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Greenhouse gas emissions from different pig manure management techniques: a critical analysis

Abstract

Manure management is the primary source of greenhouse gas (GHG) emissions from pig farming, which in turn accounts for 18% of the total global GHG emissions from the livestock industry. In this review, GHG emissions (N₂O and CH₄ emissions in particular) from individual pig manure (PGM) management practices (European practises in particular) are systematically analyzed and discussed. These manure management practices include manure storage, land application, solid/liquid separation, anaerobic digestion, composting and aerobic wastewater treatment. The potential reduction in net GHG emissions by changing and optimising these techniques is assessed. This review also identifies key research gaps in the literature including the effect of straw covering of liquid PGM storages, the effect of solid/liquid separation, and the effect of dry anaerobic digestion on net GHG emissions from PGM management. In addition to identifying these research gaps, several recommendations including the need to standardize units used to report GHG emissions, to account for indirect N₂O emissions, and to include a broader research scope by conducting detailed life cycle assessment are also discussed. Overall, anaerobic digestion and composting to liquid and solid fractions are best PGM management practices with respect to their high GHG mitigation potential.

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Title- Greenhouse Gas Emissions from Different Pig Manure Management Techniques: A
Critical Analysis

Running Title- Greenhouse Gas Emissions from Pig Manure Management

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Abstract- Manure management is the primary source of greenhouse gas (GHG) emissions from pig farming, which in turn accounts for 18% of the total global GHG emissions from the livestock industry. In this review, GHG emissions (N_2O and CH_4 emissions in particular) from individual pig manure (PGM) management practices (European practises in particular) are systematically analysed and discussed. These manure management practices include manure storage, land application, solid/liquid separation, anaerobic digestion, composting and aerobic wastewater treatment. The potential reduction in net GHG emissions by changing and optimising these techniques is assessed. This review also identifies key research gaps in the literature including the effect of straw covering of liquid PGM storages, the effect of solid/liquid separation, and the effect of dry anaerobic digestion on net GHG emissions from PGM management. In addition to identifying these research gaps, several recommendations including the need to standardize units used to report GHG emissions, to account for indirect N_2O emissions, and to include a broader research scope by conducting detailed life cycle assessment are also discussed. Overall, anaerobic digestion and composting to liquid and solid fractions are best PGM management practices with respect to their high GHG mitigation potential.

Keywords: CH_4 ; N_2O ; storage; anaerobic digestion; composting; separation; land application

1 Introduction

Pork is the most widely consumed meat in the world [1]. Due to the rapid economic growth in populous countries such as China and India over the past 20 years, the demand for pork is expected to increase by up to 40 % by 2050 compared with 2010 levels [2]. The size of the total global swine herd has doubled since the 1970s[3], and is expected to increase by a further 25 % by 2030[4]. Meanwhile there is a growing need to reduce greenhouse gas (GHG) emissions to meet national and international emission targets such as the EU Climate Action targets, which aim to reduce GHG emissions to 40% of 1990 levels by 2030[5]. Agriculture contributes about 10 12 % to the total global anthropogenic GHG emissions [6], with piggerys accounting for 18 % of the total global GHG emissions from the livestock industry [1]. There is consequently a growing demand for practical technologies which can reduce the GHG emissions from the agricultural sector. In particular, innovative pig manure (PGM) management techniques are urgently needed for GHG emission mitigation [7].

1.1 *Physical-chemical properties of pig manure*

Varying widely in composition, PGM is characterised by high nitrogen and carbon contents and a variable total solids (TS) content (Table 1). Because pigs are monogastric animals, their manure generated typically has a higher proportion of biodegradable carbon than that of ruminant manure e.g. cattle manure [8]. As a result, PGM has a higher potential to generate CH₄ emissions during management. Due to the differences in farming and manure management practises affecting manure composition, it is necessary to assess the effects of management techniques on GHG emissions from PGM in particular.

Table 1 Range of physical-chemical characteristics of liquid PGM reported by a range of studies and summarised by Xie [9]

1.2 Pig manure management techniques

PGM can be managed in a number of different ways (Figure 1) depending on the pig farming system employed, as well as site specific environmental requirements.

Figure 1 Illustration of collection and management options for piggery wastes

Managing PGM in liquid form is the most prevalent PGM management method in Europe and North America. In Europe, liquid PGM is typically collected from pits beneath slatted floors on which the pigs are housed [10]. The manure contained in these pits is periodically emptied into long term storage pits or tanks. A similar approach is taken in North America, however the manure remains beneath the slatted unit[11]. In the EU, regulations are in place to reduce the impact of land application of manure on water courses and to ensure efficient nutrient cycling on farms. Such regulations stipulate that manure can only be spread on land during spring/summer times, typically when drier weather is likely [12]. As such, manure may be in storage for between 1 and 10 months [11]. Subsequent to storage manure is typically applied to land to realise its fertilizer value. In some cases manure may undergo dewatering processes to generate a liquid and solid fraction [13, 14]. Both fractions are typically then applied to land, however they may also undergo further treatment [15]. In the case of the solid fraction composting is common, while the liquid fraction may be treated by various wastewater treatment processes, for instance aerobic wastewater treatment for removal of nutrients from liquid manure. It should be noted that the use of aerobic wastewater treatment processes is restricted to farms located in areas where land application of PGM poses a threat to sensitive waterways.

Deep litter and open lot systems are also common pig farming techniques[1]. Such systems are perceived to be an improvement on traditional systems in terms of animal welfare [1]. In deep litter systems, PGM and bedding materials become mixed. This results in a solid manure/bedding mixture. This solid mixture is periodically removed from the pig housing unit and stored in piles prior to land application. These piles are typically housed in sheds. In some instance the solid piles will be constructed to allow for passive or active composting to occur during storage. In any case, the ultimate disposal route for this solid mixture is land application [16]. For open lot systems, solid waste is typically collected in a separate manner to any liquid runoff from the site. The treatment applied to the resulting liquid fraction is dependent on site specific conditions, while the solid fractions are typically land applied or composted [17].

1.3 Pig manure and GHGs

The two main GHGs associated with manure management are CH₄ and N₂O. Both are potent GHGs; the global warming potentials of CH₄ and N₂O are 21 and 310 times higher than that of CO₂, respectively [18]. CO₂ emissions from manure management are considered as biogenetic and are neglected when analysing the overall GHG emissions from manure management. The Intergovernmental Panel on Climate Change (IPCC) states that CO₂ emissions from manure management are assumed to be zero because CO₂ photosynthesized by plants is returned to the atmosphere as respired CO₂ [19].

CH₄ is the primary GHG of concern when manure is stored in anaerobic environments. In anaerobic environments, the available carbon in the manure is first converted by acidogenic bacteria to acetate, hydrogen and CO₂. These products can then be utilized by methanogenic archaea to generate CH₄ [20].

N₂O is a significant contributor to overall GHG emissions from PGM management [21]. Ammonium is converted to NO₃⁻ under aerobic conditions and NO₂⁻ undergoes denitrification during anaerobic conditions. Thus N₂O can be generated and emitted in the greatest volume under anoxic conditions [22]. While highly variable and dependant on a range of site specific conditions, less than 1 % of the nitrification end products are N₂O, while approximately 5% of denitrification end products are N₂O [19, 23].

NH₃ emissions are also associated with PGM management and are regularly highlighted in the literature. Although such emissions can have a significant negative effect on local air and water quality, NH₃ is not a GHG. However, NH₃ is an indirect source of N₂O emissions as it can undergo subsequent denitrification. Failure to account for such indirect N₂O emissions can result in the underestimate of net GHG emissions. Prompted by the need to comply with the EU Nitrates Directive (91/676/EEC), Waterframework Directive (2000/60/EC) and National Emissions Ceiling (2001/81/EC) several EU countries (Denmark, Germany, Sweden and the Netherlands in particular) require pig farms to control NH₃ emissions, which can have both positive and negative effects on N₂O emissions (as will be discussed in Section 2.1.3).

2 Manure management techniques and GHG emissions

2.1 Storage of pig manure

To comply with national and regional legislation governing the application of animal manure to land, PGM storage is generally required prior to land application. The duration of storage varies considerably in the EU from 3 months in drier countries like Greece to as much as 10 months in wetter countries such as Finland [11].

The type of GHGs and magnitude of the emissions generated during manure storage are dependent on duration, storage system used, temperature and PGM composition. As presented in Table 2, liquid PGM storage tanks are generally found to be in an anaerobic state, leading to CH₄ and NH₃ emissions (which results in indirect N₂O emissions) and negligible direct N₂O emissions [9, 21, 24].

Table 2 Review of farm-based studies on CH₄ and N₂O emissions from PGM during storage

2.1.1 Effects of temperature on GHG emissions from storage

Temperature plays a key role in determining the GHG emission rates from the handling and storage of liquid and solid PGM. It can also lead to significant differences in reported CH₄ emission rates from storage (and indeed land application) of manure[28]. Figure 2 shows the positive correlation of daily temperature and GHG emissions. CH₄ emission rates over a year generally follow a normal distribution around the maximum annual air temperature [25]. Maximum monthly CH₄ emission can vary by up to 3 orders of magnitude for liquid PGM and 2 orders of magnitude for solid PGM in temperate countries [25, 28]. Sommer et al. [29] found that CH₄ emissions from stored liquid PGM were 0.02 g C hr⁻¹ kg VS⁻¹ when stored at temperatures in the range of 10 °C to 15 °C, while CH₄ emissions were approximately 0.1 g C hr⁻¹ kg VS⁻¹ when stored at 20 °C. Petersen et al. [28] reported that the cumulative CH₄ emissions from stored PGM in summer were over 100 times larger than those in winter. Elsgaard et al. [30] mathematically defined the relationship between temperature and CH₄ emission rates, and found that CH₄ emission rates from cattle manure, PGM and anaerobic digestate were similarly sensitive to changes in temperatures. It is clear therefore that understanding the climatic conditions under which GHG emissions from manure storage (or indeed land application) are measured is crucial when comparing studies.

Controlling manure storage temperature is a technically feasible, albeit expensive (depending on climate) control strategy, and it has been shown to reduce GHG emissions by 21 % compare with uncontrolled manure storage [31].

Figure 2 Daily GHG emissions measured from PGM storage vs daily maximum temperature [Xie [9], Petersen et al. [28], Sommer et al. [29]]

2.1.2 Liquid manure and solid manure storage

Table 2 illustrates that CH₄ emissions from solid manure heaps are similar (in the case of Loyon et al. [27]) or higher (in the case of Husted [25]) than those from liquid storage. The higher CH₄ emissions from solid storage can be explained by the combined effect of the higher temperatures found inside solid manure piles (due to heat released from the initial stages of aerobic decomposition) and the compressed nature of piles resulting in localised partial anaerobic conditions. Both factors promote N₂O and CH₄ generation. Hansen et al. [32] observed that immediately covering piles with airtight material to promote anaerobic conditions resulted in reductions in CH₄ and N₂O emissions by 88 % and 99 % respectively. The reduction in N₂O was attributed to the anaerobic conditions preventing nitrification. The reduction in CH₄ was attributed to lower temperatures measured within the covered pile due to the lack of oxygen within the pile. This study illustrates that covering solid manure piles can reduce overall GHG emissions significantly, despite the promotion of anaerobic conditions.

The duration of storage affects GHG emission rates in uncovered piles, with CH₄ emission typically peaking within the first 30 days and N₂O emissions peaking between 30 and 60 days of storage [32]. Therefore, if the storage duration time is minimized, then it is possible to reduce GHG emission. However, it is difficult to achieve a reduced storage time for 2 reasons;

1. the mandatory closed period for land application in most jurisdictions and
2. the desire of farmers to have farm yard manure (FMY) well decomposed prior to land application.

2.1.3 NH₃ emission mitigation and its effects on GHG emissions from storage

Unlike N₂O and NH₃ losses (which can affect the agronomic value of manure through N losses), CH₄ emissions are not of concern at the farm level. Therefore, management techniques aimed at limiting CH₄ emissions are uncommon [11]. However, due to the negative effects NH₃ emissions may have on local and regional air and water quality, regulations in Europe have been introduced requiring each Member State to limit national NH₃ emissions (National Emissions Ceiling, 2001/81/EC). This has resulted in many pig farms in Europe employing measures to control NH₃ emissions from liquid manure stores [33]. Covering the stored manure with straw reduces NH₃ emissions by creating an aerobic zone on top of storage areas where nitrification can occur. Table 3 summarizes the effects on GHG emissions of covering manure stores with straw. In general CH₄ and NH₃ emissions decrease, and N₂O emissions increase particularly in areas with warm ambient air conditions [21, 28, 34]. The decrease in CH₄ emissions after covering is due to oxidation of CH₄ at the surface of the manure [35, 36], while increased nitrification of NH₃ results in increased emissions of N₂O from the nitrification-denitrification cycle.

Table 3 Effect of covering liquid PGM with straw on CH₄, N₂O and NH₃ emissions found in farm-based studies

However, the net effect on GHG emissions remains unclear and is highly dependent on the temperatures prevailing at the time of storage. The two Danish studies found an overall increase in GHG emissions due to N₂O emissions in summer. In the Canadian study, Laguerre et al. [26] reported a reduction in total GHG emissions when barley straw was applied (8.65 and 2.98 g CO₂ eq day⁻¹ kg animal mass⁻¹ with no straw and with straw respectively). They noted a far higher CH₄ emission abatement (80%) than the Danish studies, and did not detect any significant increase in N₂O emissions. The warmer winter temperatures in the Danish studies led to higher CH₄ and N₂O emission rates, which may explain the lower CH₄ abatement rates and higher N₂O emissions observed. Petersen et al. [28] suggest that due to the varying findings in relation to the effect of straw covering, the IPCC guidelines of assuming a 40% reduction in CH₄ emissions when liquid manure stores are covered may be premature.

Acidification is another method used to control NH₃ emissions, allowing it to remain in the soluble NH₄-N form in acidic conditions; it also plays an important role in controlling H₂S (a cause of odour emissions) and CH₄ emissions [37, 38]. Stevens et al. [39] observed that NH₃ emissions from PGM were reduced by 82 % when concentrated sulphuric acid was added to liquid manure. Acidifying manure to a point where pH is below 6 has been shown to effectively reduce CH₄ and N₂O emissions [40], as at this pH both nitrifying bacteria and methanogenic archaea would be inhibited. Acidification of manure reduces particle size, making manure more amenable to subsequent solid/liquid separation [38]. In addition to this, acidification can reduce GHG emissions from subsequent composting (by reducing NH₃ losses) and can boost CH₄ yields during anaerobic digestion [41]. However recently Gómez-

Muñoz et al. [42] found that, in a study where PGM was acidified during storage and subsequently dewatered, acidification increased N₂O emissions from the land application of the liquid fraction of PGM.

An emission inventory study by Brink et al. [43] found that the use of NH₃ control measures in EU agriculture would increase N₂O emissions by 15 %, while having no effect on CH₄ emissions. Despite this, further work is required, in particular to clarify the effects of straw covering of PGM stores on GHG emissions.

Agitation of liquid manure stores prior to land application is a practise undertaken to ensure that manure is well mixed and that solids remain in suspension during transport and land application of manures. It is more common in cattle manure management than PGM management. Agitation is likely to have significant effects on GHG emissions, with the potential for increases in CH₄, N₂O and NH₃ emissions. VanderZaag et al. [44] found that agitation of cattle manure did not affect N₂O or NH₃ emissions but resulted in an increased release of CH₄, as a consequence an increase in GHG emission from storage by 6%.

VanderZaag et al. [35], in a study of cattle manure with straw covers found that spikes in CH₄ emissions due to agitation were offset by subsequent decreases in CH₄ emissions, however significant increases in NH₃ emissions were observed. The current literature is unclear regarding the effects of agitation on GHG emissions, and merits some further research, despite the fact that manure agitation is relatively uncommon during the management of PGM.

2.2 Land application of pig manure

Land application is generally the final utilization step for PGM. The GHG emissions resulting from this practise are highly dependent on upstream management techniques. Therefore,

understanding the processes which control GHG emission rates during and after manure land application is crucial in assessing the effects of changes in upstream management techniques.

2.2.1 Effect of manure carbon and nitrogen contents on GHG emissions

While land spreading of PGM can result in CH₄ emissions directly after application, the presence of highly active methanotrophs in anoxic soils (e.g. wet soils and clay soils) [22] limits any significant CH₄ emissions [21, 45, 46]. Sherlock et al. [47] found CH₄ emission rates of PGM applied to land was 0.08 % of total carbon applied, with the majority (46 %) of CH₄ being released in the initial 6 hours after application. Similarly, Zhong et al. [48] found that 0.1% of the total carbon in composted solid PGM applied to land was released as CH₄.

The low level of CH₄ emissions from land application would suggest that minimizing CH₄ emissions in upstream processes, which in turn would result in higher concentrations of available carbon in land applied manure, would not significantly affect CH₄ emissions i.e. no pollution shifting. However, the available carbon content of the manure can have a significant effect on N₂O emissions. If there is a high organic carbon content in the manure, the microbial soil biomass can rapidly metabolize the carbon, utilizing oxygen in the process and resulting in localized anoxic conditions [49]. Such conditions promote nitrification/denitrification, causing an increased N₂O emissions [50]. This is an issue particularly in soils which have low available carbon [22].

The nitrification/denitrification cycle plays a crucial role in converting inorganic N (primarily NH₄-N) in the PGM applied to land to plant-available nitrogen. The use of nitrification inhibitors (dicyandiamide, 3, 4-dimethylpyrazole phosphate, nitrapyrin, *etc.*) as a means to reduce N₂O emissions from land applied manure has been extensively studied [21, 51].

Nitrification inhibitors are chemical supplements added to manures prior to land spreading to prevent NH_4^+ oxidation to NO_3^- , thereby breaking the nitrification/denitrification cycle and greatly reducing overall N_2O emissions. Chadwick et al. [21] and VanderZaag et al. [51] both highlight that, overall, nitrification inhibitors can effectively reduce N_2O emissions from land application of manures in soil types where aerobic conditions are present (and, therefore, N_2O emissions likely) by between 70 % and 50 %, with a recent study observing reductions of up to 88 % [52]. The effects of such inhibitors have been found to be dependent on soil type and climate [53, 54], with lower efficacy observed in sites where anaerobic conditions were more prevalent [51]. The use of nitrification inhibitors during land application of manure remains niche due to the considerable costs and high application rates necessary with some inhibitors (dicyandiamide and 3, 4-dimethylpyrazole phosphate) as well as health and safety aspects associated with others (nitrapyrin) [51].

2.2.2 Manure application method

The methods of manure application can have a significant effect on the magnitude of GHG emissions from land application. Table 4 summarises the emission rates of N_2O when PGM is applied to land. The table illustrates that direct N_2O emission factors for liquid PGM are typically between 1 % and 5 % of the total N applied to land.

Table 4 Experimentally derived N₂O emission factors (EF) for liquid PGM land application (amended, expanded and updated from Chadwick et al. [21]).

Several recent studies have confirmed that injection resulted in higher N₂O emissions than surface application. According to Velthof and Mosquera [62], direct N₂O emissions from PGM injection (3.6 % of N applied) were significantly higher than the levels found from surface application (0.9 %). The reason postulated for this difference is that injection increases the amount of mineral N entering the soil, which is subsequently available for conversion to N₂O. Injection can also result in an increase in the prevalence of anoxic conditions within the soil. This promotes denitrification, the most significant N₂O emission pathway [21, 22]. Similar results were observed by Velthof et al. [63] and Vallejo et al. [61] when band spreading was compared with soil injection. However, contrary to this, Lovanh et al. [66] found that surface application generated higher N₂O and CH₄ fluxes than row injection, while Weslien et al. [64], Rodhe et al. [34] and Sommer et al. [67] found that the application method had no significant effect on N₂O emissions. Chadwick et al. [21] and Montes et al. [22] point out that the variability in reported effects of the land application type may be linked to the effects of site specific conditions such as weather, soil type, ambient temperature and changes in manure composition.

Injection or incorporation techniques can reduce NH₃ losses due to less direct contact with air. This would reduce the potential for indirect N₂O emissions [21, 66] and preserve the total N content in PGM. Provided that the applied N loading rates are reduced to account for the higher levels of N entering soils from injection and incorporation of manures, the use of injectors and similar application techniques will reduce direct N₂O emissions also. Therefore, injection of manure may result in a net decrease in GHG emitted from the land application of PGM [68].

309

310 2.3 Solid liquid separation/dewatering

311 It is increasingly common for farmers in Europe to employ solid liquid separation techniques
312 when managing liquid PGM [69]. As of 2003, the use of separation technologies varied
313 somewhat across Europe, with Greece (90 % of PGM), Italy (40 % of PGM), and Spain
314 (10 % of slurries) being the countries where it is the most prevalent [11]. Separation of the
315 solid and liquid fractions of liquid PGM may allow, in some instances, farmers to reduce the
316 cost associated with managing PGM. Due to its high nitrogen content, liquid PGM should not
317 be applied to sites which are approaching 170 kg ha⁻¹ year⁻¹ nitrogen limit imposed by the EU
318 Nitrates Directives (91/676/EEC). Additionally in areas with a high soil P, the P content of
319 unseparated PGM can limit its land spreading [32]. Therefore, farmers may be required to
320 travel considerable distances to dispose of PGM [70]. This increases the overall cost and
321 GHG emissions from PGM management [13, 71]. By employing separation the nutrient rich
322 (particularly P) solid fraction of PGM may be spread further from the farm, on lands which
323 have a P requirement. The liquid fraction can be spread near the farm, without risk of
324 breaching P limits. Thus, in some instances, it may reduce costs and potentially GHG
325 emissions [10] associated with PGM management. Other advantages associated with
326 employing separation (as a unit process in a PGM treatment train) are making manure more
327 suitable for subsequent treatment (such as composting or biological wastewater treatment)
328 and a reduction in odour emissions from the manure handling process.

329 Table 5 summarises the results of several studies which looked at the effect of solid liquid
330 separation on GHG emissions from manure storage and land application.

331

Table 5 Summary of the effects of solid/liquid separation on GHG emissions compared with untreated manure

2.3.1 Effects of solid/liquid separation on GHG emissions during storage

While the GHG emissions from the solid and liquid fractions of PGM are dependent on storage techniques (and indeed the efficacy of separation), studies have shown that the storage of solid and liquid fractions of separated manures results in net higher GHG emissions (between 20 %-100 % higher) than that of unseparated manures [15, 72]. This is primarily due to N₂O emissions from the storage of the concentrated separated solid fraction of manure being orders of magnitude higher than the N₂O emissions from liquid fractions or unseparated manure. Fangueiro et al. [72] also noted a 37 % decrease in GHG emissions from the separated liquid fraction relative to the unseparated fraction of PGM, which can be attributed to the lower concentrations of carbon and nitrogen in the liquid fraction resulting in lower CH₄ and N₂O emissions.

2.3.2 Effects of solid/liquid separation on GHG emissions during land application

Several studies assessed the net impact of separation on subsequent GHG emissions from land application and found it had no significant overall effect on GHG emissions [73, 74, 76]. This is despite some studies observing greater N₂O emissions from the separated liquid fraction and the lower emissions from the separated solid fraction. A decrease in N₂O emission factors (on a % N applied basis) has been found when the separated solid fraction of PGM is land applied relative to unseparated PGM [55, 60], due to a reduction in the prevalence of anoxic conditions reducing denitrification.

There are few studies which assess the overall effect of solid/liquid separation on GHG emissions from PGM management. Amon et al. [75] assessed the net GHG emission effects of a range of manure management options. They found that solid/liquid separation of PGM led to an overall 36.7 % reduction in GHG emissions when compared to untreated/raw manure (when all samples were stored and land applied in an identical manner). This was due to reduced N₂O emissions observed during the land application of solid PGM.

2.4 Anaerobic digestion

Anaerobic digestion may mitigate GHG emissions from PGM management. It achieves this by generating renewable energy which replaces fossil fuels, by reducing demand for chemical fertilizers (the production of chemical fertilisers generates significant quantities of GHGs) and by reducing emissions from subsequent manure handling, storage and land application [49]. There are several methods by which anaerobic digestion is applied to manure, as described by Cantrell et al. [77]. The three most common configurations are complete stirred tank reactors (CSTRs), plug-flow reactors and covered lagoons. Of the three mentioned, CSTRs are the most complex but generate the highest CH₄ yields and VS destruction, while covered lagoons are the simplest but can suffer from poor CH₄ yields and VS removal rates [77], and consequently higher GHG emissions. Table 6 illustrates that anaerobic digestion (AD) can significantly reduce GHG emissions from liquid PGM management.

Table 6 GHG mitigation potential of anaerobic digestion of liquid PGM via continuously stirred tank reactor (CSTR) biogas plants

CH₄ utilization is the most widely reported factor contributing to the GHG mitigation potential of AD in PGM management. Kaparaju and Rintala [78] found that employing anaerobic digestion on pig farms could lead to between 17.35 and 94 kg CO_{2 eq} t⁻¹ PGM being mitigated through the use of biogas generated. This study seems to be an outlier. Prapasongsa et al. [79], Xie [9] and De Vries et al. [81] all found that the AD of liquid PGM can result in a mitigation of between 15 and 20 kg CO_{2 eq} t⁻¹ PGM when biogas utilization and reduced emissions from manure storage were considered. Maraseni and Maroulis [80] reported a considerably lower mitigation value of 7.5 kg CO_{2 eq} t⁻¹ PGM in their study of an Australian pig farm however this study did not consider the effect the heat generated from the combustion (via CHP) of the biogas generated would have on GHG emissions.

2.4.1 Digestate storage

GHG emissions from the storage of digestate have been found to be half that of PGM [49]. This is due to AD systems removing between 40 % to 80 % of the VS in PGM [82]. The magnitude of GHG emission avoided from AD is greatly dependant on the ambient temperature (a key factor in determining the GHG emission from manure storage). Table 6 shows an Australian study [80] which calculated that 60-85 % of GHG emission mitigation potential was from a reduction in CH₄ emissions from storage, which was much higher than the 39% found in an Finnish study [78]. The higher annual mean temperatures in Australia results in relatively higher CH₄ emissions during storage. For this reason, the overall GHG emissions from PGM management are reduced to a greater extent than in cooler climates. Wang et al. [84] illustrated that provided storage temperatures of digestate remained < 15 °C

emissions of CH₄, NH₃ and N₂O all remained low, however above this, in addition to CH₄ emissions, significant N₂O emissions occurred.

2.4.2 Land application of digestate

In general, digestate appears to result in lower N₂O emissions (thus overall GHG emissions) during land application than untreated manures.

The primary mechanism responsible for the decrease in N₂O emissions from the land application of digestate is that the reduced VS content in the manure due to AD can result in a decreased microbial activity and, therefore, a reduction in both the rate of nitrification and denitrification [22, 60].

2.4.3 Chemical fertilizer replacement

Due to the mineralization of amino acids, digestate tends to have a higher NH₄-N concentration than that in raw PGM. A review by Möller and Müller [85] on the topic, found that N availability in digestate is increased by 10-25 % relative to untreated manure. This results in a more efficient fertilizer, thereby reducing the amount of chemical fertilizer required by farmers. The specific improvement in N use efficiency between PGM and digestate determines the amount of chemical fertilizer use avoided. Production of such chemical fertilizer results in GHG emissions. As illustrated in Table 6, studies have found that between 9 and 20 kg CO₂ eq t⁻¹ PGM may be avoided through replacement of chemical fertilisers as a result of AD.

2.4.4 Fugitive CH₄ emissions

It should be noted that AD systems can result in fugitive CH₄ emissions. The IPCC estimates that approximately 10 % of the total CH₄ flow can be lost to the atmosphere [19]. In a study of a manure-fed AD plant, Flesch et al. [86] found that 3% of the overall CH₄ generated is lost to the atmosphere. While in a survey of a range of German manure-fed AD plants, [Liebetrau et al. [87]] found the average fugitive emission rate was approximately 6 % of the total flow, with the majority of these emissions occurring during digestate storage. The high global warming potential of CH₄ means that such leaks can drastically affect the GHG mitigation potential of AD, and therefore need to be considered.

2.5 *Composting*

Composting of PGM has been proposed as a means of treating the solid fraction of separated PGM. A supplementary carbon rich bulking agent (generally in the form of straw or sawdust) must be added to the manure prior to composting [89]. Composting is utilized primarily to reduce the mass and volume of manure, reduce the levels of pathogenic organisms in manure and to stabilize the volatile fractions of the manure [89]. Composting results in a reduction in the available C and N content of the manure, thus direct N₂O and CH₄ emissions [90] are reduced in subsequent management stages. However, direct N₂O and CH₄ emissions occur during the composting process, in addition to significant emission of NH₃. Table 7 provides an overview of the emission rates found from composting of PGM.

Table 7 CH₄ and N₂O emissions from selected PGM composting studies

2.5.1 Active and passive composting

Composting can be classified into two broad categories. Active composting is where air is forced through the pile using blowers or periodic mechanical turning (in the case of windrow

composting) to ensure aerobic conditions prevail. Passive composting relies on the temperature gradient within the pile to cause passive aeration [94]. Active composting generates stable compost at a faster rate than passive systems [94]. Poorly aerated compost piles (passive composting for example) generate higher temperatures and may contain anoxic zones, which both contribute to higher N_2O emissions and higher CH_4 emissions observed in passive composting [91]. High NH_3 emissions may occur from actively aerated compost piles [94] and up to 25 % of total N in manure may be lost through NH_3 volatilization and N_2O emissions [92].

2.5.2 Pile management

The composition and management of piles are critical in determining the GHG emissions from composting. Sommer and Møller [95] studied a range of piles which were comprised of varying rates of straw addition from deep litter pig management systems. In piles with a reduced litter density (i.e. higher straw content), no CH_4 or N_2O emissions were detected. Such piles typically have larger pore spaces, improving air circulation [90]. However, both CH_4 and N_2O emissions were detected in the denser, more anoxic piles. It is, therefore, clear that reducing the litter density reduces GHG emissions from manure composting. The effect of litter type has been assessed for such systems. Cabaraux et al. [96] and Nicks et al. [93] found that the use of straw litter resulted in lower GHG emissions than sawdust litter. This increase in GHG emissions was attributed to the higher levels of N_2O (up to 90 %) emissions from sawdust use. Jiang et al. [97] found that adding additional bedding significantly reduced CH_4 emissions [97].

The reduced C and N contents of composted manure results in reduced GHG emissions during land application [48]. The dry nature of compost promotes aerobic conditions, which further contributes the lower GHG emissions when composted is land applied (see Section 2.3.2)

In a study where active and passive aeration of manure (co-composted with wheat straw) was compared, Thompson et al. [90] found that passive aeration generated 300 % more GHG than the liquid storage, while an actively aerated pile generated 30 % less. This indicates that passively managed piles results in increases in GHG emissions, while active aeration may result in GHG emission reductions (compared with storage of liquid manure).

It should be noted that none of the studies cited in this section account for the emissions associated with the growth, harvesting and transport of the wheat straw used as a bulking agent, and therefore the reductions in GHG observed may be offset when supply chain of the system is analysed.

2.6 Aerobic wastewater treatment methods

The aerobic treatment of liquid manure is not a common practise. In some instances aeration of manure storage pits is undertaken to reduce odour, and to reduce NH_3 emissions through the promotion of nitrification [41]. As mentioned in previous sections, PGM is applied to agricultural land to provide a nitrogen rich fertilizer. However, in regions where the addition of nitrogen fertilizer has the potential to cause damage to nearby water courses, this disposal route may be unavailable locally [10]. This is particularly the case in regions in Europe

designated as Nitrate Vulnerable Zones (NVZs) by the EU Nitrates Directive (EC 91/676/EEC)[98]. Aerobic wastewater treatment can reduce the N content of PGM by up to 75 % making it suitable for subsequent land application locally [98]. Typically aerobic treatment comprises of a two stage anoxic –aerobic process [99]. This allows nitrification and denitrification to occur, with the $\text{NH}_4\text{-N}$ in PGM ultimately being converted to N_2 and, to a far lesser extent N_2O . N_2O emissions from aerobic wastewater treatment are as a result of ammonium oxidizing bacteria (AOBs), nitrite oxidizing bacteria (NOB)s and denitrifiers [100].

There have been studies assessing the GHG mitigation effects of undertaking aerobic treatment of piggery waste [27, 98, 101]. Béline et al. [98] found that in comparison to direct land application, aerobic wastewater treatment reduces NH_3 emission from PGM by 30-52 % and reduces total GHG emissions by 55 %, despite causing an increase in N_2O emissions (1 % of N entering system). Similar N_2O emission rates for aerobic treatment were found by Loyon et al. [27]. Vanotti et al. [101] assessed the GHG mitigation potential of an advanced piggery wastewater treatment train consisting of separation, aerobic treatment, chemical flocculation/precipitation, composting (of solid fraction) and disinfection (of liquid fraction) compared to storage and land application of manure. The authors found that the advanced treatment train resulted in a 97 % reduction in GHG emissions, attributed to a 99.6% reduction in CH_4 emissions (which were generated during storage in the baseline scenario) and a 75.2 % reduction in N_2O emissions. However, the authors failed to account for the high, GHG intensive energy required to operate such as system, or the carbon emissions from the materials used in its construction.

These three studies find that aerobic treatment may reduce overall N₂O emissions by reducing the quantity of N applied to land which would subsequently be emitted as N₂O. This is despite N₂O emissions occurring during aerobic wastewater treatment. However, such treatment systems are expensive and, therefore, will only be employed in areas where direct land application of PGM is no longer possible. Therefore, it has limited applicability as a GHG mitigation PGM management technique. Further to cost, the studies above did not consider the GHG emissions associated with the infrastructure, electricity and chemical use necessary in such plants all of which would have a major effect on net reductions in GHG emissions achieved.

3 Effective GHG emission reduction methods and future research directions

3.1 Minimizing GHG emissions from pig manure management

Based on this review of the currently available literature, several practises which minimize GHG emissions from PGM management have been identified. While changes to operation of common management techniques which can reduce GHG emissions have been highlighted in Section 2 (such as cooling of manure stores, direct injection of manures during land application *etc.*), two key management practises have been identified, one to liquid manure management, and one to solid manure management, which can be employed specifically to reduce net GHG emissions during both storage and land application.

3.1.1 Liquid manure storage and AD

Compared to a baseline scenario of uncovered storage followed by land application, AD is a technique which would reduce GHG emissions from liquid PGM management. As discussed in Section 2.4, if PGM undergoes AD prior to storage then, due to the lower level of VS in the resulting digestate, CH₄ emissions, overall GHG emissions from storage will be reduced. In addition, the methane captured and utilized during AD, and the higher N availability in the digestate which increases its substitutability for chemical fertilizer, can further offset GHG emissions. This GHG emission reduction and mitigation is predicated on eliminating fugitive CH₄ emissions during AD.

3.1.2 Solid manure storage and AD

AD may also be applied to solid manures via dry anaerobic digestion. Unlike the more common wet AD systems (which typically treat manures with TS content <15 %), dry AD systems can treat solid manures and the separated solid fraction of manures [88]. While such systems can mitigate GHG emissions, no comprehensive studies have yet been undertaken to examine the effects of dry AD on net GHG emissions from PGM. As the prevalence of solid/liquid separation and pig farming systems which generate solid manure is increasing, there is a need to evaluate this technology to determine its GHG emission mitigation potential.

3.1.3 Solid manure storage and composting

Unmanaged solid manure piles have the potential to generate higher GHG emissions than liquid manure stores. However, Section 2.5 illustrates that well managed, actively aerated composting can achieve major reductions in GHG emissions, compared with unmanaged storage. Similarly, compared with solid manure, composted manure generates lower GHG emissions when land applied. Therefore, composting is an effective PGM management technique for GHG mitigation in deep litter or feedlot farming systems.

3.2 *Future research directions*

While the measurement of GHG emissions from individual manure management processes is relatively mature, this review has identified several areas which merit further attention. In particular:

- The net effect of applying straw covering to liquid manure stores on GHG emissions remains unclear. Further research, potentially studies in varying climatic conditions, would help to determine the scope of this practise.
- There is a lack of consensus on the net effect of storage and subsequent land application of the separated fractions of PGM on land on GHG emissions when compared to unseparated manures. As the manner in which such fractions are stored (e.g. actively managed composting of solid fraction) and land applied will greatly influence the net effect of separation on GHG emissions, broader studies assessing these factors would provide further insight.
- The effect of dry anaerobic digestion of solid manures on net GHG emissions remains largely unstudied. As a batch process, there is greater potential for such a system to release fugitive CH₄ emissions, however the dry, low carbon nature of the resulting digestate could potentially result in reduced GHG emissions from both storage and land application.

In addition to these points, this review has identified several broader issues and areas which should be addressed in future.

3.2.1 Standardising emission units

The variation in units used to report emissions from management techniques means often only qualitative comparisons can be made between techniques. For example CH₄ emissions from manure storage is most commonly reported as kg m⁻³ year⁻¹, but can also be reported as kg head⁻¹ year⁻¹ or g d⁻¹ kg⁻¹ animal mass, while land application emission rates are typically reported as losses expressed as % N or % C applied. Studies on composting, aerobic wastewater treatment and fugitive emissions from AD systems report emissions in a wide range of units. Standardisation of units would allow for insights to be made into the effects of changes to PGM management practices, and contextualise increases or decreases in GHG emissions reported from changes in management practises i.e. whether they have a major impact on a broader scale.

3.2.2 Indirect NH₃ emissions

As an indirect GHG emission, accounting for the effects of NH₃ emissions is crucial when assessing the GHG emissions from PGM management techniques. However, while studies typically do quantify changes in NH₃ emissions due to specific treatments, few apply estimates of the emitted NH₃ likely to be converted to N₂O during transport and deposition, and, therefore, likely underestimate net GHG emissions [48, 55, 59]. The IPCC provides equations which quantify the rate of volatilization of NH₃ from manure, under specific management conditions at specific ambient temperatures. Applying such equations, in lieu of more sophisticated NH₃ deposition and conversion models, in future studies should improve the accuracy in estimating net GHG emission rates.

3.2.3 Life cycle analysis

Work completed thus far has been targeted in its approach; comparing the GHG emissions of a baseline scenario (typically storage and land application) with a management system where an additional techniques is applied. While such studies provide valuable insight in relation to the effect of specific treatments, there is a need for more expansive studies which assess the effects of a range of management options (AD, separation, composting *etc.*) running in parallel and in sequence. Such studies would allow for more direct comparisons of the net effect of specific techniques on the GHG emissions, and would reveal synergistic or antagonistic effects of combining various techniques on GHG emissions.

Life Cycle Assessment (LCA) is a useful tool to account for changes to entire management systems. Crucially, in addition to quantitatively assessing overall changes in direct emissions, it can account for the emissions associated with the construction and the operation of management techniques [102], an area which has not often been addressed in the literature thus far. LCA also provides a framework for quantifying the uncertainty associated with conclusions about net GHG emissions. Several of the studies which have undertaken an LCA approach to quantifying the effects of various PGM management techniques have focused on the use of intensive multi-step treatment trains [81, 102], where the LCA framework can provide a clear mechanism to quantify the effects of many feedbacks between upstream and downstream processes. Other authors have taken an LCA approach to assess the effects of changing various manure management practises on GHG emissions and other areas (such as land use, nutrient cycling *etc.*). These have been reviewed by McAuliffe et al. [103]. These studies further highlight the significant role LCA may play, in addition to broader empirical studies, when assessing changes to unit processes within management systems, and wholesale manure management system changes. The use of LCA methodology to assess the effects of

changes or additions of single unit processes (as opposed to large-scale multi-step treatment trains [81, 102]) would be a useful contribution to understanding the GHG mitigation potential of common manure management techniques.

4 Conclusion

This review provides an overview on GHG emissions and GHG emission mitigation potential of a range of common PGM management practices. This review also identified key research gaps in the literature including the effect of straw covering of liquid PGM stores, the effect of solid/liquid separation, and the effect of dry anaerobic digestion on net GHG emissions from PGM management. The future need to standardize units used to report GHG emissions, to account for indirect N₂O emissions, and to include a broader research scope by conducting detailed life cycle assessment has also been highlighted.

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Table 1 Table 1 Range of physical-chemical characteristics of liquid PGM reported by a range of studies and summarised by Xie [9]

parameters	range
total solids (TS), %	0.78~9.95
total volatile solids (VS), %	0.30~8.16
total Kjeldahl nitrogen (TKN), mg L ⁻¹	1217~6698
ammonium nitrogen (NH ₄ ⁺ -N), mg L ⁻¹	540~3875
chemical oxygen demand (COD), mg L ⁻¹	7138~174300
soluble COD, mg L ⁻¹	1112~74700
5-day biochemical oxygen demand (BOD ₅), mg L ⁻¹	702~23600
total phosphorus (P), mg L ⁻¹	352~2720
total potassium (K), mg L ⁻¹	790~3530
pH	7.01~7.91

Table 2 Review of farm-based studies on CH₄ and N₂O emissions from PGM during storage

CH ₄	N ₂ O	units	remarks	location	Duration	reference
573.3	-	kg CO ₂ eq head ⁻¹ year ⁻¹	solid PGM	Denmark	1 year	Husted [25]
186.9	-	kg CO ₂ eq head ⁻¹ year ⁻¹	liquid PGM	Denmark	1 year	Husted [25]
0.6-9.3	-	g CO ₂ eq day ⁻¹ kg animal mass ⁻¹	liquid PGM stored in uncovered concrete tanks	Canada	3 years, spring to fall only	Laguee et al. [26]
140.7	1.1	kg CO ₂ eq head ⁻¹ year ⁻¹	liquid PGM stored in cold climate	Canada	2 years	Park et al. [24]
49.8-56.9	-	g [C] m ⁻³ d ⁻¹	liquid PGM	France	2 years	Loyon et al. [27]
53.4-50.4	-	g [N] m ⁻³ d ⁻¹	separated solid fraction of liquid PGM	France	2 years	Loyon et al. [27]
6.3-13.4	< LOD	g CO ₂ eq m ⁻³ day ⁻¹	liquid PGM during winter storage	Denmark	58 days	Petersen et al. [28]
1084-1269	0.53	g CO ₂ eq m ⁻³ day ⁻¹	liquid PGM during summer storage	Denmark	45 days	Petersen et al. [28]

Note: Where possible units were converted to allow some cross study comparison. To do this the following assumptions were made; Global warming potential of 21 g CO₂ eq g⁻¹CH₄ and N₂O 310 g CO₂ eq g⁻¹N₂O. < LOD- less than limit of detection.

Table 3 Effect of covering liquid PGM with straw on CH₄, N₂O and NH₃ emissions found in farm-based studies

straw type	CH ₄	N ₂ O	NH ₃	net effect on GHG emissions	location	reference
wheat	9-29 % reduction	0-99 % increase	50-88 % reduction	increase in summer only	Denmark	Petersen et al. [28]
barley	80 % reduction	ns	-	reduction observed	Canada	Laguee et al. [26]
not specified	ns	increase in warm periods	-	-	Denmark	Rodhe et al. [34]

ns= no significant difference.

Table 4 Experimentally derived N₂O emission factors (EF) for liquid PGM land application (amended, expanded and updated from Chadwick et al. [21]).

N ₂ O emission factor (% of N applied)	indirect N ₂ O emission factor (% of N applied)	soil type	application method	duration studied (d)	reference
4.8	-	clay soil	incorporated	58	Bertora et al. [55]
0.12-.44	-	sandy loam	surface applied	22	Chadwick et al. [56]
1.30	0.02 ^a	clay sand	injection	ns	De Vries et al. [57]
2 ^b	0.0175 ^c (% of N lost)	na	na	na	Intergovernmental Panel on Climate Change [19]
0.35-0.37	0.06-0.11 ^a	clay loam	band spreading	28	Meade et al. [58]
1.35 0.46	-	silt clay loam	band spread band spreading and harrowing	15	Rodhe et al. [34]
2.10	0.225 ^a	silt loam	broadcast	90	Sherlock et al. [47]
1.45-3.65	-	silt loam	surface applied	158	Sistani et al. [59]
2.3-3.45	-	loamy sand	injection		
0.06-1.5	-	loamy sand	trailing hose and broadcasting	30	Thomsen et al. [60]
1.60	-	loamy sand	surface applied	199	Vallejo et al. [61]
2.95	-	sandy soil	injected		
3.6	-		injected	1,035	Velthof and Mosquera [62]
0.9	-		surface applied		
7.30	-		incorporated		
4.90	-	sandy soil	surface applied		
6.90	-		placed at 5cm depth	93	Velthof et al. [63]
3.40	-		placed at 10cm depth		
0.27	0.06-0.01		trenching		Weslien et al. [64]
0.38	0.01-0.002	sandy loam	shallow injection	44	
0.31-0.45	0.19-0.04		band spreading		
0.20	-	-	incorporated	68	Zhong et al. [48]
3-4.5	-	clay loam	incorporated	99	Zhu et al. [65]

^a note the indirect N₂O emission factor is based on using the IPCC emission factor for the conversion of NH₃ emissions to indirect N₂O emissions. ^b Tier 1 default emission factor

EF_{3PRP,CPP}. ^c combined emission factor for NH₃ volatilization (EF₃), and runoff/leaching (EF₄). ns= no significant difference, na=not applicable.

Table 5 Summary of the effects of solid/liquid separation on GHG emissions compared with untreated manure

change relative to non-separated manure	CH ₄ emissions	N ₂ O emissions	net GHG emissions
solid fraction storage	decrease [15, 72] ^a	increase [15, 72] ^a	increase [15, 72] ^a
liquid fraction storage	decrease [15, 72]	decrease [15, 72] ^a	decrease [15, 72] ^a
solid fraction land application	-	slight decrease [55] ^a [60] ^b	[72, 73] ^a [74] ^b
liquid fraction land application	-	no difference found in lab study[55] ^b	-
net effect of solid/liquid separation	decrease [75] ^b	decrease [75] ^b	decrease [75] ^b

^a lab scale. ^b field scale.

Table 6 GHG mitigation potential of anaerobic digestion of liquid PGM via continuously stirred tank reactor (CSTR) biogas plants

total kg CO ₂ eq t ⁻¹ PGM mitigated	% GHG mitigated via biogas utilization (heat and electricity)	% GHG mitigated via reduced chemical fertilizer use	% GHG mitigated via lower emissions in storage and land application	location	reference
87.7- 125.6	40-60	28- 33	12-27	Finland	Kaparaju and Rintala [78]
20	100	-.	-	Desk-Based	Prapasongsa et al. [79]
45.3	34.6	65.4	-	Ireland	Xie [9]
68.3	11 (electricity only)	14- 20	60-85	Australia	Maraseni and Maroulis [80]
16†	-	-	-	Desk-Based	De Vries et al. [81]

† this figure describes the net effect anaerobic mono-digestion has on PM management, not the contribution in terms of GHG mitigation potential AD provides.

Table 7 CH₄ and N₂O emissions from selected PGM composting studies

CH ₄ emissions	N ₂ O emissions	compost system	units	bulking agents	duration	scale	
0.002- 0.045	0.044- 0.063	forced aeration	kg m ⁻² yr ⁻¹	wheat straw	21-24 days	farm scale	Thompson et al. [90]
0.023- 0.033	0.346- 0.65	passive					
0.107- 0.164	0.05- 0.098	forced aeration	kg m ⁻² yr ⁻¹	saw dust	85 days	farm scale	Park et al. [91]
0.236	0.249	turn over					
1.09	0.359	passive	g m ⁻³	-	40 days	lab scale	Osada et al. [92]
0.6-385	1.9-71.9	forced aeration ^a					
0.77	1.39	passive	g pig ⁻¹ d ⁻¹	saw dust	ns	farm scale	Nicks et al. [93]
1.58	0.36		g pig ⁻¹ d ⁻¹	wheat straw			

^aAeration rate varied