final report

Feedlot

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Literature review of non-enteric methane emissions from red meat production

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Executive Summary

What the report is about

This report summarises a review of public domain literature regarding the emission of methane from feedlot manure, grazed soils, and pasture manure deposits. Emissions processes are examined, measured emissions summarised, and potential mitigations highlighted.

Who is the report targeted at?

This report is directed at informing Meat and Livestock Australia staff and industry representatives. The literature review is also conducted as an initial step in a experimental research program.

Background

The range of Australian industries is set to encounter considerable pressure for decreased greenhouse gas emissions. While the effect of emissions regulation on red-meat production remains unclear, the lead time to develop the technologies required to make an impact on emissions necessitates that R&D should commence promptly. Even if the agricultural sector is exempted from a carbon pollution reduction scheme, carbon offset opportunities within the sector may be considerable and a potential means of decreasing the costs associated with the introduction of the carbon pollution trading scheme to other sectors.

In order to implement a program of research into greenhouse gas emissions it is necessary to have an accurate grasp of the current state of understanding, and the problems that face the red-meat production sector.

Objectives

The objectives of this project were to conduct a comprehensive review of:

- The manure methane emissions processes and controlling factors likely to operate in Australian systems;
- Available data regarding grazing and feedlot enterprise manure methane emissions in published literature;
- Knowledge gaps with regard to the red-meat production sector;
- The opportunities to mitigate manure methane emissions in these production systems; and
- How published data compares with current inventory calculations techniques.

Conclusions

The carbon and nitrogen cycles are inextricably linked and nitrous oxide emission mitigation cannot be conducted without consideration of methane emissions.

Methane itself is both formed and consumed in manure management systems. Methane formation is favoured by warm (30 to 40°C), moist conditions combined with low oxygen supply, and a degradable organic material. The process also tends to proceed under near neutral conditions in terms of pH. Methane consumption in manure systems, is also favoured by warmer conditions. However, methane consumption requires oxygen and increases where pH is slightly above neutral, where ammonia concentration is low, and phosphorus and potassium are available.

Manure methane emissions from Australian grazing systems are likely to be relatively unimportant.

While data is incomplete, it is likely that nitrous oxide emissions from feedlots (direct plus indirect) exceed methane emissions from their manure management system. International data tends to indicate that the major candidates for emission mitigation are the manure pad plus enteric source, stockpiles, and composting. However the available data is sparse and often inappropriate as a basis for Australian industry decisions.

Variables that contribute strongly to differences in manure methane production are temperature, moisture, diet of the animal, drying conditions, and manure handling. A range of feed-pad mitigations related to the above factors may prove effective and economical.

Composting studies include a range of practices that perform extremely poorly in terms of methane emissions. This is a surprising result considering that from current process understanding, composting should be a mitigation practice. Improved aeration, turning, carbon:nitrogen ratios, windrow covering, and moisture management practices may allow the emissions benefits of composting to be realised.

Inventory enteric emissions estimates for feedlots are within the range of values reported internationally for the sum of enteric and manure emissions from pens but are more than 50 % higher than the mean of published data. On the other hand, the international data on cumulative manure management losses reviewed in this paper tend to exceed those calculated using the Australian inventory method. Australian data from specific sources is still largely lacking.

The greatest weakness of the National Greenhouse Gas Inventory approach is the lack of ability to recognise emission decreases related to improved management.

Recommendations from the manure methane review

Realistic Australian methane emissions data is urgently required for manure on feedpads and compacted stockpiles.

There is a need to identify how methane emissions can be minimised from composting operations through managements that affect aeration, carbon:nitrogen ratio, changing moisture, manipulating free air space, and facilitating methane oxidation.

The magnitude of manure methane emissions at the feed pad is currently unknown. If the recommended methane emissions study reveals this as a substantial source, a range of mitigations are likely to be effective and may provide a good opportunity to decrease emissions — and should be investigated.

Improved emissions algorithms are required to allow a management-responsive inventory calculation approach to be developed (Tier III).

Contents

1 Introduction

Activities across the spectrum of industries are set to encounter considerable pressure for decreased greenhouse gas emissions. The final form of regulatory effects on the agricultural sector remains uncertain. However, the lead time to develop the technologies required to make an impact on emissions necessitates that R&D should commence promptly.

Anecdotally amongst lot-feeders, regulators, and industry advisors, it is well known that a high proportion of manure mass is lost at each step of the manure management system. A proportion of this is known to be in the form of various greenhouse gasses.

Unfortunately, Australia-specific data is scarce, though some initial studies have been completed, including feedlot pad emissions measurements (McGinn et al., 2008, Denmead et al., 2008). More data will be vital to inform debate around the level of exemptions extended to red-meat production from the emissions trading scheme. The collection of this vital data may require a substantial investment from Meat and Livestock Australia, and it is therefore important that the implications of previous studies are well understood and knowledge gaps effectively targeted.

In intensive systems, when considering emissions up to the point of land application, the manure and effluent waste streams are likely to be high emission sources as well as relatively easy targets for mitigation. Emissions mitigations from manure are likely to be facilitated by the fact that manure and effluent is initially concentrated on feed pads, stockpiles, and composting operations. However, baseline emissions data for the effluent and manure management systems of lot-feeds is incomplete — both internationally and for Australian conditions, especially for the more tropical conditions where the majority of feedlots are located.

For extensively grazed systems, management options will be of a completely different nature. Grazing system data is more readily available, though it is dominantly from overseas systems.
Despite the dearth of data, current Department of Climate Change guidelines contain techniques for the calculation of emissions (National Greenhouse Gas Inventory Committee, 2007). In addition to concerns related to the suitability of the protocol, these calculation techniques do not reward successful mitigation managements. This is a major disincentive to innovation. Ultimately, the needs for the red meat industries in this area are to:

- Accurately quantify emissions;
- Quantitatively understand the processes that lead to them;
- Identify the most cost effective points of mitigation, and
- Develop and extend mitigations to producers where required.

The primary objective of this literature review is to provide the first step in this R&D process — a review of currently available data on methane emissions from the red meat industry.

2 Methane emission processes

Methane can be formed or consumed in soils and manures. Substantial research has been conducted into the processes that result in emissions and the conditions that favour them. Much of this research has been conducted in systems other than red-meat production, though it is likely that an understanding of these processes in better known systems (e.g. in soils and effluent treatment) will assist process understanding in the manure management area.

Brief summary of this section:

Methane formation is favoured by warm (30 to 40°C), moist conditions combined with low oxygen supply, and a source of degradable organic material. The process also tends to proceed under near neutral conditions in terms of pH.

At the high methane concentrations that may occur in manure management systems, the biological consumption of methane is also favoured by warmer conditions — but conditions where oxygen is available. Consumption of methane increases where pH is slightly above neutral, where ammonia concentration is low, and phosphorus and potassium are available. Salinity inhibits methane consumption more than it inhibits methane production.

There is a general lack of quantitative methane emission and consumption process research, and the manure emission models represent a very restricted range of conditions.

2.1 Methane formation: methanogenisis

Methane formation from organic matter occurs under strictly anaerobic conditions (redox potentials of < -200 mV), in sulphate and nitrate deficient environments, and may follow a process similar to for glucose breakdown (Equation 1; Saggar et al., 2004).

$$C_6H_{12}O_6 \rightarrow 3CO_2 + 3CH_4 \qquad \textbf{Equation 1}$$

The activities of four different microbial communities are required to produce methane by degrading organic matter mediating four critical processes (Le Mer and Roger, 2001):

1. Biological polymers are hydrolysed into monomers through an aerobic, facultative, or strictly anaerobic process;
2. These monomers are converted into volatile fatty acids, alcohols, hydrogen, and carbon dioxide through facultative or anaerobic processes;
3. These volatile fatty acids are then converted to acetic acid, carbon dioxide, and hydrogen; and
4. Methanogens then convert acetate or hydrogen plus carbon dioxide to methane. Conversion of acetate is regarded as being responsible for two thirds of methane production (Le Mer and Roger, 2001).

The rate of these processes is controlled by a range of factors (Saggar et al., 2004, Le Mer and Roger, 2001):

- *Temperature.* Optimum temperatures for methane formation occur between 30 and 40°C (Le Mer and Roger, 2001). The rate of change of methane production reactions with temperature is different to that for the sulphate and iron reducing microbes with which methanogens compete for degradable organic materials. The relative increase in production rate with a 10°C increase in temperature (the Q_{10} value) for methanogens is about 4.6 times in anaerobic soils, while for sulphate reducers it is 1.6 and for iron reducers it is 2.4 (van Bodegom and Stams, 1999).
- *Availability of the suite of compounds useable as an energy source by methanogens* (step 4 above), including hydrogen together with carbon dioxide, acetate, formate, methanol, methylated amines, and carbon monoxide.

- *Quantity of degradeable organic matter.* Commonly expressed in terms of substrate biological or chemical oxygen demand. This is supported by observations that previous anaerobic digestion of manure slurries decreases methane emission after land application — the easily degradable organic matter is already depleted (Wulf et al., 2002, Wulf et al., 2001). In non-saline rice field soils, a strong correlation has been demonstrated between soil organic matter content and methane formation (Le Mer and Roger, 2001, Garcia et al., 1974). Soil texture (clay content) plays a role here, through the tendency of kaolinite clays to protect organic matter from microbial action (Neue et al., 1990 as cited in Le Mer and Roger, 2001).

- *Gas diffusion rate,* as a control on oxygen supply and the exit of any methane produced. This is influenced by permeability characteristics, moisture content, and in soils, soil texture (e.g. clay has lower permeability that sand). Soil texture will also alter the depth of the oxidising layer in the soil in which methanogenesis is likely to be outweighed by methane consumption (methanotrophism, Le Mer and Roger, 2001). In addition, soils rich in swelling clays tend to have greater methane production. High cation exchange capacities can also be associated with methane production even in the presence of oxygen through the development of micro-scale anoxia (Wagner et al., 1999).

- *Redox potential and oxygen availability*. Redox potentials of < -200 mV tend to maximise methane formation, while methane does not form below a redox potential (Eh) of around -150 to -160 mV in soils (Wang et al., 1993a). This maximum Eh value may actually be related more to the availability of oxygen and a toxic effect on methanogens than to the Eh itself (Yu et al., 2007), since methanogenic activity has been initiated at Eh values up to 420 mV (Fetzer and Conrad, 1993).

- *Competition with denitrification, sulphate reduction, and iron reduction.* A high iron content of soils is known to favour methane formation (Wang et al., 1993a, Joulian et al., 1997). Ferric iron is implicated in decreasing the aerobic zone surrounding plant roots by being re-oxidised, and can increase carbon oxidation into carbon dioxide, one of the feed-stocks to methanogens (Yao et al., 1999, Frenzel et al., 1999). The presence of oxidised iron may continue to support its own microbial reduction delaying availability of carbon substrates to methanogenic bacteria (Wassmann et al., 1993). Competition between methanogens and sulphate reducers for hydrogen (H_2) can result in decreased methane formation in soils with high sulphate content (Jermsawatdipong et al., 1994, Le Mer and Roger, 2001).

- *System moisture*. In soil environments, submersion allows anaerobic conditions to develop, promoting methanogensis, and inhibits aerobic consumption of methane (methanotrophism). Warm, dry conditions where manure is applied to land have resulted in less methane emission than application to wet cold soils (Chadwick et al., 2000). Application to dry soils resulted in emission only of the entrained methane from pre-application in another trial (Wulf et al., 2002). In a study of fen and bog columns, water table level was the dominant control over methane flux in the fen columns, likely through its effect in decreasing methane oxidation rates (White et al., 2008), though pore water chemistry and plant productivity were predominant controls over methane flux in the bog columns.

- *Salinity*. The inhibitory effect of high salinity on methane formation has been observed in a range of rice soils (Garcia et al., 1974). A three to four fold decrease in methane emission was observed from rice soils subjected to an increase in salinity (up to 4 dS m-1, Denier Van der Gon and Neue, 1995). Similar effects have been observed for Australian sub-tropical wetlands (Allen et al., 2007).

- *Acidity*. In anaerobic conditions, a near neutral pH was found to maximise methane production (Wang et al., 1993a) — though it seems likely this may reflect the tendency of soil pH to approach neutrality when subjected to anaerobic conditions (Ponnamperuma, 1972). Similar optimal pH conditions are suggested in a more recent review of Archeal habitats (optimal pH range 5–7.5, Chaban et al., 2006, as cited in Dalal et al., 2008). However, methane production has also proven very sensitive to small manipulations of pH in anaerobic soils, with methane formation maximised in the neutral to slightly alkaline range (Wang et al., 1993a). Data from sixty rice soils suggested no correlation between air dried soil pH and methane emission (Wang et al., 1993b).

These factors are likely to be the most influential on manure methane and grazed soil emissions in Australia, though this is yet to be established for relevant systems.

Much of the research conducted on methane emissions processes has been completed for rice soils, with discussions of plant structural influences on emission and other effects that are not relevant to a consideration of manure management or grazing emissions. Negative relationships have also been observed for 29 soils between methanogenic potential and each of the following soil characteristics: soil conductivity, chlorine content, clay content, and C/N ratio (Garcia et al., 1974, as cited in Le Mer and Roger, 2001).

Factors related to manure characteristics are also know to influence emissions (Saggar et al., 2004):

- *Physical form of manure or mass being degraded.* Slurries, effluents, and solids may produce methane at different rates. It seems likely that this is related to gas diffusion rates, and aeration.
- *Manures from different animals may produce methane at different rates*. For example, beef manures may produce less methane than pig manures (e.g. Chadwick et al., 2000).

2.2 Methane consumption: methanotrophy

A range of microbes oxidise methane in soils under aerobic conditions (Urmann et al., 2009, Hanson and Hanson, 1996), decreasing global emissions by over 50% (Reeburgh, 2003, as cited in Urmann et al., 2009). In addition, the same process results in global aerated soils being a sink for methane emissions — an important consideration in grazing emission inventories. The reaction involved can be simplified as:

$$CH_4 + (2-x)O_2 \rightarrow (1-x)CO_2 + xCH_2O + (2-x)H_2O. \qquad \textbf{Equation 2}$$

The organisms responsible usually fall into one of two major taxonomic and functional groups (Hanson and Hanson, 1996, Urmann et al., 2009):

- Type I organisms (including a subgroup labelled “X”), belonging to *Gammaproteobacteria.* This group appear to be dominant in environments in which methane concentration is low and combined nitrogen and copper levels are relatively high. This group may be analogous to Le Mer and Roger’s (2001) description of “high affinity oxidisers”. High affinity oxidisers are ubiquitous in soils that have not been exposed to high ammonia concentrations (Topp and Hanson, 1991, as cited in Le Mer and Roger, 2001), though this form of consumption contributes only about 10 % of total global consumption (Topp and Pattey, 1997).
- Type II organisms, belonging to *Alphaproteobacteria*. The growth of type II bacteria appears to be favoured in environments that contain relatively high levels of methane, low levels of dissolved oxygen and nitrogen, and limiting concentrations of combined nitrogen and/or copper. Presumably this group has some similarities to the “low affinity oxidisers” described by

Le Mer and Roger (2001). This group are the key oxidation mechanism where methane concentrations are greater than 40 ppm (Le Mer and Roger, 2001), which occur in all soils with a pH higher than 4.4 (Topp and Hanson, 1991, as cited in Le Mer and Roger, 2001). This is the type of methane oxidation that will occur in methanogenic environment (Le Mer and Roger, 2001) and may be important in manure management.

This knowledge of the conditions that promote methane biological methane oxidation presents a range of potential mitigations. However, methanotrophs may also contribute to nitrous oxide formation (Bodelier and Frenzel, 1999), and this may detract (albeit to an unknown extent) from their usefulness in mitigating methane emissions from manure management systems. Variation in methanotrophy has been observed to dominate soil methane emission variation (variation in the ratio of methanotrophy to methanogenesis was most influenced by changes in methanotrophy, Joulian et al., 1997), though the influence on emissions from manure management is not fully evaluated.

Control factors in soils include:

- *Temperature,* though relationships vary. Temperature affects community structure (Mohanty et al., 2007, Borjesson et al., 2004) and reaction rates of enzymes, microbial activity, and possibly growth (Nedwell and Watson, 1995, Borjesson et al., 2004, Nozhevnikova et al., 2001). The Q_{10} values observed ranged from 3–4 (Borjesson et al., 2004). Subsequent research has found that temperature influence in some systems is of minor importance (Urmann et al., 2009). At atmospheric methane concentrations, temperature dependence appears to be inconsistent but usually small (Borken et al., 2006, Castaldi and Fierro, 2005, Tyler et al., 1994, reviewed by Dalal et al., 2008). At temperatures between -5–10°C this factors influence was increased, and Q_{10} values varied between 1.1 and 4.8 for the temperature range 4 to 40 degrees in a range of studies (Park et al., 2005, De Visscher et al., 2001, King and Adamsen, 1992). At elevated methane concentrations, greater temperature dependence was observed (Jäckel et al., 2005, De Visscher et al., 2001). This is probably more representative of the conditions encountered in digesters or some manure management systems.
- *Organic carbon content.* In 22 rice soils, methanotroph population was found to be very strongly related to organic carbon content, though activity was not (Joulian et al., 1997).
- *Available P, K, and N.* Increased available P correlated well with increased methanotrophic activity in rice soils (Joulian et al., 1997). Phophorus addition to rice soils has been observed to decrease methane emissions (Lu et al., 1999). Additions of urea-N has also been observed to stimulate methanotrophic activity (Kruger and Frenzel, 2003), though plant production also increased, increasing methane emission (*via* increased organic matter availability, though the balance was a decrease in methane emission). Similar effects are related to K additions, combined with methanogenic inhibition in flooded rice soils (Babu et al., 2006). Under conditions of elevated atmospheric carbon dioxide, increasing N additions resulted in a reversal of the balance between methane creation and consumption (Xu et al., 2004).
- *Acidity.* Soils with pH greater than neutral tended to have increased methanotrophic activity (Joulian et al., 1997). However, methanotrophs have been observed to function at pH values as low as 3.5 (Benstead and King, 2001, Price et al., 2004).
- *Clay content.* Low clay content was correlated with increased methanotrophy, but not with methanotroph population (Joulian et al., 1997). The relationship with methanotrophy may reflect the relationship between oxygen diffusion rate and clay content.

- *Active Mn,* was found to be positively correlated with methanotroph population, but not activity (Joulian et al., 1997).

- *Water content, gas diffusion rates, and aeration—three related factors.* Anything that decreases the size of aerobic zones is likely to influence methanotrophic activity, since methane oxidation is dominantly an aerobic process (Amaral and Knowles, 1995). Methanotrophy tends to be favoured by the presence of oxidised zones (Le Mer and Roger, 2001), though methane oxidation maximises at field capacity (LeMer et al., 1996, Czepiel et al., 1995). As water content rises beyond field capacity, methane oxidation becomes decreases (Werner et al., 2006). It is proposed (Dalal et al., 2008) that this is due to the limited methane diffusion observed in wet soils (Ball et al., 1997, Del Grosso et al., 2000). However, methanotrophs remain viable through periods of anaerobic conditions, and with the return of favourable conditions will once again become active (Le Mer and Roger, 2001). Indeed, concurrent methane oxidation and methanogenesis can occur in wet soils (Khalil and Baggs, 2005), and net methane oxidation has been observed at water filled pore space values greater than 60% — due to localised aerobic microsites or anaerobic methane oxidation (Khalil and Baggs, 2005, Dale et al., 2006).

  Anaerobic methane oxidation is also possible, and occurs in sediments undergoing hydrogenotrophic sulphate reduction and hydrogen oxidation (Dalal et al., 2008). This is especially likely when acetate is depleted by methanogens (Valentine and Reeburgh, 2000, Dale et al., 2006, acetate is a key methanogenic substrate).

  One of the major limitations to methane oxidation in soil is the rate of diffusion from the air into the soil (Templeton et al., 2006, King and Adamsen, 1992, Tyler et al., 1994, Grant, 1999, Smith et al., 2000). Soil texture, compaction, and bulk density are related to air filled pore space and to rates of gas diffusion (a range of supporting references are cited in Dalal et al., 2008). Methane diffusion rates in soil water are also likely to be much lower than diffusion rates in soil air (Dalal et al., 2008, evidence cited as Whalen et al., 1992, however, no support found in this reference).

  The redox potential of rice-field soils is also influenced strongly by the presence of active Fe and organic matter (Neue and Roger, 1994, as cited in Le Mer and Roger, 2001). The organic matter relationship probably reflects increased respiration of decomposing microbes — and the ease of decomposition of the organic matter is important (Moore and Dalva, 1993).

- *Salinity* more effectively inhibits methane oxidation than it inhibits more general microbial respiration (Saari et al., 2004) or methane production (Denier Van der Gon and Neue, 1995). Dalal et al. (2008) suggests that this is related to moisture stress (Schnell and King, 1996, Saari et al., 2004) or specific chloride and ammonia inhibition (Price et al., 2004).

- *Substrate limitation.* Methane oxidation is an enzyme controlled reaction (as is methane production), and so the availability of the raw materials controls the rate of reaction. For the maximum reaction rate to be achieved, required methane concentrations are in the range ($\mu L\ L^{-1}$) of 5–30 for soil (Gulledge and Schimel, 1998, Price et al., 2004, Saari et al., 2004), 70–800 in wetland soils (Saari et al., 2004, Knief et al., 2006), and 29–84 for landfills (Park et al., 2005). Methane oxidation rates are known to be methane limited (Chan and Parkin, 2001), and increased methane concentrations can overcome some of the rate limitations imposed by gas diffusion rates through increasing soil water contents (Khalil and Baggs, 2005). Concentrating methane may be a means to increase methane oxidation efficiency.

- *Mineral nitrogen and nitrogen fertiliser.* The understanding of methane consumption and nitrogen effects continues to develop, and some authors suggest that there has been an over-emphasis of the inhibitory roles rather than the importance of stimulatory effects (Bodelier and

Laanbroek, 2004). The oxidation of methane and the oxidation of ammonia compete for oxygen (Hanson and Hanson, 1996) or alternative electron donors (Dale et al., 2006). A range of published observations suggest that soil or fertiliser ammonia inhibit methane oxidation through competitive processes (Powlson et al., 1997, Bedard and Knowles, 1989, Gulledge and Schimel, 1998, Bykova et al., 2007, Chu et al., 2007, Veldkamp et al., 2001). Conflicting observations have also been published, where ammonia did not have this effect, or where ammonia applications increased methane oxidation (Dalal et al., 2008), possibly reflecting methanotrophic diversity. Nitrogen fertiliser application tends to inhibit methane consumption by Type II methanotrophs, but enhances consumption by type I organisms (Mohanty et al., 2006). The inhibitory effects of ammonia additions are likely to be short lived due to the rapid transformation of ammonia to nitrate (Dalal et al., 2008, Chu et al., 2007). In rice soils at higher initial methane concentrations, the duration of the initial inhibitory effect can be decreased by increasing initial methane concentrations — and be replaced by stimulated oxidation (Cai and Mosier, 2000).

Nitrite also inhibits methane oxidation (Wang and Ineson, 2003), but this effect is likely to be short-lived in soils due to the normal lack of persistence of nitrite (Dalal et al., 2008). Nitrate has been found inhibitory only in very high concentrations, which likely give rise to osmotic effects (Bodelier and Laanbroek, 2004), or through reduction to nitrite (Wang and Ineson, 2003).

- *Copper and potassium additions.* Copper is an essential methanotroph enzyme co-factor, with Type II methanotrophs active at lower Cu concentrations than Type I methanotrophs (Bedard and Knowles, 1989, Myronova et al., 2006, as cited in Dalal et al., 2008). Evidence is conflicting as to whether K additions can stimulate methanotrophic activity (Babu et al., 2006, Sanhueza et al., 1994).

2.3 Process Algorithms

While it is not always necessary to develop a software based model around processes in order to answer important questions, it is usually beneficial to attempt to place process knowledge in a quantitative framework. Mathematical models can then allow understanding to be effectively validated on a more satisfactory, quantitative basis than is possible without them. Without these mathematical models, research is likely to amount to yes/no answers to merely qualitative research questions. For example, we can go beyond the question “does temperature affect methane emissions?” to “does temperature dependence follow the Arrhenius process model?”. The usefulness of this approach is not limited to the development of simulation models, but allows more intelligent development of process-based mitigations and the relative benefits of various approaches to be estimated.

A mixed empirical/mechanistic model is available for manure management systems, and provides a good basis for this algorithm discussion (Sommer et al., 2004b). The model is designed for European systems and has methane components to cover emissions from housing, storage, digestion, and land application for both cattle and pigs. A major departure from Australian lot-feed systems is the occurrence of the manure materials as slurry. Since moisture content influences methane emissions processes, this is not a trivial difference. However, the underlying process representation is useful, though the parameter values described in the publication probably are not.

Degradeable volatile solids (DVS) are considered to be the major driving force for emissions. This content is modified for residence times and temperatures in the manure management system components. A proportion of the volatile solids are considered to be non-degradeable (NDVS). The ratio of DVS to VS is related to the ratio of total methane emission during anaerobic digestion (B_o):potential methane yield($CH_{4.potential}$; units are g kg^{-1}):

$$\frac{DVS}{VS} = \frac{B_1}{CH_{4.potential}}. \qquad \textbf{Equation 3}$$

The $CH_{4.potential}$ is a quantity that can be estimated from the manure content of fat, carbohydrate, and protein (Symons and Bushwell, 1933, as cited in Sommer et al., 2004b):

$$\mathbf{C_nH_aO_b} \rightarrow \left(\frac{n}{2}+\frac{a}{8}-\frac{b}{4}\right)\mathbf{CH_{4\ potential}}. \qquad \textbf{Equation 4}$$

Sommer et al. (2004) account for temperature dependence by replacing B_0, by b_1 for DVS and b_2 for NDVS, and using an Arrhenius-type relationship (determined in the ambient temperature range from 12-31°C, Khan et al., 1997):

$$\textbf{Emission rate} = DVS \times b_1 \times \exp(\ln A - E \times (1/RT)) + NDVS \times b_2 \times \exp(\ln A - E \times (1/RT)), \qquad \textbf{Equation 5}$$

where emission rate (g methane d^{-1}) is dependent on A (g methane kg^{-1} VS h^{-1}), the Arrhenius parameter, the apparent activation energy (E, J mol^{-1}), the gas constant, and temperature (T in °K).

However, while the Arrhenius relationship is likely to apply over restricted temperature ranges in specific systems (for example Sommer et al. 2004, with a mean annual temperature range of 4 to 7.5°C), this is unlikely to be the case for broader temperature ranges. For example, Farquharson and Baldock (2008) emphasised the poor relationship between nitrous oxide emission and the Arrhenius equation for broader temperature ranges. They attributed this to emissions being the composite temperature response of several processes. In the case of methane emission from lot-feed manure management systems, in addition to individual microbe activity changes with temperature, changed temperature could also alter microbial populations, and change anaerobic zone distribution through influence on respiration rates (DeKlein and VanLogtestijn, 1996).

These composite relationships often result in emissions optima with temperature — rather than a continued increase in emission rates as temperature rises. Temperatures in lot-feed manure management systems can be driven much higher than the ambient atmospheric temperature and it is likely that methane emissions optima will be encountered within the range. In addition, emissions are likely to become de-coupled from the atmospheric temperature (T in Equation 5), and more dependent on manure temperatures in these heat-producing manure managements.

However, broad temperature range methane emission trials appear to be lacking. Temperature optima have not been identified for manure probably because wide temperature ranges have not been investigated. For example the maximum temperatures observed in one of the cattle manure emission trials were only 45°C (Husted, 1994). The Q_{10} values observed in their trial were 2.7 to 10.3, depending on the range of temperature variation. However, a temperature optima for methane emission was observed for pig solid manure, occurring between 35 and 45°C.

Likewise, the model does not account for the other factors listed in Sections 2.1, nor does it account for methane oxidation. Methane oxidation has been observed to be the dominant influence on fluctuations in methane emission from rice soil systems (Joulian et al., 1997). While this omission from representations of highly anoxic systems may be appropriate, the impact may be more pronounced in aerobic manure managements and grazing systems.

3 Methane emissions from manure

The following sections summarise the scant available data regarding methane emissions from manure — deposited in grazing lands or during grain-feeding. In general this data has been collected from overseas or is confounded by a lack of separation of enteric emissions from manure emissions.

Brief summary of this section:

For Australian grazing systems it is likely that mitigating the emission of methane from deposited manure may well be irrelevant:

- *Total methane emissions of around 230 g of CH_4 animal^{-1} day^{-1} have been observed in an Australian grazing system.*
- *Manure emissions may only be around 3% of the level of enteric emissions.*

Even under a reasonably low stocking density, e.g. at 8 ha [450 kg steer]$^{-1}$, enteric methane emissions are likely to dominate methane production and consumption processes in extensively grazed systems.

At first glance, for grain fed beef production, the apparent trend of data is that enteric emissions greatly exceed those emissions from the manure management system (Table 1). However, the studies conducted have not separated these two sources, and have not represented the wet, warm conditions that may lead to maximal emissions in Australian feedlot systems.

Table 1. The apparent trend of the data for methane emissions by source is that enteric+pen manure emissions may dominate. However, note the small number of studies conducted. The pen manure values are reliant on a single study conducted for two 24 hour periods. Moist conditions would accelerate methane emissions from pen surfaces. Data taken from studies summarised in Tables 2 to 5, where suitable, assuming 1 t of manure animal^{-1} year^{-1} where necessary.

	Mean	Minimum	Maximum	No. data points
	Kg CH_4 animal^{-1} (year of emission)$^{-1}$			
Grazing manure	0.4	0.1	1.5	14
Pen enteric + pen manure	74	53	118	5
Pen manure	2.9	2.2	3.5	2
Stockpile	11	3	18	2
Composting	12	1	29	11
Land application of lot-feed manure	Net consumption			1

3.1 Methane emissions from grazed pastures

Methane emissions from grazed pastures should be considered as the product of at least two processes, including methane production and methane emission. Before delving into the detail of the magnitude of these two processes in pastoral soils, a rough appreciation of the magnitude of these processes is important:

- Data from beef grazing around Canberra suggests total methane emissions of around 230 g of CH_4 $animal^{-1}$ day^{-1} (Harper et al., 1999).
- Methane emissions from manure deposited to pastures maybe around 3% of the level of enteric emissions (Hunter Valley dairy data Williams, 1993).
- Methane consumption by soil is around 2 (subtropical grassland) to 13 (temperate grassland) g CH_4 ha^{-1} d^{-1} (Dalal et al., 2008).

Even under a reasonably low stocking density, e.g. at 8 ha [450 kg steer]$^{-1}$ (e.g. subtropical savannah areas, O'Reagain et al., 2009), enteric methane emissions are likely to dominate methane production and consumption processes in extensively grazed systems. Mitigating manure methane emissions may therefore be irrelevant.

Brief summary of this section:

Eneric emissions probably dominate grazed system emissions, exceeding both the potential methane consumption of grazing lands and deposited manure emissions.

3.1.1 Methane production from manure deposited on grazed pastures

Dung deposited during grazing contains methane from enteric processes, but also is a source of methanogenisis — and the conditions of deposited manure are favourable for methane formation (wet, warm, microbially active, with high available carbon). Far more studies have been conducted with cattle than with sheep or other commercially grazed species (Saggar et al., 2004).

These emissions may be insignificant relative to enteric emissions, with one study finding deposited dung emissions of about 0.5 % of the emissions observed from the rumen (Flessa et al., 1996).

Emissions from manure deposited in grazing systems (Table 2) is influenced by:

- *Dung type and animal diet.* Striking differences have been observed between emissions from sheep dung and that from cattle, while generation of methane from deposited cattle manure is confirmed (e.g. Williams, 1993), it appears that much of the methane emissions from sheep manure may be entrained enteric methane (methane formed in the gut of the animal Carran et al., 2003a, as cited in Saggar et al., 2004). The lack of methane generation from sheep dung may be the result of its dryer character, and lower volume than cattle manure. Significant differences have been observed between diary cows, heifers, calves, and steers fed a range of differing diets (Jarvis et al., 1995).
- *Initial moisture content and drying processes.* In addition to the observed relationship between dung type and moisture content and emissions, more direct evidence of the effect of drying is evident. Methane emission rates have been observed to decline rapidly as dung dries (Flessa et al., 1996). A similar drying effect was observed in New South Wales with methane emissions ceasing within 2 (summer) or 3 (winter) days (Williams, 1993). The influence of moisture content is consistent with the process understanding described in sections 2.1 and 2.2 and the influence of moisture contents on gas diffusion and development of anaerobic conditions.
- *Ambient temperature and rainfall,* have been hypothesised as the origin of the high variability in emission rates from some manure pat studies (Yamulki et al., 1999). This is consistent with process understanding and observations of the effect of drying.

- *Period since deposition*. Peak emissions tend to occur within the first few weeks following manure deposition (Saggar et al., 2003b, or days Williams, 1993), with emissions occurring for up to 1.5 months (Saggar et al., 2003b). Sherlock (2003, as cited in Saggar et al., 2004) observed similar behaviour with emissions peaking 7 days after application, declining rapidly, but continuing for a period of 35 days. They suggested that crusting of the dung pat tended to act as a barrier to drying.

Dung pats produce their own micro-environment, and methane production or consumption can occur beneath them (Saggar et al., 2003b). Saggar et al. (2004) highlight the effect of N additions on methane consumption (e.g. Mosier and Schimel, 1991), and points out similar caveats to this as a general conclusion (Kruger and Frenzel, 2003) as noted and expanded further in Bodelier and Laanbroek (2004). It appears that fertiliser nitrogen additions can have a positive effect on methane oxidation rates.

In contrast to Saggar et al. (2003) some studies have indicated that there was little interaction between the dung pat and the soil beneath it, resulting in the soil having no effect on emissions (Jarvis et al., 1995).

Table 2. Methane emissions from grazing lands. Where possible, measurements have been re-calculated to common units.

Environment	Stock	Measurement Period	Dung Character	Methane Emission	Units	Comments	Reference
U.K., poorly drained soil. Permanent grassland.	Dairy Cows at milking	10 days	38.6 C, 2.5 N, 84 water (%)	1702 [1277]	mg CH_4 m^{-2} of pat [mg kg^{-1} manure]	Stock fed concentrates, grass, and clover. Manure applied manually to form a pat. Emissions continued at a low rate to 10 days.	(Jarvis et al., 1995)
	Grazed calves		38.0 C, 2.9 N, 84 water (%)	1655 [1241]	As above	N-fertilized grass. Emissions continued at a low rate to 10 days.	
	Grazed heifers		38.6 C, 2.5 N, 84 water (%)	1143 [857]	As above	Grass-clover	
			38.6 C, 2.5 N, 84 water (%)	423 [317]	As above	N-fertilized grass	
	Grazed beef steers		27.4 C, 1.6 N, 80 water (%)	406 [305]	As above	Grass-clover	
			29.6 C, 1.6 N, 85 water (%)	503 [377]	As above	N-fertilized grass	
			35.7 C, 1.7 N, 82 water (%)	300 [225]	As above	Unfertilized grass	
	Housed dairy cows at milking		41.9 C, 2.5 N, 89 water (%)	716 [537]	As above	Silage+concentrates	
	Housed sheep		39.5 C, 2.7 N, 75 water (%)	598 [449]	As above	Hay+concentrates	
	Grazing dairy cows		32.0 C, 2.5 N, 90 water (%)	2040 [1530]	As above	Fertilized grass+concentrates	
	Rough grazing		37.5 C, 2.6 N, 88 water (%)	922 [692]	As above	Rough upland grazing	
Southern Germany, coarse-loamy dystric Eutrochrept	Cattle grazing	1 days droppings observed for 78 days	86 % water	0.778±0.065	g CH_4 [animal unit]$^{-1}$day^{-1}	Average live-weight was 600 kg, 48.4 animal units ha^{-1}. Perennial ryegrass and common foxtail pastures. Mean dung patch weight of 1.5 kg (applied to 0.7065 m^{-2}), with each animal producing 6kg of fresh moist manure daily.	(Flessa et al., 1996)
North of Sydney, Hunter Valley, cows fed on lucerne and oats	Holstein-Friesian cross dairy cows	Up to 2 days for a dropping		< 2.7	g CH_4 cow^{-1} day^{-1}	Assumes 10 droppings per day. Maximum emissions were only about 3% of enteric emissions, and deposited manure emissions were probably insignificant. Temperature dependent emissions observed. Drying halted emissions.	(Williams, 1993)

Environment	Stock	Measurement Period	Dung Character	Methane Emission	Units	Comments	Reference
Devon, England. Poorly permeable silty clay loam, pH 5.5. Dairy cow manure	Grazed and supplemented dairy cows, manure collected and re-applied	100 days	2.88 % N (dry basis), 42.97 % C (dry basis), 85 % moisture (wet basis)	0.45	g CH_4 Cow^{-1} day^{-1}	1.2 kg dung samples applied to 20 cm diameter plots. Dung collected from dairy cows fed on grass silage, kale, and grazing	(Yamulki et al., 1999)
Denmark.	Grazed or housed heifers. Manure remixed and hand applied	15-18 days	88.1% water (wet basis, grazed); 82.6% water (housed).	158, 170, 37 [grazed, housed, grazed second campaign] or 2.03, 2.18, 0.47	ml of CH_4 kg^{-1} or g cow^{-1} day^{-1}, assuming 750 kg of manure $month^{-1}$	1 kg dung pats. Emission only 0.8 to 4% of the emission likely if manure stored as a liquid during the same period. Chamber technique applied did not meet recent recommendations for accuracy for nitrous oxide measurement (Rochette and Eriksen-Hamel, 2008), much of which probably applies to methane measurements. However, substantial technique development is reported in Holter (1997). Differences between climatic conditions may account for large difference between first and second campaign.	(Holter, 1997)
New Zealand, outdoors, 10-20°C	Sheep			0.167	g CH_4 [kg dung-C]$^{-1}$	Outdoor *in vitro* study	(Joblin and Waghorn, 1994, as cited in Saggar et al., 2004)
New Zealand, indoors, 37°C	Sheep			2.836	As above	Indoor *in vitro* study	(Joblin and Waghorn, 1994, as cited in Saggar et al., 2004)
New Zealand,	Sheep	10 days		0.854-3.236	As above	*Iin situ* manure on pasture. Emissions appeared to be largely the result of entrained enteric methane.	(Carran et al., 2003b, as cited in Saggar et al., 2004)
New Zealand, Palmerston North, Tokomaru and Karapoti soils	Dairy cattle	90 days		2.567	As above	*In situ* manure on pasture	(Saggar et al., 2003a, Saggar et al., 2003b, as cited in Saggar et al., 2004)
Lincoln, New Zealand, Templeton soil	Dairy cattle	About 90 days		2.268	As above	Reconstituted dung pats on pasture	(Sherlock et al., 2003, as cited in Saggar et al., 2004)

3.1.2 Methane oxidation in grazed soils

The processes controlling the consumption of methane in manure and soil systems were reviewed in Section 2.2. In grazed rangelands, the oxidation of methane is likely to be more relevant than in intensively grazed systems due to the low rates of methane production. A review of the magnitudes of methane oxidation for a range of environments has recently been completed (Dalal et al., 2008) and is used as the basis for the following data on grazed systems.

Rate of methane consumption in rangeland systems are around (as reviewed by Dalal et al., 2008):

- 55µg m^{-2} h^{-1} in temperate grassland systems, which is a rate of consumption similar to that of temperate forests.
- 8 µg m^{-2} h^{-1} in sub-tropical grassland systems — and are thus much lower than the rates of consumption in temperate grasslands, and are also much lower than the rates of consumption in the adjacent sub-tropical forests.

As is consistent with process understanding (Section 2), higher consumption appears to be supported by dryer conditions in a given environment, which tends to result in aerobic soil conditions.

While animal camping may locally affect methane oxidation rates (due to compaction related decreases in methane oxidation, Livesley et al., 2008), at the landscape scale, this does not appear to be an important consideration (van den Pol-van Dasselaar et al., 1999). However, N fertilisation may decrease methane oxidation (Mosier et al., 1991, van den Pol-van Dasselaar et al., 1999). Given the mechanisms of this effect (section 2.2), it is understandable that ammonia based fertilisers have a stronger effect than nitrate fertilisers (Veldkamp et al., 2001), however, Dalal et al. (2008) suggests that this is also related to an acidifying effect of regular ammonium sulphate applications on soil.

While rangelands in temperate climates appear to have comparable methane oxidation rates to adjacent forests, soils under arable cropping tend to have methane consumption rates of only 30% of those under forests.

3.2 Methane emissions from manure management at feedlots

Anecdotally amongst lot-feeders, regulators, and industry advisors, it is well known that a high proportion of manure mass is lost at each step of the manure management system. Some of the implied carbon loss occurs as methane. The gaps in the available data on methane emissions are as wide as those for nitrous oxide emissions from the same systems.

The apparent trend of this data (in terms of methane emissions) is that enteric emissions may be more important than sources of emissions in the manure management system (Table 1). However, the few multi-week measurements of emissions from pens have not separated enteric emissions from manure emissions. The single pen manure study so far completed (Boadi et al., 2004) only investigated emissions over two 24 hour periods, and may not include the wet pen conditions that are likely to greatly increased methane emissions (Section 2). Research suggests that pen manures may produce 7 times as much methane under moist conditions than occurs under dry conditions (laboratory incubations, Lodman et al., 1993). While Lodman et al. (1993) did not observe high emissions from feed-pad manure the day after rain in cold conditions (14°C), their observations of wet and dry patches at another feedlot demonstrated twice the rate of emission from wet patches

(30°C). Measurements conducted one day after rainfall in cold conditions may not allow enough time for the wetting effect on methanogens to be seen, or peak emissions to be reached (Section 2).

Despite the scarcity of data it is possible to make an estimate of the relative importance of methane emissions compared to nitrous oxide emissions, given a range of critical assumptions.

- Feedpad emissions dominate methane and nitrous oxide emissions.
- Feedpad manure plus enteric emissions from feedlots are around 0.146 to 0.166 kg of CH_4 animal^{-1} day^{-1} — equivalent to 3.7 to 4.2 kg of CO_2 animal^{-1} day^{-1} (Table 5).
- A 475 kg average animal weight on feed, and manure nitrogen production of 0.34 kg day^{-1} (American Society of Agricultural Engineers, 2003).
- Assuming 2 % of excreted N is emitted as N_2O, though this relies on the unverified inventory method (National Greenhouse Gas Inventory Committee, 2007). This amounts to around 6.7 kg of CO_2-equivalent animal^{-1} day^{-1} in the form of nitrous oxide.
- Assuming that 75 % of N is lost as ammonia from the feed pad, and 1.25% of this is re-emitted as N_2O (unverified inventory technique, National Greenhouse Gas Inventory Committee, 2007).

Given these assumptions it is apparent that, because emissions are likely the sum of pad plus in manure management methane emissions are likely to be less than nitrous oxide emissions from the same system. In fact, direct plus indirect nitrous oxide emissions from the feed pad (about 1700 kg CO_2-equivalent animal^{-1} year^{-1}) exceed pad enteric plus manure methane emissions (about 1600 kg CO_2-equivalents animal $^{-1}$ year^{-1}).

Table 1 provides a brief summary of the limited available data which has been largely sourced from overseas:

- *Composting results in a wide range of emissions — from minimal emissions to greater emissions than from manure stockpiling. Methane emissions tend to be maximised where composts are actively turned. Manipulating the factors known to control methane formation and consumption is likely to provide mitigation managements.*
- *Data is largely lacking on composted stockpile emissions.*
- *It is likely that sedimentation basins and feedlot effluent ponds contribute only a minor fraction of total emissions, though some initial field measurements should be collected.*
- Land application is probably not a major source of methane emissions.

3.2.1 Pen manure emissions

Little data is available on methane emissions from feedpads (Table 3), though several of the available methane data points have been collected at Australian feedlots (Loh et al., 2008, McGinn et al., 2008).

The measurement programs conducted do not distinguish between enteric and manure methane emissions. The combined values range from 0.146 to 0.323 kg of CH_4 emission animal^{-1} day^{-1}. These collective enteric+manure pad values appear to be cattle class dependent (increasing with animal weight, McGinn et al., 2008) and are influenced by diet (Boadi et al., 2004, McGinn et al., 2008).

One of the two pen manure emission studies (Boadi et al., 2004) suggests that pad manure methane emissions may only be a few percent of enteric emissions (4–5%). However the relevance of this data collected at an average manure pack temperature of 4.3°C is limited. Boadi et al (2004) also collected this data over a brief period (two 24 hour periods), and used a chamber technique that did not meet recent chamber recommendations (Rochette and Eriksen-Hamel, 2008).

Boadi et al. (2004) also observed that methane emissions increased as manure pack depth and temperature increased.

Diet may have a strong effect on pen manure emissions. While Boadi et al. (2004) observed no difference due to diet in emissions, this probably reflected the fact that diets were formulated to have similar N and energy content, they suggest that diet could have a strong effect on pen manure emissions. Diet determines the carbon :nitrogen ratio of the pen manure pack, which also influences the extent of CH_4 released from manure (Sections 2.1 and 2.2). Higher rates of CH_4 release from manure are associated with higher N content of the diet (Jarvis et al., 1995). Jarvis et al. (1995) observed over 7.5 times more CH_4 emitted from manure from grain-fed compared to hay-fed cattle. This is similar to the findings of other researchers (Lodman et al., 1993), who observed that the variables which contributed most to the differences in methane production were temperature, moisture and diet of the animal. Manure handling factors and drying conditions may be added to this list (Gonzalez-Avalos and Ruiz-Suarez, 2001). Lodman et al. (1993) attributed increased methane emissions to an increase in readily fermentable carbohydrates in faeces from high-grain diet animals. However, these differences were observed in the laboratory and not seen during field conditions. Boadi et al. (2004) suggest that research is required to investigate the emission effect of the large variation that may exist within the manure packs as a result of trampling and defecation by animals.

The rate of methane emission from the manure pad falls far short of the emission potential seen when manures are anaerobically digested. Lodman et al. (1993) found that 1 to 7% of potential methane conversion occurred in drying and wet-condition incubations, and field data suggested that 0.1 to 0.2 % of potential methane production from a single day's manure deposition occurred daily (though much more manure had accumulated than one day's manure production). Methane emissions increased strongly with temperature.

Data is currently not available to confirm these observations for Australian conditions, conditions that are known to be substantially different from those in overseas studies (e.g. temperature and humidity regimes, as described in McGinn et al., 2008). Reliable data values on the range of emissions from the pen manure (isolated from enteric emissions) are required.

Mitigations to decrease pen manure methane emission

Good pen design — particularly with respect to good drainage — is likely to be a mitigation management for pen manure pack methane emissions. The effect of rapid drying in decreasing methane emissions is well documented (Lodman et al., 1993, Boadi et al., 2004, Gonzalez-Avalos and Ruiz-Suarez, 2001). Where composting practices are developed to a sufficient standard to decrease total GHG emissions below those experienced from the pen surface, increased cleaning frequency may decrease overall emissions. Current data on pen surface emissions and composting are currently not sufficient to make this conclusion.

In well managed and relatively dry pen surface conditions, aerobic decomposition of organic matter may (speculatively) be more readily achieved than in stockpiled systems or in poorly aerated composting systems. Pad cleaning regimes that prevent deep manure accumulation may decrease methane generation, by decreasing the prevalence of anaerobic conditions.

Maximising aerobic microbial breakdown at the pad surface through pad cleaning practices may also decrease emissions from down-stream management. Likewise preventing manure loss to sedimentation basins and ponds may decrease overall anaerobic decay of organic matter that could lead to methane generation.

Liming is a known method of increasing microbial turnover of organic matter (Murayama and AbuBakar, 1996, Rangel-Castro et al., 2005). Achieving greater aerobic breakdown to carbon dioxide, however, would be decreasing the carbon offset potential of the manure products. Following experimental data collection to fill the gaps in understanding, a system-wide evaluation would be required to decide the relative benefits of increasing ammonia (as discussed in the associated review, “Grazing and lot feed nitrous oxide emissions”, Matt Redding) and carbon losses at the feed pad need to be considered. Phosphogypsum applications may be useful to further decrease methane emissions at the feedpad, though no study has been conducted on this application, and hydrogen sulphide emissions may increase.

Table 3. Greenhouse gas emissions and ammonia volatilisation from manure management at the feedpad.

Reference	Emission Type	CH_4		N_2O		Ammonia		Location	Observation Period	Comments
IPCC (1997)						20	% of N excreted			
National Greenhouse Gas Inventory (2007)		5 and 1.5 + enteric emissions	% of 0.17 X volatile solids for Queensland/NT (first value) and other states (second value)	2	% of excreted N					Value includes together solid storage and feedpad. Using ASAE volatile solids production and 475 kg average steer weight, estimate of about 7kg of CH_4 SCU^{-1} $year^{-1}$
Denmead et al. (2008)	Feedpad with cattle					69 and 24	g N [animal $day]^{-1}$ Victoria and Queensland	Victoria and Queensland	1 month	Ammonia emission factor of 0.59% of N excreted.
Flesch et al. (2007)	Feedpads with cattle					149 and 151	g NH_3 [head $day]^{-1}$, 2004, 2005	Texas	2 months	63 and 65% of total N input. 14 m^2 $animal^{-1}$
Loh et al. (2008)	Feedpads with cattle	0.146 and 0.166	kg CH_4 [animal $day]^{-1}$ Victoria and Queensland			125 and 253	g NH_3 [animal $day]^{-1}$ Victoria and Queensland	Victoria and Queensland	1 month	Queensland cattle weights from 265 to 620 kg. Victorian from 280 to 700 kg.
Todd et al. (2008)	Feedpads with cattle					131 and 118	g NH_3 [head $day]^{-1}$; 2004 and 2005	Texas	10 weeks	62 to 64% of N fed
Boadi et al. (2004)	Feedpad without cattle	16.5 and 26.5	g CO_2-equivalent [day $animal]^{-1}$; low forage:grain; high forage:grain	50.5 and 46.0	g CO_2-equivalent [day $animal]^{-1}$; low forage:grain; high forage:grain			Canada	126 days/experiment, but only 3 measurements.	112, 252 kg steers commenced feeding. 49 m^2 $head^{-1}$. Divide methane value by 25 and N_2O value by 298 to obtain methane and nitrous oxide emission masses. Manure pack methane was a few percent of the magnitude of enteric methane emissions
Lodman et al. (1993)	Feedpad emissions without cattle	0–0.0032	kg CH4 m^{-2} d^{-1}					Colorado, U.S.		Chamber approach.
van Haarlem et al. (2008)	Pad with cattle emissions	0.323	kg CH4 $animal^{-1}$ d^{-1}			0.318	kg NH_3 $animal^{-1}$ d^{-1}	Canada		Strong diurnal variability — minimised at sunrise, maximised at sunset.

Reference	Emission Type	CH_4		N_2O	Ammonia	Location	Observation Period	Comments
McGinn et al. (2008)	Pad with cattle emissions	0.214	kg CH4 $animal^{-1}$ d^{-1}			Alberta, Canada	1 month, September	Mean 475 kg steers
McGinn et al. (2008)	Pad with cattle emissions	0.166	kg CH4 $animal^{-1}$ d^{-1}			Queensland	1 week, February	Mean 442 kg steers. Lower daytime emissions than Alberta, possibly due to heat stress, lipid feeding, and lighter weight cattle at this feedlot

3.2.2 Manure storage and composting emissions

Data available on methane emissions from composting and stockpiling are very limited, restricted to overseas studies, and systems of questionable relevance to Australian lot-feeding operations. The studies that are available have observed a wide magnitude of emissions from composting operations. As discussed below, this appears to be related to aeration practice (turning versus passive aeration) and moisture regimes. In terms of emissions processes (Section 2), it might be expected that composting would result in lower emissions than stockpiling operations. However with some overseas composting practices this is not always the case. The reason for this disparity is not yet clear. It is known that the stockpiling studies available for comparison are probably not representative of Australian feedlot practice.

Stockpiling and composting cause losses of carbon, nutrients, and total mass. Canadian data comparing uncompacted 50 t conical stockpiles (maintained undisturbed for 100 to 155 days) to 50 t turned windrows (1.6 x 3 x 28 m), found considerable mass losses (Larney et al., 2006). Composting led to higher dry matter losses (39.8%) compared to stockpiling (22.5%). Carbon losses were greater with composting (66.9% of initial) than with stockpiling (37.5%). Total carbon content of the material declined from 31.4% for fresh manure, to 24.8% with stockpiling, or 16.1% with composting. In another trial of composting, carbon losses of 61% were recorded (Larney et al., 2008a).

These carbon losses occur dominantly as carbon dioxide and methane. Data collected in the course of this review suggests that methane emissions from overseas cattle manure composting results in emission of 1.14 to 8.93 kg of CH_4 t^{-1} of manure, with the mean for 6 values being around 5.1 kg of CH_4 t^{-1} of manure (Tables 1 and 4).

A recent review of composting versus potential greenhouse gas emission from anaerobic ponds and landfills suggested that emissions from composting were minimal — and substantial emission avoidance credits would result (Brown et al., 2008). Their conclusions took into account the range of emissions and energy inputs, and they also suggested that increasing compost solid content and C:N ratio would further minimise emissions.

However Brown et al.'s (2008) comparison of composting and land-filling or anaerobic ponds is not very relevant to Australian lot-feeding enterprises, where the alternative to composting is stockpiling or direct land application of pad manure. Very little emissions data is available for beef cattle manure stockpiles (Table 4, 3 values), and no data is available that is representative of compacted stockpiles. In fact, mean emission values for uncompacted beef manure stockpiles were lower than for composted operations (about 3.6 kg of CH_4 t^{-1}), though ranges overlapped. However, Pattey et al. (2005) found that uncompacted stockpiles produced 1.46 times the greenhouse gas equivalents that composting produced. Notably, in this case passive aeration was used, so enhanced methane losses during turning were not generated.

Turning events are associated with accelerated losses of some gasses from composting operations. For example, ammonia emissions increased immediately following turning (Parkinson et al., 2004). The influence on methane losses is not known. However, the surface of uncompacted stockpiles and compost windrows are also likely to act as methane oxidising media (Brown et al., 2008), through the mechanisms outlined in Section 2.2. High internal pile methane concentrations have been observed, compared to very low surface concentrations (Hao et al., 2001). Researchers have identified heat-loving methane consuming organisms in composts (Jäckel et al., 2005) and composts have previously been used in experimental biofilters to oxidise CH_4 (Mor et al., 2006).

This data also suggests that it is important to investigate methane emissions relative to compost turning events in composting operations — and their relative contributions to overall methane emissions since they allow methane to bypass windrow surface oxidation processes.

Stockpile compaction is also likely to be a major factor in methane emission. El Kader et al. (2007) suggested that water addition and compaction could be used as a means to mitigate nitrous oxide emission and ammonia volatilisation from manure stockpiles (data for dairy cowshed stockpiles and composting, and spent turkey litter). Decreases in free air space through wetting or mild compaction in turkey manure composts were observed to decrease nitrous oxide emissions and ammonia volatilisation. These free air space effects on nitrous oxide emission and ammonia volatilisation may be related to decreases in gas diffusion rates — something known to be a factor controlling methane emission and consumption (Saggar et al., 2004, Le Mer and Roger, 2001, Dalal et al., 2008).

A similar increase in nitrous oxide emissions with turning was observed for a comparison between turned and passively aerated composting of lot-feed manure in Canada (Hao et al., 2001).

Mitigations to decrease methane emissions from stored manures

Some authors regard composting as a mitigation management to decrease methane (and GHG emission in general) from stockpiles (Pattey et al., 2005). It appears that this is most likely to be effective where passive aeration techniques rather than active turning are used. The lack of data on stockpiles, compacted stockpiles, and the effect of surface layer methane oxidation make a definitive answer to the effectiveness of composting as a mitigation management unavailable.

It is important to note that the nutrient losses normally experienced in composting also decrease the fertiliser value of the manure (Peigne and Girardin, 2004). However, a range of technologies have potential to decrease ammonia volatilisation from composting and stockpiling (as discussed in the review developed by Matt Redding, “Grazing and lot feed nitrous oxide emissions”, for MLA). Techniques could include use of urease inhibitors — though inhibitor activity would need to be maintained from the pen to the composting operation, since urea hydrolysis is a very rapid process after urine deposition.

These types of managements are also likely to have an impact on methane emissions. Modification of pH to a more acid range to support greater ammonia retention may tend to decrease methane and nitrous oxide production (though it may also decrease methane consumption where ammonia concentrations in the external aerobic layer of the compost pile is increased). Where pH modification is completed with ferrous or aluminium ion- containing materials, carbon mean residence times may simultaneously be increased (these possibilities are discussed in the associated nitrous oxide review “Grazing and lot feed nitrous oxide emissions”, Redding, 2009). The potential for these mitigation managements, the interaction between the greenhouse gasses emission processes, and their economic practicality have not been investigated.

A range of other measures to decrease methane emissions from composting operations have been proposed (Brown et al., 2008):

- Manipulating composting feedstock to have a high carbon to nitrogen ratios (C:N >30:1), low moisture contents (moisture <55%), and aerating the system.
- Covering the compost windrow with a layer of moistened finished compost, the gas emission potential can be cut by 50 % for methane. Recent research suggests that this type of

management may consume up to 161 g methane m^{-2} d^{-1} (1 m flow length, Scheutz et al., 2009).

From the literature reviewed it seems that using aeration techniques that do not involve active compost turning in addition to the two measures suggested by Brown et al. (2008) is likely to result in composting approaches that emit less methane than stockpiling.

Data for a range of types of agricultural, municipal, and industrial composting operations suggests that where these measures are not employed emissions of at least 2.5 % of initial C will occur as methane, and 1.5 % of initial N will occur as nitrous oxide (Brown et al., 2008). It is notable that Hao et al.'s (2001) data for nitrous oxide is less than this minimum, despite the low C:N ratio of the initial material, though Brown et al.'s (2008) estimate was correct for methane. Questions remain as to whether active turning and active aeration are a significant impediment to composting as a methane emission decreasing management (Hao et al., 2001). Active turning also redistributes mineral N compounds (including nitrate) formed in the aerobic compost surface layer to deeper layers where denitrification is more likely to occur (Hao et al., 2001).

Additions of phosphogypsum are known to decrease methane emissions from composting operations (up to a 97 % decrease in methane emission, Hao et al., 2005) possibly through inhibition or competition between sulphate reducing bacteria and methanogens for organic carbon and energy sources.

The type of carbon substrate added to composts may affect methane emissions. Some differences were observed for straw versus wood-chip additions (Hao et al., 2004), though overall cumulative emissions were not significant different. The differences in emissions timing for the two materials, and the differences in total CO_2 emissions suggest that other studies with other materials may form more conclusive results.

Brown et al. (2008) suggested that mitigation managements amount to manipulations of some of the factors controlling methane production and oxidation (Sections 2.1 and 2.2), particularly the redox status, and probably the effect of excess ammonia on methane consumption. Additions of phosphogypsum capitalise on the known competition between sulphate reduction and methane formation (Section 2.1). A wide range of other mitigations are conceivable that capitalise on other controlling factors, such as temperature, pH, manganese content and salinity. Opportunity in this area should be explored.

Table 4. Manure solids/deep litter/manure stockpile or composting emissions from both pigs and cattle.

Reference	Emission Type	CH_4		N_2O		Location	Observation Period	Comments
Hao et al. (2001)	Canadian lot-feed manure, containing wheat straw bedding material.	6.3 and 8.1 (not significantly different)	kg CH_4-C t^{-1} (dry weight); passive versus turned	0.62 and 1.07 (significant)	% total N, passive versus turned	Canada	99 days	Passive aeration (perforated pipes) versus turning (6 turns). Manure C:N ratio of 19.3, 34..2 % carbon (dry basis).
Hao et al. (2004)	Composted straw and woodchip bedded manure	8.92 and 8.93	kg CH_4-C t^{-1} (dry weight)	0.0771 and 0.0842	kg N_2O t^{-1}	Canada	99 days	Windrows 33 m^2, 1.6 to 1.8 m high, turned 4 times in frist 49 days, 8 times in total. Initially 33.1 and 44.7 % carbon (straw and woodchip, dry basis).
Hao et al. (2005)	Straw bedded manure compost, with added phosphogypsum	2.83, 0.51, 0.44	kg CH_4-C t^{-1} (dry weight)	28.29, 21.50,30.95	g N_2O-N t^{-1} (dry weight)	Alberta, Canada	134 days.	Rates of addition of phosphogypsum were 10, 17.8, 26.9 % of manure dry weight. Windrows 3.5x 6.6x1.7 m high. Turned fibe times (days 15, 23, 43, 65, 91). Manure 30.8% C, 1.56% N, 0.44% S.
Hao et al. (2005)	Straw bedded manure compost, control treatment	15.36	kg CH_4-C t^{-1} (dry weight)	12.06	g N_2O-N t^{-1} (dry weight)	Alberta, Canada	134 days.	Windrows 3.5x 6.6x1.7 m high. Turned five times (days 15, 23, 43, 65, 91). Manure 30.8% C, 1.56% N, 0.44% S.
Pattey et al. (2005)	Canada, beef cattle manure.	2.13 [0.067]	kg CH_4-C t^{-1} (dry weight); [t CO_2-eq $head^{-1}$ $year^{-1}$]	0.034 [0.010]	g N_2O kg^{-1} dry matter; [t CO_2-eq $head^{-1}$ $year^{-1}$]	Canada	3 month period	10% bedding material. Uncompacted stockpile. Initial C:N ratio of 35.0
Pattey et al. (2005)	Canada, beef cattle manure.	0.11 [0.003]	kg CH_4-C t^{-1} (dry weight); t CO_2-eq $head^{-1}$ $year^{-1}$]	0.162 [0.049]	g N_2O kg^{-1} dry matter; [t CO_2-eq $head^{-1}$ $year^{-1}$]	Canada	3 month period	10% bedding material. Passively aerated compost treatment. Initial C:N ratio of 35.0
Pattey et al. (2005)	Canada, dairy cattle manure.	5.92 [0.416]	kg CH_4-C t^{-1} (dry weight); [t CO_2-eq $head^{-1}$ $year^{-1}$]	0.403 [0.272]	g N_2O kg^{-1} dry matter; [t CO_2-eq $head^{-1}$ $year^{-1}$]	Canada	3 month period	50% bedding material. Dairy cattle. Uncompacted stockpile. Initial C:N ratio of 20.0
Pattey et al. 2005	Canada, dairy cattle manure.	1.14 [0.080]	kg CH_4-C t^{-1} (dry weight); [t CO_2-eq $head^{-1}$ $year^{-1}$]	0.582 [0.393]	g N_2O kg^{-1} dry matter; [t CO_2-eq $head^{-1}$ $year^{-1}$]	Canada	3 month period	50% bedding material. Dairy cattle. Passively aerated compost treatment. Initial C:N ratio of 20.0
Xu et al. (2007)	Alberta, Canada. Lot feed manure composting compared to lot feed manure co-composting with mortalities	1.14 and 3.16	kg CH_4-C t^{-1}	3.74 and 5.70	% of total N; composting and co-composting of mortalities.	Canada	310 days	98.3% manure, 1.67% straw; or 88.3% manure, 1.5% straw, and 10.2% cattle mortalities. Nitrous oxide emissions from treatment turned with a front-end loader. 1 t manure produced $head^{-1}$ annually (60% moisture dry basis). Mortality rate of approximately 1.26%.

Reference	Emission Type	CH_4		N_2O		Location	Observation Period	Comments
Sommer et al. 2004 (2004a)	Canada. Circular conic stockpile. Cereal straw bedding included.	0.3	% of initial C	0.3	% of intial N	Canada	7 days	Unknown whether compacted. Very short measurement period to test measurement techniques.
Wolter *et al.* 2004	Pig deep litter stockpile	2	% of total carbon	1.9	% total N	Germany	113 days	Thermophillic, high temperature decomposition occurred; ammonia emission plus denitrification accounted for 26% of N
Amon *et al.* 2007	Staw flow, pig slurry storage	29.8 and 97.3	kg CO_2-eq m^{-3} [200 days]$^{-1}$	35.4 and 36.7	kg CO_2-eq m^{-3} [200 days]$^{-1}$	Austria	200 days	Covered and uncovered storage, straw flow system.

3.2.3 Effluent pond methane emissions

The shallow depth of lot-feed effluent/sedimentation ponds in Australia, the variability of liquid levels, and the small quantity of carbon entering these systems suggests that they are unlikely to be a major source of methane. For example, organic carbon contents of lot-feed effluents are likely to be low (data are scarce though one Canadian value suggests 0.5 g L-1, Miller et al., 2004, Australian feedlot effluents contain about 0.76% volatile solids, Watts et al., 1994), and inflow volumes are also small compared to the quantities of manure collected from pads. Estimates from a 15 000 SCU feedlot under climatic conditions similar to Dalby suggest annual runoff volumes of around 100 ML from pens and other infrastructure (pers. comm. Alan Skerman). This amounts to around 500 to 760 t of sediment entering the pond system while manure export from the pens will be around 1.0 to 1.1 t of manure [standard cattle unit]$^{-1}$ year^{-1}, amounting to around 14 000 t of manure. However, the lack of data in this area suggests that confirmation is required.

While as little as 3.5% of total manure solids enters the ponds, there is one factor that may magnify the relative importance of this feedlot infrastructure as a methane source: the prevalence of anaerobic conditions. While conditions in solid manure management may favour only a small fraction of potential methanogenic conversion being achieved (Lodman et al., 1993), conditions under prolonged saturated conditions in sedimentation basins and effluent ponds may support near complete methanogenic conversion. However, applying Sommer et al.'s (2004) estimated rate of emission (0.216 kg CO_2-equivalent [kg volatile solids]$^{-1}$), and assuming that the manure material entering the sedimentation and pond system contains around 10% volatile solids, emissions would be around 11 to 16 t CO_2-equivalent per year for the above feedlot example. This is a minor fraction of total emissions.

There is a requirement to conduct at least a minimal measurement campaign on methane emission from feedlot effluent ponds to establish the relative magnitude of this emission source within the manure management system.

3.2.4 Data regarding manure and effluent application to land

The emission of methane from manures and effluent applied to dominantly aerobic land tends to be shortlived (Dalal et al., 2008). A small number of studies have investigated this emission behaviour with dairy manures, slurries, and pig manure slurries— suggesting that land application emissions are likely to be a small proportion of the total methane emissions (Tables 1 and 5).

Table 5. Methane emissions from solid cattle manure application to agricultural land.

Site and environment	Experimental period	CH_4 Emission	Comments	Reference
Lincoln, New Zealand, Udic Dystrochrept soil, white clover-perennial ryegrass pasture with piggery effluent application	100 days	1052 g CH_4-C ha^{-1} net emission	Pig slurry 60 m^3 ha^{-1}, containing 1326 kg of C ha^{-1}. 0.08 % of the applied carbon was emitted as methane (peaking immediately on application), and this process was essentially complete within 7 days. Much of the methane emission was dissolved methane	Sherlock et al. (2002)
Sweden, Eutric Cambisol, red fescue+perennial ryegrass+smooth stalked meadow och pasture	45 days	70 g CH_4-C ha^{-1} d^{-1} for band spread; 38 g CH_4-C ha^{-1} d^{-1} for slot injected	Initial peaks of emission only. Emission had returned to background within several days, and overall methane was consumed by the soil.	Rodhe et al. (2006)
Nasu, Japan. Orchardgrass and Italian ryegrass, volcanic ash derived soil, Entic Hapluumbrepts. Dairy manure composts and dairy manure applied	Partially throughout 3 years	Methane consumption was the general rule	Manure products applied at 15 to 30 t ha^{-1}.	Mori et al. (2008)

3.3 Comparison with national greenhouse gas inventory (2007) emission factors

Using the NGGI method (2007), and ASAE volatile solids production data (American Society of Agricultural Engineers, 2003) an estimate of about 7 kg of methane production from manure management [475 kg animal]$^{-1}$ year^{-1} was developed. The cumulative manure management losses reviewed in this paper tend to exceed this value even given the probable underestimation of pad manure emissions (Table 1), except possibly where pad manure is applied directly to land. The enteric emissions estimate (calculated from National Greenhouse Gas Inventory Committee, 2007) of around 97 kg of CH_4 SCU^{-1} year^{-1} is within the range of values reported internationally for total pen emissions (manure plus enteric), but is more than 50 % higher than the mean.

Given the current lack of Australian data, however, the NGGI (2007) emission factors may be considered reasonable estimates. Further measurements of real Australian values are required as documented in this review.

The greatest weakness of the NGGI approach is the lack of ability to recognise emission decreases related to improved management. The broad range of emission values detailed in Tables 2 to 5 is an indicator of the influence of process factors that determine the magnitude of emissions. The NGGI values tend to be within these ranges or slightly above these values — but are not responsive to the factors that cause the range of emission factors observed.

This is a similar drawback to the current calculation techniques as noted by McGinn et al. (2008) for diet-related enteric emissions decreases.

4 Knowledge gaps

The following sections detail critical knowledge gaps revealed by the literature review, and research opportunities that may develop into new advantages for the industry.

4.1 The lack of relevant manure management emissions data

Anecdotally amongst lot-feeders, regulators, and industry advisors, it is well known that a high proportion of manure mass is lost at each step of the manure management system. Data are largely lacking, but it is known that mass loss during composting can be large (61% of carbon lost, Larney et al., 2008b, more than 23% of carbon lost in only 17 days, Sommer et al., 2004a).

Hypothetically, it may be possible to estimate total methane losses assuming realistic conversion rates of initial manure carbon to methane. Sommer et al. (2004a) calculated that 0.3% of initial carbon was emitted as methane in association with a loss of 23% of carbon. This corresponds to about 0.8% conversion to methane, assuming 60% of initial carbon is lost in manure management (similar to compost carbon balances by Larney et al, 2008). Despite Sommer et al.'s (2004) indication that their methane conversion rate was fairly high, the data collected for my review suggests manure emissions already greatly exceed this magnitude.

Therefore, we are left without a basis to complete this type of estimate. How much of the very high proportion of decayed manure carbon is lost as methane is unknown.

Table 1 to 5 demonstrate the extreme paucity of available data in this area, and the overall lack of data representing Australian moisture and temperature regimes. Since these two factors are key controllers of methane emissions (Section 2). This dependence is extreme. For example, the relative increase in methane formation with a 10°C increase in temperature is a factor of 4.6 in anaerobic soils (van Bodegom and Stams, 1999).

Stockpile data is limited to two un-compacted stockpiles. The fact that emissions from these systems is less than emissions from most composting operations is surprising — and reason to re-examine the stockpile sources. The lack of consideration of compacted stockpiles means that the data cannot be relied upon for Australian systems. However, fewer large manure stockpiles are maintained in the industry than were in existence a decade ago (anecdotal evidence through discussions with Queensland regulators).

There is a requirement to measure methane emission from feedlot effluent ponds to confirm the relative small magnitude of this emission source within the manure management system.

This is a critical failure in terms of informing the Australian industry, and relevant data must be collected. Such data is likely to allocate substantially more of the combined pad manure plus enteric source to the manure source. If manure management mitigations for feedlots are simpler to develop than those required to modify enteric emissions, this would totally change the industry outlook for decreasing emissions.

4.2 Optimising composting

The data available for methane emissions from manure composting suggest that composting can range from a negligible to serious methane emission source. The cause of these disparate outcomes is likely to be a difference in management, a difference that has not yet been effectively identified. There are suggestions that these outcomes may be related to aeration/turning practices, compaction, moisture management, and methane consumption in the outer aerobic layers of compost piles.
It is important to investigate methane emissions relative to aeration, moisture, free air space, and methane oxidation and the management events that influence them in composting operations. The relative contributions of these management practices to overall methane emissions must be identified so that composting management can be optimised for minimum emissions.

4.3 Pen management

Variables that contribute strongly to differences in methane production are temperature, moisture, diet of the animal, drying conditions, and manure handling. With the current lack of data on the magnitude of methane emissions from the manure on the feed pad it is not appropriate to recommend that mitigations immediately be investigated. However, if emissions at this point are found to be substantial, a range of mitigations related to cleaning practice and the above factors may be effective and should be investigated.

4.4 Incomplete process algorithms

Improved emissions algorithms are required to allow a management-responsive inventory calculation approach to be developed (Tier III).

Current algorithms such as those of Summer et al. (2004) have been determined for very limited temperature, moisture, and oxygen availability ranges. This approach allowed elegant simplifications in the anaerobic manure systems that they sought to represent. Under these restricted conditions some quite simple relationships held — such as increasing emission with increasing temperature.

For the broader temperature ranges and oxygen availability conditions encountered in feedlot manure management systems, parameter values that maximise emissions are very likely to exist. For example, emissions will not continue to increase as temperature increases — but will maximise at some value then fall as temperature increases beyond that point. Sommer et al. (2004) is also reliant on empirical rather than process based relationships, and does not represent methane consumption. This is a major drawback of the available algorithms in feedlots where methane consumption through microbial oxidation may well be the process that controls the magnitude of emissions.

5 Conclusions and recommendations

5.1 Emissions processes

The carbon and nitrogen cycles are inextricably linked and nitrous oxide emission mitigation cannot be conducted without consideration of methane emissions.

Importantly, methane is both formed and consumed in manure management systems.

Methane formation is favoured by warm (30 to 40°C), moist conditions combined with low oxygen supply, and a degradable organic material. The process also tends to proceed under near neutral conditions in terms of pH.

Methane consumption, at the high methane concentrations that may occur in manure management systems, is also favoured by warmer conditions. However, methane consumption requires oxygen and increases where pH is slightly above neutral, where ammonia concentration is low, and phosphorus and potassium are available.

Salinity inhibits methane consumption more than it inhibits methane production.

5.2 Grazing systems

In reasonably arid low density stocked areas or on low quality pastures, enteric methane emissions are likely to dominate methane production and consumption processes. Deposited manure emissions are relatively insignificant in these systems due to manure drying halting emissions processes in deposited manure. Mitigating manure methane emissions in this portion of the production system may therefore be irrelevant.

Methane consumption rates in temperate grasslands systems are around 55μg m^{-2} h^{-1}. In sub-tropical grassland systems, consumption is much less, at about 8 μg m^{-2} h^{-1}.

5.3 Feedlot beef production

Given a range of unverified assumptions it is apparent that manure management methane emissions are likely to be less than nitrous oxide emissions from the same system. In fact direct plus indirect nitrous oxide emissions from the feed pad (about 1700 kg CO_2-equivalent $animal^{-1}$ $year^{-1}$) may exceed pad enteric plus manure methane emissions (about 1600 kg CO_2-equivalents animal $^{-1}$ $year^{-1}$).

Available data (although sparse) and anecdotal evidence tends to indicate that the major candidates for emission decreases are the manure pad, stockpiles, and composting.

- There is a critical lack of data to decide how much of the manure pad plus enteric source is attributable to the manure on the pad.
- Few stockpile studies have been conducted, and none of them appear representative of Australian practice.
- Composting studies include a range of practices that perform extremely poorly in terms of methane. This is a surprising result considering that from current process understanding composting should be a mitigation practice. Other studies have realised the benefits of composting in terms of methane management. It is important to identify the practices that effectively minimise emissions. Further study may reveal that turning tends to favour greater emissions than effective passive aeration. Moisture management to ensure the prevalence of aerobic conditions and the maintenance of a mature, aerated blanket of compost over the composting pile are probably methane emission mitigation strategies.
- No data is available on emissions from effluent ponds and sedimentation basins. While as little as 3.5% of total manure solids enters the ponds the conditions under prolonged saturated conditions in sedimentation basins and effluent ponds may support near complete methanogenic conversion.

5.3.1 Compost and stockpile emission mitigations

While mitigation managements for composting are not well established it appears likely that the following managements may be developed to decrease methane emissions:

- Maintaining high carbon to nitrogen ratios (C:N >30:1);
- Low moisture contents (moisture <55%);
- Effective aeration, without active turning; and
- Covered compost piles with a layer of moistened finished compost.

An understanding of the process relationships allows additional educated guesses as to what managements may be developed to further mitigate methane emissions. These relationships suggest that there may be strong interactions between methane emissions and the mitigations proposed for nitrous oxide emissions (in the associated review by Matt Redding). These interactions may be positive or negative and current process understanding is not sufficient to adequately predict any effect.

5.3.2 Feed pad methane mitigations

Variables that contribute strongly to differences in methane production are temperature, moisture, diet of the animal, drying conditions, and manure handling. A range of mitigations related to the above factors may be effective and economical.

5.4 National Greenhouse Gas Inventory comparison

The international data on cumulative manure management losses reviewed in this paper tend to exceed those calculated using the Australian inventory method, even given the probable underestimation of pad manure emissions. However, the magnitude of the difference between these values is small compared to some of the sources involved (e.g. the pad manure plus enteric methane emissions source).

Inventory enteric emissions estimates are within the range of values reported internationally for total pen emissions (manure plus enteric), but are more than 50 % higher than the mean. Australian data from specific sources is still largely lacking. Rectifying the lack of data surrounding methane emissions directly from the pad manure is likely to increase the proportion of total pen emissions attributed to the manure — a methane source that is likely to be fairly readily managed.

The greatest weakness of the NGGI approach is the lack of ability to recognise emission decreases related to improved management. The broad range of emission values detailed in Tables 1 to 5 is an indicator of the strong influence of management factors that determine the magnitude of emissions — factors that are not accounted for in the current Tier II calculation approach.

5.5 Research recommendations

To correct the complete lack of relevant data, it is necessary to collect emissions data from:

- Manure on feedpads;
- Compacted stockpiles; and
- Effluent ponds and sedimentation basins.

There is a need to identify what management is required to minimise methane emissions from composting operations. This could be achieved by investigating how methane emissions can be controlled by managements that affect aeration, carbon:nitrogen ratio, changing moisture, manipulating free air space, and facilitating methane oxidation.

The magnitude of manure methane emissions at the feed pad is currently unknown. If the recommended methane emissions study reveals this as a substantial source, a range of mitigations are likely to be effective and may provide a good opportunity to decrease emissions — and should be investigated.

Improved emissions algorithms are required to allow a management-responsive inventory calculation approach to be developed (Tier III). If the current Tier II approach remains the standard, industry emission mitigations will not be reflected in inventory calculations. Algorithms need to reflect the wide range of temperature, moisture, carbon:nitrogen ratios, and reduction/oxidation potentials encountered in feedlot systems. These algorithms need to specifically represent methane oxidation processes which may be the most critical factor in methane emission from the mixed aerobic/anaerobic conditions that exist in feed-pad manure packs, compost piles, and stockpiles.

Emission sources and potential mitigation strategies need to be valuated in terms of CO2 equivalents, to allow the net compromise between decreasing methane and decreasing nitrous oxide emissions, to gain the best overall reduction in greenhouse gas emissions.

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