials, and various biogas end applications (Börjesson & Berglund, 2007; De Vries et al., 2012; Hamelin et al., 2011; Thyø & Wenzel, 2007). These studies highlighted that AD of solely, or fractions of, animal manure (mono-digestion) reduced GHG emissions and fossil fuel depletion due to bio-energy production compared to a reference without digestion. To boost bio-energy production and economic profitability of mono-digestion, co-substrates are added, including energy crops and wastes (co-digestion) (Hamelin et al., 2011; Thyø & Wenzel, 2007). This use of co-substrates competes with other applications, such as animal feed or the production of heat or compost and, therefore, will induce the need of a substitute for their initial use. The environmental impact of producing these substitutes, however, has so far not been considered in LCAs of AD. To further improve the insight into the consequences of such a change, LCAs of bio-energy should include other environmental impacts, such as acidification and eutrophication (Cherubini & Strømman, 2011). Additionally, LCAs of bio-energy production should account for the impact of land use change (LUC) and its related carbon (C) emissions from using various substrates. Generally direct (DLUC) and indirect land use change (ILUC) are distinguished, both included in LUC. While DLUC represents the land use changes in a given country or region associated with the expansion of a specific crop in that area, ILUC refers to global market reactions to feedstock displacement and the resultant land use changes. Accounting for LUC is important as it has the potential to undermine reductions in GHG emissions obtained by bio-energy production (Plevin et al., 2010). However, LUC is most often not addressed in LCAs of AD.

The aim here was to assess and compare environmental consequences of anaerobic mono-digestion and co-digestion of pig manure to produce bio-energy. Environmental impacts of producing a substitute for the initial use of the substrates, including the induced LUC, were accounted for. For co-digestion, five co-substrates were evaluated: ensiled whole crop maize, glycerin, beet tails, wheat yeast concentrate (WYC) and roadside grass. These co-substrates represent various product groups that are, or will be, used in agricultural digesters, i.e. energy crops, by-products from food or feed industry, animal feed products, and residual or waste products.

2 Materials and Methods

2.1 LCA approach and functional unit

The ISO-14040 standard provides the general framework for LCA, which was followed in this study (ISO-14040, 2006). A consequential approach to LCA was used to compare the environmental consequences of mono-digestion with co-digestion using alternative substrates. This implied that all processes affected by the mono- or co-digestion systems studied were included in the model (i.e. system expansion). For the affected processes the marginal suppliers were included (e.g. for electricity, heat, and mineral fertilizers) (Weidema, 2003).

To enable a comparison of scenarios, environmental impacts were related to a functional unit (FU), i.e. the main function of the system expressed in quantitative terms. As the study is focused on the use of various substrates and the substitution of their initial use, an input-related FU of one ton substrate (fresh matter) added to the digester was used. This was either pig manure or a mixture of pig manure and co-substrate(s). Studies addressing different applications of substrates, in this case bio-energy production, are recommended to use input-related FUs (Cherubini & Strømman, 2011).

2.2 System boundaries and definition of scenarios

2.2.1 System boundaries common to all scenarios

The general scope of this research was North-Western Europe. The context of the Netherlands was used to identify the involved marginal suppliers for electricity, heat, and mineral fertilizer, when establishing the composition of manure and co-substrates, and when legislation had to be taken into account (e.g. for co-digestion).

The system, the included processes, and the system boundary are illustrated in Figure 1. It was considered that digesting manure avoided the conventional management of raw manure without further processing, i.e. outside storage in a concrete-covered tank, transport, and field application. Hence, manure was stored solely inside the animal house prior to digestion where after it was transported to the digestion facility. Processes included in the system boundary, therefore, were: manure storage in the animal house; (avoided) outside raw manure storage and application (avoided); anaerobic digestion; storage and field application of digestate; produced electricity and heat from biogas; avoided production of fossil-based electricity, heat, and mineral fertilizer; production of substitutes for initial use of the co-substrates; transport between several life cycle stages; cultivation of silage maize (as a co-substrate) and the displaced barley, and impacts related to production of capital goods. Pig production was excluded from the system boundary because it was assumed to be unaffected by a change in demand for manure as a substrate for AD (i.e. pig production as main production process with manure as by-product). Similarly, the main production processes of other used by-products (i.e. glycerin, beet tails, and WYC) were excluded, as these were not affected by a change in demand for these products as co-substrates.

A centralized and average scale agricultural digestion plant was considered. The biogas it produced was used in a combined heat and power unit (CHP) for the production of electricity and heat. Produced electricity was assumed to substitute marginal Dutch electricity, i.e. 28% coal-based, 67% natural gas-based, and 5% wind-based electricity (De Vries et al., 2012). Produced heat was partly used for the digestion process and partly for substitution of marginal heat, i.e. 79% natural gas-based and 21% light fuel oil-based in the Netherlands (De Vries et al., 2012). The digestate was transported and applied to the field as fertilizer, where it substituted marginal mineral N, P, and K fertilizer. Marginal production of mineral fertilizer was assumed to be calcium ammonium nitrate for N, triple superphosphate for P_2O_5, and potassium chloride for K_2O (De Vries et al., 2012).

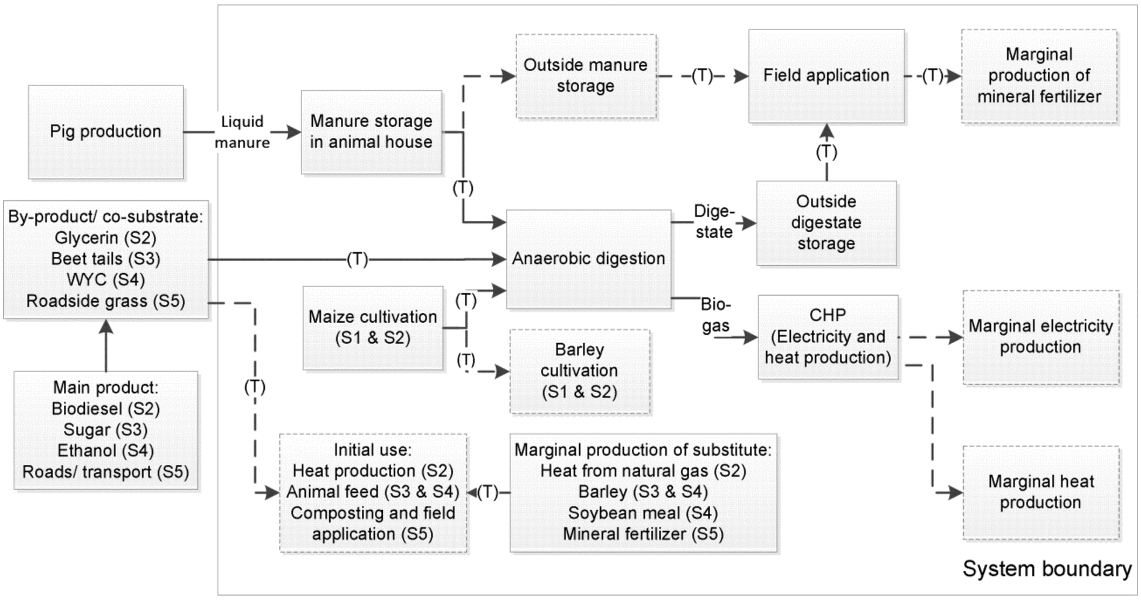

Fig. 1. Processes considered within the system boundary. Dashed boxes represent avoided processes. Black arrows indicate induced flows whereas dashed arrows indicate avoided flows. (T) represents transportation. S1 – S5 are the considered scenarios.

2.2.2 Definition of scenarios and substituting products

Mono-digestion of pig manure comprised the digestion of solely manure (1 ton wet weight). Afterwards, digestate was stored and applied to the field as fertilizer. In scenarios with co-digestion, a mixture of 50% manure and 50% co-substrate (on a wet weight basis) was assumed. The Dutch law requires a minimum input of 50% manure in AD in order to allow application of digestate to the field (DR, 2012).

Scenario 1 (S1) comprised co-digestion of manure with maize silage. Maize is the most commonly used energy crop for biogas production in Europe. It is attractive due to its high dry matter (DM) yield per ha and high CH_4 production potential (Amon et al., 2007). The maize silage was produced in the Netherlands, specifically for AD (Fig. 1), and displaced the production of spring barley (i.e. the marginal crop) (Weidema, 2003). Such displacement induced LUC (i.e. including DLUC as well as

ILUC). As the production of maize, instead of spring barley, was assumed to induce only minor changes in emissions (i.e. in fertilization, tillage, etc.), DLUC was excluded from the model. ILUC, however, was included to reflect the production of the displaced spring barley in locations outside the Netherlands, as further detailed in section 2.3.7.

Scenario 2 (S2) comprised co-digestion of manure with crude glycerin (10%) and maize silage (40%). Glycerin is a by-product of biodiesel production and significantly increases CH_4 production when added to AD, as it contains large quantities of labile organic matter. Glycerin, however, is known to inhibit the digestion process when added in high quantities (> 10 – 15%), as the concentration of volatile fatty acids becomes inhibiting for methanogenic bacteria. Addition of glycerin to the substrate, therefore, was considered to be 10% with 40% of maize silage. Glycerin was assumed to be initially used for heat production in an industrial gas-fired boiler (i.e. marginal use), where it would have replaced 0.94 GJ of natural gas per GJ of glycerin (Thyø & Wenzel, 2007). Use of glycerin in other markets, like cosmetics or the drug industry, was not considered both because these are currently saturated and given the costs for purification. As in S1, S2 included LUC for the maize silage portion digested.

Scenario 3 (S3) comprised co-digestion of manure with beet tails. Beet tails are cut off at the first cleaning of sugar beets during sugar processing, and represents about 5% of the fresh mass of beets. Beet tails are available all year long for digestion when preserved as silage. Beet tails, now used for AD, were assumed to be initially used in dairy cattle feed for their carbohydrate value, which induces the need for another carbohydrate fodder. The marginal carbohydrate fodder was assumed to be spring barley produced in the Netherlands. For one ton of beet tails, 0.11 ton of barley was required, based on the Dutch energy value for animal feed (SI). The extra production of barley on Dutch agricultural soils induced LUC that was included in the analysis.

Scenario 4 (S4) comprised co-digestion of manure with wheat yeast concentrate (WYC). WYC is a protein-rich by-product from bio-ethanol production. WYC was assumed to be initially used in animal feed for its protein value and, therefore, soy bean meal from Brazil was assumed to replace WYC in feed (Weidema, 2003). For 0.50 ton of WYC, 0.33 ton of soybeans (or 0.27 ton soybean meal) was required, based on the Dutch digestible protein value (SI). The additional soy oil produced (0.05 ton) alongside the soy meal was assumed to substitute palm oil (0.23 ton fruit bunches) (Dalgaard et al., 2008). The no longer produced carbohydrate-rich palm cake, as by-product from palm oil, was assumed to induce a demand for additional barley (3.2 kg) in order to balance carbohydrate supply on the feed market. LUC related to Brazilian soybean, Malaysian palm fruit, and Dutch barley production was considered in the analysis (specified in section 2.3.7).

Scenario 5 (S5) comprised co-digestion of manure with roadside grass, originating from maintenance of side-strips along roads. In the Netherlands, roadside grass is usually composted and subsequently applied to agricultural soils for its fertilizing and soil amending capabilities (Ehlert et al., 2010). Avoided composting was thus considered the opportunity cost of using roadside grass for AD. The compost was assumed to be substituted with mineral N, P and K fertilizer.

2.3 Life cycle inventory and assumptions

2.3.1 Chemical composition and methane yields of the substrates

Table 1 presents the chemical compositions considered for pig manure and co-substrates, before and after digestion, the fraction of organic matter (OM) decomposed during AD and the CH_4 yield per ton of substrate. A mass balance approach was used to compute all compositions and flows within the system. Manure composition after storage in the animal house was based on De Vries et al. (2012). The composition of roadside grass was represented by an average composition from harvested spring and autumn grass (Ehlert et al., 2010).

2.3.2 Storage of manure and digestate

Table 2 shows the considered emissions of N (ammonia (NH_3), nitrous oxide (N_2O), nitrogen oxides (NO_x), and nitrogen gas (N_2)) and CH_4 from manure and digestate storage. As storage duration affects CH_4 emission and because pig manure used for AD was stored in the animal house shorter (yearly average of 1 month instead of 3 month without digestion), the difference of in-house CH_4 emissions was included in the assessment based on De Vries et al. (2012). Emissions of N_2O, NO, and N_2 occurring during in-house and outside storage of manure were not included, as these were assumed the same for both storage durations. Outside storage of raw manure and digestate (yearly average of 2 months) took place in a covered concrete tank. During storage, nutrient leaching (e.g. of nitrate NO_3^-, PO_4^{3-}, and K) was assumed negligible. Energy required for pumping manure and digestate was 1.2 kWh ton^{-1}.

2.3.3 Production of substituting products

The environmental impacts of producing substituting products comprised cultivation, processing, and transport. Since detailed inventory data are presented in the SI[1], general assumptions are presented here. Background (or generic) LCA data (e.g. heat and electricity production from fossil energy, mineral fertilizer production, herbicide

[1] The supporting information (SI) is available free of charge via the online version.

production, diesel production, etc.) were taken from the Ecoinvent database v2.2 (EcoinventCentre, 2007).

Maize cultivation in S1 and S2 was based on average Dutch data. Maize was assumed to be fertilized with mineral fertilizer, as this is the marginal source for nutrients.

In S2, heat production occurred in an industrial gas-fired boiler >100 kW. The required natural gas (1.58 GJ) was computed based on an average lower heating value for glycerin of 16.5 MJ kg^{-1} and a boiler efficiency of 102% (EcoinventCentre, 2007).

In S3, barley production was based on average Dutch data (SI). As in S1 with maize, it was assumed that barley was fertilized with mineral fertilizer.

In S4, soybean production was based on Brazilian production circumstances occurring in the Central West and Southern region of Brazil (SI). Production of palm oil was based on production in Malaysia, as marginal source (SI). As in S3, barley production was based on average Dutch data.

In S5, emission data for composting of roadside grass were based on the composting of municipal food waste (Brinkman et al., 2004). During composting, approximately 60% of the wet weight of roadside grass was reduced. Emission of CH_4, was considered to be 0.20 kg ton^{-1} of grass entering the facility, N_2O 1.5% of N, and NH_3 5.24% of N. In total, 10% of the N in the roadside grass was assumed to be lost during composting and 90% of the NH_3 to be removed with a bio-filter (Brinkman et al., 2004). Per ton of grass, energy requirement of composting was 29 kWh. Leaching of NO_3^-, PO_4^{3-}, and K during composting was excluded due to a sealed concrete floor.

2.3.4 *Anaerobic digestion*

AD was considered to take place in two stages in a continuous stirred tank reactor (CSTR). Operation occurred at mesophilic temperature (around 35°C) with a hydraulic retention time of 60 days, based on common practice in The Netherlands, and required 66 MJ electricity per ton of substrate and 110 MJ heat per ton of substrate (Börjesson & Berglund, 2007; Peene et al., 2011). The CHP had an electric capacity of 500 kW with an overall energetic efficiency of 80% and an electric efficiency of 35%. Electricity used for the process was assumed to be taken from the grid, whereas produced electricity was supplied to the grid. The required heat for the digestion process resulted from the heat produced from the biogas engine. Of the remaining heat, 50% was assumed to avoid marginal heat.

During AD, organic N in the substrates is partly converted into mineral N (N_{min}). It was assumed that 20% of the organic N in the initial substrate was mineralized (De Vries et al., 2012).

Fugitive emissions of CH_4 from the digestion plant and the biogas engine were assumed to be 1.5% of total produced CH_4 (i.e. 1% from the digestion plant and 0.5% from the gas engine) (IPCC, 2006). Emission of N_2O from the biogas engine was 0.1 kg TJ^{-1} of electricity produced, whereas emission of NO_x was 0.42 g m^{-3} of biogas produced (VROM, 2010).

2.3.5 *Application of products and avoided mineral fertilizer*

Manure and digestate were assumed to be injected into arable land. For the avoided compost (S5), it is assumed that it would have been applied by a solid manure spreader to arable land, whereas mineral fertilizers would have been applied with a broadcast spreader. Data on the environmental impacts of field application processes were taken from the Ecoinvent database (EcoinventCentre, 2007).

Emissions of N and PO_4^{3-} during application and the N fertilizer replacement values (NFRVs) for manure, digestate, compost, and mineral fertilizers are presented in Table 2. The NFRVs were considered to be: 65% for manure and digestates, 15% for compost, and 100% for mineral N fertilizer (Brinkman et al., 2004; DR, 2012). The replacement value for P and K was assumed to be 100% for all products (De Vries et al., 2012).

2.3.6 Transport of products

Transport of products occurred by lorry (16 - 32 tons) between several life cycle stages (Fig. 1). For the transport of the raw manure to the AD plant a distance of 31 km was considered, whereas an average distance of 104 km was assumed for transporting the digestates to agricultural fields. This 104 km was a weighted average of local transport (34% over 31 km), external transport within the Netherlands (48% over 120 km) and transport outside of the Netherlands (18% over 200 km) (De Vries et al., 2012; Peene et al., 2011). All other products were assumed to be transported over 50 km.

2.3.7 Land use change emissions

LUC emissions related to displaced cultivation of barley in S1 – S4 were based on Tonini et al. (submitted), who related the displacement of spring barley in Denmark to the conversion of specific biomes worldwide and quantified the associated greenhouse gas emissions (SI). This was assumed to be representative for Western Europe. The net land expansion was adjusted by using the average Dutch yield for barley leading to 1.22 ha expanded per ha of displaced barley. An average LUC factor of 309 ton CO_2 ha^{-1} of displaced barley was applied, which was annualized over a 20 years period (1.55 kg

CO_2 m^{-2} y^{-1}). LUC emissions in S4 for soybean cultivation were calculated specifically for this study (i.e. 1.67 ton CO_2 ton^{-1} soybeans y^{-1}, or 0.47 kg CO_2 m^{-2} y^{-1}). For the calculations, it was assumed that 20% of the increased soybean demand resulted from yield increase, whereas 80% resulted from an increase in soybean area in the tropical open forest in Brazil (23%) and savanna in the Cerrado region (77%) (SI). Similarly, the LUC emission for reduced palm fruit production (i.e. 1.95 ton CO_2 ton^{-1} palm fruit y^{-1}, or 3.7 kg CO_2 m^{-2} y^{-1}) was assumed to result from a 30% yield increase and from 70% increase in cultivation area (i.e. 70% conversion of tropical moist forest in in Southeast Asia and 30% conversion of peat land, see SI).

2.3.8 Soil carbon storage

Soil C storage was included based on Hermann et al. (2011) (SI). Changes in soil C storage, and thus in the amount of C released as CO_2 to the atmosphere, occur due to differences in C composition among manure, digestate, and compost. During digestion, C is converted into CH_4 and CO_2 resulting in less C applied to the soil with the digestate and thus less C to be stored in the soil compared to undigested matter. Compared to manure, an increased proportion of C is converted to relatively stable humus-C in the soil by grass-based compost (Hermann et al., 2011). It was assumed that 35% of the C in the manure and digestates (representing 50% of OM) was converted into humus-C. For grass compost this was assumed to be 51%. The C converted to humus-C was assumed to remain in the soil C pool over a time horizon of 100 years (Hermann et al., 2011) (SI).

2.4 Sensitivity analysis

The sensitivity of results and conclusions to several parameters were tested: minimum and maximum values for LUC, higher fugitive methane emissions from the digestion facility, a higher electric efficiency of the biogas engine, and increased NFRV of the digestates.

2.4.1 Minimum and maximum values for land use change

LUC emission factors contain considerable uncertainty depending on the applied methods and data, and therefore will have a considerable effect on final results (Plevin et al., 2010). The uncertainty range from Tonini et al. (submitted) was used to address the minimum and maximum values for LUC emission related to displacement and production of spring barley (S1 – S4), i.e. a minimum of 140 ton CO_2 ha^{-1} (0.70 kg CO_2 m^{-2} y^{-1}) and maximum of 477 ton CO_2 ha^{-1} (2.38 kg CO_2 m^{-2} y^{-1}). Furthermore, deforestation related to soybean cultivation has shown signs of decrease in recent years due to policy changes by Brazilian government (Prudêncio da Silva et al., 2010). To consider a minimum LUC factor for soybean production, data from Prudêncio da Silva et al. (2010) were used who reported an average value of 0.058 kg CO_2-eq m^{-2} (0.28 ton CO_2-eq ton^{-1} soybeans). This factor included 1% of land used for soybean production transformed from rainforest and 3.4% transformed from Cerrado. For a minimum LUC value related to Malaysian palm fruit cultivation, data from the Ecoinvent database were used, i.e. 0.47 kg CO_2 m^{-2} (0.25 ton CO_2 ton^{-1} palm fruit produced, related to the conversion of Malaysian rainforest (SI)) (EcoinventCentre, 2007).

To consider maximum LUC values for soybean cultivation, the same method as in the baseline scenario was applied. However, instead of tropical open forest being converted, only tropical evergreen forest was assumed to be converted (i.e. includes more C and leads to higher CO_2 emission, see SI). Furthermore, the increase in demand was met by 100% expansion, i.e. no yield increase was considered. This led to a value of 14.5 ton CO_2 ton^{-1} soybeans y^{-1} (4.1 kg CO_2 m^{-2} y^{-1}). Maximum LUC values for palm fruit were calculated by assuming expansion on 50% peat land and 50% tropical moist forest (SI). Furthermore, as with soybeans, the increase in demand was met by 100% expansion. Moreover, a higher emission from peat land was considered (112 ton CO_2 ha^{-1} y^{-1}, SI) resulting in a maximum (avoided) LUC emission of 4.33 ton CO_2 ton^{-1} palm fruit y^{-1} (8.2 kg CO_2 m^{-2} y^{-1}).

2.4.2 Fugitive methane emissions from the digestion plant

Fugitive emissions of CH_4 from the digestion plant increase the amount of GHG released to the atmosphere and consequently reduce the bio-energy produced. Fugitive emissions have been reported to be as much as 5.2% of produced CH_4 under normal operating conditions (Flesch et al., 2011). To consider the magnitude of this impact, emission of CH_4 from the digestion plant was increased from 1% to 5%.

2.4.3 Electric efficiency of the biogas engine

Electric efficiency of the biogas engine is important with regard to the amount of electricity (versus heat) produced, and consequently the amount of avoided fossil fuels. The electric efficiency was increased from 35% to 45% to consider the effect of more displaced fossil-based electricity. The total efficiency was kept equal to the baseline scenario.

2.4.5 NFRV of digestates

The NFRV of digestate is often said to be higher compared to undigested manure due to mineralization of N, increasing its availability for crops (Hamelin et al., 2011). To consider the impact of a higher fertilizing capacity of the digestate, the NFRV of the digestates was increased from 65% to 75% in the sensitivity analysis.

2.5 Life cycle impact assessment

All emissions and resources used were included in the assessment and categorized under seven environmental impact categories: climate change (CC in kg CO_2-eq; including CO_2, CH_4, and N_2O), terrestrial acidification (TA in kg SO_2-eq; including sulfur dioxide (SO_2), NO_x, and NH_3), marine eutrophication (ME in kg N-eq; including NO_3^-, NO_x, and NH_3), freshwater eutrophication (FE in kg P-eq; including PO_4^{3-}); particulate matter formation (PMF in kg PM_{10}-eq; including particulates < 10 µm and NH_3, SO_2, and NO_x as precursors), fossil fuel depletion (FFD in MJ), and land use (m^2). The scenarios and impact assessments were modeled and computed in Microsoft Excel and by using the ReCiPe midpoint v.1.04 impact assessment method (Goedkoop et al., 2009).

3 Results and discussion

3.1 Impact assessment of anaerobic mono- and co-digestion

3.1.1 Climate change

Mono-digestion reduced CC by 16 kg CO_2-eq per ton of substrate, as compared to the situation where manure is not digested (Fig. 2). This was mainly due to avoided manure storage and application (i.e. altered management), but also to avoided fossil-based electricity and heat. Addition of co-substrates in S1 – S4 increased CC (36 – 105 kg CO_2-eq) mainly as a result of induced LUC. LUC contributed 104 – 199 kg CO_2-eq in S1 – S4. In S5, the reduction in CC of 89 kg CO_2-eq resulted from avoided fossil-based electricity and heat and the avoided composting. Avoided composting prevented emission of N_2O and the energy use that occurs during composting. In all scenarios, net transport contributed 16 – 19 kg CO_2-eq, whereas the AD process contributed 18 – 42 kg CO_2-eq.

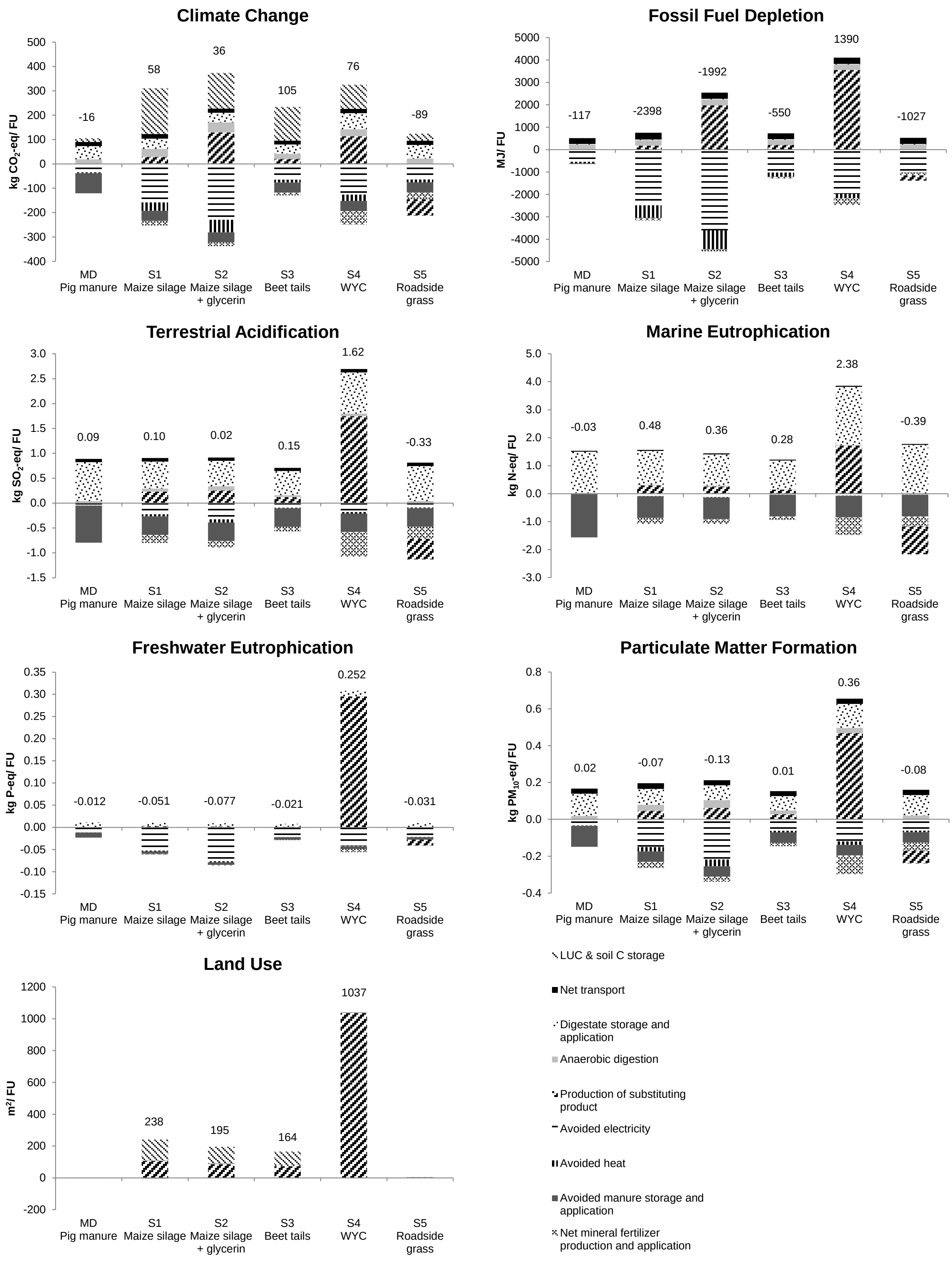

Fig. 2. Impact assessment results for the scenarios. Numbers above the bars present the net results. Net transport represents the induced and avoided transports as shown in Fig. 1 (excluding transport during co-substrate production).

3.1.2 Fossil fuel depletion

Mono-digestion reduced FFD by 117 MJ, as compared to the situation where manure is not digested, reflecting a net reduction in energy use (46 kWh electricity was produced and 51 MJ of heat avoided in S1) (Fig. 2). Addition of co-substrates in S1 – S3 and S5 increased bio-energy production and resulted in higher reductions of FFD, with S1 having the highest reduction (FFD of 2398 MJ). Despite having the highest bio-energy production (305 kWh electricity produced and 651 MJ of heat avoided), S2 did not lead to a lower FFD (-1992 MJ) than S1 (210 kWh electricity produced and 431 MJ heat avoided). This was mainly caused by the bio-energy that would have been produced by the direct combustion of the glycerin (1.65 GJ), which had to be substituted by an equivalent amount of energy from natural gas. Despite bio-energy production in S4 (167 kWh electricity produced and 332 MJ heat avoided), the scenario had a net increase in energy demand due to production and transport of soybean meal. S5 (86 kWh electricity produced and 143 MJ heat avoided) had, after S1 and S2, the highest reduction in FFD (-1027 MJ), whereas S3 (87 kWh electricity produced and 145 MJ heat avoided) showed approximately half of the reduced FFD (-550 MJ) found in S5. In S5, this result is mostly due to the shift from energy consumption during composting to bio-energy production from AD. In all scenarios, transport and AD required roughly 200 – 300 MJ of energy.

3.1.3 Terrestrial acidification

Compared to raw manure application, mono-digestion increased TA by 0.09 kg SO_2-eq, through a higher NH_3 emission from digestate application. This increase resulted from a higher N_{min} content in the digestate. Addition of co-substrates in S1 – S3 generally showed equal performance as the mono-digestion scenario. In these scenarios, increased

TA from the production of the substitute and the storage and application of the digestate was decreased by avoided electricity and heat. In S4, TA was considerably higher (1.61 kg SO_2-eq) due to emissions of NH_3, NO_x, and SO_2 from the production of the soybeans. S5 was the only scenario that reduced TA (-0.33 kg SO_2-eq), mainly due to the avoided NH_3 emissions during composting and application of compost.

3.1.4 Marine eutrophication

ME was approximately neutral for mono-digestion (-0.03 kg N-eq) as compared to the situation where manure is not digested (Fig. 2). For mono-digestion, the reduced ME from avoided manure storage and application was similar to the ME from digestate storage and application. ME increased with the addition of co-substrates in S1 – S4 (0.28 – 2.38 kg N-eq). This was mainly due to two factors: (1) emission of NO_3^- during production of the co-substrates and (2) addition of N from the co-substrates, which increased the N application from digestate. In the mono-digestion scenario as well as in S1, S2, and S3, ME was little affected (maximum of -0.2 kg N-eq) by the net mineral fertilizer production; in S4, however, reduced ME from net mineral fertilizer production was considerably higher (-0.6 kg N-eq) due to the high nutrient content in the WYC (Fig. 2). A net reduction of ME (-0.40 kg N-eq), again related to avoided production and application of compost, was seen in S5. The digestate in S5 was considered to have a higher nitrogen fertilizer value compared to compost, thus consequently reducing more mineral N fertilizer and thereby leaching of NO_3^-.

3.1.5 Freshwater eutrophication

Mono-digestion reduced FE (-0.012 kg P-eq), compared to the situation where manure is not digested, mainly as a result of avoided electricity production. Addition of co-substrates in S1 – S5 further reduced FE (-0.021 – -0.077 kg P-eq), but not in S4 due to the cultivation of soybeans contributing to FE through leaching of PO_4^{3-}, as relatively high emission per kg of dry matter occurs (SI). In S5, FE was reduced mainly through avoided electricity from energy production during AD, and through the avoided electricity from composting.

3.1.6 Particulate matter formation

Mono-digestion, as compared to the situation where manure is not digested, resulted in negligible PMF (0.02 kg PM_{10}-eq); for mono-digestion, emissions of NO_x from transport and NH_3 from digestate storage and application were counteracted by a reduction in PM_{10}, NO_x, and SO_2 emissions from the substituted fossil fuels. Addition of co-substrates in S1 – S5, decreased PMF (0.01 – -0.13 kg PM_{10}-eq), except for S4 (0.36 kg PM_{10}-eq) where more emissions of NH_3, NO_x, and SO_2 occurred during production and transport of soybean meal. In S5, reduced NH_3 emissions from composting resulted in reduced PMF.

3.1.7 Land use

Mono-digestion had negligible land use, because no co-substrates were used and, therefore, no substitute was required (land use in this case was only related to production facilities and capital goods production). S5 also had negligible land use as the roadside grass is a residual product that does not interact with crops and thus land use. Addition of co-substrates in S1 – S4 increased land use directly for cultivation of respectively maize (S1, 106 m^2 and S2, 85 m^2), barley (S3, 73 m^2) and soybeans (S4, 1037 m^2; increased cultivation of soybeans (1153 m^2), avoided oil palm cultivation (122 m^2), increased barley cultivation (4.4 m^2), and land use related to capital goods (2 m^2)). Furthermore, land use expanded in S1 – S3 due to displaced cultivation of barley (respectively 129, 103, and 89 m^2) (Fig. 2). S4 had the highest land use, which was due to the relative low DM yield per ha of cultivated soybeans compared to maize and barley (SI).

3.2 Sensitivity analysis

3.2.1 Minimum and maximum values for land use change

In case of minimum values for LUC, net CC decreased up to 109 kg CO_2-eq compared with base line results. This meant that in S1, S2 and S4, CC was reduced more than mono-digestion (Table 3). In S1, the impact of reducing LUC emissions was highest (a change of 109 kg CO_2-eq), because of the displaced barley and the related LUC emission factor, i.e. higher than for soybeans (section 2.3.7). In case of maximum LUC values, net CC increased up to 3730 kg CO_2-eq compared with base line results in S1 - S4. This increase was highest in S4 (i.e. 3654 kg CO_2-eq). These results indicate that the assumed LUC factor had a major impact on the conclusions of this study with regard to CC.

3.2.2 Fugitive methane emission from the digestion plant

Increasing fugitive CH_4 emissions from the digestion plant increased CC (11 – 75 kg CO_2-eq) and FFD (27 – 182 MJ) for all scenarios (Table 3). For mono-digestion, this offset almost all GHG reduction. These results indicate that controlling emission of CH_4 from the digestion process is essential to maximize the advantage concerning CC.

3.2.3 Electric efficiency CHP

Increasing the electric efficiency of the CHP reduced CC in all scenarios (up to 53 kg CO_2-eq, Table 3), FFD (up to 820 MJ), and, to a lesser extent, also FE (up to 0.021 kg P-eq). These results indicate an opportunity in further improving the environmental performance of AD by increasing efficient conversion of biogas into electricity.

3.2.4 NFRV of the digestate

Increasing the NFRV of the digestate by 10% decreased CC, TA, ME, and FFD in all scenarios, by up to respectively 13 kg CO_2-eq, 0.105 kg SO_2-eq, 0.16 kg N-eq, and 53 MJ. This decrease was mainly due to a higher amount of mineral N fertilizer that was substituted by the digestate, compared to the baseline results. In general, however, this did not change the ranking of the scenarios, although TA in S2 became lower than in the case of mono-digestion.

3.3 General discussion

3.3.1 Mono-digestion compared to co-digestion

Overall, mono-digestion of pig manure performed well from an environmental perspective as bio-energy was produced and most impact categories either remained neutral (ME, PMF, and land use) or were reduced (CC, FFD, and FE), compared to conventional storage and field application of raw manure. Bio-energy production by mono-digestion, however, was relatively low compared to co-digestion which was also observed by other authors (Thyø & Wenzel, 2007).

Adding co-substrates increased bio-energy production compared to mono-digestion, but showed that the environmental impact of producing the substitute was important for most impact categories. This notably applied for S4 where the addition of WYC resulted in increased environmental impact. As a protein-rich substrate, WYC competes with soy meal on the international market, and its production has a relatively high environmental impact (Prudêncio da Silva et al., 2010). Co-digestion with WYC, therefore, does not represent an attractive alternative to increase bio-energy production. On the other hand, in S5 the addition of roadside grass increased bio-energy production compared to mono-digestion and reduced all environmental impact categories. For roadside grass, moving to AD compared to composting represented improved management. As other studies have confirmed, anaerobic (co-)digestion is a better environmental alternative than composting for waste or residue management (Hermann et al., 2011; Patterson et al., 2011). This indicates that from an environmental perspective, such wastes and residues are preferred as co-substrates.

Adding maize silage, glycerin, and beet tails in S1 – S3 also represented attractive alternatives to increase bio-energy production and reduce FFD, FE, and PMF compared to mono-digestion. However, these scenarios led to increased CC, ME, and land use. For CC, LUC increased GHG emissions and reduced the attractiveness of maize silage and beet tails for co-digestion, meaning LUC must be considered when selecting a co-substrate. Moreover, using maize silage and beet tails adds nutrients to the total substrate and thus to the digestate. This may increase nutrient surpluses in areas where the digestate is produced, especially in cases where there is already a surplus of nutrients (i.e. in intensive livestock production areas, such as the Netherlands).

Overall, anaerobic mono-digestion of pig manure and co-digestion with wastes or residues presents a good opportunity to produce bio-energy and reduce environmental impact. However, co-digestion with potential animal feed stocks, increases the impact by inducing the need of a substitute and should, therefore, be avoided. Furthermore, to increase bio-energy production from mono-digestion, additional manure processing by means of, e.g. separation or pretreatment can be applied. Such treatment can be used to concentrate the decomposable organic matter in a single fraction or increase the fraction of decomposable organic matter. Including such additional processing should be evaluated from a life cycle perspective as high energy demands may counteract the produced bio-energy (De Vries et al., 2012; Hamelin et al., 2011).

3.3.2 Sensitivity of the results

Uncertainty of the LUC magnitudes on CC appeared to be crucial for the co-digestion scenarios with maize silage and WYC (S1, S2, and S4), as conclusions for CC were altered for these. It is, thus, of great importance to improve LUC estimations through further analysis in order to understand if the "real life" values are closer to the minimum or to the maximum of the range presented in Table 3. In any case, however, LUC contribution to CC is not zero, and as such should be addressed where crops or any substrate inducing a need for more crop production are used as substrates for bio-energy production (Plevin et al., 2010).

Fugitive CH_4 emissions from the digestion plant should be kept to a minimum, as these affect both CC and bio-energy production. These effects were also observed by Patterson et al. (2011) who pinpointed the effect of fugitive CH_4 emissions on CC during upgrading of the biogas. Current data of fugitive emissions, however, are generally based on rough estimates or few measurements (Flesch et al., 2011). Verification by further measurements on different types of digesters would, therefore, be essential for future LCAs on biogas production.

With respect to the assessed scenarios, increased electric efficiency of the gas engine did not change the ranking and conclusions. However, improving the electric efficiency of biogas engines, as this sensitivity analysis highlights, represents one option to enhance the overall environmental performance of biogas installations (more than 100% improvement for CC and FFD).

Increased NFRV of the digestates also did not alter the conclusions of the study. Nevertheless, increasing the fertilizing capacity of the digestate and its related management will improve the overall environmental performance of AD scenarios. However, NFRV levels are strongly dependent on local factors, such as climate, soil type and crop rotation and the related management of the digestate (i.e. method and timing of application) and should be considered site specific (De Vries et al., 2012).

Other parameters affecting final results, but not included in the sensitivity analysis, include methane yields of the co-substrates and the initial use and related marginal suppliers for the substitutes. Methane yields will differ upon the quality and origin of the substrate, but also the type and management of the digestion process. In general, higher yields will mainly lead to lower CC and FFD and vice versa.

The environmental impact related to producing the substitute needed for the co-substrate used for AD will depend on variables, such as the extent of substitution and the product properties (Weidema, 2003). When, for example, the WYC in S4 has a low quality and as a result cannot be utilized for animal feed, it becomes a waste that would otherwise be composted or landfilled. In such a case, using WYC for bio-energy production by AD would lead to a much more environmentally sustainable result, as no interaction with feed would occur. Furthermore, the substitution ratio of co-substrates initially used as animal feed (i.e. how much feed is displaced per kg of WYC or beet tail taken away for AD) also depends on factors, such as digestibility, market prices, etc., and not solely on the protein and energy content. In most cases, a new feed ration will be computed in order to optimize prices, and product characteristics. In S3, for example, beet tails in animal feed might also be compensated by adding maize and grass silage, if these happen to be cheaper or more practical. The initial use of co-substrates should therefore be analyzed for each site-specific, geographical, temporal and economical context.

3.3.3 Substrate availability and potential for bio-energy production and GHG mitigation

The availability of the substrates is limited (in 2010 in the Netherlands: approximately 11800 kilotons (kt) of pig manure, 125 kt of beet tails, 4 kt of glycerin, 250 kt of WYC, and 700 kt of roadside grass), as these are constrained resources depending on the production of another main product. This will limit both the total bio-energy production potential by means of AD of these substrates, but also economic viability is limited due to competition with other markets and because bio-energy is highly subsidized. Currently, prices of co-substrates are increasing strongly due to higher demand. This strengthens the point made earlier that greater focus on mono-digestion is needed, in particular on the development of technologies to enhance biogas production from manure. However, the total bio-energy production potential and GHG reduction potential of mono-digestion remains relatively low. If all pig manure in the Netherlands were mono-digested, this would represent about 0.5% of the total electricity use (117 billion kWh used in 2010 or 2.2% of the electricity consumed by households, 24.7 billion kWh in 2010) and a reduction of 0.1% of total emitted GHGs (211 Mton emitted in 2010 or 2.2% of agricultural GHG emissions, 8.6 Mton in 2010). Including the digestion of all other substrates in this study (assuming only half of the maize silage produced), and additionally all cattle manure, would roughly double the electricity production and reduce GHGs slightly more. Hence, anaerobic mono- and co-digestion of manure and co-substrates provides a potential to produce bio-energy and reduce environmental consequences, but on its own cannot fulfill increasing future bio-energy demands.

4 Conclusions

Anaerobic mono-digestion of pig manure produced bio-energy and improved overall environmental performance as compared to conventional manure management, but represents a limited source for bio-energy. Co-digestion with maize silage and beet tails, which compete with animal feed, and glycerin increased bio-energy production (up to 568%) and reduced terrestrial acidification, particulate matter formation, and freshwater eutrophication, but increased marine eutrophication, land use and climate change (through LUC). Co-digestion with wheat yeast concentrate, a protein-rich co-substrate substituted with soymeal, increased all environmental impacts. Co-digestion with roadside grass, a residual or waste product, appeared most environmentally sustainable for increasing bio-energy production of mono-digestion.

Acknowledgements

This research was financed by the Dutch ministry of Economic Affairs, Agriculture and Innovation (programs: KB-13 and KB-III). We would like to thank our colleagues: Hendrik Jan van Dooren (Wageningen UR Livestock Research), Wim Corré (Plant Research International), and Kor Zwart (Alterra) for their useful input prior and during the writing of this research article. We thank Corina van Middelaar for providing us with data for the analysis. Finally, we acknowledge Linda McPhee (Linda McPhee Consulting) for technical correction of the manuscript.

References

Amon, T., Amon, B., Kryvoruchko, V., Machmüller, A., Hopfner-Sixt, K., Bodiroza, V., Hrbek, R., Friedel, J., Pötsch, E., Wagentristl, H., Schreiner, M., Zollitsch, W. 2007. Methane production through anaerobic digestion of various energy crops grown in sustainable crop rotations. *Bioresource Technology*, **98**(17), 3204-3212.

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**(5), 326-344.

Brinkman, A.J.F., Van Zundert, E.H.M., Saft, R.J. 2004. Herziening levenscyclusanalyse voor GFT-afval. Herberekening LCA bij het MER-LAP. Grontmij, IVAM, Amsterdam, the Netherlands.

Cherubini, F., Strømman, A.H. 2011. Life cycle assessment of bioenergy systems: State of the art and future challenges. *Bioresource Technology*, **102**(2), 437-451.

Dalgaard, R., Schmidt, J., Halberg, N., Christensen, P., Thrane, M., Pengue, W. 2008. LCA of soybean meal. *The International Journal of Life Cycle Assessment*, **13**(3), 240-254.

De Vries, J.W., Groenestein, C.M., De Boer, I.J.M. 2012. Environmental consequences of processing manure to produce mineral fertilizer and bio-energy. *Journal of Environmental Management*, **102**(0), 173-183.

Dekker, P.H.M., Stilma, E.S.C., van Geel, W.C.A., Kool, A. 2009. Levenscyclusanalyse meststoffen bij gebruik in de biologische en gangbare landbouw. Praktijkonderzoek Plant en Omgeving, Wageningen University and Research Centre, Lelystad, the Netherlands.

DR, 2012. Regulations for the use of organic manure: http://www.hetlnvloket.nl/onderwerpen/mest, Dienst Regelingen, Ministry of Economic Affairs, Agriculture and Innovation. The Hague, the Netherlands. Accessed April 23 2012.

Duynie, 2008. Tarwegistconcentraat. Productomschrijving: http://www.duynie.nl/Basis.aspx?Tid=2&Lid=46&Lit=TEKST&Hmi=2576&Smi=2641&STIJL=11&Act=1480, Duynie B.V. Alphen aan den Rijn, the Netherlands. Accessed February 6 2012.

EcoinventCentre, 2007. Ecoinvent data v2.0 Final reports econinvent 2007, Swiss Centre for Life Cycle Inventories. Dübendorf, Switzerland.

Ehlert, P.A.I., Zwart, K.B., Spijker, J.H. 2010. Biogas uit bermmaaisel. Duurzaam en haalbaar? Alterra-report 2064. Alterra, Wageningen University and Research Centre, Wageningen, the Netherlands.

EU, 2009. Directives on the promotion of the use of energy from renewable sources. http://europa.eu/legislation_summaries/energy/renewable_energy/en0009_en.htm. Directive 2009/28/EC, European Parliament. Brussels, Belgium. Accessed April 23 2012.

Flesch, T.K., Desjardins, R.L., Worth, D. 2011. Fugitive methane emissions from an agricultural biodigester. *Biomass and Bioenergy*, **35**(9), 3927-3935.

Goedkoop, M., Heijungs, R., Huijbregts, M.A.J., de Schryver, A., Struijs, J., van Zelm, R. 2009. ReCiPe 2008. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and endpoint level. First edition.

Ministry of Spatial Planning and Environment (VROM), The Hague, the Netherlands.

Groenestein, C.M., Van Bruggen, C., De Haan, B.J., Hoogeveen, M.W., Huijsmans, J.F.M., Van De Sluis, S.M., Velthof, G.L. 2012. NEMA: Dutch inventory of ammonia emission from livestock production and focus on housing and storage. in: *Emission of gas and dust from livestock, June 10 - 13, 2012*. Saint Malo, France.

Hamelin, L., Wesnæs, M., Wenzel, H., Petersen, B.M. 2011. Environmental Consequences of Future Biogas Technologies Based on Separated Slurry. *Environmental Science & Technology*, **45**(13), 5869-5877.

Hermann, B.G., Debeer, L., De Wilde, B., Blok, K., Patel, M.K. 2011. To compost or not to compost: Carbon and energy footprints of biodegradable materials' waste treatment. *Polymer Degradation and Stability*, **96**(6), 1159-1171.

IPCC, 2006. Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and Other Land Use, Geneva, Switzerland.

ISO-14040, 2006. Environmental Management - Life Cycle Assessment - Principles and Framework. International Organisation for Standardisation (ISO), Geneva, Switzerland.

Kool, A., De Boer, H.C., Van Dooren, H.J.C., Timmerman, M., Van Dun, B., Tijmensen, M. 2005. Kennisbundeling covergisting. CLM, Culemborg, the Netherlands.

KWIN, 2009-2010. *Kwantitatieve Informatie Veehouderij 2009-2010*. Wageningen UR Liverstock Research, Lelystad, the Netherlands.

Patterson, T., Esteves, S., Dinsdale, R., Guwy, A. 2011. Life cycle assessment of biogas infrastructure options on a regional scale. *Bioresource Technology*, **102**(15), 7313-7323.

Peene, A., Velghe, F., Wierinck, I. 2011. Evaluatie van de vergisters in Nederland. NL Agency. Ministry of Economic Affairs, Agriculture and Innovation, The Hague, the Netherlands.

Plevin, R.J., O'Hare, M., Jones, A.D., Torn, M.S., Gibbs, H.K. 2010. Greenhouse Gas Emissions from Biofuels' Indirect Land Use Change Are Uncertain but May Be Much Greater than Previously Estimated. *Environmental Science & Technology*, **44**(21), 8015-8021.

Prudêncio da Silva, V., van der Werf, H.M.G., Spies, A., Soares, S.R. 2010. Variability in environmental impacts of Brazilian soybean according to crop production and transport scenarios. *Journal of Environmental Management*, **91**(9), 1831-1839.

Santibáñez, C., Teresa Varnero, M., Bustamante, M. 2011. Residual glycerol from biodiesel manufacturing, waste or potential source of bioenergy: a review. *Chilian Journal of Agricultural Research **71**: 469 - 475*.

Stehfest, E., Bouwman, L. 2006. N_2O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. *Nutrient Cycling in Agroecosystems*, **74**(3), 207-228.

Thomassen, M.A., Zwart, K.B. 2008. Development sustainability index. Co-digestion animal manure with byproducts. Report 148. Animal Sciences Group, Lelystad, the Netherlands.

Thyø, K.A., Wenzel, H. 2007. Life Cycle Assessment of Biogas from Maize silage and from Manure. Institute for Product Development, Aalborg, Denmark.

Timmerman, M., Dooren, H.J.C.v., Biewenga, G. 2005. Mestvergisting op boerderijschaal. Animal Sciences Group, Wageningen UR, Lelystad, the Netherlands.

Tonini, D., Hamelin, L., Wenzel, H., Astrup, T. submitted. Bioenergy production from perennial energy crops: a consequential LCA of 12 bioenergy chains including land use changes. *Submitted to Environmental Science & Technology*.

Van Bruggen, C., Groenestein, C.M., De Haan, B.J., Hoogeveen, M.J., Van der Sluis, S.M., Velthof, G.L. 2011. Ammoniakemissie uit dierlijke mest en kunstmest in 2009. Berekend met Nationaal Emissiemodel voor Ammoniak (NEMA). WOt werkdocument 251, Wageningen University, Wageningen, the Netherlands.

Velthof, G.L., Mosquera, J. 2010. Calculation of nitrous oxide emission from agriculture in the Netherlands; update of emission factors and leaching fraction. Alterra, Report 2151, Wageningen, the Netherlands.

VROM, 2010. Besluit emissie-eisen middelgrote stookinstallaties milieubeheer. http://wetten.overheid.nl/BWBR0026884/geldigheidsdatum_12-10-2010, Ministry of Infrastructure and the Environment, The Hague, the Netherlands. Accessed February 6 2012.

Weidema, B. 2003. Market information in life cycle assessment. Danish Environmental Protection Agency, Project No. 863, Copenhagen, Denmark.

Tables

Table 1. Composition of manure and co-substrates (kg ton^{-1} fresh matter) before and after digestion, OM decomposed and CH_4 produced

	DM	OM	N_{tot}	N_{min}	N_{org}	P_2O_5	K_2O	% OM decomposed	m^3 CH_4 produced ton^{-1} FM
Before digestion									
Pig manure after storage in animal house	90.3	60.3	7.35	4.35	3.0	4.2	7.2	38[e]	14.0[e]
Pig manure after outside storage[a]	90.0	60.0	7.20	4.20	3.0	4.2	7.2	-	-
Maize silage[a]	301	290	3.80	0.03	3.77	1.4	4.3	80[b]	115[g]
Glycerin[b]	842	794	n.d.	n.d.	n.d.	n.d.	n.d.	90[b]	406[h]
Beet tails[c]	136	110	2.30	0	2.30	0.7	2.3	80[b]	39.1[c]
Wheat yeast concentrate[d]	240	198	11.0	0.32	10.6	5.2	7.0	80[b]	88.6[d]
Roadside grass[f]	496	363	8.04	0	8.04	2.8	10.4	60[f]	38.5[f]
Composted roadside grass[f]	697	363	18.1	0	18.1	7.0	26.0	60[f]	-
After digestion									
Pig manure	67.4	37.4	7.52	5.06	2.46	4.3	7.4		
Maize silage	69.4	57.9	4.95	0.98	3.96	1.8	5.6		
Glycerin	127	79.4	n.d.	n.d.	n.d.	n.d.	n.d.		
Beet tails	48.0	22.0	2.52	0.50	2.02	0.8	2.5		
Wheat yeast concentrate	84.5	38.8	13.0	2.52	10.5	6.2	8.3		
Roadside grass	279	145	10.3	2.06	8.22	3.6	13.3		

n.d. = no data, '-' = not included, DM = dry matter, OM = organic matter, N_{min} = mineral nitrogen, N_{org} = organic nitrogen FM = fresh matter.

[a] KWIN (2009-2010); [b] Assumptions based on Thomassen & Zwart (2008); [c] Kool et al. (2005); [d] Duynie (2008); [e] Timmerman et al. (2005); [f] Ehlert et al. (2010); [g] Amon et al. (2007); [h] Average of Thomassen and Zwart (2008) and Santibáñez et al. (2011).

Table 2. Emissions during storage and field application of products

	CH_4	NH_3-N	N_2O-N_{dir}	NO-N	N_2-N	N_2O-N_{ind}		NO_3-N	PO_4-P
	kg ton^{-1}	% TAN	% N	% N	% N	% NH_3-N + NO_x-N	% N leached	% N	% of P
Storage									
Pig manure storage in animal house	1.33[*,a] 0.29[#,a]	-	-	-	-		-	-	-
Pig manure outside storage	0.17[a]	2% N[b]	-	-	-	1.0[c]	-	-	-
Digestate outside storage	0.17[a]	2% N[b]	0.1[c]	0.1[b]	1.0[b]		-	-	-
Field application									
Pig manure	-	2.0[d]	1.3[f]		-			20.6[h]	
Digestate	-	2.0[d]	1.3[f]		-			19.8–22.5[h]	
CAN	-	2.5[d]	1.0[f]	0.55[g]	-	1.0[c]	0.75[c]	16.6[h]	0.6[i]
Urea	-	15.0[d]	1.0[c]		-			-	
Compost	-	5.8[e]	1.25[e]		-			26.6[h]	

'-' = not included, N_2O-N_{dir} = direct emission, N_2O-N_{ind} = indirect emission.

[a] De Vries et al. (2012),[*] = 3 months, [#] = 1 month storage; [b] Groenestein et al. (2012); [c] IPCC (2006); [d] Van Bruggen et al. (2011); [e] Brinkman et al. (2004); [f] Velthof and Mosquera (2010); [g] Stehfest and Bouwman (2006); [h] Based on Dekker et al. (2009), 19.8% (raw manure), 21.4% (S1), 21.2% (S2), 20.9% (S3), 22.5 (S4), and 22.3% (S5); [i] Based on the EDIP, 2003 method assuming that 10% of P was leached and 6% reached the aquatic environment.

Table 3. Results of the sensitivity analysis in absolute values per functional unit (1 ton substrate) for the scenarios (only most affected impact categories are presented)

		Values for land use change emissions		Fugitive CH_4 emission from AD 5%		Electric efficiency +10%			NFRV digestates +10%			
		Minimum	Maximum									
Scenario		CC	CC	CC	FFD	CC	FE	FFD	CC	TA	ME	FFD
MD	bsl	-16		-16	-117	-16	-0.012	-117	-16	0.09	-0.03	-117
	se	-	-	-5	-90	-24	-0.015	-240	-27	0.003	-0.16	-160
S1	bsl	58		58	-2398	58	-0.051	-2398	58	0.10	0.48	-2398
	se	-51	166	110	-2272	22	-0.066	-2963	50	0.037	0.39	-2430
S2	bsl	36		36	-1992	36	-0.077	-1992	36	0.02	0.36	-1992
	se	-51	123	111	-1810	-17	-0.098	-2812	29	-0.041	0.27	-2022
S3	bsl	105		105	-550	105	-0.021	-550	105	0.13	0.28	-550
	se	30	180	126	-498	90	-0.027	-782	98	0.075	0.20	-578
S4	bsl	76		76	1390	76	0.252	1390	76	1.61	2.38	1390
	se	-17	3730	117	1490	47	0.241	940	63	1.51	2.22	1337
S5	bsl	-89		-89	-1027	-89	-0.031	-1027	-89	-0.33	-0.39	-1027
	se	-	-	-68	-976	-103	-0.037	-1258	-99	-0.41	-0.52	-1072

'-' = no change, MD = mono-digestion, bsl = baseline results, se = results of the sensitivity analysis, CC = climate change (kg CO_2-eq), FFD = fossil fuel depletion (MJ), FE = freshwater eutrophication (kg P-eq), TA = terrestrial acidification (kg SO_2-eq), and ME = marine eutrophication (kg N-eq), NFRV = nitrogen fertilizer replacement value.