Good science for improving policy: greenhouse gas emissions from agricultural manures

Chris Pratt[A,C], *Matthew Redding*[A], *Jaye Hill*[A], *Andrew Shilton*[B], *Matthew Chung*[B] *and Benoit Guieysse*[B]

[A]Department of Agriculture, Fisheries and Forestry, 203 Tor Street, Toowoomba, Qld 4350, Australia.
[B]School of Engineering, Massey University, Palmerston North 4472, New Zealand.
[C]Corresponding author. Email: christopher.pratt@daff.qld.gov.au

Abstract. Australia's and New Zealand's major agricultural manure management emission sources are reported to be, in descending order of magnitude: (1) methane (CH_4) from dairy farms in both countries; (2) CH_4 from pig farms in Australia; and nitrous oxide (N_2O) from (3) beef feedlots and (4) poultry sheds in Australia. We used literature to critically review these inventory estimates. Alarmingly for dairy farm CH_4 (1), our review revealed assumptions and omissions that when addressed could dramatically increase this emission estimate. The estimate of CH_4 from Australian pig farms (2) appears to be accurate, according to industry data and field measurements. The N_2O emission estimates for beef feedlots (3) and poultry sheds (4) are based on northern hemisphere default factors whose appropriateness for Australia is questionable and unverified. Therefore, most of Australasia's key livestock manure management greenhouse gas (GHG) emission profiles are either questionable or are unsubstantiated by region-specific research. Encouragingly, GHG from dairy shed manure are relatively easy to mitigate because they are a point source which can be managed by several 'close-to-market' abatement solutions. Reducing these manure emissions therefore constitutes an opportunity for meaningful action sooner compared with the more difficult-to-implement and long-term strategies that currently dominate agricultural GHG mitigation research. At an international level, our review highlights the critical need to carefully reassess GHG emission profiles, particularly if such assessments have not been made since the compilation of original inventories. Failure to act in this regard presents the very real risk of missing the 'low hanging fruit' in the rush towards a meaningful response to climate change.

Additional keywords: manure management, methane, mitigation, nitrous oxide.

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Introduction

Agriculture represents 5–15% of Australia's and 60% of New Zealand's (NZ) export revenues (New Zealand Government 2013). This economic powerhouse has an unfortunate environmental downside because agriculture is also a major source of the greenhouse gases (GHG) nitrous oxide (N_2O) and methane (CH_4). There are three main sources of agricultural GHG in Australia and NZ: (1) enteric CH_4, (2) soil N_2O and (3) manure management CH_4 and N_2O. Most of the agricultural emissions in these countries occur as enteric CH_4 and soil N_2O. In NZ, annual enteric CH_4, soil N_2O and manure management emissions account for ~20 mega tonnes (Mt), 10 Mt and 1 Mt as carbon dioxide equivalents, (CO_2-e), respectively, (New Zealand Government 2013). By comparison, the annual emissions of enteric CH_4, soil N_2O and manure management CH_4 and N_2O amount to ~55 Mt, 12 Mt and 4 Mt of CO_2-e, respectively, in Australia (Australian Government 2013).

The knowledge of causes, magnitude, and mitigation potential of enteric CH_4 and soil N_2O emissions is now well established for Australia, NZ, and overseas (de Klein *et al.* 2001; Dalal *et al.* 2003; Saggar *et al.* 2004; Eckard *et al.*

2010; Cottle *et al.* 2011). By contrast, agricultural manure management emission estimates have not been verified by a rigorous review of the available literature since they were developed more than 15 years ago.

A review of manure management GHG emissions in Australia and NZ is therefore needed for assessing the accuracy of the countries' GHG inventories for this category. First, it will allow for an evaluation of current inventory estimates. In NZ, Chung *et al.* (2013) demonstrated that CH_4 emissions from dairy effluent are potentially underestimated by 400%, and Hill (2012) showed that CH_4 emissions from pig effluent are being overestimated by a factor of ~200%. These inaccuracies were caused by the inappropriate adoption of international default factors as well as the omission of key emission sources by NZ's GHG Inventory. The studies by Chung *et al.* (2013) and Hill (2012) revealed that a GHG emission profile within a nation's inventory can shift from minor to major, or vice versa, upon careful inspection of emission factors and management practices. A second reason a review of manure management GHG emissions is timely is because these emissions are point sources. Thus, mitigation options for these gases are more

amenable to development and implementation than for diffuse, non-point CH_4 and soil N_2O emissions, which occur at the paddock-scale.

Upon critically assessing manure management GHG emission estimates in Australia and NZ it is also possible to examine whether appropriate weighting is currently given to mitigating these emissions. This is a third reason why a review of this subject is important. Hence, in this paper we evaluate (1) manure management GHG emission estimates in Australia and NZ by comparing inventory emission factors with published country-specific data, where available, (2) mitigation options for emissions from this sector focusing on barriers to and drivers for uptake and (3) whether funded research into agricultural GHG mitigation is currently balanced to achieve greatest effect.

GHG emissions from key livestock sectors

Fig. 1 shows emission profiles for Australia's and NZ's main manure management GHG sources, based on inventory data. In this review, we use Australia's and NZ's definition of 'manure management', which encompasses CH_4 and direct and secondary N_2O emissions from all manure management systems as well as CH_4 emissions from pasture-deposited livestock manure. Global warming potentials (GWP) of 21 and 310 were used to convert CH_4 and N_2O emissions, respectively, to units of CO_2-e. These GWP are adopted by the most recent versions (2013) of the Australian and NZ GHG inventories. In the following discussion we examine the robustness of each of the GHG profiles in Fig. 1.

GHG emissions from dairy cattle 'manure management'

According to best available estimates, CH_4 emissions from NZ dairy farm anaerobic effluent ponds are the largest GHG source in NZ and Australia (Fig. 1) for the 'manure management' emission category. For example in NZ ~1.3 of the 1.4 Mt of CO_2-e/year of CH_4 from dairy farm manure comes from anaerobic ponds compared with just 0.1 Mt from manure deposited on pastures [determined from data in Saggar *et al*. (2004)]. It is important to note that the term 'effluent pond' is used to encompass a wide range of systems from small (<100 m^3) pits with just a few weeks' effluent storage capacity to large (>2000 m^3) lagoons for up to 1 year's storage capacity. Both the NZ and Australian GHG inventory apply blanket definitions to their dairy effluent ponds and use the Intergovernmental Panel on Climate Change (IPCC) CH_4 conversion factors developed by Mangino *et al*. (2001) to determine CH_4 emission rates on a temperature basis. Current research being collaboratively undertaken by several NZ Government research organisations will elucidate relationships between pond design and CH_4 emissions.

The input values used to determine pond CH_4 emissions are shown in Table 1. This table also includes calculation parameters for all the major GHG sources discussed in this paper. Table 2 summarises the emission calculation formulae. On a liveweight basis, Chung *et al*.'s (2013) estimate for NZ pond CH_4 emissions matches reasonably well with an NZ pond emission rate published by Craggs *et al*. (2008). These are, however, much higher than the emission rate reported by McGrath and Mason
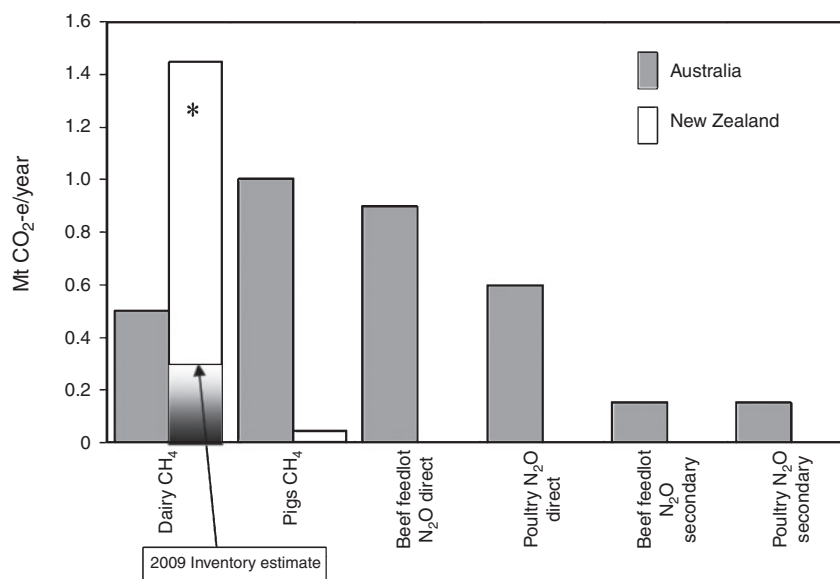
*For NZ's Dairy CH_4, the main source of emissions is anaerobic effluent ponds. For these ponds, a recently published CH_4 emission estimate (1.3 Mt CO_2-e/year) by (Chung et al., 2013) was considered more accurate than the national inventory value (0.3 Mt CO_2-e/year), which contains several key mathematical errors (its input paramters are incorrect and it's affected by an erroneous geometry calculations), as well as omitting several key potential CH_4 sources (such as waste feed and additional manure from feed pads). Hence, the estimate from Chung et al. (2013), which was calculated using Intergovernmental Panel on Climate Change (IPCC) methodology and NZ inputs, is included in this figure.

Fig. 1. Greenhouse gas emission profiles for major sources in the 'manure management' categories for Australia and New Zealand. The profiles are based on the most recently available inventory calendar year data (2011). New Zealand dairy methane is an exception, which was based on the publication of Chung *et al*. (2013).

Table 1.   Input values used to calculate emissions from the largest 'manure management' greenhouse gas sources in Australia and New Zealand
B_0 = biochemical methane potential, MCF = methane conversion factor, VS = volatile solids. References: (1) = New Zealand Government (2013), (2) = Chung *et al.* (2013), (3) = IPCC (2006), (4) Ledgard and Brier (2004), (5) = Australian Government (2013), (6) = Külling *et al.* (2003), (7) = Rotz (2004), (8) = Hutchings *et al.* (2001), (9) = Bouwman *et al.* (2002a), (10) = Bouwman *et al.* (2002b), (11) = Akiyama *et al.* (2005), (12) = Stehfest and Bouwman (2006)

	Anaerobic dairy effluent ponds NZ – CH_4	Anaerobic dairy effluent ponds Aust. – CH_4	Anaerobic pig ponds Aust. – CH_4	Beef feedlots Aust. – N_2O	Poultry Aust. – N_2O
(A) Livestock population ($\times 10^6$)	4.35 (1, 2)	2.56 (4)	2.25 (4)	1.17 (4)	95.86 (4)
(B) B_0 manure (m^3 CH_4/kg VS)	0.24 (2, 3)	0.24 (3, 5)	0.45 (3, 5)	–	–
(C) B_0 feed (m^3 CH_4/kg VS)	0.31 (2)	–	0.45 (5)	–	–
(D) B_0 milk (kg CH_4/m^3 milk)	48 (2)	–	–	–	–
(E) MCF (frac B_0)[A]	0.72 (2, 3)	0.9 (3, 5)	0.9 (3, 5)	–	–
(F) m^3 waste milk/head.year	0.067 (2)	–	–	–	–
(G) kg VS from milking shed manure/head.year	70[B] (2, 3, 4)	61[B] (5)	–	–	–
(H) kg VS from feed + standoff pad manure/head.year	24 (2)	–	–	–	–
(I) kg VS from waste feed/head.year	14 (2)	–	–	–	–
(J) kg manure + feed from house flushing VS/head.year	–	–	78[C] (3, 5)	–	–
(K) Density CH_4 (g/L)	0.67 (2, 3)	0.67 (3)	0.67 (3)	–	–
(L) N excretion (kg/head.year)	–	–	–	82.88 (5)	0.69 (5)
(M) N_2O emission factor (kg N_2O/kg N excreted)	–	–	–	0.02 (3, 5, 6)	0.02 (3, 5, 6)
(N) Volatilisation factor (frac kg NH_3-N/kg N excreted)	–	–	–	0.3 (3, 5, 7, 8)	0.3 (3, 5, 7, 8)
(O) Secondary N_2O factor (frac NH_3-N converted to N_2O-N)	–	–	–	0.01 (3, 5, 9, 10,11, 12)	0.01 (3, 5, 9, 10, 11, 12)
(P) Global warming potential CH_4	21 (3)	21 (3)	21 (3)	–	–
(Q) Global warming potential N_2O	–	–	–	310 (3)	310 (3)

[A]Methane conversion factors for 'lagoons' from Mangino *et al.* (2001) used in inventories.
[B]Based on 6% of manure from milking shed entering ponds.
[C]Back-calculated from inventory's VS excretion rates, B_0, MCF, population data and nationwide emission rate.

Table 2.   Formulae used to calculate greenhouse gas (GHG) emissions from largest 'manure management' GHG sources in Australia and New Zealand (refer to Table 1 for symbols)

GHG source	Emission calculation formulae
Anaerobic dairy effluent ponds NZ – CH_4	Nationwide emissions, Mt CO_2-e/year (E) = $\sum$: $(A \times G \times B \times E \times K/1 \times 10^{\wedge 9} \times P)$; $(A \times H \times B \times E \times K/1 \times 10^{\wedge 9} \times P)$; $(A \times I \times C \times E \times K/1 \times 10^{\wedge 9} \times P)$; $(A \times F \times D/1 \times 10^{\wedge 9} \times P)$
Anaerobic dairy effluent ponds Aust. – CH_4	$E = (A \times G \times B \times E \times K/1 \times 10^{\wedge 9} \times P)$
Anaerobic pig ponds Aust. – CH_4	$E = (A \times J \times B \times E \times K/1 \times 10^{\wedge 9} \times P)$
Beef feedlots and poultry Aust. – direct N_2O	$E = (A \times L \times M/1 \times 10^{\wedge 9} \times Q \times 44/28)$
Beef feedlots and poultry Aust. – secondary N_2O	$E = (A \times L \times N \times O/1 \times 10^{\wedge 9} \times Q \times 44/28)$

(2004) upon which the NZ Inventory is based (Table 3). This is perhaps not surprising given the McGrath and Mason (2004) study employed a methodology that required an individual to stand beside a pond and not only manually count the CH_4 bubbles but also to categorise their size.

The estimate by Chung *et al.* (2013) also incorporated the CH_4-producing potential of wastes which enter ponds in addition to the milking shed manure. These include: manure from feed and standoff pads (i.e. hard surfaces used for providing supplementary feed and keeping cows off sodden pastures in wet periods); waste milk; and waste supplementary feed residues. It is significant to note that neither the NZ GHG Inventory nor indeed the generic international IPCC methodology make any account for these additional sources of waste that enter anaerobic ponds. While these wastes might have been assumed to have minor gas-emitting potential when the inventory was compiled, this is certainly not the case as evidence now shows that they can generate as much CH_4 as the milking shed manure itself (Chung *et al.* 2013).

The use of feed and standoff pads is also a critical issue when quantifying pond CH_4 emissions. Using survey data, Chung *et al.* (2013) estimated feed pad use is currently ~20% across all NZ dairy farms. However, the rate of feed pad use is rapidly increasing and in some areas, is already nearing 100% (Northland Regional Council, pers. comm., 2012). Moreover, the Ministry for Agriculture and Forestry (pers. comm., 2012) stated that the use of feed, wintering, and stand-off pads will undoubtedly increase in coming years. In NZ, effluent from these pads is generally flushed into an anaerobic storage/treatment system, thereby potentially boosting CH_4 emissions

Table 3. Relationship between inventory estimate of emissions and field-measured emission rates for various 'manure management' sources, calculated from published literature

Author	Emission source	Measurement technique/duration	Emission factor (EF) (kg CO_2-e/kg liveweight.year)	EF given by national inventory
Williams (1993)	Dairy CH_4 Aus. – ponds	Vented chamber/1 day during dry period	0.66[A]	0.52
Chung *et al.* (2013)	Dairy CH_4 NZ – ponds	Using IPCC inputs	0.70[B]	0.16
Craggs *et al.* (2008)	Dairy CH_4 NZ – ponds	Floating cover/1 year	0.71[C]	0.16
McGrath and Mason (2004)	Dairy CH_4 NZ – ponds	Observational bubble counting/3 months	0.34[D]	0.16
Maraseni and Maroulis (2008)	Pigs CH_4 Aus. – ponds	Not described	6.01[E]	8.13
Craggs *et al.* (2008)	Pigs CH_4 NZ – ponds	Floating cover/1 year	1.20[F]	2.03

[A]Holstein-Friesian crossbreeds on farm, same as for national population (Dairy Australia 2013). No weight-adjustment applied in calculations.
[B]Liveweights from NZ GHG Inventory used.
[C]Breed on studied farm not reported, assumed same as national population. No weight-adjustment applied in calculations.
[D]Breed on studied farm Friesian-dominated, average bodyweights similar (Piccand *et al.* 2013) to inventory estimate (460 kg/head), no weight-adjustment applied in calculations.
[E]Weight-corrected for 53 kg/animal for studied farm to 60 kg/head for national population (Australian Government 2013).
[F]Type of pigs not reported, no weight-adjustment applied in calculations.

by an order of several Mt CO_2-e/year. Without doubt, accurate and updated survey data on feed and standoff pad use across NZ are crucial from a GHG emission perspective. Yet this potentially large additional CH_4 source is currently not considered by NZ's GHG Inventory, or indeed by IPCC methodology.

Similar to NZ, the main GHG emitted from manure on Australian dairy farms is CH_4 generated from anaerobic effluent ponds. The Australian GHG Inventory estimates that ~0.5 Mt CO_2-e/year of CH_4 are emitted from dairy effluent in Australia (Fig. 1): ~0.4 Mt from ponds, the other 0.1 Mt coming from manure deposited directly on pastures (Australian Government 2013). The Australian Inventory purports that 6% of dairy cattle manure is deposited at the milking shed; the same as for NZ. However, information from the Victoria government (pers. comm., 2013) suggested that this value is more likely 15–20%. Clearly, more survey data are needed to define this parameter as uncertainty in this one factor alone represents a potential error of 200–300% for the nationwide emission estimate.

One publication was identified reporting CH_4 emissions from an Australian dairy effluent 'pond' (Williams 1993). Emissions from this 'pond', a waterlogged depression, were higher per kg liveweight than the Australian Inventory's estimate of effluent pond CH_4 (Table 3). This higher emission rate could be attributable to farm-to-farm variations in animal genetics, feed regimes and environmental conditions. Importantly, it may also result from the omission of waste milk and feed in the Australian GHG Inventory's methodology. Chung *et al.* (2013) showed that waste milk, which enters ponds from pipe washdown and tank cleaning, could account for ~25% of CH_4 emissions from dairy farm effluent ponds.

Dairy Australia (2013) report that feed pad use is less than 2% on Australian dairy farms. However, Gourley *et al.* (2007) noted that from the last decade, farming operations are intensifying so the trend to increased feedlot use is expected. Indeed, a continuing increase in feed pad use was confirmed by an industry representative (Department of Environment and Primary Industries, Victoria, pers. comm., 2013). This will

have the potential to increase CH_4 emissions from dairy effluent, with more manure being stored under anaerobic conditions, yet as for NZ, the additional wastes entering ponds from feed pads are not accounted for in the Australian GHG Inventory.

To summarise, the following significant concerns exist regarding dairy cattle emission estimates:
(1) The basis of the NZ Inventory has numerous problems including the validity of its source data (it also contain several key mathematical errors; refer to Fig. 1 for details).
(2) There are conflicting reports as to the amount of manure deposited at the milking shed with estimates from NZ and Australia that vary by 200–300%.
(3) The impact of waste milk discharges and wasted feed that enters anaerobic ponds is currently overlooked and could represent a considerable CH_4 source.
(4) While the dairy industry is moving towards far greater use of feed pads, the resultant increased manure load is currently unaccounted for by GHG inventories.

Further to the above issues pertaining to pond CH_4 emissions, there also appear to be inconsistencies regarding CH_4 emissions from pasture-deposited manure reported in the Australian and NZ GHG inventories (Australian Government 2013; New Zealand Government 2013). Emissions from this source are almost the same in both countries (0.1 Mt CO_2-e/year). Although both countries have the same pasture disposal rate (94%) NZ has almost twice as many dairy cows as Australia. Clearly there are serious questions over the accuracy of dairy effluent CH_4 emissions for what is the largest GHG 'manure management' source for NZ and Australia combined.

GHG emissions from pig 'manure management' (Australia)

Methane is the main GHG produced on pig farms and is generated by the anaerobic degradation of effluent in ponds. The Australian GHG Inventory reports an annual CH_4 production of ~1 Mt CO_2-e from pig effluent (Fig. 1) based on the input data and methodologies shown in Tables 1 and 2.

The inventory's estimate of CH_4 emissions is ~8.13 kg CO_2-e/kg liveweight.year. Maraseni and Maroulis (2008) reported a similar emission rate, 6.01 kg CO_2-e/kg liveweight.year, from the pond of a 15 000-head farm in Victoria (Table 3). This direct experimental evidence provides some verification that the national calculations are grounded in accurate CH_4 emission factors. In addition to accurate emission factors, an understanding of the breakdown of pig 'manure management' practices in use across the country is essential because this can considerably affect nationwide GHG emissions. For the Australian pig industry, the inventory estimates that ~80–90% of effluent is discharged to ponds and that the remaining fraction is managed via a combination of dry storage/litter techniques to which very low CH_4 conversion factors (5% of biochemical CH_4 potential) are ascribed. By comparison Australia Pork Limited (APL, pers. comm., 2013) noted that ~91% of pig effluent is treated in either conventional housing (where all effluent is managed in ponds) or a combination of conventional housing and deep litter (where some to all effluent is managed in ponds). The Australian Inventory's partitioning of 'manure management' systems on Australian pig farms is, thus, apparently based on sound 'manure management' data.

GHG emissions from beef feedlots (Australia)

Feedlots are commonly used for finishing beef cattle from pastures and croplands in Australia. The Australian GHG Inventory reports that ~5% of Australia's beef cattle population [25–30 million, MLA (2013)] is on feedlots at any time (i.e. 1.17 million cattle) and that ~0.95 Mt of CO_2-e/year as direct N_2O is emitted from these feedlots (Fig. 1); almost one-third of total 'manure management' emissions. But are the emission factors which are used to calculate this estimate accurate? The direct N_2O emission factor (0.02 kg N_2O/kg N excreted) represents the average outcome from complex biological mechanisms depending on ecological and environmental conditions, and was derived from IPCC expert judgement and research by Külling *et al*. (2003). Külling *et al*.'s (2003) study was conducted in Switzerland using dairy rather than beef cattle manure. Moreover, the key processes affecting the N_2O emission factor, which include moisture (Montes *et al*. 2013), manure physical parameters (Chadwick 2005) and temperature (Dobbie and Smith 2001), are likely very different in Switzerland than Australia. For example, emissions from the manure in Külling *et al*.'s (2003) study were assessed at 20°C. By contrast, temperatures on Australian beef feedlot surfaces of 45°C are common (Redding *et al.* in press) and can reach up to 60°C (Queensland Department of Agriculture, Fisheries and Forestry, pers. comm., 2013). Manure on feedlot surfaces is often cracked and moisture content varies considerably from the surface to the base, in contrast with the manure assessed by Külling *et al*. (2003), which was a wet slurry. In short, the vastly differing conditions of Külling *et al*.'s (2003) study compared with beef feedlots in Australia raise serious questions concerning the use of the northern hemisphere default factors in the Australian GHG Inventory.

Few published studies have verified the appropriateness of the above emission factors for Australia and no publications were identified reporting direct N_2O emissions from beef feedlots in the country. Two studies conducted in the USA documented direct N_2O measurements from beef cattle feedlots. Rahman *et al*. (2013), using a wind tunnel, reported an N_2O emission rate of 4.67 kg CO_2-e/kg liveweight.year at a North Dakota feedlot, over four seasons. This rate is more than three times higher than the Australian GHG Inventory emission estimate (1.53 kg CO_2-e/kg liveweight.year). By contrast, Borhan (2011), using flux chambers, reported 0.13 kg CO_2-e/ kg liveweight.year (as direct N_2O emissions) from a Texan beef feedlot over summer. The variability between these field measurements may have been due to differences in management practices, environmental conditions and measurement techniques. Rochette and Eriksen-Hamel (2008) noted that N_2O measurements determined using chambers can yield highly variable results from site to site, and techniques applied influence the emission rate measured. Nonetheless, the emission fluxes reported from the two studies were not only markedly different from each other (×30 difference); they also differed considerably variable from the Australian GHG Inventory estimate (×3 and ×12 difference). This highlights the need for actual field measurements of direct N_2O emissions from beef feedlots in Australia. Even a single field study, conducted over a sufficiently long period, could be used to develop a more appropriate emission factor than currently adopted for Australia. Without such measurements it is difficult to ascertain the appropriateness of the default emission factors currently used by the inventory. Consequently, the estimate of direct N_2O emissions from Australian beef feedlots given in Fig. 1 remains unverified.

Secondary N_2O emissions from beef feedlots in Australia have also been estimated using IPCC default factors from the 1995 Second Assessment Report. The NH_3 volatilisation factor (0.3 kg NH_3-N/kg N excreted) used in Australia's GHG Inventory is sourced from publications by Hutchings *et al*. (2001), who focussed on Denmark's gas inventory and Rotz (2004), in their review on farm N management. Two Australian studies measured NH_3 emissions from beef feedlots: Denmead *et al*. (2008) thus reported 66 g NH_3-N/kg liveweight.year, while Loh *et al*. (2008) documented an average 108 g NH_3-N/kg liveweight.year. Both studies used feedlots from Victoria and Queensland and employed open-path micrometeorological techniques to measure emissions. The NH_3 fluxes reported are higher than the national GHG inventory estimate of 48 g NH_3-N/ kg liveweight.year, which suggests that a nationally relevant NH_3 volatilisation rate is more likely in the range of 0.4–0.7 kg NH_3-N/kg N excreted. Further process work would be required to confirm this.

The secondary N_2O emission factor (0.01 kg N_2O-N/NH_3-N), adopted from the IPCC, is sourced from Butterbach-Bahl *et al*. (1997); Brumme *et al*. (1999); Denier van der Gon and Bleeker (2005); and Corre *et al*. (1999). These publications are all based on northern hemisphere studies. Butterbach-Bahl *et al*. (1997) and Brumme *et al*. (1999) determined secondary N_2O emissions from German native forests, based on assumed atmospheric N deposition rates. Corre *et al*. (1999) did the same for various land-use types in Canada while Denier van der Gon and Bleeker (2005) determined secondary N_2O emission factors for the Netherlands based on literature estimates of N deposition rates. Actual N deposition rates were not measured in any of these investigations.

Moreover, the studies did not specifically examine the conversion of ammonia-deposited N to N_2O from manure applications. Hence, as is the case with the direct N_2O emission factors, the relevance of secondary N_2O emission factors to agricultural systems in Australia is questionable.

In addition to unverified N_2O emission factors, there is uncertainty regarding Australia's beef feedlot population. The Inventory's estimate of 1.17 million for 2011 differs from other sources: Muir *et al.* (2011) noted that ~680 000 cattle are on Australia's feedlots at any time, while the industry itself reported a feedlot population of 788 000 in June 2011 (ALFA 2013). These variations can affect Australia's beef feedlot manure GHG emission estimate by up to 100%. Thus, it is crucial that the inventory adopts the most accurate population estimate, which should be that given by the industry.

Another important consideration in estimating beef feedlot manure GHG emissions is the practice of stockpiling (Fig. 2). Manure from feedlots can be cleaned-out regularly (<1 week) or as infrequently as <6 months (Dantzman *et al.* 1983) and is commonly stockpiled or composted before being applied to agricultural land (Queensland Department of Agriculture, Fisheries and Forestry staff, pers. comm., 2013). If feedlot manure is frequently removed, the corresponding GHG emissions will be low. However, emissions from the manure once it is applied to land may be high and will contribute to agricultural soils emissions (12 Mt CO_2-e/year for Australia). If manure is stockpiled or composted for a prolonged period, it could emit significant quantities of N_2O, NH_3 and CH_4 under suitable environmental conditions. For example, Pattey *et al.* (2005) demonstrated that GHG (N_2O and CH_4 combined) emissions from poorly managed beef manure compost can be seven times greater than from the same manure in well aerated composting piles; although additional secondary N_2O emissions through NH_3 volatilisation would need to be considered.

Emissions from stockpiled or composted manure are not quantified in the Australian GHG Inventory. Moreover, as no published field-measured emissions from manure stockpiling exist it is difficult to ascertain whether these storage systems are significant GHG sources. Work is needed to elucidate GHG emissions from stockpiled and composted beef feedlot manure in Australia.

GHG emissions from poultry litter management (Australia)

According to the Australian GHG Inventory, direct and secondary N_2O emissions from poultry shed litter are responsible for ~0.75 Mt CO_2-e/year (Fig. 1). The input parameters used to calculate these N_2O emissions are summarised in Tables 1 and 2. No published studies have reported N_2O or NH_3 fluxes from Australian poultry sheds. Robertson *et al.* (2002) recorded an average NH_3 flux of 0.2 kg/head.year from a meat-poultry shed in the UK. Using the default emission factor of 0.01 for secondary N_2O emissions (in lieu of more applicable data) this equates to ~0.1 Mt CO_2-e/year when transposed to Australia's national poultry population; similar to the Australian GHG Inventory's estimate of poultry secondary N_2O emissions (Fig. 1). Yet, as was demonstrated in the beef 'manure management' discussion, published country-specific data are required to validate the use of the northern hemisphere factors for predicting poultry manure N_2O emissions because conditions in Australian poultry sheds could be very different from those in Europe. We are aware of work underway measuring N_2O and NH_3 emissions in poultry sheds across Australia. Data from this project, which is expected to be completed in mid-2015, will provide valuable insight into the accuracy of the emission factors currently adopted by the Australian GHG Inventory.

Other sources

The remaining 'manure management' GHG emission sources in Australia and NZ each produce less than 0.1 Mt CO_2-e/year, according to the respective national Inventories. Some of these emission profiles are well developed, such as NZ's estimate for pig effluent pond CH_4, which is based on emission factors verified by field measurements (Craggs *et al.* 2008) and recent manure management survey data (Hill 2012). Other sources, however, may emit greater quantities of GHG than currently estimated by inventory protocols. For example, secondary N_2O emissions from dairy effluent ponds in NZ are not accounted for in the national GHG inventory. Yet, in a study by Pratt *et al.* (2012), measured NH_3 emissions from an anaerobic dairy effluent pond indicated a nationwide secondary N_2O production rate of 0.4 Mt CO_2-e/year (using Table 1's secondary N_2O emission
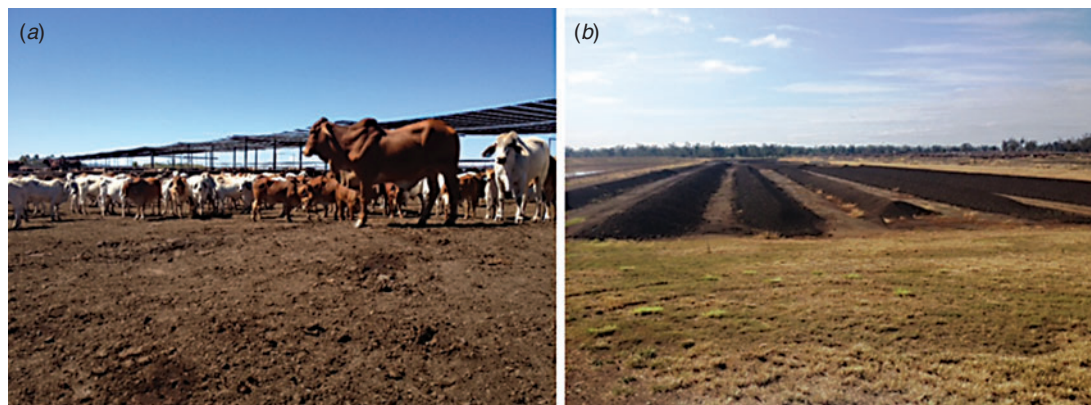
Fig. 2. Manure that is collected on beef feedlot surfaces (*a*) is often removed regularly and either stockpiled or composted in windrows (*b*) before being applied to land.

factor). The accuracy of this estimate is compromised by the extremely short monitoring period (2 days) on a 4-m^2 section of one pond, and the questionable appropriateness of the secondary N_2O emission factor. Nonetheless, the NH_3 emissions reported on this pond warrant further field measurements in this area.

GHG mitigation options and implications for research policy

Mitigation options across the sectors

There are many practical options for mitigating CH_4 emissions from dairy effluent ponds including those summarised in Table 4. In NZ, agriculture has been exempted from the country's emissions trading scheme and therefore there is a lack of incentive to implement any significant technology changes for the sake of addressing greenhouse gas emissions *per se*. However, the economics of these technologies are sensitive to volumetric loading rates (i.e. the amount of wastes anaerobically treated per unit of pond volume) and with research have significant potential to be further optimised. Furthermore, changes in management practice such the increase in the use of feed pads, and thus more collected manure, significantly increase the production of biogas; this can significantly improve the profitability of schemes such as CH_4 recovery for energy (Shilton *et al*. 2009).

In Australia the principal manure GHG source is CH_4 emissions from pig farms and the current estimate of these emissions appears to be well supported by industry data and field measurements. The potential for efficient energy recovery, and therefore CH_4 mitigation, from pig ponds is typically greater than for dairy ponds due to their higher CH_4 production rates. Several pig farms in Australia, and NZ, have deployed gas combustion systems on their effluent ponds, and the quantities of energy recovered have made this option economically viable (Fig. 3). This follows years of experience in viable energy recovery from livestock farms in Europe; Wilkinson (2011) reports ~4000 farm-scale digesters in Germany alone. In addition to energy recovery, Australia's Carbon Farming

Initiative – a federal government program which enables farmers and landowners from all farming industries to claim financial reward based on demonstrable GHG reductions – entails further potential incentive for pig farmers to capture their effluent CH_4 emissions (Maraseni and Maroulis 2008). Thus, there is the real possibility that emissions for this source could soon be greatly reduced.

Direct and secondary N_2O emissions from beef feedlots and poultry sheds are the other major sources of 'manure management' GHG emissions in Australia. Strategies to

Table 4. Estimated efficiencies of anaerobic dairy effluent pond methane (CH_4) mitigation options (adapted from Shilton *et al*. 2009)

Mitigation approach	% CH_4 reduction achievable	Comments on confidence in effectiveness of technology
Cover pond and flaring/ with or without energy recovery	97	Approach has been implemented successfully on other livestock farms (e.g. pig farms). Effectiveness of flaring required validation
Biofilter cover, comprising methane-eating bacteria, on pond surface	98	Biofilter cover design has not been field tested. Percentage CH_4 reduction shown is based on preliminary study (Pratt *et al*. 2013)
Conversion of anaerobic to aerobic/facultative ponds	73	Easily achievable for farms, which have 2-pond treatment systems (i.e. ~40% of NZ farms). % CH_4 reduction based on theoretical calculations, field measurements required
Solids separation – i.e. physically preventing solids from entering anaerobic ponds and depositing them in aerobic conditions on pastures instead	80	% CH_4 reduction based on theoretical calculations, field measurements required
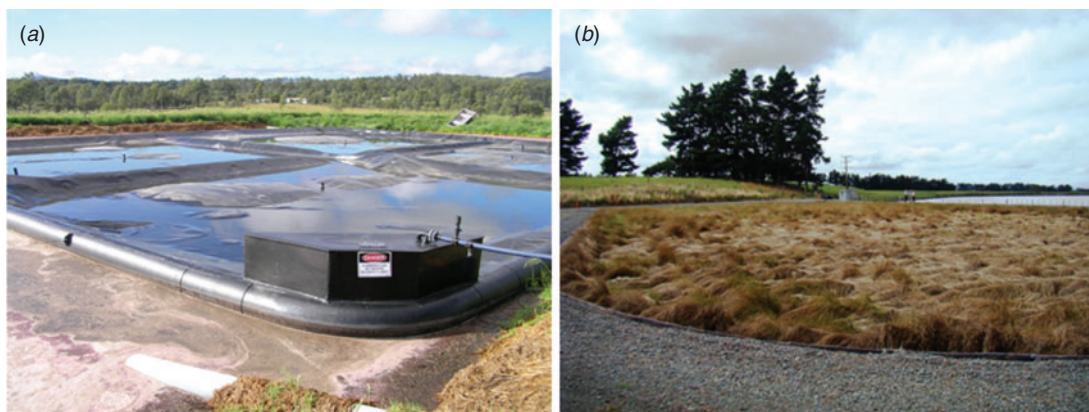
Fig. 3. Examples of methane (CH_4) mitigation options for effluent ponds (*a*) a gas capture system on a piggery effluent pond in Australia; (*b*) a biofilter for reducing nitrogen leaching on an industrial wastewater treatment pond in New Zealand. Research (Pratt *et al*. 2013) has shown that CH_4-consuming bacteria could be embedded in the filter and achieve almost 95% CH_4 reductions from dairy ponds.

mitigate these emissions may be implemented at the farm-management level. For example, de Klein *et al.* (2001) suggest tightening the supply of manure N to crops as a way to restrict direct and secondary N_2O emissions. Reducing the amount of N excreted from livestock through dietary modifications could reduce gaseous emissions. Several researchers have shown that a decrease in excreted N decreases ammonia emissions, which would likely lead to reduced secondary N_2O emissions (Canh *et al.* 1998; Hayes *et al.* 2004). However, research is needed to conclusively demonstrate a link between decreased N excretion rates and lower direct N_2O emissions. In terms of technical mitigation options, the nitrification inhibitor dicyandiamide has been shown to reduce pastoral N_2O emissions by ~25% in NZ (Monaghan *et al.* 2013) and could conceivably be applied to mitigate feedlot direct N_2O emissions. However, recent concerns have emerged regarding the detection of dicyandiamide in milk and its effects on animal and human health are unknown. Moreover, this compound is less effective in warm climates, so its efficiency in the Australian environment is likely suboptimal.

Options to reduce NH_3 emissions (the precursor to secondary N_2O) from feed pads and poultry sheds include the use of sorbing materials, which could be incorporated into the manure management systems through direct application or by adding to the animals' feed. Such materials have been shown to be effective in preliminary trials (Redding 2011). Varel *et al.* (1999) showed that urease inhibitors can reduce NH_3 emissions from feedlot cattle waste. It is also conceivable that appropriate management of vegetation around intensive farming systems (such as beef feedlots and poultry sheds) could reduce indirect N_2O emissions via tighter cycling of NH_3, which is volatilised from manures. Nonetheless, it should be borne in mind that discussing the wider potential strategies to mitigate N_2O from Australian beef feedlots and poultry farms is premature given the uncertain emission estimates for these sources.

Policy implications of inaccurate estimates: considerations in striving for best returns on research investment

Currently, no technologies are being widely deployed to mitigate 'manure management' GHG emissions in Australia or NZ. To some extent this can be attributed to the impression that enteric GHG emissions greatly overshadow manure emissions, given by inventories of beef-dominant countries. However, for countries with strong dairy and pig industries, such as Australia and NZ, this is not the case. The lack of emphasis on mitigating manure emissions can also be ascribed to the inaccurate and significant underreporting of emissions from these sources as discussed above. To illustrate this point we consider the following example regarding NZ's dairy farming industry where the dogma exists that effluent pond CH_4 emissions are negligible compared with enteric emissions and, thus, no funding is directed to manure emission mitigation research. Table 5 shows recent literature values of maximum CH_4 reduction possible from various enteric CH_4 mitigation assessments.

The NZ Government recently invested ~$50 million for the next decade for research into agricultural GHG mitigation (NZAGRC 2012). Based on the breakdown of funding reported for 2011 (NZAGRC 2011), it appears that ~35% ($17 million) will be allocated to mitigating enteric CH_4 emissions. By contrast, presently there is zero funding allocated to pond CH_4 mitigation technologies. However, in terms of the 'state-of-readiness' of the technologies presented in Table 5, Eckard *et al.* (2010) noted that most enteric mitigation options are far from developed to the stage of being adopted. They comment that rumen manipulation technologies require much more research as vaccine use is controversial and enzymes have yet to show any sustained positive results. Patra (2012) remarked that most enteric mitigation strategies have not been tested in long-term experiments and, thus, require extensive future research. Although the magnitude of the costs

Table 5.   Upper estimates of reductions in dairy (and other livestock) enteric methane (CH_4) emissions reported in the literature for a range of experimental mitigation approaches

Reference	Mitigation approach	Description	Duration of study	Maximum CH_4 reduction (per cow, unless otherwise stated)
O'Neill *et al.* (2011)	Diet modification	Switched from grass to mixed ration diet	10 weeks	11%
Monteny *et al.* (2006)	Diet modification	Switched from intensive to extensive grazing	1 year	21%
Beauchemin and McGinn (2006)	Diet modification	Added oils and acids to diet	6 months	0%
Boadi *et al.* (2004)	Diet modification	Improved feed quality intake	Based on review	26%
	Diet modification	Switched grazing forage	Based on review	21%
	Diet modification	Altered feeding regime	Based on review	9%
	Diet modification	Added palm kernel oil to feed	Based on review	34%
Grainger *et al.* (2008)	Rumen manipulation [A]	Added monesin	5 months	0%
Boadi *et al.* (2004)	Rumen manipulation [A]	Added bacteriosin	Based on review	50%
	Rumen manipulation [A]	General estimate on potential of vaccines	Based on review	20%
Eckard *et al.* (2010)	Rumen manipulation [A]	Vaccine for sheep	Based on review	8%
	Selective breeding	General estimate	Based on review	10–20%
Boadi *et al.* (2004)	Selective breeding	NZ study	Based on review	8–9%
	Selective breeding	Canadian study	Based on review	21%

[A]Rumen manipulation targets the micro-fauna in livestock to reduce CH_4 production. This is generally achieved by the injection of vaccines or enzymes, which inhibit methanogens in the rumen.

Table 6.   New Zealand dairy methane (CH_4) mitigation opportunities

CH_4 source	Enteric	Anaerobic ponds
Projected quantity of CH_4 potentially mitigated (Gg CO_2-e per decade)	19 700	13 870
Projected value of CH_4 mitigation ($ million/ decade)[A]	492[B]	336[C]
Allocated research funding over next 10 years ($ million)	17	0
Broad cost/benefit ratio of research investment	29	No investment[D]
State-of-readiness (risk)	'require many years of research before practical and commercially viable products and strategies are available for use on farms' Eckard *et al.* (2010)	'close to market' readiness

[A]Based on CO_2-e price of $25 per tonne, does not factor in implementation costs.
[B]Assumes 20% reduction of emissions from enteric processes (through vaccines, diet manipulation or selective breeding).
[C]Assumes 97% reduction for waste emissions (through covering ponds and burning the gas).
[D]With $1 million investment over next decade, becomes 336 (>10 times higher than for enteric emissions).

associated with enteric mitigation technologies is not yet known, it is likely that they will be significant. Eckard *et al.* (2010) stated that it will be many years before practical and commercially viable technologies are ready for use on farms. Some mitigation approaches may be used in combination to produce a cumulative effect in CH_4 reduction. Even factoring in this 'combined effect' Eckard *et al.* (2010) noted that maximum CH_4 reductions that could be achieved for pasture-based systems (through selective breeding and grazing management) would be just 20%.

By contrast to enteric mitigation technologies, the 'state-of-readiness' for pond CH_4 mitigation strategies is high but further research is needed to optimise and practically demonstrate the technologies, as their uptake is currently low. The numbers in Table 6 show that, with a more balanced research portfolio, the NZ Government could achieve significant reductions in both enteric and pond CH_4 emissions.

The case study on the current research investment portfolio for NZ dairy CH_4 emissions has an important implication for the wider GHG 'manure management' category in Australia and NZ. It highlights the need to develop accurate emission profiles as a prerequisite to balance GHG mitigation research efforts to best-effect. In this respect, it is crucial to verify N_2O emission estimates from beef feedlots and poultry farms in Australia because, based on current data, we simply do not know if these are significant agricultural GHG sources. This highlights the wider importance for countries to scrutinise all of their major GHG emission profiles, particularly if such assessments have not been made since the compilation of original inventories.

Conclusions

GHG emissions from livestock 'manure management' in Australia and NZ are more than 5 Mt CO_2-e/year combined, according to best-available estimates. Methane from NZ and Australian dairy farms, CH_4 from Australian pig farms and N_2O from Australian beef feedlots and poultry sheds appear to be the major sources of livestock manure emissions. While the estimate of pig farm CH_4 emissions appears reasonably accurate (based on industry data and field measurements), CH_4 emission estimates from dairy farms are underestimated, and in particular increasing use of feed and standoff pads will lead to greater quantities of manure yielding CH_4 from treatment/ storage under anaerobic conditions. Consequently, 'manure management' in these countries could be a much more significant GHG emission source than currently reported. The accuracy of N_2O emission estimates from beef feedlots and poultry sheds in Australia remains uncertain because they are based entirely on northern hemisphere emission factors whose appropriateness for Australia is questionable and unverified. Therefore, most of Australasia's key livestock 'manure management' GHG emission profiles are apparently unsubstantiated. Encouragingly, there is a range of options available to mitigate manure GHG emissions. However, opportunities to mitigate emissions from the largest 'manure management' GHG source, i.e. dairy effluent pond CH_4 in NZ, are currently being missed due to imbalanced investment into mitigation research by the government, which stems from an underreporting of manure CH_4 emissions by the NZ GHG Inventory. This is particularly alarming as this emission point source has several 'close-to-market' abatement solutions and therefore represents a real opportunity for meaningful action soon compared with the more difficult-to-implement, long-term alternative strategies that are currently being invested in. Internationally, this review demonstrates the critical importance for countries to re-scrutinise their GHG emission profiles, particularly if such assessments have not been made since the compilation of original inventories. Failure to act in this regard presents the very real risk of missing the 'low hanging fruit' in the rush to a meaningful response to climate change.

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