Technical, environmental and cost-benefit assessment of manure management chain: A case study of large scale dairy farming

Sefeedpari, P., Vellinga, T., Rafiee, S., Sharifi, M., Shine, P., & Pishgar-Komleh, S. H.

Technical, environmental and cost-benefit assessment of manure management chain: a case study of large scale dairy farming

Paria Sefeedpari[1,2,3*], Theun Vellinga[2], Shahin Rafiee[1*], Mohammad Sharifi[1], Philip Shine[4], Seyyed Hassan Pishgar-Komleh[5]

[1] Department of Agricultural Machinery, Faculty of Agricultural Engineering and Technology, University of Tehran, Karaj, P.O. Box 4111, Iran.

[2] Wageningen Livestock Research, Wageningen University and Research, PO Box 338, 6700 AH, Wageningen, The Netherlands.

[3] Institute of Soil Science and Plant Cultivation—State Research Institute, P.O. Box 24 100 Puławy, Poland

[4] Department of Process, Energy and Transport Engineering, Cork Institute of Technology, Cork, Ireland

[5] Department of Economics, Faculty of Economics and Management, Czech University of Life Sciences Prague, Kamýcká 129, 165 00 Prague, Suchdol, Czech Republic

*Corresponding author email: pariasefeedpari@ut.ac.ir

Postal address: Department of Agricultural Machinery Engineering, Faculty of Agricultural Engineering and Technology, University of Tehran, P.O. Box 4111, Karaj 31587-77871, Iran

Tel: (+98) 2632801011

Fax: (+98) 2632808138

Abstract

Improper management of livestock manure has resulted in loss of nutrients and organic matter available in manure in addition to negative environmental impacts. This study developed and compared eight manure management scenarios across their entire life cycles, rom excretion to transport to land, considering technical, environmental and economic aspects. The scenarios based on combinations of collection, sand separation, solid/liquid (S/L) separation, anaerobic digestion (AD), composting, and storage were compared. Mass balances, costs and benefits and greenhouse emissions were evaluated. The model framework was tested and validated for a large-scale dairy farm with 9,000 heads of cattle and daily manure production of approximately 505 t in Iran. The study indicated that sand separation and S/L separation did not contribute to a change in manure nutrients or emissions but reduced sand, maintenance cost, and transport requirements. AD followed by separation achieved the highest emission reduction (27.7 kg CO_{2eq} t^{-1}) due to the avoided emissions from replacing fossil fuels by renewable energy. Composting method had the lowest costs; however it resulted in a low nutrient recovery efficiency and high nitrous oxide emission. The assessment revealed that AD is a promising management option yielding a high potential greenhouse gas savings, nutrients recovery and nitrogen availability in fertilizer for plants. In spite of the high investment costs of AD, it could be a profitable strategy due to the high subsidies paid to renewable energy projects in Iran. In conclusion, this study showed that the choice of manure treatment method has a strong influence on nutrients, profitability and greenhouse gas balances by performing sensitivity analysis. The results of this study and the application of this model further indicate the need to consider various significant impacts, farm specifications and local conditions to decide the best manure management options.

Keywords: Manure management; Environmental analysis; Cost-benefit analysis; Anaerobic digestion; Dairy.

1. Introduction

Manure is an output of livestock production systems that is being increasingly sought after as a valuable resource to supply nutrients to soil, as a replacement to costly synthetic fertilisers (Audsley and Wilkinson, 2014). Manure treatment strategies such as manure to energy practices can help return organic matter to the soil, while concentrating nutrients for easier and less costly uses. These benefits offer opportunities for extending the concept of nutrients circularity in agriculture and livestock production in particular, shifting perception from manure as a problem to manure as a resource.

Manure is one of the key agricultural greenhouse gas (GHG) sources (second important GHG source after enteric fermentation) and accounts for about 10% of on-farm emissions (O'Mara, 2011). Although the livestock industry is not considered a major producer of CO_2, it is considered as the source of other greenhouse gases, such as methane (CH_4) and nitrous oxide (N_2O) (Chianese et al., 2009). Processing of manure would result in the release of emissions to air, soil and water (Pratt et al., 2015). The most suitable manure management strategy depends on a combination of factors including economic feasibility, governmental policies, financial incentives, and social acceptance (Montalvo, 2008). Selecting the most efficient manure management strategy must ensure the profitability of investment (Burney et al., 1980). Minimizing the total cost of a chain is desired while the constraints are fulfilled (Gharaei et al., 2017). As the main aim of manure management is to adapt the business goals and strategies to the competitive market, it is essential to conduct investments within a reliable supply chain (Hoseini Shekarabi et al., 2018). In a real world scenario, such chains can be modelled and optimized in order to minimize the cost of the chain (Gharaei et al., 2017).

A simplified manure management chain (MMC) is shown in Fig. 1. It illustrates that the N, P and K content, organic matter and volume of the initial stream is defined as excreted and collected on the dairy farm. These will be affected by manure processing, which in turn will define the quality of the end products (e.g. fertilizer).

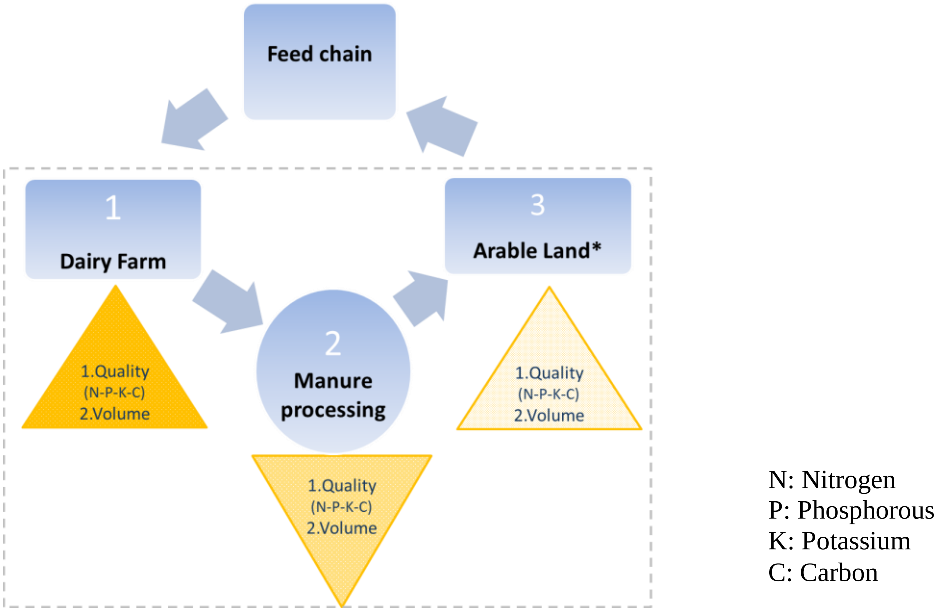


Fig. 1. The simplified nutrient cycling system in livestock production. The dotted line depicts the system boundary of the chain under study

One critical issue with intensification of dairy farming systems in Iran, i.e. high stocking densities of dairy farms, is due to the alienation between increasing the farm size and the environmental and economic challenges. This problem has been driven by a number of management adjustments, such as improved manure management systems. This improvement has not been addressed yet in academic research projects in Iran and as the observations showed, all these improvements are carried out without being technically feasible, economically and environmentally justified. These problems are due to lack of adequate manure management policies. In most cases, manure is left on land, sun-dried and possibly sold for applying on arable farms. On the other hand, manure has a farm gate value in Iran, i.e. farmers can receive a payment for selling manure as an organic fertilizer or bioenergy resource.This has caused manure processing to increasingly becoming attractive to dairy farmers.

Modelling approaches for nutrient flows and mass balance (MB) of materials in and out of MMC (especially N and P) have been studied at the global, regional, and national levels all over the world. Petersen et al. (2007) stated that a whole farm perspective taking the on-farm interactions into account is needed. A regional based study on manure management systems in the European Union (EU-27) has been assessed by Oenema et al. (2007). Chadwick et al. (2015) reviewed manure management practices both at the national and regional scale. There are some models, which have focused on the single emissions of livestock production such as ammonia (NH_3) (Reidy et al., 2008), and CH_4 and N_2O (Sommer et al., 2004). Sommer et al. (2004) developed a dynamic model to calculate CH_4 and N_2O emissions through a whole chain of manure management from collection to application.

A wide range of studies have been conducted to analyse the environmental and/or economic benefits of a single manure treatment method. These include: De Vries et al. (2012) studied environmental consequences of anaerobic digestion (AD) using life cycle assessment (LCA), Mezzullo et al. (2013) studied LCA of a small-scale AD, ten Hoeve et al. (2014) conducted a LCA of mechanical separation, and Torquati et al. (2014) assessed the economic viability of biogas production from manure. Similar assessments of a single manure treatment method have been conducted by Junior et al. (2015) on GHG emissions from AD of beef manure, Blumenstein et al. (2018) on economic profitability of AD, Yazan et al. (2018) on economic sustainability of manure based biogas supply chains, and Rennie et al. (2018) on modelling assessment of liquid manure storage. Other studies have focused on a combination of manure treatment practices. Aguirre-Villegas et al. (2014) assessed four different manure-processing pathways for their environmental impacts on different sustainability indicators. These pathways included solid/liquid (S/L) separation, AD, AD plus S/L separation and a base-case pathway of no manure treatment and direct land application of manure. Integrated manure management methods were evaluated by Aguirre-Villegas and Larson (2017) using LCA tools. Selecting among different manure management strategies were studied in Polish pig farms. This study proposed that the best manure treatment methods involved applying half of the pig manure on arable farms, processing of the other half by filtration, while excluding the storage of manure in lagoons (Makara and Kowalski, 2018).

A number of studies have applied supply chain modelling and optimization. Gharaei and Pasandideh (2017) modelled the costs in an integrated supply chain. Based on their findings, the optimization of an integrated supply chain in co-operation with the supply chain managers can prove beneficial and applicable for them to minimize the total inventory cost of the chain.

Although many researchers have focused on manure management, no studies in literature have considered assessing manure management across the entire chain from manure excretion to land application, while carrying out a farm-level technical, environmental and economic analysis. No research has been conducted in Iran related to manure management and different technologies from a whole chain perspective. To illustrate the originality of the current study compared to the previous works in the literature review are presented in Table S1.

As discussed above, there is lack of comprehensive knowledge and information among dairy farmers and stakeholders related to manure management strategies in Iran. On this basis, this paper aims to demonstrate the current situation of manure management in Iran, while analysing the environmental and economic impacts of eight potential manure management strategies. Potential and quality assessment of manure production was carried out at farm-level and based on the current real situation on the farms, feed ration and herd characteristics including body weight, energy requirements and milk quality. This study aims to fill the gap between understanding the effects of selecting different manure management strategies and the up-take of this knowledge by farmers during their decision making process. This research will address a key question in manure management: what are the economic and environmental impacts of different manure management methods for a specific farm? To answer this question, this study is elaborated at four different levels: (i) developing a spread-sheet framework to calculate the potential of manure production and its quality; (ii) analysing different manure treatment strategies based on three dimensions (technical, environmental and economic); (iii) tracking and accounting the nutrients flows, emissions and losses within the MMC; and (iv) testing and validating the framework for the case under study in Iran under different eight alternative scenarios.

The authors designed and tested a framework for conducting all calculations of the whole manure management chain from technical aspects through assessing different scenarios, environmental impact through assessing GHG emissions and economic impact through cost and benefit analysis. Integrating these three aspects and studying different manure management systems in large-scale dairy farming in Iran are other novelties of this paper. Estimating the potential of manure production and its characteristics based on herd structure, cattle characteristics in a farm and dietary highlight the novelty of this work (Table S1).

The paper is structured as follows: firstly, an overview of the framework is given, before the manure treatment processes are described. This is then followed by a description of the economic methods and the description of different scenarios. Results of applying the model for assessing the environmental and economic impacts of each scenario are then presented and discussed before the paper ends with concluding remarks.

2. Material and methods

2.1. Model description

In this study, the Technical, Environmental, and Cost-Benefit Analysis of Manure Management Chain (TECBA-MMC) model framework (Fig. 2) was developed based on a process-based analysis and a MB approach. The TECBA-MMC model framework calculates: (i) N, P, K and C flows, volatile solids (VS) and degradation rates, losses through emissions to air, soil and water during collection, transport, S/L separation, long-term storage and final transport to arable land, (ii) the quality of the final product(s) and (iii) environmental impacts, costs and benefits along the whole MMC. The model framework is represented in a modular way. Within this framework, the quality of the ingoing manure can be calculated or can be defined by the user. The technologies and methods of manure treatment can be selected by the user of model. For example, one can go from Transport 1 (as shown in Fig. 2) to composting directly. The effluent of one step is the influent to the next step, whereby within each step, a MB is calculated, keeping track of all changes in N, P, K, C and total mass due to conversions, losses and emissions. Manure quality may be calculated after each step by dividing nutrient amounts by the total mass. The detailed calculation methodology and parameters used are presented in the Supplementary Information (SI).

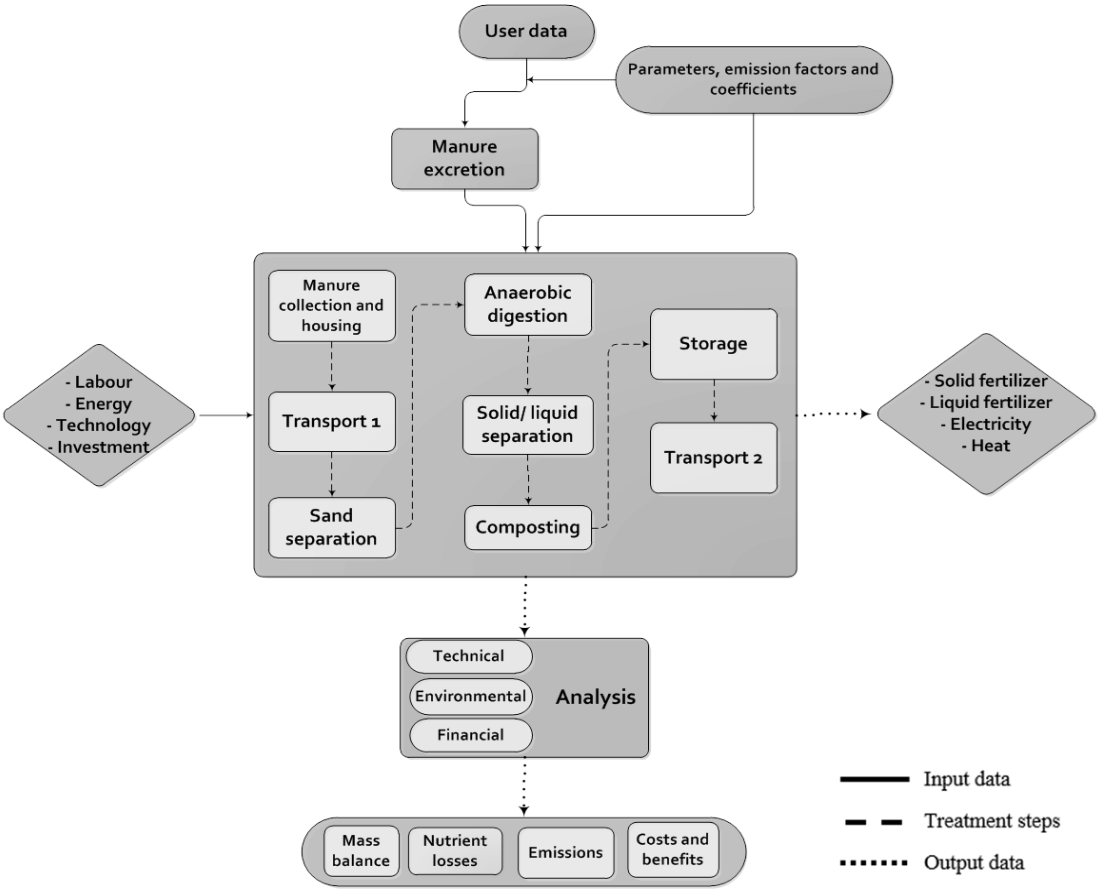


Fig. 2. Modelling framework for the TECBA-MMC model

2.2. *Manure excretion*

Accurate estimation of the quality and amount of inflow is essential for an accurate design of manure processing and treatment, as well as the quality of the final product. The developed framework allows manure production to be calculated on the basis of animal nutrition for cattle farms. A general method of estimating nutrient excretion is indicated as Eq. (1):

$$\textit{Nutrient Excretion} = \textit{Nutrient Intake} - \textit{Nutrient Retention} \tag{1}$$

The nutritional calculations of feed compositions were based on the method used in IPCC (2006), NRC (2001) and the GLEAM model (MacLeod et al., 2018). The equations and detailed data on herd composition and characteristics are listed in the SI (Eqs. (S1-S8) and Tables S2 and S3). The excretion of nutrients via dung and urine were calculated from the amount of nutrients available in the feed intake minus the nutrients retention in body tissue and nutrients export through milk or beef (Nennich et al., 2005) (Eqs. (S9 and S10) and Table

S4). Measured manure characteristics can also be fed to the model as user defined data. Due to the variation of bedding material from farm to farm (IPCC, 2006), the effect of bedding materials quality on the characteristic of manure has been excluded. More information on nutrients retention coefficients and characteristics is presented in the SI file.

VS of manure are the organic material in livestock manure and consist of both biodegradable and non-degradable fractions (Sommer et al., 2004). The VS content of manure was estimated based on diet, as VS equals the fraction of undigested feed excreted as feces and can be mixed with urine, as indicated in Eq. (S11). VS is formulated by IPCC (2006) (Tier 2) based on the feed intake and digestibility. The degradable and non-degradable volatile solids (VS_d and VS_{nd}) were calculated using the Eqs. (S12 and S13). To calculate the total solids (TS) on the basis of volatile solids content (VS_{tot}), conversion coefficients were utilised on the basis of the animal category (Rotz et al., 2016) (Table S5).

Nitrogen in excreta exists in different forms. Fecal N was assumed to be organic N (the indigestible fraction of crude protein in feed), and urinary N (ammoniacal N) was considered to be urea (Eqs. (S14 and S15)) (Reijs, 2007). In Vonk et al. (2016), the total ammoniacal nitrogen (TAN) excretion was defined as the sum of excreted urine N and a fraction of net mineralized N in feces. This implies that the amount of N-org (organic N) is lower than the amount of N in dung (Eqs. (S16 and S17)).

2.3. Description of manure treatment methods

The six manure processes considered in the model framework were: collection and housing, mechanical sand separation, AD, mechanical S/L separation, composting, storage and manure hauling. Detailed descriptions of these processes are as follows:

2.3.1. Collection and housing

In this model, the user may choose the type of bedding based on the type of housing system. The amount of bedding for each type can be a constant parameter or defined by the user. Collection methods, e.g. blade and loader, scraper, vacuum machine, capacity, power requirement, collection frequencies, manure temperature in housing and initial costs of collection equipment were inputs to the model. Based on this information, the parameters for bedding volume (Table S6), the total bedding volume, total fuel consumption, carbon dioxide (CO_2) emission from diesel use and costs of manure collection were calculated as formulated in Eq. (S18).

Ammonia (NH_3) emissions in housing were estimated on a TAN excretion basis. This method assumed a linear relation between TAN contents and NH_3 emission (Eq. (S19)). Nitrous oxide (N_2O) emissions from animal management processes were calculated based on Vonk et al. (2016). Detailed information related to N emissions is provided in the SI (Eqs. (S20-S23) and Table S7). It should be noted here that CO_2 emissions from livestock were not estimated since annual net CO_2 emissions were assumed to be zero; i.e. the CO_2 photosynthesized by plants is returned to the atmosphere as respired CO_2. CH_4 emissions in housing were calculated considering the type of housing system (free stall and loose housing system) and the ambient temperature as presented in Eqs. (S24-S26). P and K losses were ignored during the collection and hauling process. Particulate matter ($PM_{2.5}$ and PM_{10}) emissions particularly from animal housing systems were calculated for different animal categories (Eq. (S27) and Table S8).

2.3.2. Mechanical sand separation

In this study, a mechanical separation technique consuming water and electricity was included. To facilitate sand sedimentation, the recycled water returned to the cycle in a dilution rate of 1:1 (one-part water to one-part sand-laden manure). Separated sand, water, and manure effluent were the outputs of this process. The production of sand was not included in the analysis. The sand collected along with manure was quantified as it affects the stream down to the next process. No biotic emissions were assumed for sand, and its contribution to the nutrient flows was zero. The GHG emissions (CO_2, CH_4 and N_2O) from the production and use of electricity, obtained from the electricity production mix, was 0.91 kg CO_{2-eq} per kWh including 15% losses in grid (Ecoinvent3.3, 2016) (Table S9).

2.3.3. Anaerobic digestion (AD)

In this model, the AD pathway was a continuous mesophilic processAll information about AD process are provided in Table S10. The source of GHG emissions were fossil fuels used to generate multiple energy streams in the CHP plant, combustion of CH_4, and the gas leakage from the digester walls. The GHG emissions from the CHP were not considered in the MB analysis since it is up taken via photosynthesis (Whiting and Azapagic, 2014). The potential for energy production was calculated based on the energy content (lower heating value) of produced CH_4 minus the CH_4 leakage (Eq. (S28)). The electricity production was limited by either the capacity of the generator, the operating time or the amount of available biogas (Eq. (S29)). The heat requirement to maintain the digester in its desired mesophilic operating temperature was calculated using Eqs. (S30 and S31). The VS amount in the effluent was calculated using the Eqs. (S32 and S33). During AD, organic matter converts to biogas (CH_4 and CO_2 gasses). Considering carbon conversion, the C content of digestate was calculated. This calculation was used for estimating the changes in VS_d. With respect to the recommended dry matter of influent to AD process, water may be required for diluting manure (Eqs. (S34 and S35)) (Jørgensen, 2009).

2.3.4. Mechanical solid/liquid separation

Mass, nutrient and dry matter balances were carried out through applying and extending the model developed by Melse and de Buisonje (2015) for screw press separation. This framework calculated the distribution of minerals and organic matter between different fractions using the certain characteristics such as nutrient contents in kg t^{-1} of the effluent, separation efficiency (%), DM content (kg t^{-1}) of the solid fraction produced and manure volume to the separator (m^3 d^{-1}). GHG emissions due to the separation process were only due to the electricity use of the separation equipment.

2.3.5. Composting

In this stage, different types of composting equipment can be selected, relevant to the type of composting in vessel, static pile, intensive windrow and passive windrow, each with specific labour and fuel for machinery requirements. During composting, organic matter transformation takes place, whereby NH_3 may be emitted due to acidification, and GHG emissions (CH_4, NH_3, N_2O) are released (Table S12). The losses of N, P, K and C during composting will affect the agronomic value of final product as a soil amendment. In Table S13, the CH_4 conversion factors during composting are listed. The GHG emissions in this treatment may be attributed to the diesel use. CH_4 emission during composting is related to the method of composting and temperature (IPCC, 2006).

2.3.6. Storage

The final product of the manure management system is stored prior to land application. Storage types may be categorized into lagoon, pit (with or without cover) and heap (with or without cover) based on the type of the manure (solid, liquid and slurry) (Tittonell et al., 2010). Manure temperature (Monteny et al., 2001) and the storage duration (Chianese et al., 2009) greatly affect the amount of CH_4 produced. The effect of wind velocity and technical factors such as the C: N ratio and pH on the rate of losses in open storage was ignored in this model. The model was run for a seasonal temperature dependency of CH_4 emission due to the fact that a quadratic relationship was found between the temperature and CH_4 emission factors (EFs) obtained from (IPCC, 2006) (Fig. S1). Coefficients for losses as a percentage of initial influent entering storage and the EFs of NH_3 and N_2O losses are presented in Tables S14 and S15.

2.3.7. Manure hauling

The most common method of transporting solid manure was assumed to be by truck. The fuel use for transport by truck and nurse tank was estimated based on the distance and truck specific fuel consumption. These data were obtained from farmers and hauling agencies (truck operators). For road transport of solid manure, the hauling distance determines GHG emissions and the cost of transport (Araji and Stodick, 1990). It is worth mentioning that in this study, the liquid fraction was assumed to stay at the farm for nearby land applications, thus no transport of liquid manure was considered. All the parameters and coefficients used in different manure treatment processes are presented in Table S16.

2.4. Manure Nutrients Recovery Efficiency

To compare the nutrients recovery efficiency of different manure management strategies, the manure nutrient recovery efficiency (MNRE) index was introduced. MNRE is the fraction of nutrients in the final product divided by the nutrients in the initial product (Eq. (S36)).

2.5. Cost- benefit analysis

For the cost-benefit analysis, all costs of activities (including investment and production costs) were required. These included data related to the total investment costs (TIC) (cost of land (or its opportunity cost), equipment and machinery, site preparation, civil works and pre-production costs), and total production costs (TPC) (energy, labour, maintenance depreciation, insurance and contingency costs). Total revenues (TR) were calculated by multiplying the amount of final products by the prices and correspondingly the gross profit was gained (GP = TR-TPC). The required data and the price of other by-products like electricity were collected from different sources such as farmers, manure processors, transport agencies and statistics. The emissions due to sand mining and transport by the buyer were not incorporated in this study. Avoided costs of sand recovery, buying sand and lower maintenance were included in the calculations. To assess the feasibility of future managerial decisions, profitability indicators including net present value (NPV), internal rate of return (IRR), benefit to cost ratio (BCR), payback period (PBP) and profitability index (PI) were evaluated (Eqs. (S37-S42)). More constant values and economic coefficients are listed in Table S17.

2.6. Scenarios description

To test the model, eight scenarios were investigated and assessed to explore the capability of the model when estimating the consequences of different manure management strategies (Table 1) on a large-scale dairy cattle farm in Iran that owns over 9,000 cows (4,500 lactating cows). To give an increased insight into the farm structure and technical conditions, some characteristics of the target farm are presented in Table S18. In this target case, there is a market price for solid, liquid, composted manure and other products, as listed in Table S19.

To implement the model, real data from dairy farms were collected based on the input data requirements of the model. These data included herd characteristics, technical information related to housing type, manure collection, temperature, feed ingredients, etc. The manure treatment approaches were: manure collection (MC), transport 1 (T1), sand separation (Sand Sep.), AD, solid/liquid separation (S/L Sep.), composting (Comp.), storage (S), and transport to land (T2). A description of scenarios considered in this study is shown in Table 1.

Table 1. Scenarios of process analysis of a large-scale dairy farm in Iran

Activities	Sc. 1 (BC)	Sc. 2	Sc. 3	Sc. 4	Sc. 5	Sc. 6	Sc. 7[4]	Sc. 8[5]
MC	×	×	×	×	×	×	×	×
T1	×	×	×	×	×	×	×	×
Sand Sep.[1]	-	-	-	×	×	×	×	×
AD	-	-	-	-	-	×	×	×
S/L Sep.[2]	-	×	×	×	×	×	×	×
Comp.[3]	-	-	×	-	×	-	×	×
S[6]	×	×	×	×	×	×	×	×
T2[7]	×	×	×	×	×	×	×	×
Scenario type	Baseline	Current	Current	Future	Future	Future	Future	Future

'×' indicates included processes whereas '-' indicates excluded processes.
[1] Mechanical sand separation
[2] Screw press
[3] Intensive windrow on bare soil; Turning operation is done by loaders on composting site.
[4] In Scs. 3, 5 and 7, 50% of manure is separated and 50% is composted.
[5] In Sc. 8, 100 % of digested manure is separated and 100% of solid manure is composted.
[6] In Scs. 3, 5 and 7, storage is applied to the solid separated manure.

[7] In Scs. 3, 5 and 7, transport is applied to the merged solid separated and composted manure.

Sc. 1 is the baseline case (BC), whereby no manure treatment except storage is carried out (Table 1). The two following scenarios (Sc. 2 and 3) show the effect of S/L separation (S/L Sep.) for the total volume (Sc. 2) and for half of the volume in Sc. 3. In the surveyed dairy farms, it was observed that composting slurry was preferred over separation due to the higher cost of separation. Sand separation was added to Sc. 4 and 5. In scenarios 6 to 8, the impact of AD and an improved sand separation were investigated when different post digestion treatment techniques were combined. In Sc. 6, all digestate was separated and stored; whereby in Sc. 8 all digestate was separated, the solid fraction was composted and the liquid fraction was stored. Sc. 7 was an intermediate between Sc. 6 and Sc. 8 where 50% of manure was separated and stored, and the remained 50% was composted. For all scenarios, it was assumed that the solid separated fraction was stored for a definite period (two months), while the composted manure was not stored and the merged solid separated and composted manure was transported to arable land.

2.7. Sensitivity analysis

Sensitivity analysis (SA) was carried out in this study to find out whether changes in surrounding conditions have impact on the manure management systems that is being observed. Conditions that may have effect on manure management systems of this study can be set into two categories. Firstly, changes in policies and modelling decisions (different alternative scenarios), namely passive windrow composting instead of intensive windrow and storage with cover instead of storage without cover (Hamelin et al., 2014). Second type was a one-at-a-time (OAT) approach (local sensitivity analysis) (Groen, 2016). This SA was performed by varying some input parameters contributed to the technical, environmental and economic outputs of future scenarios (Scs.6-8) by ±10%. The input parameters for sensitivity analysis were selected accurately to consider all influential parameters. Details behind the modelling of sensitivity analyses are presented in the SI.

3. Results

Different scenarios of manure management on a large-scale dairy farm in Iran were investigated using a framework model, assessing the technical (mass balance), environmental and economic aspects. The results are presented for different above mentioned aspects as follows:

3.1. Mass balance of nutrients

The characteristics of the excreted manure (N, P, K, C content) and manure properties are presented in Table S20. For the selected farm, the total manure excretion, amounts of N, P, K, fraction of TAN in the excreted N and the C: N ratio were calculated as 504.6, 2.9, 0.24, 1.8 t d^{-1}, 0.7 and 7.467.

A MB for the total mass and manure quality for scenarios 1, 2 and 3 is presented in Fig. 3, and for scenarios 4-8 in Figs. S2 and S3. The values of the total mass are expressed as t d^{-1} and the quality of manure is expressed as kg t^{-1} of manure at the end of each step. The balance shows how different methods of manure processing affect losses compared to the other scenarios. Sand separation saved large amounts of sand estimated at over 164 kg t^{-1} of excreted manure in Sc. 4 and 82 kg t^{-1} in Sc. 5, reduced the manure volume and caused no nutrients to be lost. S/L separation hardly affected nutrient losses while it reduced further transport problems, as the liquid fraction could be applied at the dairy farm, whereas the solid fraction could be exported. In Sc. 6, 7 and 8 where AD was applied, there was a significant reduction in carbon compared to the other pathways (over 50% in the final product). Although, the carbon decomposition rate was high during composting, a large amount of carbon was oxidized and emitted as CO_2. Comparing Sc. 6, 7 and 8 shows that about 28%, 24% and 29% of nitrogen was decomposed and volatilized, which increased the N loss. The losses from storage were affected by the type and quality of the manure, while the loss of nutrients in the solid fraction was lower than the slurry. For example, comparing Sc. 2 with the baseline case, the total loss was reduced by about 18%.

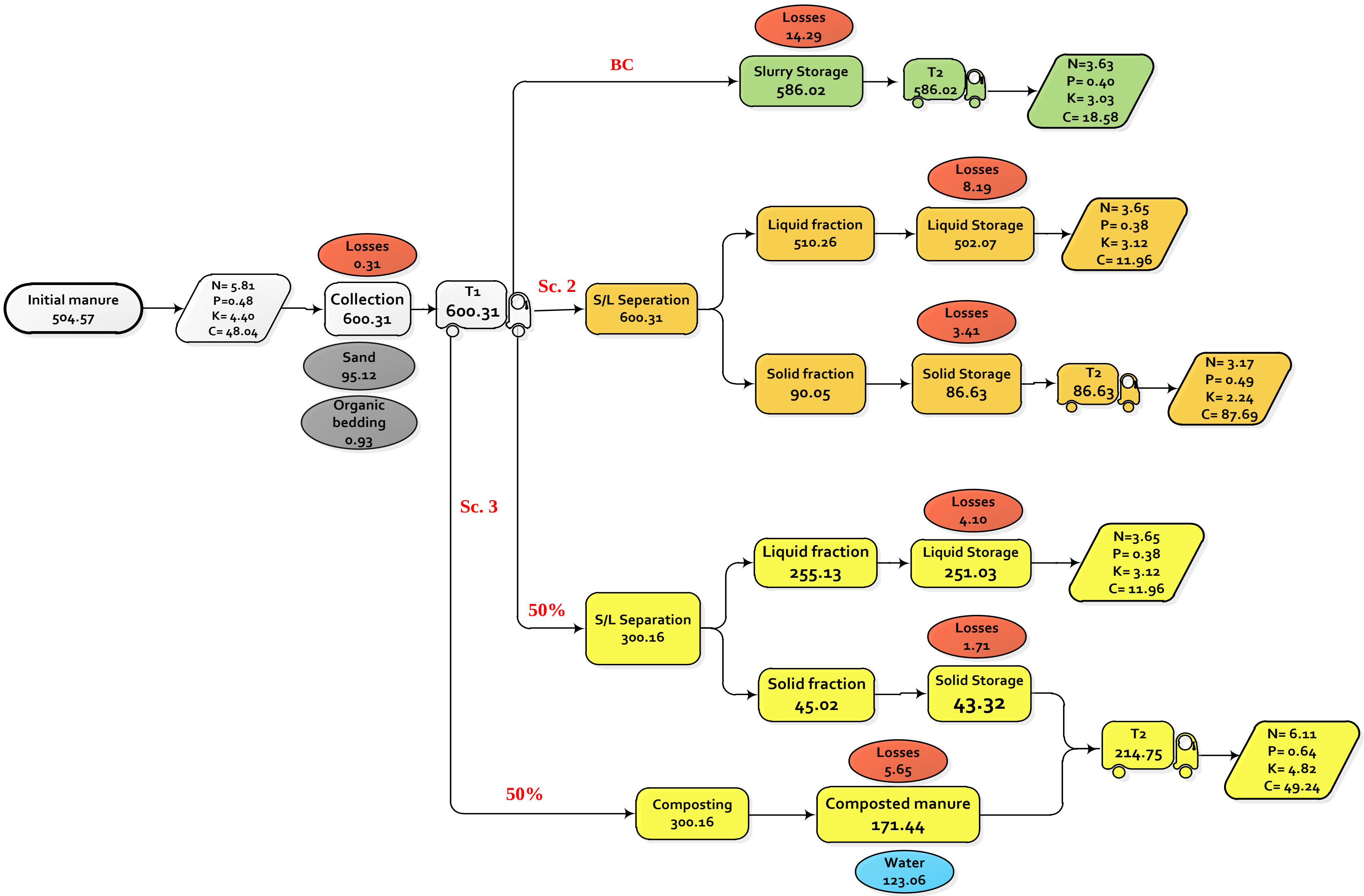


Fig. 3. Mass balance through Sc. 1, Sc. 2 and Sc. 3. All values are in t d^{-1} except for the values in trapezoids in kg t^{-1}.

3.2. Environmental assessment

GHG emissions are presented in two categories: fossil based and biotic based emissions. GHG emissions were converted to the 100-year time horizon global warming potential (GWP) using the appropriate coefficients (298 and 34 kg CO_{2eq} for N_2O and CH_4) (Stocker et al., 2013). Results are expressed per t of excreted manure. Fig. 4 represents the global warming potential of all scenarios.

3.2.1. Fossil carbon dioxide emissions

The fossil CO_2 emissions ($CO_{2\text{-}f}$) were in the range of 0.8 – 4.5 kg CO_{2eq} t^{-1} that have contributed least to the total emission. In the case of AD, 42.4 kg CO_{2eq} may be avoided by using produced biogas to generate electricity and heat by a CHP.

3.2.2. Biogenic methane emissions

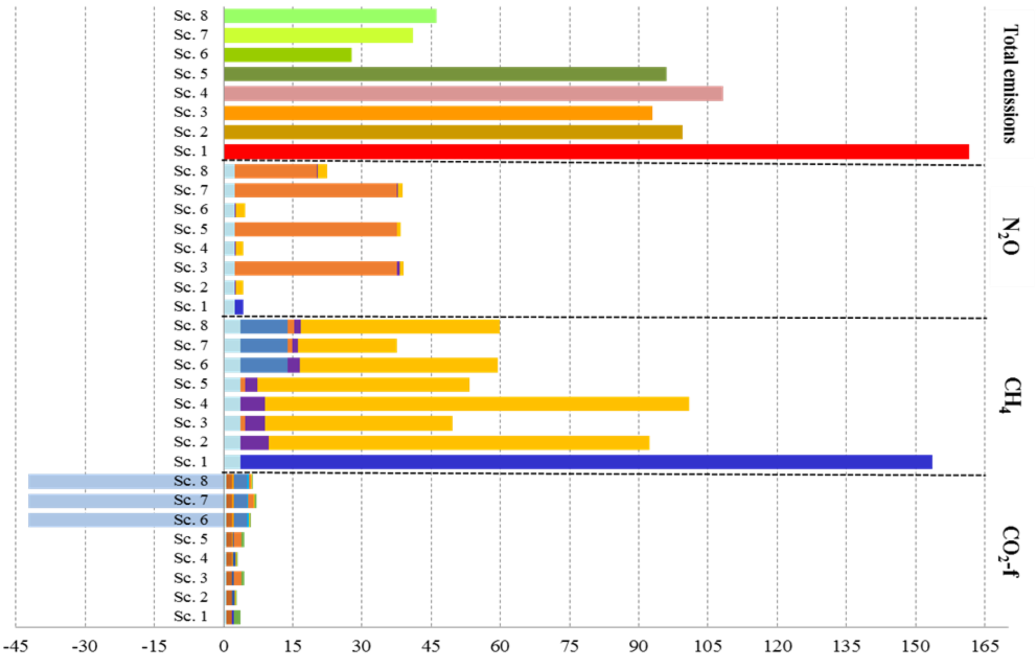
CH_4, mainly produced by storage of slurry in Sc. 1, was the major source of emissions, while Sc. 2 and 4 ranked next in emitting CH_4 from the liquid manure storage. During Scs. 6, 7 and 8, results showed that there was a potential for CH_4 formation in the storage of liquid fraction following AD. This considerable share of CH_4 was mainly due to the anaerobic conditions in pits/silos and the limited retention time in the AD equipment (26 days). Separation of manure reduces the methane emission as shown in all other scenarios. Composting of the manure, which is practised for 50% of the volume in Scs. 3, 5 and 7 showed a reduction in CH_4 emissions, compared to their neighbouring scenarios 2, 4 and 6, where all manure was separated to liquid and solid fraction. A reduction of CH_4 in these schemes was related to the type of the composting treatment. The CH_4 emission calculated for scenario 8 was comparable to scenario 7, due to the fact that all digestate was separated in Sc. 8 and the liquid fraction had a large contribution to CH_4 emissions.

3.2.3. Biogenic nitrous oxide emissions

N_2O was substantially emitted through scenarios where composting was applied. N_2O was estimated to be 35.1 kg CO_{2eq} t^{-1} in Sc. 3, 5 and 7 and 17.8 kg CO_{2eq} t^{-1} in Sc.8 caused by a lower N content of the solid fraction, compared to the liquid fraction. Separating manure to the solid fraction (C-rich and N-poor fraction) and liquid fraction (C-poor and N rich fraction) followed by solid manure composting reduced N_2O emissions and reduced the amount of CH_4 in the liquid manure storage tank. The sum of N_2O emissions from the storage of solid and liquid fractions was equal in scenarios 2 and 4, whereby 16% of that was attributed to the storage of the solid fraction. The same trend was observed in scenarios 6 to 8 showing the efficiency of separation slurry into N-rich liquid fraction.

3.2.4. Total emissions

Results indicated that scenarios including AD (Scs. 6, 7 and 8) had the lowest total emissions due to the capture of methane emissions in the digester and avoidance of fossil-fuel-related emissions due to generating electricity from biogas. Of these three scenarios, Sc. 6 had the lowest GWP impact amounted to 27.7 kg CO_{2eq} t^{-1} and significantly lowest N_2O emissions. Emissions from Sc. 1 were estimated to be 162 kg CO_{2eq} t^{-1}, contributing most to the GWP impact.



CO_2

	Sc. 1	Sc. 2	Sc. 3	Sc. 4	Sc. 5	Sc. 6	Sc. 7	Sc. 8
Total emissions								
Collection	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Transport1	1.4	1.4	1.4	1.4	1.4	1.4	1.4	0
Sand Seperation				0.2	0.1	0.3	0.3	0.3
S/L Seperation1		0.5	0.3	0.5	0.2			
Anaerobic digestion						2.9	2.9	2.9
S/L Separation2						0.4	0.2	0.4
Composting			1.6		1.6		1.3	0.4
Solid storage	0	0	0	0	0	0	0	0
Liquid storage	0	0.2	0.1	0.2	0.1	0.2	0.1	0.2
Slurry storage	0.3							
Transport2	1.5	0.2	0.5	0.2	0.5	0.2	0.4	0.1
Avoided						-42.4	-42.4	-42.4

CH_4

	Sc. 1	Sc. 2	Sc. 3	Sc. 4	Sc. 5	Sc. 6	Sc. 7	Sc. 8
Total emissions								
Collection	3.7	3.7	3.7	3.7	3.7	3.7	3.7	3.7
Transport1	0	0	0	0	0	0	0	0
Sand Seperation				0	0	0	0	0
S/L Seperation1		0	0	0	0			
Anaerobic digestion						10.2	10.2	10.2
S/L Separation2						0	0	0
Composting			1		1		0.9	1.4
Solid storage	0	6	4.2	5.2	2.6	2.6	1.3	1.5
Liquid storage	0	82.7	40.7	92	46	43	21.5	43
Slurry storage	150	0	0					
Transport2	0	0	0	0	0	0	0	0
Avoided								

N_2O

	Sc. 1	Sc. 2	Sc. 3	Sc. 4	Sc. 5	Sc. 6	Sc. 7	Sc. 8
Total emissions								
Collection	2.4	2.4	2.4	2.4	2.4	2.4	2.4	2.4
Transport1	0	0	0	0	0	0	0	0
Sand Seperation				0	0	0	0	0
S/L Seperation1		0	0	0	0			
Anaerobic digestion						0	0	0
S/L Separation2						0	0	0
Composting			35.1		35.1		35.1	17.8
Solid storage	0	0.3	0.6	0.3	0.1	0.3	0.2	0.3
Liquid storage	0	1.6	0.8	1.6	0.8	2	1	2
Slurry storage	1.8	0	0					
Transport2	0	0	0	0	0	0	0	0
Avoided								

Total emissions

	Sc. 1	Sc. 2	Sc. 3	Sc. 4	Sc. 5	Sc. 6	Sc. 7	Sc. 8
Total emissions	162	99.5	92.9	108	96.1	27.7	41.0	46.1

Fig. 4. Process based analysis of GHG emissions (kg CO_{2eq} t^{-1}) for all steps of the MMC scenarios.

3.3. Manure Nutrients Recovery Efficiency

The comparison of the MNRE index across all scenarios is illustrated in Fig. 5. In this graph, 100% refers to a situation whereby all the inputs of a component remain in the final product(s). The total nitrogen recovery ranges between 71 and 76%, where Sc. 8 and Sc. 7 had the lowest and highest recovery levels. In scenarios where composting was performed on half of the manure, less nitrogen was exposed to N decomposition during storage and higher N recovery was attained. In scenarios involving AD, the ratio between N_{org} and N_{min} changed in favour of plant-available ammonium. All the scenarios showed a constant and high level of P recovery since phosphorus is hardly soluble and leached. The level of K recovery was less than P (about 80%). When separation and composting was applied to the AD digestate (Sc. 8), the recovery decreases to about 78%. The higher recovery rate in Scs. 3, 5 and 7, compared to Scs. 2, 4 and 6, may be due to the fact that in the latter 3 scenarios, less manure was stored, whereas more K may have been lost through runoff and leaching. The MNRE of C was very low for all scenarios suggesting that carbon conversion took place in all types of manure treatment, whereas the carbon conversion in the AD scenarios (Scs. 6, 7 and 8) showed a stronger decrease of carbon recovery. The lowest MNRE rates for C were found in Sc. 8 (35%) with separation and intensive composting of digestate, which ended up with greater losses. In this scenario, AD was applied followed by mechanical separation and composting of the C-rich solid fraction where C conversion via composting was intensified. Meanwhile, the liquid fraction of the manure was stored. The liquid fraction had a large contribution to CH_4 emissions with a reduced C recovery. Regarding C recovery, the highest MNRE was found in scenario 2 (56%), where manure separation was practised without active composting.

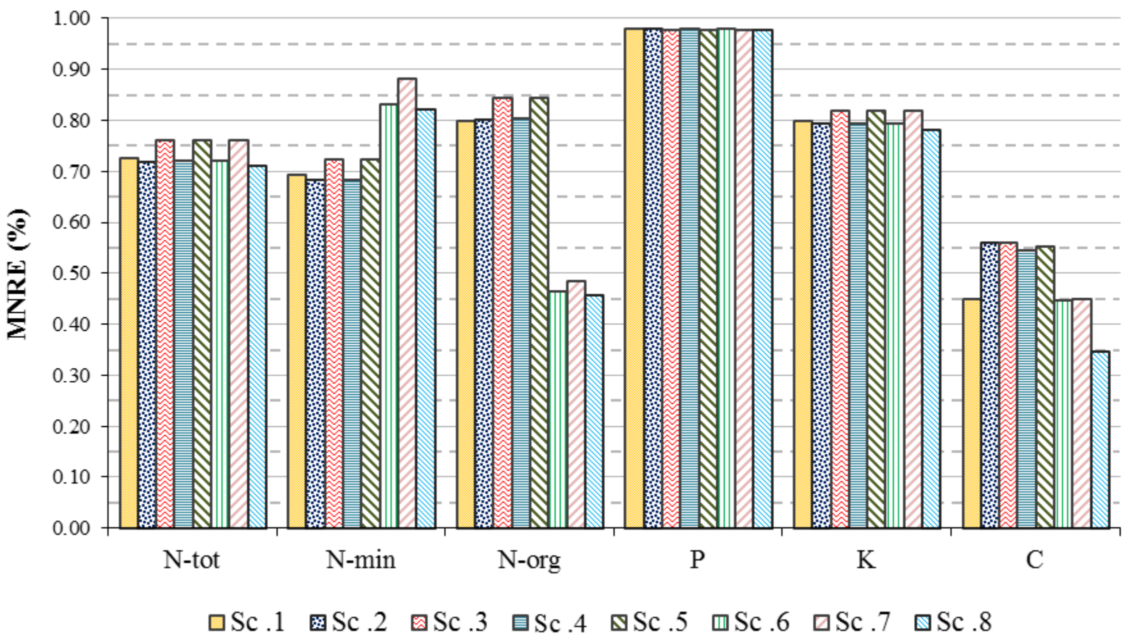


Fig. 5. The Manure Nutrient Recovery Efficiency for N (total, organic and mineral), P, K and C at the farm level for different scenarios.

3.4. Cost-benefit analysis of the scenarios

A cost-benefit analysis was performed by the TECBA-MMC tool using the data provided by the study farm. The results of this section are presented in Table 2. Financial indices were calculated to investigate the feasibility of each scenario. The attractiveness of the investment projects is clearly shown by the positive net present value (NPV), the higher (greater than one) profitability index (PI), and the higher internal rate of return (IRR) (greater than the discount rate of 20%, in this case). As shown in Table 2, scenarios that excluded AD showed higher profitability indices, while more GHG emissions were produced. The highest NPV was achieved in Sc. 1 as the lowest total investment cost (TIC) was charged. As shown in Table 2, Scs. 4 and 5 which include sand separation had lower economic indices, while the avoided costs due to the sand recovery and maintenance had been taken into account. Therefore, the added weight of sand mixed with manure sold to arable lands, regardless of manure quality, would lead to the higher profitability. Comparing Scs. 6-8 for cost-effectiveness of AD showed the financial feasibility of these scenarios. This economic viability was due to the high price of electricity sold to the grid with the current high subsidy policy on renewable energy production in Iran. This has motivated dairy farmers, especially those on large-scale farms, to start investing on such projects with an average payback period of two years. There are other benefits to biogas production such as social impacts, which have not been quantified in this study, such as improved sanitation on the farm, health implications, more job opportunities and the economic growth of the area. The post digestion treatment techniques and the corresponding by-products resulted in the difference in profitability of these projects. For example, Sc. 7 was shown to be more profitable than Sc. 6 and 8 since a greater amount of solid digestate was produced (about 60% and 78% more than the other two scenarios).

Table 2. Calculated GHG, total investment costs and financial indices of the manure treatment techniques

Criteria	Unit	Sc. 1	Sc. 2	Sc. 3	Sc. 4	Sc. 5	Sc. 6	Sc. 7	Sc. 8
Separation	%	0	100	50	100	50	100	50	100*
Composting	%	0	0	0	0	0	0	50	100**

GHG total	kg CO_2 t^{-1}	161.6	99.5	92.9	108.2	96.1	27.7	41	46.1
TIC	K€	43.11	66.67	148.94	92.34	174.6	2558.8	2639	2629.2
BCR	-	113.73	22.08	24.21	18.03	20.74	2.34	4.77	2.09
IRR	%	50	38	39	35	38	22	27	22
PBP	y	0.01	0.07	0.06	0.1	0.07	2.46	1.08	2.74
PI	-	347.98	62.83	75.7	43.16	63.11	1.72	3.9	1.55
NPV	K€	15724.8	3560	4008.0	3369.3	3901.5	3032.3	9494.1	3312.2

* This is applied to 100% of digestate

** This is applied to 100% of solid fraction

3.5. Sensitivity analysis results

Two types of sensitivity analyses were carried out to determine the robustness of the results by changing two alternative manure treatment methods and varying sensible parameters. The effect on GWP, avoided emissions, NPV and nutrient availability for Scs. 6-8 were reported. Results for alternative composting are represented in Fig. 6 and for other scenarios are presented in Figs. S4-S7.

Passive composting instead of intensive composting would be an advantage for better environmental performance since passive method is a type of windrow composting with much less turning schedule. For passive composting performed in Scs.3, 5 and 7, Fig. 6 highlights a decrease in total GWP through all scenarios (around 34%). This result is in agreement with the findings of Hao et al. (2001). Higher N_2O emission for the intensive composting has been decreased in much larger magnitude during passive composting method (around 87%). Less N_2O emissions was reported when static pile with less frequent aeration (passive) method was compared with intensive composting (Szanto et al., 2007). The detailed results are described in the SI.

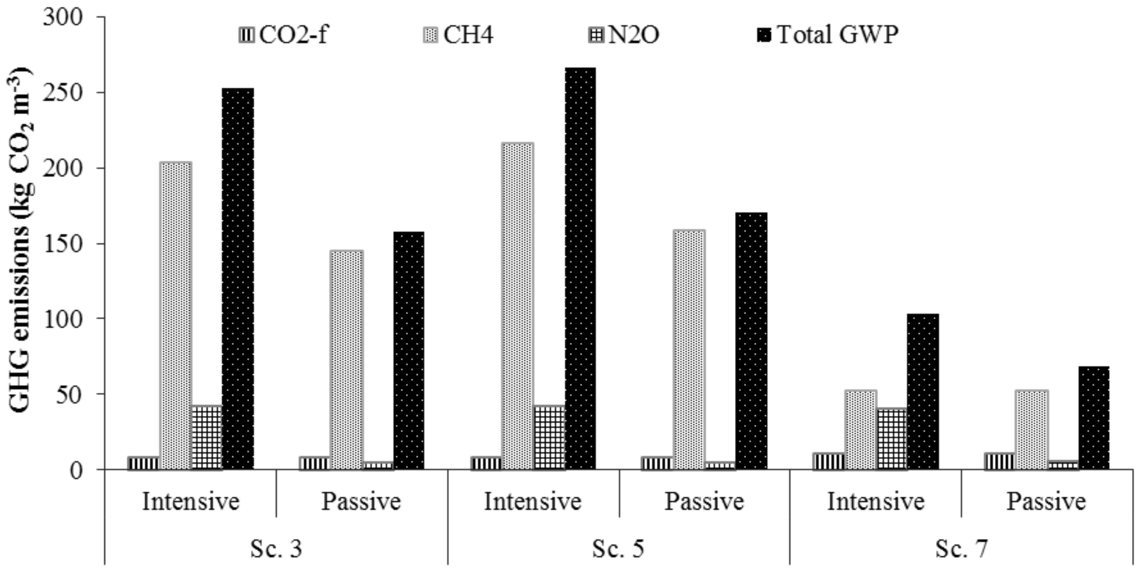


Fig. 6. Results for GHG emissions for the sensitivity analysis performed with passive composting instead of intensive composting

4. Discussion

The technical, environmental and economic aspects of different manure management systems were assessed so that the eco-friendly scenarios promoting circular economy and recovering the essential nutrients in organic fertilizers were identified. Providing the subsequent technical, environmental and economic analysis, a comprehensive overview of the evaluated scenarios using the TECBA-MMC model is given in Table 3. The negative and values show deteriorations and improvements when comparing each scenario with the BC. It should be noted that a negative NPV does not reflect a net loss; instead, it demonstrates the change to the BC.

Table 3. Changes in the performance of each scenario compared to the BC*

Scenario	Unit	Sc. 2	Sc. 3	Sc. 4	Sc. 5	Sc. 6	Sc. 7	Sc. 8
Recovery of N_{tot}	%	-0.68**	4.8	-0.68	4.8	-0.68	4.8	-2.1
Recovery of P	%	0	-0.2	0	-0.2	0	-0.2	-0.41
Recovery of K	%	-1.25	2.5	-1.25	2.5	-1.25	2.5	-2.5
Recovery of C	%	24.4	24.4	20	22.2	0	0	-24.4
Total GHG	%	-38.5	-42.6	-32.2	-40.7	-83	-74.7	-71.5
NPV	%	-77.4	-74.5	-78.6	-75.2	-80.7	-39.6	-78.9

* Difference= [B-A/A] ×100
** Negative values shows deterioration and positive values shows improvement.

The results showed that although the sand separation process (in Sc.4 and Sc. 5) did not significantly affect emissions (compared to Sc. 2 and Sc. 3), it saved bedding sand, transport and equipment amortization costs. Taking into account the avoided costs and emissions of sand mining could affect the results of this study. Sand separation is a crucial process for the efficiency of the AD process due to sand sedimentation and time losses for removing sediments in a continuous AD process. Regarding sand separation, no relevant evidence in previous studies was found for comparison purposes.

The large reduction of emissions in the AD scenarios (71.5% - 83%) may have been caused by capturing methane emissions and saving fossil fuels. Since the capture of methane by AD can be considered as avoided emissions, it cannot be used as a negative value in the calculations. The AD process optimised methane production from manure. The savings in fossil fuels due to producing electricity and utilising heat was used as a negative figure in the calculations. Comparing the GHG emissions of the AD in scenario 6 with the baseline scenario showed a reduction of emissions to 134.3 kg CO_{2eq} t^{-1}. Pardo et al. (2017) reported a reduction of 101 kg CO_{2eq} t^{-1} for a one stage mesophilic AD followed by open storage in temperate climatic conditions. Based on these results, the carbon conversion and CH_4 capturing occurred in the AD lead to a reduced decomposition rate of carbon in the subsequent storage process. In this study, a reduction of 9% VS in digestate, relevant to the change in C amount, was estimated. Pardo et al. (2017) calculated this reduction to be 7%. The shift from N_{org} to N_{min} during the AD process was estimated to be 36%, 40%, and 36%, showing the higher NH_4 content in the digested slurry. This change has been reported by Aguirre-Villegas et al. (2014) (68%) and Möller et al. (2008) (24%).

Of the options found to reduce emissions and costs, separation of the slurry or the digested slurry into solid and liquid fraction was best. Enhancing the quality of manure streams is an additional benefit of this technique. Nutrients such as N and P may be removed along with the solid stream and can be further used as fertilizer or bedding. The same implication was reported in previous studies by Hjorth et al. (2010) and Aguirre-Villegas and Larson (2017). Another important aspect of this was the fate of the liquid fraction of mechanical separation. When applied at the dairy farm, a certain area of land is needed for application in an agronomic way and it has to match the crop requirements. Otherwise, it should be transported to neighbouring lands or sent to further separation devices for reverse osmosis to produce water that can be discharged to surface water. Insufficient land availability runs the risk of an over application of nutrients.

The results show that composting of manure may be a cost saving action, due to the volume reduction for transportation, caused by evaporation of water and the oxidation of carbon. This is done at the expense of nitrogen and potassium recovery and increasing nitrous oxide emissions, and in the meantime reducing methane emissions. Tiquia et al. (2002) reported great C losses in composting methods included windrow turnings. There are additional studies, which have reported the high C and N losses during composting (Hao et al., 2001; Sommer, 2001). Due to investment limitations, farmers may prefer to select composting. This method is not recommended when the nutrient recovery in the final manure is substantial due to higher risk of N and K losses. There exists a trade-off between these conflicting objectives, i.e. maximum nutrients recovery efficiency, minimum environmental impacts and maximum financial profitability.

This study showed that the developed model may be utilised as a helpful tool for assessing technical, environmental and economic effects of manure management options; however, this model still has its limitations regarding ensuring the best solution due to a number of uncertainties. The main uncertainties in the results of this study are related to the robustness of the data derived from literature and the emission factors used. There were a number of limitations related to data requirements. In this regard, local data collection for the feed ration of cows, nutritional value of the feed, investment costs and capacities to implement the new technologies is necessary. Selecting suitable coefficients and emissions factors to specifically meet the dairy farming situation in Iran was the other limitation of this study.

The application of the model in evaluating the performance of eight scenarios throughout the MMC from manure collection to transport of manure to arable lands represents the cost-effectiveness and environmental impacts in supporting decision makers for choosing the best options that meet agronomic, environmental and economic goals. The approach adopted and the model developed can be applied for other managerial decisions, herd compositions, feedstock mixtures, while considering technical, environmental and economic aspects. This study presents an assessment of different manure management scenarios considering three individual perspectives and it still lacks the integration of three aspects. In this context, this study may be extended to a multi criteria decision-making analysis for selecting the best scenarios considering these three aspects. For future works, the model may be extended to incorporate more alternative strategies of manure processing. For example, different manure separation techniques such as reverse osmosis and ultrafiltration, storage facilities, co-digestion of manure with other locally available organic wastes could be considered. Social assessment is recommended to be integrated into this model in future works to achieve a sustainable life cycle of manure management.

The sensitivity analyses performed in this study showed the most effective parameters on the most important indicators of this study. Dry matter was influential on GWP and the annual discount rate and loan rate on biogas projects had a great impact on NPV. Covered storage achieved higher environmental impact reduction. The overall effects on GWP, NPV and other management indicators will be analysed in future works.

5. Conclusion

Manure management and processing scenarios in large-scale dairy farming were evaluated by developing a process-based model considering the technical, environmental and economic performance of an entire chain. The model was shown to operate effectively and has shown its value in a described case study. The results of applying the model to the study farm presents the efficiency of manure treatment techniques in a whole MMC, which will aid farmers, decision makers and investors to understand the performance of each scenario and improve their practices to reduce GHG emissions. The main key factors for the best manure management methods which were analysed in this study consisted technical, environmental and economic aspects.

To predict the quality of the final product of each scenario, a mass-balance approach was employed. As a result, a decision maker (or farmer) can select the suitable manure treatment scenario considering the farm condition such as soil characteristics and plant requirement. Based on the results, sand separation did not significantly affect emissions, quality of the end-product and could be quite efficient in reducing energy consumption related to the mining, processing and transporting of sand, and reducing maintenance and transport costs. In terms of GHG emissions, scenarios 1-3 had reasonable increase of environmental emissions. In contrast, the results showed no contribution of sand separation to environmental burdens. Composting reduced transport costs and resulted in low recoveries of C, N and K and high levels of nitrous oxide emissions that reduces the agronomic value of the final compost. Composting would be a good solution when manure needs to be exported and the quality of compost is not important.

In general, S/L separation was very beneficial to emission mitigation especially methane, reducing transport cost and preventing over fertilization but still greatly contributing to nitrous oxide emission when combined with composting. AD was found as the best option to reduce emissions. Although AD is a costly manure strategy, the current situation in Iran makes it profitable. AD not only keeps nutrients in the digestate but also makes the nitrogen uptake by plants easier due to nitrogen mineralization.

To assess the robustness of the findings, sensitivity analyses were performed. The sensitivity scenarios with a high DM content, with covered storage, passive composting and different leakage, electricity price, loan rate and annual discount rate resulted in different ranges of the considered economic and environmental impacts.

With respect to the findings of the process-based scenario analyses, this study is useful for farmers, investors and policy makers of manure management and renewable energy production to have a clear understanding related to the consequences of each decision (manure management methods) and further plan or decide how to improve manure management systems at both farm and regional level. It would enable to consider the trade-off between the conflicting objectives such as maximum nutrients recovery efficiency, maximum financial profitability and/or minimum environmental impacts. To get the most reliable results that may help farmers, investors and policy makers in the decision making process, it is important to consider the most characteristic (and state-of-the-art) technology combined with information representing the farm in question.

Acknowledgments

The authors express their appreciation to Dr. Karin Groenestein, Dr. Roland Melse, and Ing. Fridtjof de Buisonje for their stimulating discussion and remarks. We also thank to Dr. Bastiaan Meerburg, the head of Livestock and environment department, Wageningen University and Research for all of his supports in this project.

References

Aguirre-Villegas, H.A., Larson, R.A., 2017. Evaluating greenhouse gas emissions from dairy manure management practices using survey data and lifecycle tools. Journal of Cleaner Production 143, 169-179.

Aguirre-Villegas, H.A., Larson, R., Reinemann, D.J., 2014. From waste-to-worth: energy, emissions, and nutrient implications of manure processing pathways. Biofuels, Bioproducts and Biorefining 8(6), 770-793.

Araji, A., Stodick, L., 1990. The economic potential of feedlot wastes utilization in agricultural production. Biological Wastes 32(2), 111-124.

Audsley, E., Wilkinson, M., 2014. What is the potential for reducing national greenhouse gas emissions from crop and livestock production systems? Journal of Cleaner Production 73, 263-268.

Blumenstein, B., Siegmeier, T., Selsam, F., Möller, D., 2018. A case of sustainable intensification: Stochastic farm budget optimization considering internal economic benefits of biogas production in organic agriculture. Agricultural Systems 159, 78-92.

Burney, J., Lo, K., Carson, W., 1980. Dairy waste management system selection by network analysis. Canadian Agricultural Engineering 22(1), 41-47.

Chadwick, D., Wei, J., Yan'an, T., Guanghui, Y., Qirong, S., Qing, C., 2015. Improving manure nutrient management towards sustainable agricultural intensification in China. Agriculture, Ecosystems and Environment 209, 34-46.

Chianese, D., Rotz, C., Richard, T., 2009. Simulation of carbon dioxide emissions from dairy farms to assess greenhouse gas reduction strategies. Transactions of the ASABE 52(4), 1301-1312.

De Vries, J., Vinken, T., Hamelin, L., De Boer, I., 2012. Comparing environmental consequences of anaerobic mono-and co-digestion of pig manure to produce bio-energy–a life cycle perspective. Bioresource Technology 125, 239-248.

Ecoinvent3.3, 2016. Ecoinvent®Swiss Center for Life Cycle Inventories. http://www.ecoinvent.org/database/ecoinvent-33/ecoinvent-33.html. (Accessed 06 June 2018).

Gharaei, A., Pasandideh, S.H.R., 2017. Four-echelon integrated supply chain model with stochastic constraints under shortage condition. Industrial Engineering & Management Systems 16(3), 316-329.

Gharaei, A., Pasandideh, S.H.R., Arshadi Khamseh, A., 2017. Inventory model in a four-echelon integrated supply chain: Modeling and optimization. Journal of Modelling in Management 12(4), 739-762.

Groen, E.A., 2016. An uncertain climate: the value of uncertainty and sensitivity analysis in environmental impact assessment of food. PhD thesis. Wageningen University, Wageningen, the Netherlands.

Hamelin, L., Naroznova, I., Wenzel, H., 2014. Environmental consequences of different carbon alternatives for increased manure-based biogas. Applied Energy 114, 774-782.

Hao, X., Chang, C., Larney, F.J., Travis, G.R., 2001. Greenhouse gas emissions during cattle feedlot manure composting. Journal of Environmental Quality 30(2), 376-386.

Hjorth, M., Christensen, K.V., Christensen, M.L., Sommer, S.G., 2010. Solid–liquid separation of animal slurry in theory and practice. Agronomy for Sustainable Development 30(1), 153-180.

Hoseini Shekarabi, S.A., Gharaei, A., Karimi, M., 2018. Modelling and optimal lot-sizing of integrated multi-level multi-wholesaler supply chains under the shortage and limited warehouse space: generalised outer approximation. International Journal of Systems Science: Operations & Logistics, 1-21.

IPCC, 2006. Agriculture, forestry and other land use, Guidelines for National Greenhouse Gas Inventories, vol. 4, chapter 10. http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html (Accessed 06 June 2018).

Jørgensen, P.J., 2009. Biogas– green energy, Process, Design, Energy supply, Environment, 2 ed. Faculty of Agricultural Sciences, Aarhus University, Danmark.

Junior, C.C., Cerri, C.E., Pires, A.V., Cerri, C.C., 2015. Net greenhouse gas emissions from manure management using anaerobic digestion technology in a beef cattle feedlot in Brazil. Science of the Total Environment 505, 1018-1025.

MacLeod, M., Vellinga, T., Opio, C., Falcucci, A., Tempio, G., Henderson, B., Makkar, H., Mottet, A., Robinson, T., Steinfeld, H., 2018. Invited review: A position on the Global Livestock Environmental Assessment Model (GLEAM). Animal 12(2), 383-397.

Makara, A., Kowalski, Z., 2018. Selection of pig manure management strategies: case study of polish farms. Journal of Cleaner Production 172, 187-195.

Melse, R., de Buisonje, F., 2015. Quick calculation tool for manure separation and distribution of mass, minerals and dry matter. http://sdddc.org/en/download/detail-155.aspx. (Accessed 06 June 2018).

Mezzullo, W.G., McManus, M.C., Hammond, G.P., 2013. Life cycle assessment of a small-scale anaerobic digestion plant from cattle waste. Applied Energy 102, 657-664.

Möller, K., Stinner, W., Deuker, A., Leithold, G., 2008. Effects of different manuring systems with and without biogas digestion on nitrogen cycle and crop yield in mixed organic dairy farming systems. Nutrient Cycling in Agroecosystems 82(3), 209-232.

Montalvo, C., 2008. General wisdom concerning the factors affecting the adoption of cleaner technologies: a survey 1990–2007. Journal of Cleaner Production 16(1), S7-S13.

Monteny, G., Groenestein, C., Hilhorst, M., 2001. Interactions and coupling between emissions of methane and nitrous oxide from animal husbandry. Nutrient Cycling in Agroecosystems 60(1-3), 123-132.

Nennich, T., Harrison, J.H., Vanwieringen, L.M., Meyer, D., Heinrichs, A., Weiss, W.P., St-Pierre, N.R., Kincaid, R., Davidson, D.L., Block, E., 2005. Prediction of manure and nutrient excretion from dairy cattle. Journal of Dairy Science 88(10), 3721-3733.

NRC, 2001. Nutrient requirements of dairy cattle, 7 ed. National Academies Press, Natl. Acad. Press, Washington, DC.

O’Mara, F.P., 2011. The significance of livestock as a contributor to global greenhouse gas emissions today and in the near future. Animal Feed Science and Technology 166, 7-15.

Oenema, O., Oudendag, D., Velthof, G.L., 2007. Nutrient losses from manure management in the European Union. Livestock Science 112(3), 261-272.

Pardo, G., Moral, R., del Prado, A., 2017. SIMSWASTE-AD-A modelling framework for the environmental assessment of agricultural waste management strategies: Anaerobic digestion. Science of the Total Environment 574, 806-817.

Petersen, S.O., Sommer, S., Béline, F., Burton, C., Dach, J., Dourmad, J., Leip, A., Misselbrook, T., Nicholson, F., Poulsen, H.D., 2007. Recycling of livestock manure in a whole-farm perspective. Livestock Science 112(3), 180-191.

Pratt, C., Redding, M., Hill, J., Jensen, P.D., 2015. Does manure management affect the latent greenhouse gas emitting potential of livestock manures? Waste Management 46, 568-576.

Reidy, B., Dämmgen, U., Döhler, H., Eurich-Menden, B., Van Evert, F., Hutchings, N., Luesink, H., Menzi, H., Misselbrook, T., Monteny, G.-J., 2008. Comparison of models used for national agricultural ammonia emission inventories in Europe: Liquid manure systems. Atmospheric Environment 42(14), 3452-3464.

Reijs, J.W., 2007. Improving slurry by diet adjustments: a novelty to reduce N losses from grassland based dairy farms. PhD thesis. Wageningen University, Wageningen, The Netherlands.

Rennie, T.J., Gordon, R.J., Smith, W.N., VanderZaag, A.C., 2018. Liquid manure storage temperature is affected by storage design and management practices—A modelling assessment. Agriculture, Ecosystems & Environment 260, 47-57.

Rotz, C.A., Corson, M.S., Chianese, D.S., Montes, F., Hafner, S.D., Bonifacio, H.F., Coiner, C.U., 2016. Integrated Farm System Model: Reference Manual. Vol. 4.3. Pasture Systems and Watershed Management Research Unit, Agricultural Research Service, USDA, United States.

Sommer, S.G., 2001. Effect of composting on nutrient loss and nitrogen availability of cattle deep litter. European Journal of Agronomy 14(2), 123-133.

Sommer, S.G., Petersen, S.O., Møller, H.B., 2004. Algorithms for calculating methane and nitrous oxide emissions from manure management. Nutrient Cycling in Agroecosystems 69(2), 143-154.

Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M., 2013. Climate Change 2103: the Physical Science Basis. , 1 ed., New York, USA.

Szanto, G., Hamelers, H., Rulkens, W., Veeken, A., 2007. NH3, N2O and CH4 emissions during passively aerated composting of straw-rich pig manure. Bioresource Technology 98(14), 2659-2670.

ten Hoeve, M., Hutchings, N.J., Peters, G.M., Svanström, M., Jensen, L.S., Bruun, S., 2014. Life cycle assessment of pig slurry treatment technologies for nutrient redistribution in Denmark. Journal of Environmental Management 132, 60-70.

Tiquia, S., Richard, T., Honeyman, M., 2002. Carbon, nutrient, and mass loss during composting. Nutrient Cycling in Agroecosystems 62(1), 15-24.

Tittonell, P., Rufino, M.C., Janssen, B.H., Giller, K.E., 2010. Carbon and nutrient losses during manure storage under traditional and improved practices in smallholder crop-livestock systems—evidence from Kenya. Plant and Soil 328(1-2), 253-269.

Torquati, B., Venanzi, S., Ciani, A., Diotallevi, F., Tamburi, V., 2014. Environmental sustainability and economic benefits of dairy farm biogas energy production: A case study in Umbria. Sustainability 6(10), 6696-6713.

Vonk, J., Bannink, A., van Bruggen, C., Groenestein, C., Huijsmans, J., van der Kolk, J., Luesink, H., Voshaar, S.O., Sluis, S., Velthof, G., 2016. Methodology for estimating emissions from agriculture in the Netherlands. Wageningen University & Research, Wageningen, The Netherlands.

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

Yazan, D.M., Cafagna, D., Fraccascia, L., Mes, M., Pontrandolfo, P., Zijm, H., 2018. Economic sustainability of biogas production from animal manure: a regional circular economy model. Management Research Review 41(5), 605-624.

Supporting Information

Technical, environmental and cost - benefit assessment of manure management chain: case study of large scale dairy farming

Paria Sefeedpari[a,b,c*], Theun Vellinga[b], Shahin Rafiee[a*], Mohammad Sharifi[a], Philip Shine[d],

Seyyed Hassan Pishgar-Komleh[e]

[a] Department of Agricultural Machinery, Faculty of Agricultural Engineering and Technology, University of Tehran, Karaj, P.O. Box 4111, Iran.
[b] Wageningen Livestock Research, Wageningen University and Research, PO Box 338, 6700 AH, Wageningen, The Netherlands.
[c] Institute of Soil Science and Plant Cultivation—State Research Institute, P.O. Box 24 100 Puławy, Poland
[d] Department of Process, Energy and Transport Engineering, Cork Institute of Technology, Cork, Ireland
[e] Department of Economics, Faculty of Economics and Management, Czech University of Life Sciences Prague, Kamýcká 129, 165 00Prague, Suchdol, Czech Republic

*Corresponding author email: pariasefeedpari@ut.ac.ir

1. Introduction

Table S1. Comparison of the most relevant studies that reported models for evaluating manure management systems

Reference	TE[1]	EN[2]	EC[3]	Single MMC	Multiple MMC	Region	Method
Sommer et al. (2004)	×	✓	×	×	✓	Europe	Dynamic modelling
Schils et al. (2007)	✓	✓	✓	✓	×	Netherlands	Empirical modelling
Menind and Olt (2009)	×	×	✓	✓	×	Estonia	Cost-benefit analysis
Prapaspongsa et al. (2010)	×	✓	×	×	✓	Denmark	LCA
De Vries et al. (2012)	×	✓	×	×	✓	Netherlands	LCA
ten Hoeve et al. (2014)	×	✓	×	✓	×	Denmark	LCA
Torquati et al. (2014)	×	×	✓	✓	×	Italy	Economic modelling
Aguirre-Villegas et al. (2014)	×	✓	×	×	✓	Wisconsin	Mechanistic modelling
Hamelin et al. (2014)	×	✓	×	✓	×	Denmark	Consequential LCA
Pardo et al. (2017)	×	✓	×	×	✓	Europe	Mechanistic modelling
Makara and Kowalski (2018)	✓	×	×	×	✓	Poland	MCDM
Ström et al. (2018)	✓	×	✓	×	✓	Cambodia	Socio-economic analysis
Current study	✓	✓	✓	✓	✓	Iran	Process-based modelling framework

[1] Technical analysis [2] Environmental analysis [3] Economic analysis [4] Cost-benefit analysis [5] Multi-Criteria decision making

2. Material and methods

2.1. Manure excretion

To estimate the feed intake for animal subcategories, the following equations were used (IPCC, 2006).

$NE_m = Cf_i \times LW^{0.75}$	NE_m= net energy required for the maintenance (MJ d^{-1}) Cf_i= a coefficient corresponded to animal category (MJ d^{-1} kg^{-1}) (Table S2) LW= live weight of animal (kg)	(S1)
$NE_a = C_a \times NE_m$	NE_a= net energy for animal activity (MJ d^{-1}) C_a= a coefficient corresponded to feeding situation (Table S2)	(S2)
$NE_g = 22.02 \times (BW/C \times MW)^{0.75} \times WG^{1.97}$	NE_g= net energy needed for growth (MJ d^{-1}) BW= the average live body weight (kg) MW = the mature live body weight of, kg WG = the average daily weight gain, kg d^{-1} C= a coefficient corresponded to the animal sex (Table S2)	(S3)
$NE_l = Milk \times (1.47 + 0.40 \times MF)$	NE_l= net energy for lactation (MJ d^{-1}) Milk= amount of milk produced (kg d^{-1}) MF= fat content of milk (%)	(S4)
$NE_p = C_p \times NE_m$	NE_p= net energy required for pregnancy (MJ d^{-1}) C_p= pregnancy coefficient (Table S2) NE_m= net energy required for the maintenance (MJ d^{-1})	(S5)
$REM = \left[1.123 - (4.092 \times 10^{-3} \times DE\%) + [1.126 \times 10^{-5} \times (DE\%)^2]\right] - (25.4/DE\%)$	REM= ratio of net energy in a diet for maintenance to digestible energy consumed DE%= digestible energy expressed as a percentage of gross energy (Table S3)	(S6)
$REG = [1.64 - (5.160 \times 10^{-3} \times DE\%] + [1.308 \times 10^{-5} \times (DE\%)^2] - (37.4/DE\%)$	REG= ratio of net energy available for growth in a diet to digestible energy consumed DE%= digestible energy expressed as a percentage of gross energy	(S7)
$GE = \left[(NE_m + NE_a + NE_l + NE_p)/REM\right] + (NE_g/REG)/DE\%/100$	See Eqs. (S1-S7)	(S8)

Table S2. Coefficients for estimating feed (IPCC, 2006)

C_{fi}: Cattle/Buffalo (non-lactating cows)	MJ d^{-1} kg^{-1}	0.322
C_{fi}: Cattle/Buffalo (lactating cows)	MJ d^{-1} kg^{-1}	0.386
C_{fi}: Cattle/Buffalo (bulls)	MJ d^{-1} kg^{-1}	0.37
C_a: Ranging	(-)	0.36
C_a: Grazing	(-)	0.17
C_a: Stall	(-)	0
C_g: female cattle	(-)	0.8
C_g: castrates	(-)	1.0
C_g: bulls	(-)	1.2
C_p: Pregnancy dairy cows	(-)	0.1

Table S3. Feed characteristics of the cattle farm

Feed LCI- Standards	DM	DE	CP	GE	P	K	Na
	g kg^{-1}	%	g kg^{-1}	MJ kg^{-1}	g kg^{-1}	g kg^{-1}	g kg^{-1}
Standard concentrate	894	80.94	157.72	16.1	6.89	10	0.2
Maize silage	370	71.9	186	19.1	1.7	10.4	0.1
Alfalfa	894	61.8	203.58	18.2	2.6	24.6	0.2
Wheat Straw	910	48.4	46.15	18.5	0.7	11.2	0.1
Cotton meal	922	81.7	49	21.2	12.40	16.60	0.9
Molasses	730	79.7	75.34	14.7	0.7	51	2.4
Cotton seed	923	62.8	236.18	23.8	5.9	12	0.1

DM= Dry matter content, DE= Digestibility, CP: Crude protein, GE= Gross energy, P= phosphorus, K=potassium, Na= Sodium (Feedipedia (2017), Standard concentrate from statistical data)

The amount of nitrogen (N) excreted by cattle was calculated as the difference between the total nitrogen taken in by the animal and the total nitrogen retained for milk production and growth. N intake is calculated as follows (Eq. (S9)) (IPCC, 2006):

$$N_{intake} = GE/GE_{feed} \times (CP/6.25)/100 \qquad \text{(S9)}$$

where,

N_{intake}= daily N consumed per animal, kg N $animal^{-1}$ d^{-1}

GE = gross energy intake of the animal, based on Eq. (S3), and IPCC (2006) constants, MJ animal^{-1} d^{-1}

GE_{feed} = gross energy for dietary per kg of dry matter, MJ kg^{-1}, from Table S3.

CP = crude protein content of diet, g kg^{-1}

6.25 = conversion from kg of dietary protein to kg of dietary N, kg feed protein (kg N)$^{-1}$

The total retention of N is derived from the following equation (IPCC, 2006):

$$N_{retention} = \left[Milk \times \left(\frac{Milk\ P\%}{100}\right)/6.38\right] + \left[WG \times \left[268 - \left(\frac{7.03 \times NE_g}{WG}\right)\right]/6.25\right] \quad \text{(IS10)}$$

where,

$N_{retention}$= daily N retained per animal, kg N animal^{-1} d^{-1}

Milk = milk production, kg animal^{-1} d^{-1} (applicable to dairy cows only)

Milk P% = percent of protein in milk (input data to the model while it can be calculated as [1.9 + 0.4 * milk fat%], where milk fat% is an input, assumed to be 4% for cow milk.

6.38 = conversion from milk protein to milk N, kg Protein kg N^{-1}

WG = weight gain, kg d^{-1}

268 and 7.03 = constants, dimensionless (NRC, 2001)

NE_g = net energy for growth, calculated from Eq. (S3), MJ d^{-1}

Phosphorous (P) and Potassium (K) retention in milk and tissue are estimated using the following coefficients in Table S4.

Table S4. P and K retention in milk and tissue of cattle (WUM, 2009)

	Weight	P	K
Milk	600	1	1.6
Growth		7.4	2

Once gross energy (GE) intake and its fractional digestibility are estimated, the VS excretion rate is estimated as (IPCC, 2006):

$$VS = \left[GE_{feed} \times \left(1 - \frac{DE\%}{100}\right) + (UE \times GE)\right] \times \left[\left(\frac{1 - ASH}{18.45}\right)\right] \quad \text{(S11)}$$

where,

VS = volatile solid excretion d^{-1} on a dry-organic matter basis, kg VS $animal^{-1}$ d^{-1}

GE_{feed} = gross energy intake, MJ d^{-1}

DE% = digestibility of the feed in percent (e.g. 60%)

UE = urinary energy as fraction of GE (assumed to be 0.04*GE for most ruminant while as the amount of grain in diet increases, this fraction reduces to 0.02)

GE_{feed} = gross energy for dietary per kg of dry matter, MJ kg^{-1} (Table S3)

ASH = ash content of manure (assumed to be 0.08 for cattle)

From Sommer et al. (2004), the degradable and non-degradable volatile solids (VS_d and VS_{nd}) are calculated as:

$$VS_d = VS \frac{E_{CH4act}}{E_{CH4pot}} \quad \text{(S12)}$$

where,

VS= total VS in the manure (g),

E_{CH4act} = achievable emission of CH_4 during anaerobic digestion (g kg^{-1} VS) (Table S5)

E_{CH4pot} = potential CH_4 yield of the manure (g kg^{-1}VS) (Table S5)

Table S5. Parameters and coefficients for manure characteristics

Urinary energy	% of GE	2-4	(IPCC, 2006)
Ash content	%	8	(IPCC, 2006)
Volatile solids content [1]	g VS g TS^{-1}	0.68, 0.698, 0.726	(Rotz et al., 2016)
Actual CH_4	g CH_4 kg^{-1} VS	0.2	(Sommer et al., 2004)
Potential CH_4	g CH_4 kg^{-1} VS	0.48	(Sommer et al., 2004)
Carbon content	% of OM	45.37	(Bianchi et al., 2008)

[1] Values for lactating cows, dry cows, heifers

Non-degradable VS is calculated as follows (Aguirre-Villegas et al., 2014):

$$VS_{nd} = VS - VS_d \quad \text{(S13)}$$

TAN is calculated by the following equations (Vonk et al., 2016):

$$N_{dung} = N_{tot} * (100 - DE)/100 \quad \text{(S14)}$$
$$N_{Urine} = N_{tot} - N_{dung} \quad \text{(S15)}$$
$$TAN\ (N_{urine}) = (N_{tot} - N_{dung}) + (N_{mineralization} * N_{tot}) \quad \text{(S16)}$$
$$N_{org} = N_{dung} - (N_{mineralization} * N_{tot}) \quad \text{(S17)}$$

where,

N_{dung}= total nitrogen excreted in dung (kg farm^{-1} d^{-1})

N_{tot} = total nitrogen excretion (kg farm^{-1} d^{-1})

TAN = total ammoniacal nitrogen (kg farm^{-1} d^{-1})

$N_{mineralization}$= percentage of mineralization of organic N in manure (%), (10% of N_{org} during liquid storage and in the animal house (Beline et al., 1998; Vonk et al., 2016).

2.2. Description of process-based analysis

1.2.1. Collection and housing

Table S6 lists approximate bedding requirements for a cow of 680 kg of body weight.

Table S6. Bedding volume for 680 kg of BW

Manure solids	kg d^{-1}	1.395	(Lorimor et al., 2004)
Sand	kg d^{-1}	20	(Gooch et al., 2003)
Organic	kg d^{-1}	1.395	(Lorimor et al., 2004)
Straw	kg d^{-1}	2	(Lorimor et al., 2004)
Chopped bedding	kg d^{-1}	1.8	(Lorimor et al., 2004)

Generally, the average fuel consumption of a tractor is calculated using the following formula (Grisso et al., 2004). The same approach was used for fuel consumption during manure collection.

$$Q_{avg} = 0.223 \times P_{pto} \quad \text{(S18)}$$

where,

Q_{avg} = average gasoline consumption, L h^{-1}

P_{pto} = maximum PTO power, kW

The CO_2 emission factor of diesel was considered to be 3.17 kg CO_{2eq} MJ^{-1} (Daneshi et al., 2014).

$$NH_{3\ emission\ in\ housing} = \left(TAN + N_{org} \times N_{mineralization}\right) \times N_{tot} \times housing\ share \times EF\ NH_3 \times 17/14 \quad \text{(S19)}$$

where,

$NH_{3\ emission\ in\ housing}$= ammonia emission (kg NH_3 d^{-1})

N_{org} = organic N fraction in manure (Eq.(S17))

$N_{mineralization}$= fraction of N_{org} that mineralized, (10%)

Housing share= share of housing systems per livestock category (%), user defined data

EF NH_3= ammonia emission factor (% of TAN) for animal housings (Table S7)

17/14= conversion factor from NH_3 to NH-N based on molecular weight

$$N_2O_{emissions} = \left(N_2O_{\ direct+}N_2O_{indirect}\right) \times {}^{44}/_{28} \quad \text{(S20)}$$

$$N_2O_{emissions} = N_{tot} \times EF\ N_2O \times {}^{44}/_{28} \quad \text{(S21)}$$

where,

$N_2O_{emissions}$ = nitrous oxide emissions (kg N_2O d^{-1})

$N_2O\text{-}N_{direct}$ = direct nitrous oxide emission (kg N_2O-N d^{-1}) from manure management process,

$N_2O\text{-}N_{indirect}$ = indirect nitrous oxide emission (kg N_2O-N d^{-1}) following atmospheric deposition of NH_3 and NO_x from manure management

N_{tot} = the total N of manure (kg N farm d^{-1})

EF N_2O = the emission factor adapted from IPCC (2006) (kg N_2O kg N^{-1})

44/28 = conversion factor from N_2O-N to N_2O.

Direct N_2O emissions from livestock manure are calculated as follows (Vonk et al., 2016):

$$N_2O - N_{emissions-direct} = N_{tot} \times EF\ N_2O_{\ direct} \times MMS\ share \quad \text{(S22)}$$

where,

$N_2O_{emission\text{-}direct}$= direct nitrous oxide emission (kg N_2O-N $farm^{-1}$ d^{-1})

N_{tot} = total nitrogen (kg N $farm^{-1}$ d^{-1})

EF $N_2O_{\ direct}$ = emission factor for manure management system in kg N_2O-N kg N^{-1} influent

MMS share= share of manure management method (%)

To estimate indirect nitrous oxide emission, the sum of total NH_3 and NO_x emissions from manure management process are multiplied with an emission factor (Vonk et al., 2016).

$$N_2O_{emission\text{-}indirect} = (NH_3 \times 14/17 + NO_x \times 14/30) \times \mathrm{EF}\ N_2O\ \mathrm{indirect} \tag{S23}$$

where,

$N_2O_{emission\text{-}indirect}$= indirect nitrous oxide emission (kg N_2O-N farm^{-1} d^{-1}) due to atmospheric deposition of NH_3 and NO_x within all manure management system

NH_3 = ammonia emission (kg NH_3 farm^{-1} d^{-1}) within all defined MMS

14/17= conversion factor from NH_3 to NH_3-N

NO_x = nitrogen monoxide emissions (kg NO_x) within all defined MMS

14/30= conversion factor from NO_x (expressed as nitrogen monoxide) to NO_x-N

EF N_2O $_{indirect}$ = emission factor for indirect emission in N_2O-N kg N^{-1} emitted as NH_3 and NO_x

Table S7. Emission factors of N_2O, NO_x, N_2 and NH_3 in different housing systems (kg kg^{-1} N_{tot}) (IPCC, 2006)

Loose housing	NO_x-N	N_2-N	N_2O-N_{direct}	N_2O-N $_{indirect}$ (per kg N as NH_3 and NO_x)	NH_3 (% TAN)
	0.005	0.025	0.02		
Liquid*	0.002	0.02	0.002	0.01	12.67
Solid	0.005	0.025	0.005		

* For slurry stream an average is taken.

From Chianese et al. (2009), an empirical equation for estimating CH_4 emission to the ambient temperature (R^2 = 0.48) was used to calculate CH_4 emission in free stalls (Eq. (S24)):

$$E\ CH_{4\,floor-free\ stall} = max(0, 0.13 * T) * \frac{A_{barn}}{1000} \tag{S24}$$

where,

E $_{CH4\ freestall\ floor}$ = daily rate of CH_4 emissions from barn (kg CH_4 d^{-1})

T = ambient temperature (^{o} C)

A_{barn} = floor area exposed to manure (m^2)

Following equation was utilized to estimate CH_4 emissions in loose housing system (IPCC, 2006).

$$E\ CH4_{floor-\ loose\ housing} = \mathrm{VS}\ (B_o)\ (0.67)\ MCF \quad \text{(S25)}$$

where,

$ECH_{4\ floor\text{-}loose\ hosing}$ = daily CH_4 emission, kg CH_4 d^{-1}

VS = volatile solids excreted in manure, kg VS

B_o = maximum CH_4 producing capacity for dairy manure, 0.24 m^3 CH_4 (kg VS)$^{-1}$

0.67 = density of CH_4 (Table S11)

MCF = CH_4 conversion factor for the manure management system, % (MCF is limited to a minimum value of 0). In the following, the MCF is modeled as a linear relationship with ambient outdoor temperature in a loose housing system.

$$MCF = 0.0625\, T_a - 0.25 \quad \text{(S26)}$$

where,

T_a = ambient temperature, ^{o}C

Emission of particulate matter production is calculated as follows:

$$PM_{emission} = EF_{PM\ 2.5\ /\ 10} \times housing\ share \times No.of\ animals/1000 \times 365 \quad \text{(S27)}$$

where,

$PM_{\ emission}$ = Particulate matter emission (kg PM y^{-1})

$EF_{PM\ 2.5/\ 10}$ = Particulate matter emission factor of housing system and animal category, kg PM animal^{-1} d^{-1}

1000 = conversion factor from gram to kilogram

365= conversion factor from year to day

Table S8. Emission factors (EF) of particular matter 2.5 and 10 (Vonk et al., 2016)

Housing type	Lactating cow	Dry cow	Heifer
Free & tie stall			
10	80.8	80.8	0
2.5	22.3	22.3	0
Traditional			
10	147.5	147.5	37.7
2.5	40.6	40.6	10.4

143 *1.2.2. Mechanical separation*

Table S9. Electric grid data (Anonymous, 2011)

Electric grid fuels mix	Electricity delivered (%)
Steam	31
Natural gas	26.9
Combined cycle	36
Hydro	5
Diesel/ wind/ renewable sources	0.1
Nuclear	1.1

144 *1.2.3. Anaerobic digestion (AD)*

Table S10. Assumptions and input data to AD module

CAPg - CHP size-electric capacity	kW_{el}	999	Assumed[1]
CHP size- heat capacity	Kw_{th}	1050	Assumed
Electricity use of AD	% of m^3 influent	6	(FNR, 2006; Frey et al., 2013)
CH4 leakage from installation	% of biogas	1	(Hou, 2016; Pardo et al., 2017)
CH4 leakage from CHP	% of biogas	1.5	(Hou, 2016; Pardo et al., 2017)
LHV of methane-mass	MJ kg^{-1}	50	(Masters, 2013)
E_g - Electricity conversion efficiency	%	38	(Lansche and Müller, 2012)
Heat conversion efficiency	%	46	(Lansche and Müller, 2012)
Specific energy of diesel	MJ kg^{-1}	45.6	(Anonymous, 2017)
E_{vs} - Conversion efficiency of digester	%	35	(Møller et al., 2004)
Methane productivity	kg CH_4 kg^{-1} VS	0.35	(Hill, 1984; Møller et al., 2004)
CH_4 yield in AD	%	65	(Jørgensen, 2009)
CO_2 yield in AD	%	35	(Jørgensen, 2009)
F_{run} - CHP running time	%	85	Assumed
Specific heat capacity of manure	KJ Kg^{-1} $°C^{-1}$	2.8	(Nayyeri et al., 2009)
Mesophilic Temperature	°C	38	(Yu et al., 2014)

TS of substrate	%	8	(Jørgensen, 2009)
Energy requirement for H_2S cleaning	kWh h^{-1}	0.4	(Anonymous, 2013)

[1] Assumed for the modeled large scale biogas plant.

The power available in biogas is calculated as follows:

$$P_{bg} = E_{CH4}\,(1 - \alpha)\,(CH4) / (3.6 \times 100) \quad \text{(S28)}$$

where,

P_{bg} = power available in the biogas produced, kWh d^{-1}

E_{CH4} = lower heating value of methane, MJ kg^{-1} (Table S10)

α = biogas leakage rate, %

CH_4= methane production rate, kg d^{-1} (flow rate of volatile solids into digester (kg d^{-1}) × Methane productivity × Conversion efficiency of digester) (Table S10)

3.6 = conversion from MJ to kWh

The amount of biogas that can be converted to electricity is a function of operating time, efficiency of the CHP generator and is formulated as follows (Rotz et al., 2016):

$$ELECT = min\,(24 \times F_{run} \times CAP_g, E_g \times P_{bg} / 100) \quad \text{(S29)}$$

where,

ELECT = electricity produced, kW h d^{-1}

F_{run} = portion of time engine-generator sets are running, % (Table S10)

CAP_g = electric generation capacity, kW (Table S10)

E_g = efficiency of electric generation, % (Table S10)

The heat demand of digester can be calculated considering that the temperature difference of ambient temperature and the digester (different values for summer and winter season):

$$Q = mC\Delta T \quad \text{(S30)}$$

where,

Q = heat requirements, kJ

m = manure mass, kg

C = specific heat capacity of manure, KJ Kg^{-1} $^{\circ}C^{-1}$

ΔT = temperature change (ambient temperature – digester temperature in °C)

The dry matter of AD effluent is calculated as follows (Rotz et al., 2016):

$$Q_e = Q_m - E_{vs}\ Q_{vs} \qquad (S31)$$

where,

Q_e = digester effluent dry matter entering long term storage, kg d^{-1}

Q_m= loading rate of manure dry matter, kg d^{-1}

Q_{vs} = flow rate of volatile solids into digester, kg d^{-1}

E_{vs} = efficiency of volatile solids conversion, % (Table S10)

Table S11. Density values used in the model

Density of diesel	kg L^{-1}	0.83
Density of manure	kg m^{-3}	1000
Density of CH_4	kg m^{-3}	0.67
Density of biogas	kg m^{-3}	1.15
Density of CO_2	kg m^{-3}	1.98

Then, the degradable fraction of the effluent (the volatile solids leaving the digester) is determined as the difference between the VS entering and the VS decomposed in the digesters:

$$VS_d = (B_o\ /\ ECH4_{pot} - E_{vs})\ Q_{vs} \qquad (S32)$$

where,

VS_d = degradable volatile solids in effluent, kg d^{-1}

B_o = actual methane productivity during anaerobic digestion, g (kg VS)$^{-1}$ (Table S5)

ECH_{4pot} = potential methane productivity during storage of the manure, g (kg VS)$^{-1}$ (Table S5)

Accordingly, the non-degradable fraction in the effluent is as follows:

$$VS_{nd} = (1 - B_o / ECH_{4pot})\, Q_{vs} \quad \text{(S33)}$$

where,

VS_{nd} = non-degradable volatile solids in effluent, kg d^{-1}

Water requirement of digester is calculated as follows:

$$\textit{Total influent (kg/d)} = \textit{Total solids of fresh discharge (kg/d)} \times 100/8 \quad \text{(S34)}$$

$$\textit{Required water (kg/day)} = \textit{Total influent (kg/day)} - \textit{Total solids of fresh discharge (kg/d)} \quad \text{(S35)}$$

1.2.4. Composting

Table S12. Differences in physical and nutrient content changes during composting of manure

Composting method	DM[a] (%)	N_{tot}[b] (%)	TAN[c] (%)	N[c] (%)	P[d] (%)	K[e] (%)	C[f] (%)	NH_3[c] (%)	N_2O (kg N_2O kg^{-1} N-tot)[g]
In vessel	34	15	65	2.6	1.7	8.2	40	16.7	0.006
Static pile	38	18	70	2.3	1.8	11.2	40	14.9	0.6
Intensive windrow	41	11.6	67	2.9	2.4	15.5	44	7.2	0.1
Passive windrow	45	28	70	3.4	1.7	13.8	48.5	12	0.001

[a] Sommer (2001); Eghball et al. (1997)

[b] Eghball et al. (1997); Michel Jr et al. (2004); Sommer (2001)

[c] Gibbs et al. (2002); Michel Jr et al. (2004); Sommer (2001)

[d] Michel Jr et al. (2004); Sommer (2001)

[e] Eghball et al. (1997); Parkinson et al. (2004); Sommer (2001)

[f] Michel Jr et al. (2004); Sommer (2001)

[g] IPCC (2006)

Table S13. CH_4 conversion factors during different methods of composting (%) (IPCC, 2006)

Annual Ave. temperature	<14	15	16	17	18	19	20	21	22	23	24	25	26	27	28
In vessel	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
static pile	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Intensive windrow	0.5	1	1	1	1	1	1	1	1	1	1	1	1.5	1.5	1.5
Passive windrow	0.5	1	1	1	1	1	1	1	1	1	1	1	1.5	1.5	1.5

1.2.5. Storage

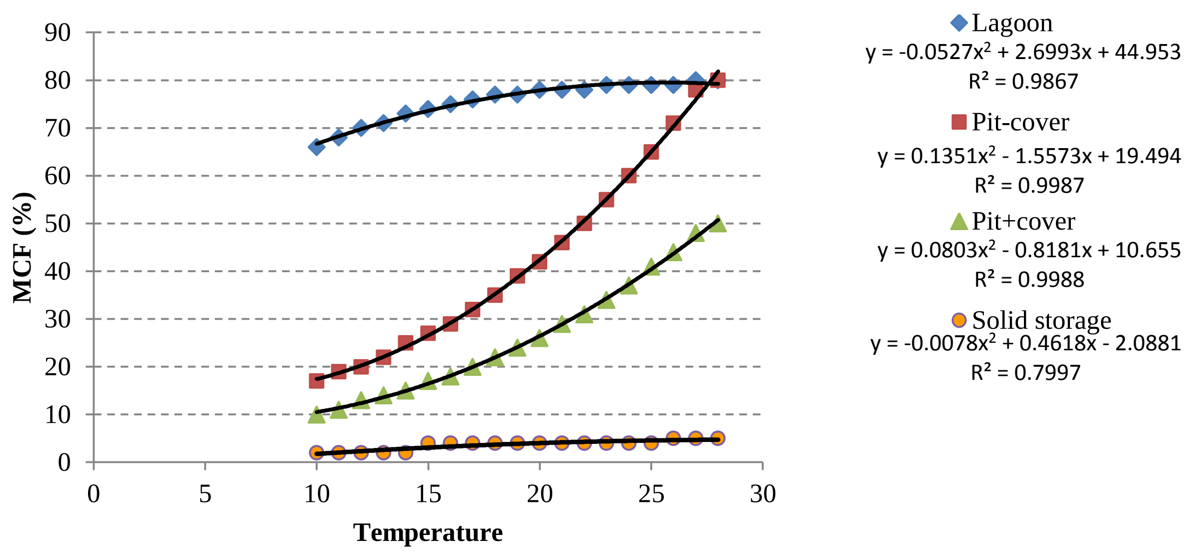


Fig. S1. CH_4 conversion factors during different methods of storage (%)

Table S14. Changes to physical quality and nutrient content of manure for different storage types (%)

Difference to initial	<2 mon					2-4 mon					> 4 mon				
	Lagoon	Pit+cover	Pit-cover	Heap+cover	Heap-cover	Lagoon	Pit+cover	Pit-cover	Heap+cover	Heap-cover	Lagoon	Pit+cover	Pit-cover	Heap+cover	Heap-cover
VS	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
TS	120	120	120	120	120	120	120	120	120	120	120	120	120	120	120
N-tot	50	95	80	85	75	25	92.5	72.5	75	65	30	90	65	65	55
N-min	80	70	80	50	70	70	60	80	60	70	70	60	70	70	85
P	65	20	50	35	40	75	30	60	45	50	60	50	50	60	60
K	60	60	60	50	50	60	60	60	50	50	60	80	70	50	70
C	60	50	55	25	30	70	60	65	35	40	70	65	70	85	70

Chadwick (2005); Rotz (2004); Tittonell et al. (2010)

Table S15. Emission factors (EF) and correction factors (CF) for different storage conditions

Factors	EF	CF of storage type					CF of storage time (month) [e]		
	% NH4-N	Lagoon[b]	P-C*	P+C* [c]	H-C* [b]	H+C*	<2	2-4	>4
NH_3-N	0.15[a]	3	1	0.5	0.2	0	0.5	0.8	1
N_2O-N	0.01[d]	0.1	0.1	2	2[e]	0	0.5	0.8	1

* P-C= Pit-cover, P+C= Pit+cover, H-C= Heap- cover, H+C= Heap+ cover

[a] Clemens et al. (2006); Misselbrook et al. (2015); Rotz (2004)

[b] Misselbrook et al. (2015); Pardo et al. (2017); Rotz (2004)

[c] Amon et al. (2006); Pardo et al. (2017)

[d] Clemens et al. (2006); Pardo et al. (2017)

[e] Petersen et al. (1998)

Table S16. Parameters for manure treatment processes

Parameter	Unit	Value	Reference
sand recovery efficiency	%	87	(Wedel, 2012)
sand recovery efficiency for AD	%	95	(Wedel, 2012)
Sand Sep. electricity use- 87%	kWh m^{-3}	0.15	(Wedel, 2012)
Sand Sep. electricity use- 95%	kWh m^{-3}	0.26	(Wedel, 2012)
Emission factor of electricity	$CO_{2\text{-eq}}$ kWh^{-1}	0.91	(3.3, 2016)
N mineralization in AD	%	43	(Jørgensen, 2009)
P loss in AD	%	0	(Bachmann et al., 2011; Möller and Stinner, 2010; Pardo et al., 2017)
K loss in AD	%	0	(Möller et al., 2010)
Liquid fraction of S/L Sep.	%	85	(Melse and de Buisonje, 2015)
Solid fraction of S/L Sep.	%	15	(Melse and de Buisonje, 2015)
Screw press electricity use	kWh m^{-3}	0.5	(Fleming and MacAlpine, 2003; Møller et al., 2000)
Composting labour use	h m^{-3}	0.175	(Anonymous, 1996)
Composting fuel use	L m^{-3}	1.73	(Anonymous, 1996)
Storage agitation electricity use	kWh m^{-3}	0.25	(Aguirre-Villegas et al., 2014)

2.3. MNRE index

$$MNRE\ (\%) = Q_{nutrients\ recycled}\ /\ I_{nutrients\ excretion} \times 100 \qquad \text{(S36)}$$

2.4. Cost- benefit analysis

$$NPV = -I + \sum_{t=0}^{n} CF_t/(1+r)^t \qquad \text{(S37)}$$

where,

NPV= Net present value

I = Initial capital investment cost

CF = Expected cash flow at time t

r = Discount factor

$$0 = P_0 + P_1/(1+IRR) + P_2/(1+IRR)^2 + \cdots + P_n/(1+IRR)^n \qquad \text{(S38)}$$

where,

P_0, P_1, . . . P_n= Cash flows in periods 1, 2, . . . *n*.

IRR= internal rate of return.

$$BCR = GP/TPC \qquad \text{(S40)}$$

where,

GP= Gross profit

TPC= Total production cost

$$PBP = TIC/GP \qquad \text{(S41)}$$

where,

TIC= Total investment cost

$$\mathrm{PI} = (\mathrm{TR}/(1+\mathrm{r})^{\mathrm{n}})/TIC \qquad \text{(S42)}$$

where,

TR= Total revenue

n= number of years for which the cash flows is calculated

Table S17. Parameters and coefficients for financial analysis

Maintenance		
(a) building and constructions	%	3
(b) equipment	%	5

Separation using sand bedding	%	10
Duration of investment	y	20
Salvage rate	% of initial cost	10
Life span		
(a) building and constructions	y	20
(b) equipment	y	12
Pre-production costs	% of initial cost	10
Contingencies	% of production cost	10
Administrative costs	% of production cost	5
Insurance cost	% of initial cost	0.5
Tax rate	%	25
Annual discount rate	%	20
Loan rate	%	70
Interest rate on loan	%	17
Loan repayment period	y	5
Loan type	-	Fixed amount

2.5. Description of the case and scenarios

Table S18. Characteristics of the studied farm

Breed	Holstein	No. of cattle	9062
Actual capacity (Lactating)	4500 heads	Milk yield (daily)	42 kg head^{-1}
Milk fat	3.2%	Milk protein	2.7%
LW-Lactating and dry	680 kg head^{-1}	LW-heifer	515 kg head^{-1}
Age at first calving	2.08 y	Fertility rate	0.95
Avg. temperature- summer	22.16 ˚C	Avg. temperature- winter	7.5 ˚C
Housing	Freestall/ Loose	Floor type	covered
Bedding	Sand/organic material	Collection method	Vacuum machine/blade and loader
Collection capacity	20 m^3	Transportation	Truck
Transport capacity	20 m^3	Diesel fuel consumption	0.16 L m^{-1}
Distance to collection site	1 km	Avg. distance to land	30 km
Liquid storage	Pit without cover	Solid storage	Heap without cover
Avg. dimension of pit	90×10×1.7 m	Avg. dimension of heap	12×4×6 m

Table S19. Current market prices of manure products

Market prices	Unit	€[1]
Solid fertilizer	€ kg^{-1}	0.03
Liquid fertilizer	€ kg^{-1}	0.003
Bio-electricity	€ kWh^{-1}	0.09
Heat	€ kWh^{-1}	0.004
Compost	€ kg^{-1}	0.02
Solid manure	€ kg^{-1}	0.02

[1] Euro

2.6. Sensitivity analysis

Sensitivity analyses (SA) were performed for future scenarios and their impact to different indicators regarding the condition change in SA. The SA modeling details are described below:

2.6.1. Passive windrow composting instead of intensive windrow composting

In this sensitivity analysis, the alternative method for composting was considered to be passive windrow composting. This method of composting involves decomposing of material over a long time with no or little agitation of animal manure. Therefore, it consumes less labour and equipment. Although the composting process is slow, it may have less environmental burdens and greater potential for odour reduction (at the expense of anaerobic composting). The results were calculated for scenarios including composting treatment (Sc. 3, 5 and 7 based on Table 1). The alternative coefficients and emission factors are presented in Tables S12 and S13.

2.6.2. Storage of manure in heaps and pits with cover instead of storage without cover

In this sensitivity analysis a covered storage of manure in pits and heaps was assumed. Storage of liquid manure in pit without cover and solid manure in pyramidal heaps with rectangular basis less than 2 m height and 4 m width was considered. A cover (or crust) creates aerobic conditions on the surface and reduces CH_4 and NH_3 emissions but may increase N_2O emissions (Rotz et al., 2016). Total GHG emissions are higher without this cover. The relevant coefficients and emission factors are presented in Table S14.

2.6.3. *Impact of changing some input parameters*

In this type of sensitivity analysis, four major influencing parameters were changed by +10%. These parameters included DM, leakage from biogas installations, electricity price and interest rate and their effects on GWP, nutrients recovery and NPV indicators were analysed and described. To select parameters with strong impact on the results by more than ±1%, a cut-off criterion was adopted.

The nutrients content and GWP were calculated proportionally to DM content changes. In this study, the sensitivity scenarios of the higher and lower DM content were compared with their corresponding treatment scenarios.

However, the impact of greater than 1% leakage may not be so realistic in large scale AD plants, the change of this parameter was analysed and the impacts on GWP and avoided emissions were estimated.

The sensitivity of NPV affected by changing electricity price and interest rate by ±10%, were discovered. The price of electricity from CHP is a tariff set by the Ministry of Power in Iran (70% loan on green energy projects). The results of this sensitivity analysis may be helpful for policy makers to see the effect of changing electricity tariffs on biogas projects from the farmers' perspective.

3. Results

3.1.Mass balance of nutrients

Table S20. Estimated dairy manure properties and characteristics

Properties	Cow subcategories (kg animal^{-1}d^{-1})			Total amount (kg d^{-1})
	Lactating cows	Dry cows	Heifers	
N intake via feed	0.64	0.17	0.22	3,963.22
N retention milk and growth	0.21	0.00	0.01	1,041.46
N excretion	0.43	0.17	0.21	2,931.06
P intake via feed	0.08	0.02	0.02	456.57
P retention milk and growth	0.04	0.00	0.00	216.60
P excretion	0.04	0.02	0.02	239.97
K intake via feed	0.29	0.11	0.18	2,132.10
K retention milk and growth	0.07	0.00	0.00	319.61
K excretion	0.23	0.11	0.18	1,812.50
VS	6.57	1.86	2.56	41,795.66
VS_d	2.74	0.77	1.07	17,414.86
VS_{nd}	3.83	1.08	1.49	24,380.80
C excretion	3.81	1.08	1.49	24,241.48
TS	9.66	2.66	3.53	60,548.80
TME	80.48	22.19	29.40	504,573.35
TAN fraction	0.62	0.74	0.73	0.70
C:N ratio	8.79	6.47	7.13	7.46

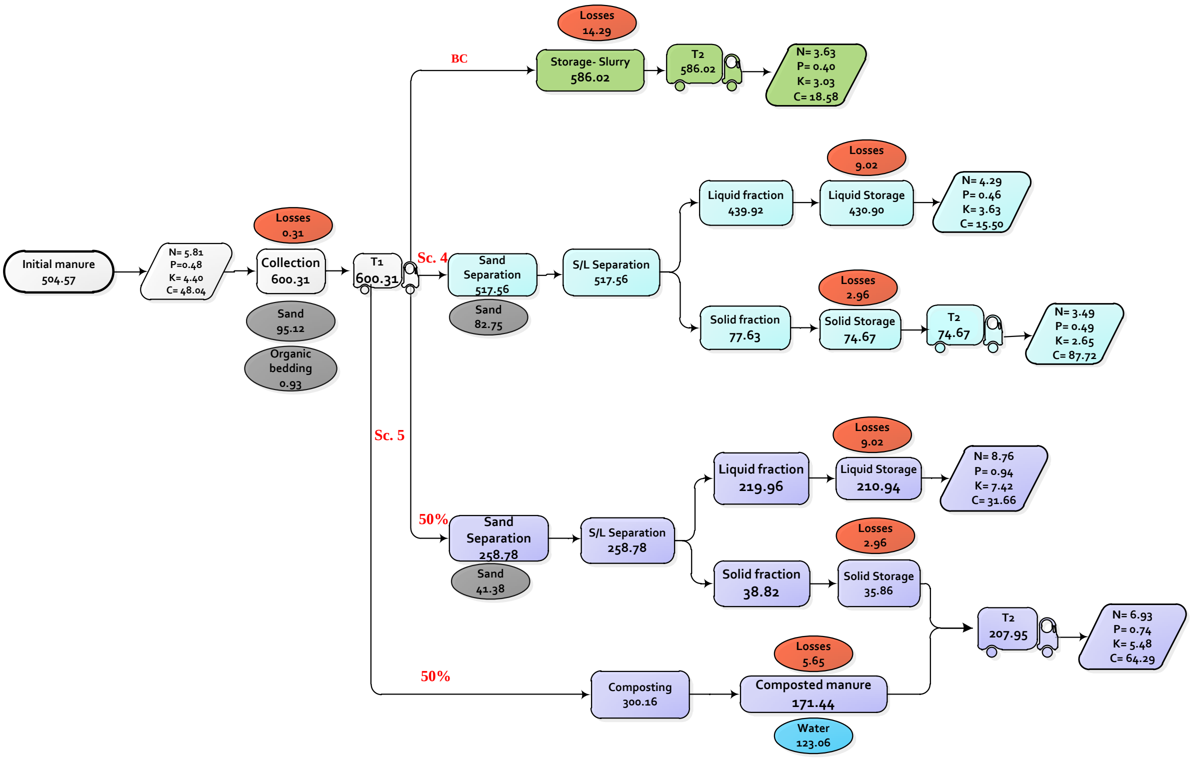


Fig. S2. Mass balance through Sc. 1, Sc. 4 and Sc. 5 (the unit for all values are t d^{-1} except for values in trapezoids are kg t^{-1}).

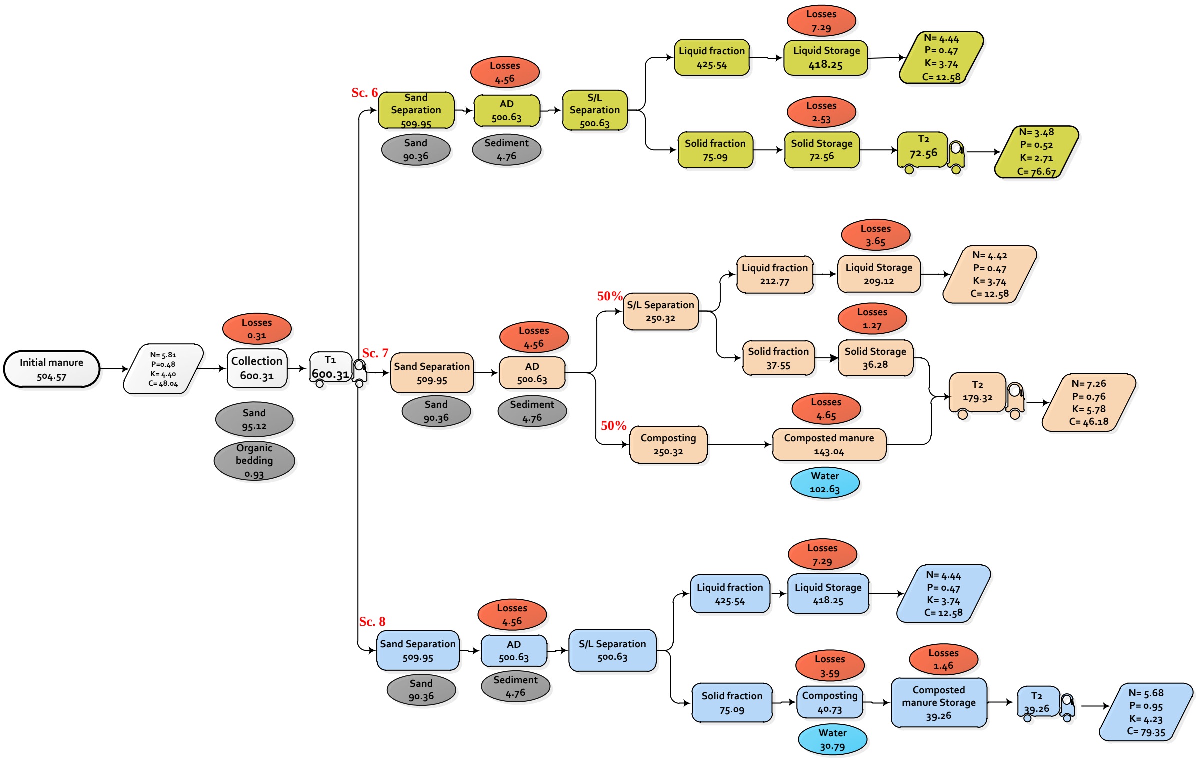


Fig. S3. Mass balance through Sc. 6, Sc. 7 and Sc. 8 (the unit for all values are t d^{-1} except for values in trapezoids are kg t^{-1}).

3.2. Sensitivity analysis results

The detailed results for the sensitivity analysis on the alternative scenarios and on the parameter changes are presented in Figs. S4–S7. As expected, storing manure in covered area caused reducing emissions for all scenarios. The difference between covered and uncovered storages in terms of environmental impacts is depicted in Fig. S4.

Sensitivity to dry matter is presented for GWP and nutrients recovery. As depicted in Fig. S5 (a-c), K recovery and GWP are more sensitive to change of DM. C recovery was found more sensitive in Sc. 7 and 8 where composting was carried out. Nitrogen and phosphorus showed less sensitivity to DM change. It was showed that DM content had impact on environmental outcomes and nutrient contents by Prapaspongsa et al. (2010).

This sensitivity analysis emphasized the importance of leakage losses in environmental performance of biogas production. This is further detailed in the SI (Fig. S6). As expected, greater (smaller) leakage rate (±10%) led to more (less) CH_4 losses and GWP impact as opposed to 1% in its corresponding initial scenario and resulted in less emissions captured during AD process.

The impact of electricity price, loans on renewable energy projects and interest rates were examined (Fig. S7). A 10% change in the r would have the biggest influence on NPV of the projects, with greater impact on Sc. 8. NPV is negatively related to the annual discount rate (r) which indicates an increase in r caused to NPV reduction. NPV is sensitive to the loan rate and electricity price. NPV is more influential in scenarios that involved all manure (100% of digestate) passing through the solid/liquid separation, than scenario 6 with 50% composting. This is due to the efficiency of S/L Sep. in separating slurry into solid fraction.

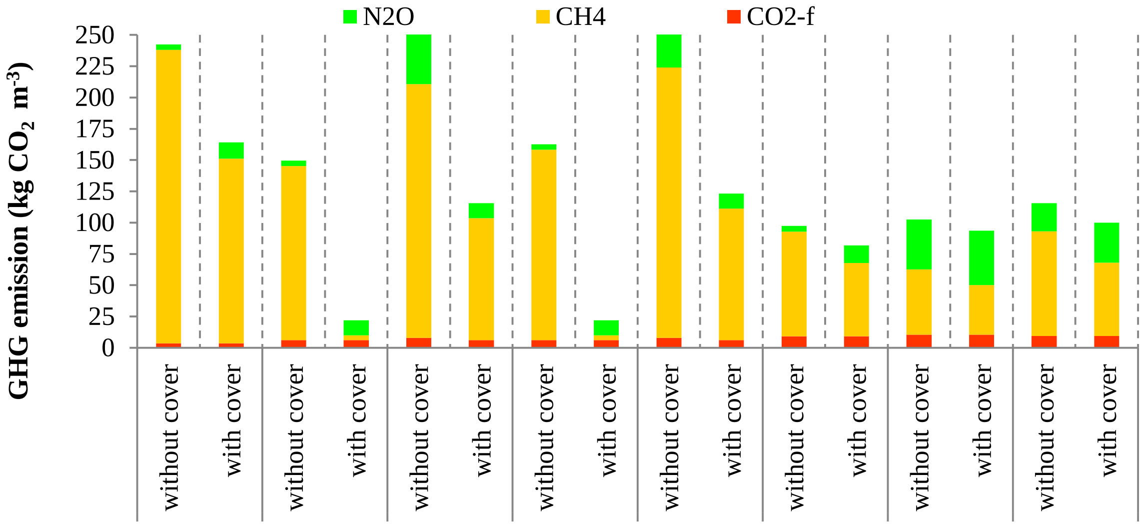


Fig. S4. Breakdown of the covered storage impact for the GHG emissions

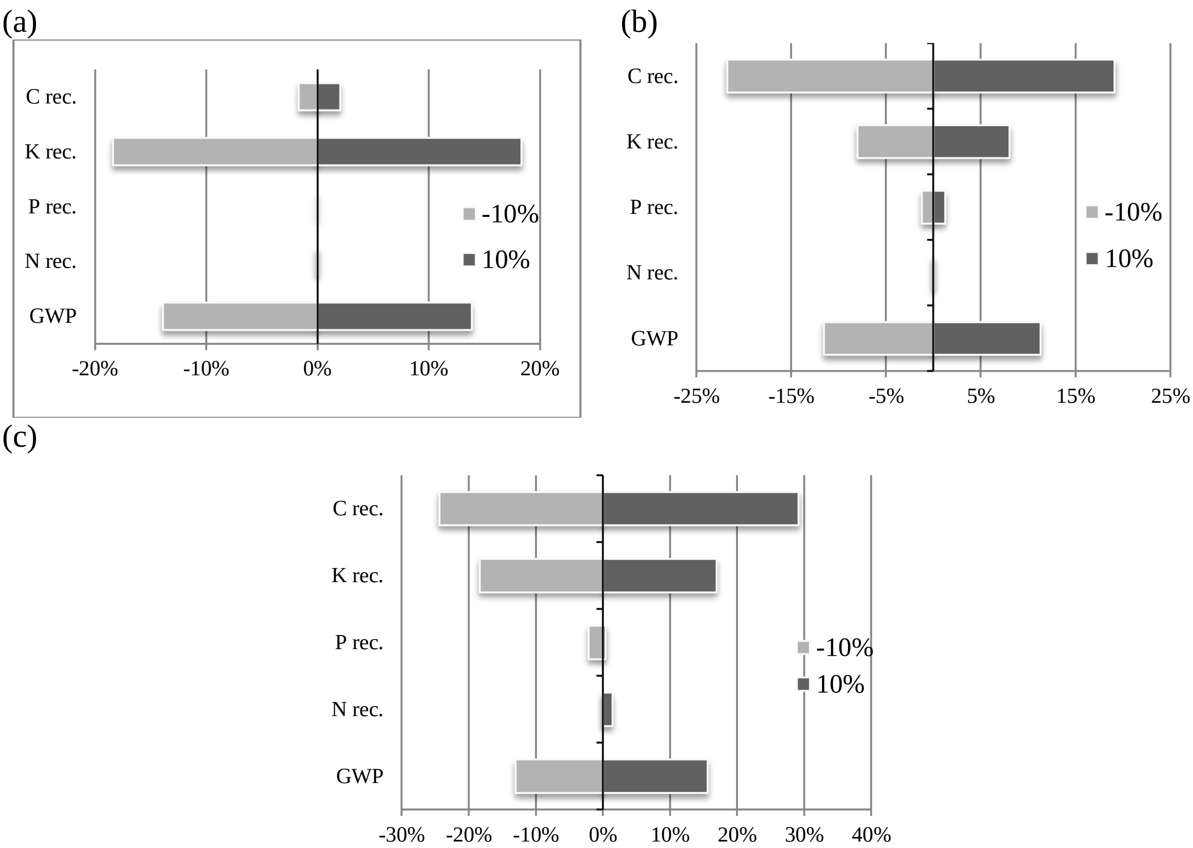


Fig. S5. Changes in GWP, N, P, K and C recovery for a ±10% change of parameters for a) scenario 6 b) scenario 7 and c) scenario 8.

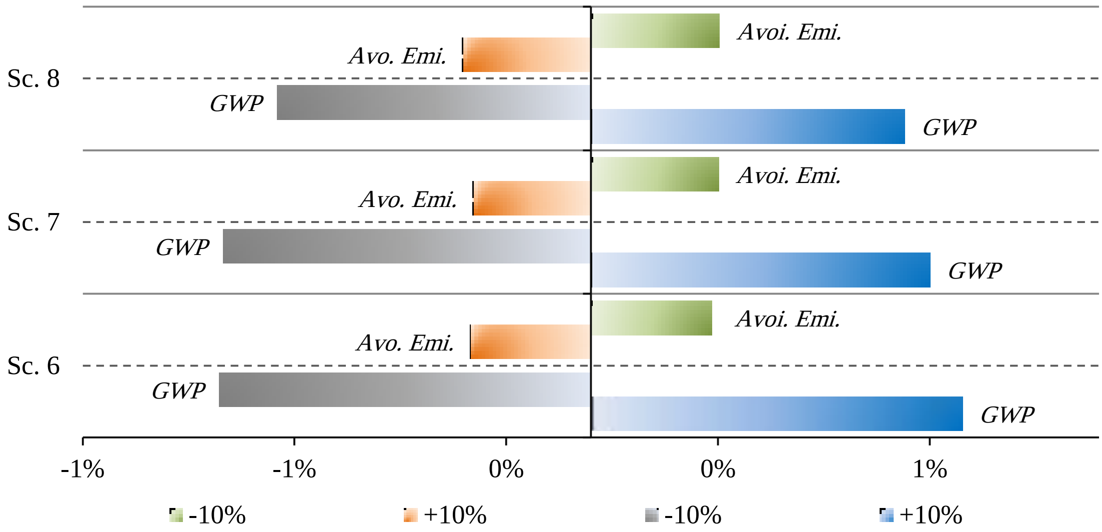


Fig. S6. Changes in GWP and avoided emissions (Avo. Emi) for a ±10% change of leakage for Scs. 6-8.

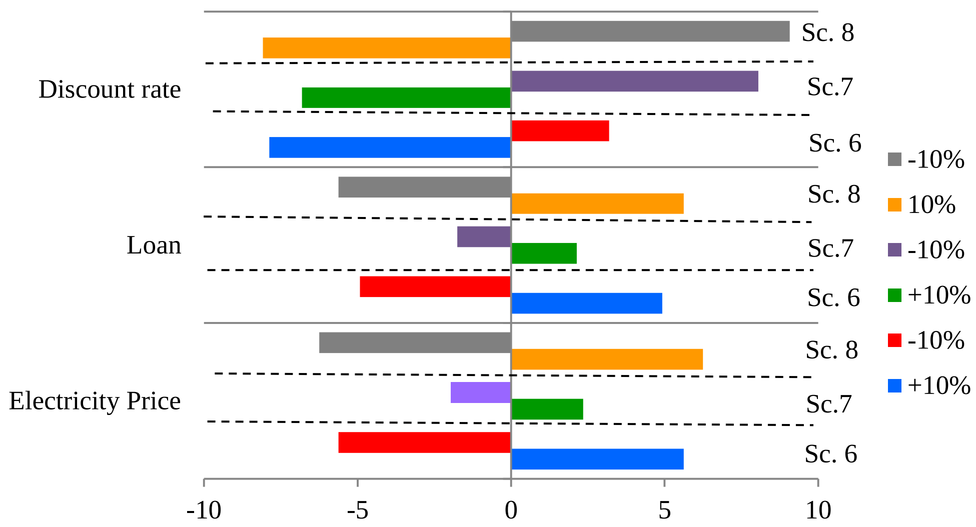


Fig. S7. Changes in NPV for a ±10% change of electricity price, loan and discount rate for Scs. 6-8.

References

3.3, E., 2016. Ecoinvent®Swiss Center for Life Cycle Inventories. http://www.ecoinvent.org/database/ecoinvent-33/ecoinvent-33.html. (Accessed 06 June 2018).

Aguirre-Villegas, H.A., Larson, R., Reinemann, D.J., 2014. From waste-to-worth: energy, emissions, and nutrient implications of manure processing pathways. Biofuels, Bioproducts and Biorefining 8(6), 770-793.

Amon, B., Kryvoruchko, V., Amon, T., Zechmeister-Boltenstern, S., 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agriculture, Ecosystems & Environment 112(2-3), 153-162.

Anonymous, 2011. Statistical Report for Electricity Generation. Iran Power Generation, Transmission & Distribution Management Company. Ministry of Energy (In Persian).

Anonymous, 2017. Energy density. https://en.wikipedia.org/wiki/Energy_density. (Accessed 06 June 2018).

Anonymous, 1996. Windrow composting systems can be feasible, cost effective. https://www.cias.wisc.edu/windrow-composting-systems-can-be-feasable-cost-effective/. (Accessed 06 June 2018).

Anonymous, C.H.a.P.P.C., Catalog of CHP technologies, 2013. Combined Heat and Power Partnership (CHP), Catalog of CHP technologies. https://www.epa.gov/sites/production/files/2017-09/documents/catalog-of-chp-technologies-section-7.pdf. (Accessed 06 June 2018).

Bachmann, S., Wentzel, S., Eichler-Löbermann, B., 2011. Codigested dairy slurry as a phosphorus and nitrogen source for Zea mays L. and Amaranthus cruentus L. Journal of Plant Nutrition and Soil Science 174(6), 908-915.

Beline, F., Martinez, J., Marol, C., Guiraud, G., 1998. Nitrogen transformations during anaerobically stored 15N-labelled pig slurry. Bioresource Technology 64(2), 83-88.

Bianchi, S.R., Miyazawa, M., Oliveira, E.L.d., Pavan, M.A., 2008. Relationship between the mass of organic matter and carbon in soil. Brazilian Archives of Biology and Technology 51(2), 263-269.

Chadwick, D., 2005. Emissions of ammonia, nitrous oxide and methane from cattle manure heaps: effect of compaction and covering. Atmospheric Environment 39(4), 787-799.

Chianese, D., Rotz, C., Richard, T., 2009. Simulation of carbon dioxide emissions from dairy farms to assess greenhouse gas reduction strategies. Transactions of the ASABE 52(4), 1301-1312.

Clemens, J., Trimborn, M., Weiland, P., Amon, B., 2006. Mitigation of greenhouse gas emissions by anaerobic digestion of cattle slurry. Agriculture, Ecosystems & Environment 112(2-3), 171-177.

Daneshi, A., Esmaili-Sari, A., Daneshi, M., Baumann, H., 2014. Greenhouse gas emissions of packaged fluid milk production in Tehran. Journal of Cleaner Production 80, 150-158.

De Vries, J., Vinken, T., Hamelin, L., De Boer, I., 2012. Comparing environmental consequences of anaerobic mono-and co-digestion of pig manure to produce bio-energy–a life cycle perspective. Bioresource Technology 125, 239-248.

Eghball, B., Power, J.F., Gilley, J.E., Doran, J.W., 1997. Nutrient, carbon, and mass loss during composting of beef cattle feedlot manure. Journal of Environmental Quality 26(1), 189-193.

Feedipedia, 2017. Animal Feed Resources Information System. http://www.feedipedia.org. (Accessed 06 June 2018).

Fleming, R., MacAlpine, M., 2003. Evaluation of mechanical liquid/solid manure separators. http://www.ridgetownc.uoguelph.ca/research/documents/fleming_Separatortesting.pdf. (Accessed 06 June 2018).

FNR, 2006. Handreichung Biogasgewinnung und – Nutzung. – 3. überarbeitete Auflage; Fachagentur Nachwachsende Rohstoffe (In German). https://www.big-east.eu/downloads/FNR_HR_Biogas.pdf. (Accessed 06 June 2018).

Frey, J., Grüssing, F., Nägele, H., Oechsner, H., 2013. Cutting the electric power consumption of biogas plants: the impact of new technologies. Landtechnik 68(1), 58-63.

Gibbs, P., Parkinson, R., Misselbrook, T., Burchett, S., 2002. Environmental impacts of cattle manure composting, Microbiology of Composting. Springer, pp. 445-456.

Gooch, C., Wedel, A., Karszes, J., 2003. Economic analysis of mechanical sand-manure separation of flushed sand-laden dairy manure, 2003 ASAE Annual Meeting. American Society of Agricultural and Biological Engineers.

Grisso, R.D., Kocher, M.F., Vaughan, D.H., 2004. Predicting Tractor Fuel Consumption. Biological Systems, Engineering. Papers and Publications. Paper 164, 553-561.

Hamelin, L., Naroznova, I., Wenzel, H., 2014. Environmental consequences of different carbon alternatives for increased manure-based biogas. Applied Energy 114, 774-782.

Hill, D., 1984. Methane productivity of the major animal waste types. Transactions of the ASAE 27(2), 530-0534.

Hou, Y., 2016. Towards improving the manure management chain. Wageningen University.

IPCC, 2006. Agriculture, forestry and other land use, Guidelines for National Greenhouse Gas Inventories, vol. 4, chapter 10. http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html (Accessed 06 June 2018).

Jørgensen, P.J., 2009. Biogas– green energy, Process, Design, Energy supply, Environment, 2 ed. Faculty of Agricultural Sciences, Aarhus University, Danmark.

Lansche, J., Müller, J., 2012. Life cycle assessment of energy generation of biogas fed combined heat and power plants: Environmental impact of different agricultural substrates. Engineering in Life Sciences 12(3), 313-320.

Lorimor, J., Powers, W., Sutton, A., 2004. Manure Characteristics, Manure Management Systems Series. MWPS-18, section 1, second edition. Iowa State University, Ames, Iowa.

Makara, A., Kowalski, Z., 2018. Selection of pig manure management strategies: case study of polish farms. Journal of Cleaner Production 172, 187-195.

Masters, G.M., 2013. Renewable and efficient electric power systems. John Wiley & Sons.

Melse, R., de Buisonje, F., 2015. Quick calculation tool for manure separation and distribution of mass, minerals and dry matter. http://sdddc.org/en/download/detail-155.aspx. (Accessed 06 June 2018).

Menind, A., Olt, J., 2009. Biogas plant investment analysis, cost benefit and main factors. Malinovska, L., Meote, A., kronbergs, e., Nulle, i.(editors), Engineering for Rural Development, 28-29.

Michel Jr, F.C., Pecchia, J.A., Rigot, J., Keener, H.M., 2004. Mass and nutrient losses during the composting of dairy manure amended with sawdust or straw. Compost Science & Utilization 12(4), 323-334.

Misselbrook, T.H., Gilhespy, S., Cardenas, L.M., Williams, J., U., D., 2015. Inventory of Ammonia Emissions from UK Agriculture, Inventory Submission Report. London, UK.

Møller, H.B., Lund, I., Sommer, S.G., 2000. Solid–liquid separation of livestock slurry: efficiency and cost. Bioresource Technology 74(3), 223-229.

Møller, H.B., Sommer, S.G., Ahring, B.K., 2004. Methane productivity of manure, straw and solid fractions of manure. Biomass and Bioenergy 26(5), 485-495.

Möller, K., Schulz, R., Müller, T., 2010. Substrate inputs, nutrient flows and nitrogen loss of two centralized biogas plants in southern Germany. Nutrient Cycling in Agroecosystems 87(2), 307-325.

Möller, K., Stinner, W., 2010. Effects of organic wastes digestion for biogas production on mineral nutrient availability of biogas effluents. Nutrient Cycling in Agroecosystems 87(3), 395-413.

Nayyeri, M., Kianmehr, M., Arabhosseini, A., Hassan-Beygi, S., 2009. Thermal properties of dairy cattle manure. International Agrophysics 23(4), 359-366.

NRC, 2001. Nutrient requirements of dairy cattle, 7 ed. National Academies Press, Natl. Acad. Press, Washington, DC.

Pardo, G., Moral, R., del Prado, A., 2017. SIMSWASTE-AD-A modelling framework for the environmental assessment of agricultural waste management strategies: Anaerobic digestion. Science of the Total Environment 574, 806-817.

Parkinson, R., Gibbs, P., Burchett, S., Misselbrook, T., 2004. Effect of turning regime and seasonal weather conditions on nitrogen and phosphorus losses during aerobic composting of cattle manure. Bioresource Technology 91(2), 171-178.

Petersen, S.O., Lind, A.-M., Sommer, S.G., 1998. Nitrogen and organic matter losses during storage of cattle and pig manure. The Journal of Agricultural Science 130(1), 69-79.

Prapaspongsa, T., Christensen, P., Schmidt, J.H., Thrane, M., 2010. LCA of comprehensive pig manure management incorporating integrated technology systems. Journal of Cleaner Production 18(14), 1413-1422.

Rotz, C., 2004. Management to reduce nitrogen losses in animal production 1. Journal of Animal Science 82(13_suppl), E119-E137.

Rotz, C.A., Corson, M.S., Chianese, D.S., Montes, F., Hafner, S.D., Bonifacio, H.F., Coiner, C.U., 2016. Integrated Farm System Model: Reference Manual. Vol. 4.3. Pasture Systems and Watershed Management Research Unit, Agricultural Research Service, USDA, United States.

Schils, R., De Haan, M., Hemmer, J., Van den Pol-van Dasselaar, A., De Boer, J., Evers, A., Holshof, G., Van Middelkoop, J., Zom, R., 2007. DairyWise, a whole-farm dairy model. Journal of Dairy Science 90(11), 5334-5346.

Sommer, S.G., 2001. Effect of composting on nutrient loss and nitrogen availability of cattle deep litter. European Journal of Agronomy 14(2), 123-133.

Sommer, S.G., Petersen, S.O., Møller, H.B., 2004. Algorithms for calculating methane and nitrous oxide emissions from manure management. Nutrient Cycling in Agroecosystems 69(2), 143-154.

Ström, G., Albihn, A., Jinnerot, T., Boqvist, S., Andersson-Djurfeldt, A., Sokerya, S., Osbjer, K., San, S., Davun, H., Magnusson, U., 2018. Manure management and public health: Sanitary and socio-economic aspects among urban livestock-keepers in Cambodia. Science of the Total Environment 621, 193-200.

ten Hoeve, M., Hutchings, N.J., Peters, G.M., Svanström, M., Jensen, L.S., Bruun, S., 2014. Life cycle assessment of pig slurry treatment technologies for nutrient redistribution in Denmark. Journal of Environmental Management 132, 60-70.

Tittonell, P., Rufino, M.C., Janssen, B.H., Giller, K.E., 2010. Carbon and nutrient losses during manure storage under traditional and improved practices in smallholder crop-livestock systems—evidence from Kenya. Plant and Soil 328(1-2), 253-269.

Torquati, B., Venanzi, S., Ciani, A., Diotallevi, F., Tamburi, V., 2014. Environmental sustainability and economic benefits of dairy farm biogas energy production: A case study in Umbria. Sustainability 6(10), 6696-6713.

Vonk, J., Bannink, A., van Bruggen, C., Groenestein, C., Huijsmans, J., van der Kolk, J., Luesink, H., Voshaar, S.O., Sluis, S., Velthof, G., 2016. Methodology for estimating emissions from agriculture in the Netherlands. Wageningen University & Research, Wageningen, The Netherlands.

Wedel, A., 2012. Sand-manure separation, Got Manure? Liverpool, NY, USA.

WUM, 2009. Dierlijke mest en mineralen 1990–2008. CBS, IKC-Veehouderij, LAMI, LEI-DLO, RIVM en SLM (In Dutch).

Yu, L., Wensel, P.C., Ma, J., Chen, S., 2014. Mathematical modeling in anaerobic digestion (AD). Journal of Bioremediation and Biodegradation 5(S4), 1-12.