Manure management: Implications for greenhouse gas emissions

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ABSTRACT

Slurry, farmyard manure and poultry manure are an inevitable consequence of livestock products generated from housed animals. These manures are recycled back to land for plants to use the nutrients they contain. However, since they contain inorganic N, microbially available sources of C and water, they provide the essential substrates required for the microbial production of N_2O and CH_4. These greenhouse gases can be produced and emitted at each stage of the 'manure management continuum', being the livestock building, manure stores, manure treatment and manure spreading to land. The contribution that manure management makes to total national agricultural emissions of N_2O and CH_4 vary, but can exceed 50% in countries reporting to the UNFCCC in 2009. On farm management decisions interact with environmental controls such as temperature and water availability of key microbial processes (i.e., nitrification, denitrification, methanogenesis, CH_4 oxidation), affecting the magnitude of emissions from each stage of the manure management continuum. We review the current understanding of how manure management influences direct and indirect N_2O emissions and CH_4 emissions, introduce new data comparing direct N_2O emissions following spreading of a range of manure types by different methods, and highlight some of the mitigations being considered by researchers and policy makers in developed and developing countries.

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1. Scope and introduction

There is much interest in understanding effects of manure management on direct and indirect source of greenhouse gas (GHG) emissions, as manures contain substantial quantities of N, much of which is in inorganic forms (Anon., 2010), C (e.g.,

Table 1
Composition of some typical livestock manures.

g/kg fresh weight	Dry matter	N	Ammonium-N	Nitrate-N[a]	Uric acid-N	C:N[b]
Cattle slurry	60	2.6	1.2	0	0	9
Pig slurry	40	3.6	2.5	0	0	3
'Old' cattle FYM	250	6	0.6		0	13
'Fresh' cattle FYM	250	6	1.2	0	0	
'Old' pig FYM	250	7	1.0		0	10
'Fresh' pig FYM	250	7	1.8	0	0	
Layer manure	350	19	6.7	0	2.9	5
Broiler litter	600	30	7.5	0	3.5	7

Source: Anon. (2010). These are typical values from a national dataset in the UK. There is likely to be substantial variability about each value depending on type of feed, livestock type and age, and storage conditions. These values are only indicative and will not be relevant for many circumstances.

[a] No NO_3^- present in slurry unless aeration has taken place. No NO_3^- in fresh solid manure unless active mixing in animal house.

[b] Typical C:N ratio. No distinction made between old and fresh FYM in the study (Chadwick et al., 2000b).
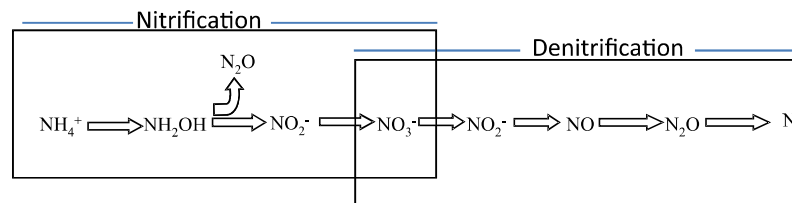

Fig. 1. Schematic chemical representation of two processes responsible for N_2O production.

Paul et al., 1993; Møller et al., 2004a) and water – three essential factors controlling processes leading to production and emissions of N_2O and CH_4. Whilst manures are a source of GHG, the management practice selected by farmers has the scope to influence the magnitude of gaseous losses, and the potential to reduce those emissions. In this paper we review the latest information and understanding of how manure management influences direct N_2O and CH_4 emissions, as well as highlight how manure management controls indirect sources of N_2O. We present emissions of these two GHG from all stages of the manure management continuum, being animal housing, yards, manure storage and treatment, and land spreading. We also introduce new data comparing direct N_2O emissions following spreading of a range of manure types using different methods. We also highlight some of the mitigations being considered. Grazing is not considered a part of the manure management chain, an approach similar to Sommer et al. (2009), even though N deposited in urine and faeces during grazing is a source of N_2O emissions (Oenema et al., 1997; Yamulki et al., 1998; Saggar et al., 2007; Thomas et al., 2008; Liebig et al., 2010). Since we cover the entire manure management continuum, it is not the intention to provide an in depth analysis of N_2O and CH_4 emissions of each phase. Rather, we highlight key papers and discuss general trends and contrasting information.

1.1. Processes of nitrous oxide and methane production

Nitrous oxide is generated by nitrification (Bremner and Blackmer, 1978) and denitrification (Firestone and Davidson, 1989), which occur in soil following addition of manure (Chadwick et al., 2000a; Sherlock et al., 2002; Fangueiro et al., 2008b,c, 2010; Singurindy et al., 2009). Emissions also occur from livestock bedding and solid manure stores (Chadwick, 2005; Thorman et al., 2006; Blanes-Vidal et al., 2008; Hassouna et al., 2008), as well as in the surface layer of stored slurry (Sommer et al., 2000). The majority of inorganic N present in slurry and fresh solid manure is in the form of ammonium (Sommer et al., 1992; Chadwick et al., 2000b; Burton and Turner, 2003; Fangueiro et al., 2008a; Anon., 2010; Table 1). Transformation from ammonium to nitrate via nitrification is a source of N_2O, and produces NO_3^- which is a source of N for denitrification, the biological reduction of nitrate to N_2 gas, where N_2O is an important product of incomplete denitrification, Fig. 1).

Most CH_4 of agricultural origin arises from enteric fermentation, with rice paddies also being a large source. Methane from manure is generated during anaerobic decomposition of organic matter in faecal matter and bedding material (Hellmann et al., 1997; Batstone et al., 2002; Batstone and Keller, 2003; Møller et al., 2004a,b). These organic compounds are degraded into other compounds such as volatile acids by acid producing bacteria. Methane producing bacteria then use the volatile acids to produce CH_4. The absence of oxygen is a precondition for production of CH_4 via microbial metabolism of organic material in livestock manure. Methane production from manure is affected by environmental factors such as temperature (Clemens et al., 2006; Sommer et al., 2007), biomass composition and management of the manure (Hill et al., 2001; Ni et al., 2008).

1.2. National legislation and greenhouse gas emissions

Emissions of CH_4 and N_2O from livestock production are regulated as part of the Kyoto Protocol under the United Nations Framework Convention on Climate Change (UNFCCC). The EU reduction target for GHG is 9% by 2008 to 2012, with reference to 1990, with a proposed further reduction target of 20% by 2020.

Other national or international legislation may impact strategies to alter direct and indirect GHG emissions from manure. For example in Europe the Nitrates Directive (91/676/EEC) concerns protection of waters against pollution caused by NO_3^- from agricultural sources. European member states have implemented action plans to meet their obligations to reduce NO_3^- losses to water. Management of livestock manures through practices such as controlling the timing of application of high available N content manures is a key aspect of these action plans and may therefore influence N_2O and CH_4 emissions from manure management.

Ceilings on national annual NH_3 emissions were included in the Gothenburg Protocol (United Nations Convention on Long-range Transboundary Air Pollution; CLRTP, 2004). Consequently, in some countries, slurry covers are required to reduce emissions and manures must be incorporated into soils rapidly after application or slurry must be applied using trailing hoses or shallow injection, to reduce NH_3 emissions (Webb et al., 2005), with subsequent implications for N_2O and CH_4 emissions.

Manures from livestock production systems are estimated to contribute 30 to 50% to the global N_2O emissions from agriculture (Oenema et al., 2005). The methodology for estimating national CH_4 and N_2O emissions from livestock manure based on emission factors was devised by the IPCC. We explored the contribution that manures make towards total national N_2O and CH_4 emissions in a number of countries using the 2009 submissions to UNFCCC of the 2007 GHG inventory (UNFCCC, 2009), an approach which uses the common reporting format for a range of countries (Table 2). Our assessment of the contribution of N_2O emissions from manures as a percentage of the total N_2O from agriculture, includes emissions from manure spreading and manure management in the animal house and the store, but excludes N_2O emissions from urine and faeces deposition by grazing/outdoor livestock. For example, in the UK, manure management and manure application to land contributed 16% of total agricultural N_2O emissions in 2007 (UNFCCC, 2009; MacCarthy et al., 2010). Actual contribution from manures is higher than that in Table 2, as the contribution from livestock manures accounts for direct N_2O emissions only and not for indirect losses from N deposition and N leaching. The proportion of N_2O from manure management can be much larger. For example emissions from manure management in Japan represent over 50% of the total agriculture emissions (Table 2). Differences in N_2O emissions from manure management among countries reflect differences in fertiliser, crop, livestock and manure management practices.

Methane losses from manure management are 12–41% of total agricultural CH_4 emissions for most countries, Australia being the exception, where few livestock are housed and little manure is managed. Differences in the proportional loss of CH_4 from manure management among countries reflect differences in the duration of manure storage, the proportion of ruminant livestock relative to other livestock types (Haeussermann et al., 2006; Sommer et al., 2009) and the extent of rice production and biomass burning.

Whilst IPCC inventory methodology can be useful in presenting the general contribution that manure management makes to total N_2O and CH_4 emissions from agricultural sources, the inventory methodology of many countries generally provides only a cursory understanding of effects of manure management on emissions, and hence how manure management might be manipulated to reduce them. Nitrous oxide emissions are affected by climate, soil type, application strategy and manure composition (Sommer et al., 2009). Whilst CH_4 emissions from manure management depends on the time manure is stored inside animal houses or in outdoor manure stores, temperature and manure composition (Sommer et al., 2004b; Haeussermann et al., 2006; Monteny et al., 2006). Therefore, there has been an increase in the number of research studies investigating effects of management and environmental conditions on GHG emissions from different stages of the manure management continuum.

1.3. The manure management continuum

Manure management is a continuum from generation by livestock to storage and treatment and finally to land spreading. There is the potential for N_2O and CH_4 emissions at each stage of this continuum (Fig. 2). For describing and estimating NH_3 emissions from the manure management continuum, a mass flow approach has been used (Webb and Misselbrook, 2004) as this allows effects of management at one phase that reduces emissions and conserves manure N to be considered as the manure passes to the next stage in the continuum. Other gaseous N losses, including N_2O, are included in this mass flow in a manner similar to that of Dämmgen and Hutchings (2008). The importance of this whole system approach is that effects of mitigation methods at one stage are considered in downstream stages (Sommer et al., 2009). However, as far as we are aware, there has not been an attempt to use a similar mass flow approach for C and CH_4 emissions.

2. Nitrous oxide emission from manure management

2.1. Nitrous oxide emissions from animal housing and collection yards

In animal houses that do not use bedding materials the slurry/faeces/urine remains in a predominantly anaerobic state with little opportunity for the NH_4^+ to be nitrified. As a result, little or no N_2O emissions are likely to occur from such systems

Table 2

Sources of N_2O (kt/yr) from selected countries demonstrating the contribution of manures to the total agricultural emissions in 2007.

kt/yr	Direct emissions							Indirect emissions		Total agric.	% from manure management
	Animal house and manure storage	Manure spreading	N fix crops	Crop residue	Grazing emission	Fert. emission	Other[a] emission	N deptn.	N leach		
Austria	2.83	2.05	0.44	0.79	0.72	1.96	0.03	0.56	3.00	12.38	39.4
Australia	5.14	2.22	1.82	2.41	12.65	9.27	0.00	11.53	8.44	53.53	13.7
Canada	15.45	7.25	0.00	16.85	12.56	22.79	0.28	9.02	25.29	109.69	20.7
Denmark	1.90	3.71	0.68	1.03	0.66	3.75	0.29	1.44	6.31	20.13	27.9
Japan	15.68	3.40	0.27	2.14	0.04	4.64	0.00	4.21	5.39	36.13	52.8
Portugal	1.85	1.09	0.05	0.48	2.43	0.96	0.01	0.60	2.42	9.89	29.7
Spain	9.61	8.74	3.68	2.21	5.23	17.54	0.74	3.45	22.07	73.27	25.0
UK	5.43	7.51	0.41	7.19	13.77	19.41	0.56	4.92	20.83	80.52	16.1

Source: National inventory submissions 2009 to the UNFCCC (http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/4771.php).

[a] Austria = sewage sludge spreading; Denmark = industrial waste and sewage sludge spreading; Portugal = not specified; Spain = domestic wastewater sludge and municipal solids waste compost; and UK = N fixed by improved grassland.
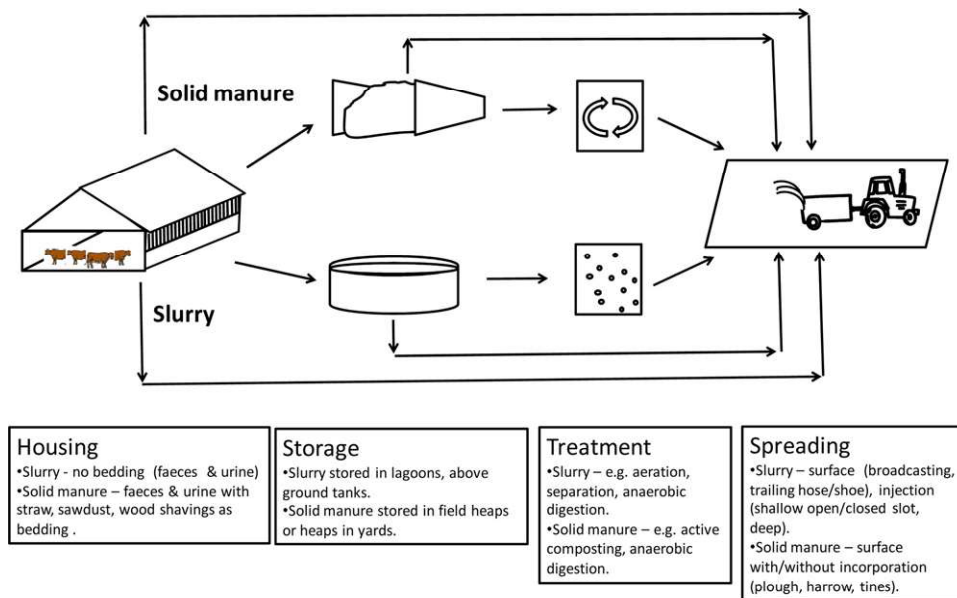
Fig. 2. A schematic representation of the sources of N_2O and CH_4 from the manure management continuum.

(Zhang et al., 2005). Thorman et al. (2003) compared N_2O and NH_3 emissions from cattle and pigs housed in slurry-based and straw bedded buildings and demonstrated that whilst there were N_2O emissions from cattle housed with straw bedding, 4–5 mg N_2O-N m^2/d, there were little or no N_2O emissions from slurry-based cattle or pig buildings.

Jungbluth et al. (2001) quoted studies which found emissions from cattle houses of 0.14–2.0 g N_2O/livestock unit (LU)/d, which corresponds to 0.05–0.7% of N excreted. Emissions of 0.66–3.62 g N_2O/LU/d have been measured from slurry-based pig houses with fully slatted floors (Costa and Guarino, 2009). Much higher emissions may occur frm deep litter systems with fattening pigs. For example, Groenestein and Van Fassen (1996) reported values of 4.8 and 7.2 g N_2O/LU/d, representing 50–60% of total N emissions from the litter with ammonia being the other major source of gaseous N loss. Mechanical mixing of deep litter in the building has been reported to further increase N_2O emissions (Groenestein et al., 1993), due to increased rates of nitrification followed by denitrification of NO_3^-.

There have been few measurements of N_2O (or CH_4) emissions from hard standing areas (*e.g.*, cattle collection yards). Those few measurements suggest very low or no N_2O emissions (Ellis et al., 2001; Misselbrook et al., 2001), due to the lack of transformation of NH_4^+ to NO_3^- under conditions of many UK animal collection yards where faeces and urine would normally be wet and predominantly anaerobic.

2.2. Nitrous oxide emissions from manure storage and treatment.

Solid manure stores provide aerobic and anaerobic conditions within close proximity (Hansen et al., 2006), and they can be a source of N_2O production/consumption and emission. Emissions of N_2O are typically from <1% to 4.3% (Table 3) of the total N in stored cattle and pig farmyard manure (FYM) heaps, but emissions as high as 9.8% have been reported (Webb et al., in press). Others have found that between 0.2 and 0.8% of total heap N was lost as N_2O from stored poultry manure heaps (Thorman et al., 2006) with covering of heaps lowering NH_3 emissions, but having no effect on N_2O emissions. However, Chadwick (2005) showed that covering and compacting cattle FYM heaps has the potential to markedly reduce N_2O and NH_3 emissions. Hence, maintaining anaerobic manure conditions is key to reducing N_2O emissions from solid manure heaps.

In the initial thermophilic phase of composting deep litter, production of N_2O is low (Czepiel et al., 1996; Petersen et al., 1998; Sommer, 2001) because N_2O producing nitrifying and denitrifying microorganisms are generally not thermophilic (Hellmann et al., 1997). After the thermophilic phase, N_2O production increases with N_2O production rates being substantial during the following low temperature period. Nitrous oxide emissions from stored manures with high concentrations of NH_4^+ are produced during nitrification (Hao et al., 2005; Hao, 2007; Yamulki, 2006) and as an intermediate product of denitrification (Lipschultz et al., 1981; Petersen et al., 1998).

Emissions from composting animal manure in passive aerated heaps and from turned livestock waste in wind rows is between 10 and 30.0 g N/t (Czepiel et al., 1996; Sommer, 2001). There is a tendency for N_2O emissions to increase with increasing density of composting manure heaps (Webb et al., in press), which may be due to microenvironments having low O_2 content. Nevertheless, monthly turning may promote aerobic conditions in the whole compost pile and decrease N_2O emissions from denitrification without any increase in NH_3 emissions (Szanto et al., 2007), although other studies have shown that turning solid manure heaps can stimulate N_2O (Parkinson et al., 2004) and other gaseous emissions (Hassouna

Table 3
N$_2$O emissions from solid manure stores.

	Aeration or static composting	Treatment	N$_2$O (EF)	Storage period (months)	Heap size (t)	Initial manure characteristics (fresh wt basis)					Measurement method
						DM (%)	Total N (kg/t)	Ammonium-N (kg/t)	pH	C:N	
Pig manure											
Petersen et al. (1998)	Static	Conventionally stored	0.5–2.0	3	4	24.6	11.5	1.3	–	8.2	Open chambers (0.2 x 0.7m)
Thorman et al. (2007a,b)	Static	Conventionally stored	2.63	12	4	25.4	7.8	2.2	8.4	–	Emission hood covered the entire heap
Cattle manure											
Amon et al. (2001)	Active aeration	Summer	0.36	3	7	28	6.6	1.1	7.6	14	Emission hood covered the entire heap
	Static	Summer	0.57	3	7	20	6.4	1.2	7.4	14	Emission hood covered the entire heap
	Active Aeration Static	Winter	0.45	3	7	22	6.7	0.6	8.7	16	
		Winter	0.88	3	7	21	6.3	0.4	8.2	15	
Chadwick (2005)	Static	Conventionally stored SP1	2.3	4	5	20	5.3	1.1	–	17	Emission hood covered the entire heap
	Static	Conventionally stored SP2	0.1	3	8	26	5.2	1.3	8.5	21	
	Static	Conventionally stored SP3	1.3	3	6	20	3.3	0.3	8.5	24	
	Static	Covered+compacted SP1	0.7	4	5	20	5.1	0.7	–	18	
	Static	Covered+compacted SP2	2.1	3	8	25	5.1	0.7	8.5	19	
	Static	Covered+compacted SP3	0.6	3	6	19	3.6	0.3	8.5	23	
Hao et al. (2001)	Passive aeration	Aeration pipes/windrows	0.62	3	>20	30	17.7	2.0	7.6	19	15.5 cm diameter vented chambers
	Active aeration	Turning of windrows	1.07		>20				7.6	19	
Thorman et al. (2007a)	Static	Conventionally stored	4.32	12	7	19.8	5.2	0.6	7.8	–	Emission hood covered the entire heap
Yamulki (2006)	Static	Organic	0.28	4	0.04	33.9	10.0	–	8.6	16	Emission hood covered the entire heap
	Static	Organic+Straw	0.26	4	0.04	38.1	9.8	–	8.0	19	
	Static	Conventional	0.70	4	0.04	30.0	9.1	–	8.6	15	
	Static	Conventional+Straw	0.48	4	0.04	41.7	8.0	–	8.2	19	
Poultry manure											
Thorman et al. (2006)	Static	Conventionally stored	0.17–0.81	3–6	5–6	–	–	–	–	–	Emission hood covered the entire heap
	Static	Covered (plastic sheet)	0.55–0.70								

SP: storage period.

et al., 2008). This presumably arises as a by-product of nitrification, as well as the result of generating a higher NO_3^- source via nitrification for N_2O loss via denitrification.

Yamulki (2006) and Sommer et al. (2000) conducted studies which showed that additional straw has the potential to reduce GHG emissions during solid manure storage. Yamulki (2006) demonstrated that the mixing of 50% by volume more chopped straw could reduce N_2O emissions by 32% from small scale stores of conventional cattle manure. The authors attributed this response to a higher initial C:N ratio (19 compared to 14) and dry matter (DM) content (41% compared to 30%) as a result of straw addition.

Slurry stores however, remain principally anaerobic unless O_2 is introduced as part of a treatment process, or windy conditions prevail. Nitrous oxide emission from slurry or liquid manure with no surface cover is negligible (Sommer et al., 2000). In contrast, stored slurry with a surface layer of straw or natural crust may be a source of N_2O emissions (Sommer et al., 2000; VanderZaag et al., 2009). The presence of a surface crust on cattle slurry can provide aerobic conditions in the crust and hence nitrification can occur with N_2O generation (Sommer et al., 2000). Cattle slurry will normally form a crust, whereas pig slurries rarely form crusts unless the DM content is unusually high. Some materials used to cover liquid manure stores to prevent ammonia emissions enhance crust formation and may lead to an increase in N_2O emissions during manure storage (Berg et al., 2006).

Slurry crust conditions can control N_2O emissions. For example, Sommer et al. (2000) showed that N_2O emissions increase at reduced water content of the surface layer, and may at low water content be as high as 25 mg N_2O-N m^2/h. Clemens et al. (2006) found average N_2O emissions corresponding to 30–50 mg N_2O-N m^2/h during winter and summer storage of cattle slurry, with highest rates where straw was used as a surface cover and temperatures were high. In the study of Sommer et al. (2000), there were no emissions during winter due to low temperatures and a high water content of the surface crust or surface straw layer.

Intensive aeration to remove excess N from livestock slurries has been shown to increase N_2O emissions (Willers et al., 1996; Béline et al., 1999; Loyon et al., 2007; Molodovskaya et al., 2008). Burton et al. (1993) showed that N_2O losses during intensive aeration could be up to 19% of total N in pig slurry.

Slurry separation is proposed as a treatment process to obtain a nutrient (*i.e.*, N, P) rich solid fraction and a liquid fraction (Fangueiro et al., 2008a), and to increase the potential slurry store storage capacity. However, the solid fraction is similar to untreated solid manure and has been shown to result in higher N_2O emissions during storage (Hansen et al., 2006), relative to untreated slurry due to the mix of aerobic/anaerobic conditions in the solid heap. Hansen et al. (2006) showed that as much as 4.8% of the initial total N content of separated solids from pig slurry was lost as N_2O over a 4-month period. Storage of the liquid fraction can lead to even lower N_2O emission relative to untreated slurry, and this decrease could be amplified if screw-press separation was combined with chemically enhanced settling to obtain a supernatant liquid fraction (Fangueiro et al., 2008a,b,c). But overall, slurry separation results in a marked increase in N_2O emissions during the storage phase of the different fractions, because of the large emissions from the stored solid fraction (Dinuccio et al., 2008; Fangueiro et al., 2008a).

Anaerobic digestion is designed to optimise conversion of available C into CO_2 and CH_4 (*i.e.*, biogas). Within the digester, a proportion of organic N is mineralised to NH_4^+, but there is little opportunity for nitrification of NH_4^+ to NO_3^- and N_2O emissions from stored digestate are not altered.

2.3. Nitrous oxide emissions from manure spreading

Application of manures to soil allows manure NH_4^+ to be subject to aerobic soil processes such as nitrification to generate soil NO_3^- (Chadwick et al., 2001). There is often a delay between manure application and N_2O emissions (Rochette et al., 2008; Fangueiro et al., 2010), generally attributed to the delay in mineralisation/nitrification and generation of a pool of soil NO_3^- as well as the time required for manure C to become available. There have been many reported studies of N_2O emissions following manure spreading (Chadwick et al., 2000a; van Groenigen et al., 2004; Rodhe et al., 2006; Thorman et al., 2007a; Fangueiro et al., 2008b,c; Rochette et al., 2008; Bertora et al., 2008; Sistani et al., 2010). Tables 4 and 5 summarise studies where N_2O emissions were measured after slurry and solid manure field spreading.

Emission factors (*i.e.*, cumulative N_2O-N loss as a proportion of total N applied in the manure) can range from <0.1 to 3% (Tables 4 and 5). Higher emissions (7.3–13.9%) have been measured during land application of pig slurry (Velthof et al., 2003). The range in N_2O emission factors following slurry and solid manure applications reflects differences in soil type, soil conditions (*i.e.*, temperature, water filled pore space), manure composition (*i.e.*, NH_4^+-N, C content and form) and measurement period. The low emission factors for land applied solid manure reflect the lower available N content of most of these manure types. Immediate N_2O emissions following manure application are generally the result of a source of NO_3^- within the manure (*e.g.*, stored or composted solid manure), or the effect of manure carbon fuelling denitrification of residual soil nitrate.

Tier 1 IPCC methodology ignores type of manure applied to land in providing the N_2O emission factor, as a common emission factor (EF) based on the proportion of N applied is used for all manure types. A large proportion of manure N is in the organic form (Table 1) and requires mineralisation followed by nitrification to form a manure derived NO_3^- pool for denitrification. Hence, it may not contribute much to short and medium term N_2O emissions. Also, large quantities of N are emitted via NH_3 volatilisation within 48 h (Webb and Misselbrook, 2004) following manure spreading, thus reducing the pool of N available for N_2O emission. Hence, it may be more useful to express N_2O emissions as a proportion of the inorganic

Table 4
N_2O emissions following cattle and slurry applications in field experiments.

	Application method	Site characteristics and season of application			Sampling period (d)	Slurry N application (kg/ha)		Slurry characteristics			N_2O-N emission		
		Soil texture	Season of application	Crop		Total N	Ammonium-N	DM (g/kg)	C:N	pH	Cumulative (kg/ha)	Total N EF (%)	Ammonium-N EF (%)
Cattle slurry													
Chadwick et al. (2000a)	Surface broadcast	Sandy loam	Spring	Grass	20	125	50	49	7	–	1.51	0.97	2.42
		Sandy loam	Summer	Grass	22	110	70	57	9	–	0.34	0.12	0.19
van Groenigen et al. (2004)	Row application then worked into soil	Sandy	Spring	Maize	180	104	47	99	–	–	0.47	0.31	1.00
		Sandy	Spring	Maize	180	156	70	99	–	–	1.28	0.73	1.82
		Sandy	Spring	Maize	180	209	94	99	–	–	1.70	0.75	1.81
		Sandy	Spring	Maize	180	261	117	99	–	–	1.92	0.68	1.63
		Clay	Spring	Maize	180	98	45	92	–	–	2.51	0.96	5.57
		Clay	Spring	Maize	180	147	38	92	–	–	3.01	0.95	4.45
		Clay	Spring	Maize	180	196	90	92	–	–	3.37	0.88	3.74
		Clay	Spring	Maize	180	245	113	92	–	–	6.81	2.03	6.04
Rodhe et al. (2006)	Injection	Silty clay	Summer	Grass	45	68	33	65	9	7.1	0.75	1.10	2.12
	Band spread	Silty clay	Summer	Grass	45	68	33	65	9	7.1	0.20	0.30	0.43
Rochette et al. (2008)	Incorporated	Clay	Spring	Maize	180	150	–	60	11	6.8	1.25	–	–
		Loam	Spring	Maize	180	150	–	60	11	6.8	2.12	–	–
		Clay	Spring	Maize	180	150	–	48	7	6.8	6.06	–	–
		Loam	Spring	Maize	180	150	–	48	7	6.8	1.09	–	–
Pig slurry													
Chadwick et al. (2000a)	Surface	Sandy loam	Spring	Grass	20	97	50	63	4	–	0.77	0.44	0.94
		Sandy loam	Summer	Grass	22	295	135	46	3	–	0.57	0.12	0.27
		Sandy loam	Autumn	Grass	21	300	280	25	2	–	0.74	0.24	0.26
Vallejo et al. (2005)	Surface	Loamy sand	Spring	Grass ley	215	199	186	97	4	7.1	0.78	1.60	1.72
	Injected	Loamy sand	Spring	Grass ley	215	199	186	97	4	7.1	1.05	2.95	3.18
Thomsen et al. (2010)	Trailing hose	Loamy sand	Spring 07	Cereal	30	162	140	38	5	7.9	0.49	0.30[a]	–
	Straight tine	Loamy sand	Spring 07	Cereal	30	162	140	38	5	7.9	2.51	1.50[a]	–
	Winged tine	Loamy sand	Spring 07	Cereal	30	162	140	38	5	7.9	1.86	1.20[a]	–
	Trailing hose	Loamy sand	Spring 08	Cereal	30	127	81	47	3	8.0	0.70	0.30[a]	–
	Straight tine	Loamy sand	Spring 08	Cereal	30	127	81	47	3	8.0	0.40	0.06[a]	–
Sistani et al. (2010)	Surface	Silt loam	Spring 07	No-till corn	141	200	112	–	3	–	0.73	1.56	2.78
	Injection	Silt loam	Spring 07	No-till corn	141	200	112	–	3	–	0.47	0.73	1.31
	Aeration	Silt loam	Spring 07	No-till corn	141	200	112	–	3	–	0.69	1.43	2.56
	Surface	Silt loam	Spring 08	No-till corn	158	204	50	–	4	–	0.29	0.69	2.80
	Injection	Silt loam	Spring 08	No-till corn	158	204	50	–	4	–	0.82	1.31	9.55
	Aeration	Silt loam	Spring 08	No-till corn	158	204	50	–	4	–	0.49	2.34	5.35

[a] N_2O emission factor cited as being adjusted for NH_3 losses. EF: emission factor.

Table 5
N$_2$O emissions from land-applied solid manure.

	Application method	Site characteristics and season of application			Sampling period (d)	N applied in solid manure (kg/ha)		Solid manure characteristics			N$_2$O-N emission		
		Soil texture	Season of application	Crop		Total N	Ammoniacal-N	DM (g/kg)	C:N	pH	Cumulative (kg/ha)	Total N EF (%)	Ammoniacal-N EF (%)
Cattle FYM													
Chadwick et al. (2000a)	Broadcast	Sandy loam	Autumn	Grass	21	315	5	251	15	–	0.65	0.20	12.60
Webb et al. (2004)	Ploughed 4h	Sandy loam	Autumn 99	Tillage	60	145	11	171	–	–	0.06	<0.01	–
	Ploughed 24 h	Sandy loam	Autumn 99	Tillage	60	145	11	171	–	–	0.10	0.02	–
	Surface	Sandy loam	Autumn 99	Tillage	60	145	11	171	–	–	0.92	0.33	–
	Ploughed 4 h	Sandy loam	Autumn 00	Tillage	60	110	8	167	–	8.2	0.22	0.02	–
	Ploughed 24 h	Sandy loam	Autumn 00	Tillage	60	110	8	167	–	8.2	0.12	<0.01	–
	Surface	Sandy loam	Autumn 00	Tillage	60	110	8	167	–	8.2	0.60	0.22	–
Thorman et al. (2007a)	Broadcast	Sandy loam	Summer	Tillage	75	185	<1	152	–	7.7	0.53	0.16	43.18
	Ploughed	Sandy loam	Summer	Tillage	75	185	<1	152	–	7.7	0.46	0.12	15.20
	Disced	Sandy loam	Summer	Tillage	75	185	<1	152	–	7.7	0.41	0.09	28.95
Pig FYM													
Chadwick et al. (2000a)	Surface	Sandy loam	Autumn	Grass	21	310	85	193	11	–	0.19	0.05	0.20
Webb et al. (2004)	Ploughed immediately	Sandy loam	Autumn 99	Tillage	60	180	30	255	–	8.8	0.03	<0.01	–
	Ploughed 4 h	Sandy loam	Autumn 99	Tillage	60	180	30	255	–	8.8	0.03	<0.01	–
	Ploughed 24 h	Sandy loam	Autumn 99	Tillage	60	180	30	255	–	8.8	0.03	<0.01	–
	Surface	Sandy loam	Autumn 99	Tillage	60	180	30	–	–	–	0.04	<0.01	–
	Ploughed immediately	Sandy loam	Autumn 00	Tillage	60	–	–	–	–	–	0.30	–	–
	Ploughed 4 h	Sandy loam	Autumn 00	Tillage	60	–	–	–	–	–	0.30	–	–
	Surface	Sandy loam	Autumn 00	Tillage	60	–	–	–	–	–	1.30	–	–
Thorman et al. (2007a)	Surface (stored)	Sandy loam	Spring	Tillage	75	236	5	204	–	7.7	0.07	0.00	0.01
	Ploughed (stored)	Sandy loam	Spring	Tillage	75	236	5	204	–	7.7	0.29	0.09	4.49
	Surface (fresh	Sandy loam	Spring	Tillage	75	371	61	405	–	8.6	0.07	0.00	0.00
	Ploughed (fresh)	Sandy loam	Spring	Tillage	75	371	61	405	–	8.6	3.27	0.86	5.30
Poultry manure													
Chadwick et al. (2000a)	Surface	Sandy loam	Autumn	Grass	21	416	156	432	5	–	0.19	0.05	0.11

N applied in the manure, or even as a proportion of the inorganic N remaining in the manure after NH_3 volatilisation has occurred.

To exemplify this point, Table 6 summarises a database of N_2O emissions from a range of manure types applied to soil in field studies in England (ADAS and Rothamsted Research in North Wyke, Devon) between 1994 and 2003. The selection criteria (*i.e.*, a measurement period >21 d) generated 92 EF from a range of sites under grassland and arable cropping. Ammonia emissions were also measured from these field studies, so the rapid loss of readily available N is accounted for in estimated N_2O emissions. Nitrous oxide emissions are expressed conventionally as a % of the total N applied, but also as a % of the readily available N applied and also after accounting for NH_3 losses.

Nitrous oxide EF were dependant upon manure type where EF were expressed as the % of total N applied, the % of readily available N applied and the % of total N applied remaining after NH_3 loss (Table 6), with losses being the highest from poultry manure. However, when results were expressed as the % of readily available N applied remaining after NH_3 loss, there was no difference in EF among manure types, indicating that it is the readily available N and not total N applied that drives N_2O emissions.

Several key management decisions affect the magnitude of N_2O emissions after manure spreading, as they influence interactions between the C and N content of the soil with the soil's microbiology and physico-chemical properties at the time of application. Decisions on timing of application, application rate and method of application are key factors as outlined below.

2.3.1. Manure type

Chadwick et al. (2000a) compared N_2O emissions from soil following pig or dairy slurry applied at the same NH_4^+-N application rate (Table 4). Higher emissions resulted from the dairy slurry (2.42% of applied NH_4^+-N, 0.97% of total N applied) compared to the pig slurry (0.94% of NH_4^+-N applied, 0.44% of total N applied), which the authors attributed to differences in the C content of the slurries as well as to the fine solids in the dairy slurry blocking soil pores and enhancing anaerobic soil conditions. Rochette et al. (2008) compared N_2O emissions after applications of liquid and solid manures and found no clear differences among treatments. In future studies, it is essential that a better description of manure characteristics, particularly available N and C, are reported in order to determine effects of manure applications on N_2O emissions at a given site as well as aid in meta-analyses of N_2O studies.

2.3.2. Soil type

The water holding capacity and organic matter content of clay soils, as compared to sandy soils, tends to result in higher N_2O emissions following manure application (van Groenigen et al., 2004; Table 4). Indeed, background N_2O fluxes from clay soils can be much higher than from sandy soils (van Groenigen et al., 2004). In a comparison of N_2O emissions from contrasting soils in the Netherlands, van Groenigen et al. (2004) showed emission factors were 2 times higher (*i.e.*, 1.21% of applied N) from a clay soil than a sandy soil (*i.e.*, 0.62% of applied N) after application of dairy slurry. Rochette et al. (2008) suggested that the main source of N_2O was denitrification in a clay soil and nitrification in a sandy loam and also recorded higher N_2O emissions from clay soil (Table 4). Differences in the cation exchange capacity (CEC) of contrasting soils has also been suggested as a factor in N_2O emissions (Jarecki et al., 2008), with the higher CEC of clay soils reducing N availability through increased adsorption of NH_4^+ as compared to sandy soils.

2.3.3. Timing of manure application (temperature, soil water filled pore space)

Manure applications should be timed to supply actively growing crops with essential nutrients (Anon., 2010). Excess N remaining in the soil risks loss to the environment as NO_3^- leaching or N_2O emissions. Since, N_2O production is a microbial process, timing of manure application within the agricultural calendar influences rate of N_2O emission based on effects of soil temperature and water on the activity of soil microorganisms (Dobbie et al., 1999). After autumn/winter slurry application, the quantity of N lost by NO_3^- leaching can be high and much higher than after slurry application in the spring (Chambers et al., 2000), a factor which influences the magnitude of indirect N_2O losses (Thorman et al., 2007b).

Thorman et al. (2007b) measured direct and indirect (*i.e.*, NH_3 volatilisation and NO_3^- leaching) N_2O emissions following cattle slurry application to freely draining soils. Using IPCC default EF for the fraction of leached and volatilised N lost as N_2O (2.5% and 1%, respectively; IPCC, 1996), they estimated indirect N_2O losses were higher as a result of autumn/winter slurry application (0.49% of total N applied) than from spring application (0.10% of total N applied). Direct N_2O emissions from the autumn/winter and spring slurry applications were 1.10 and 0.51% of the slurry N applied, respectively. The authors suggest that this is because slurry applied to an actively growing crop in the spring creates a bigger N sink than when it is applied in the autumn. This effect of timing of manure application on N_2O emissions corroborates earlier findings on arable land following a comparison of pig slurry application in the autumn and spring (Weslien et al., 1998).

The introduction in Europe of the Nitrate Vulnerable Zone Action Plans (Nitrates Directive (91/676/EEC)) has resulted in introduction of closed periods for spreading of manures with high available N contents and, whilst designed to reduce impacts of manure management on water quality, this directive has the added benefit of potentially reducing both direct and indirect N_2O emissions.
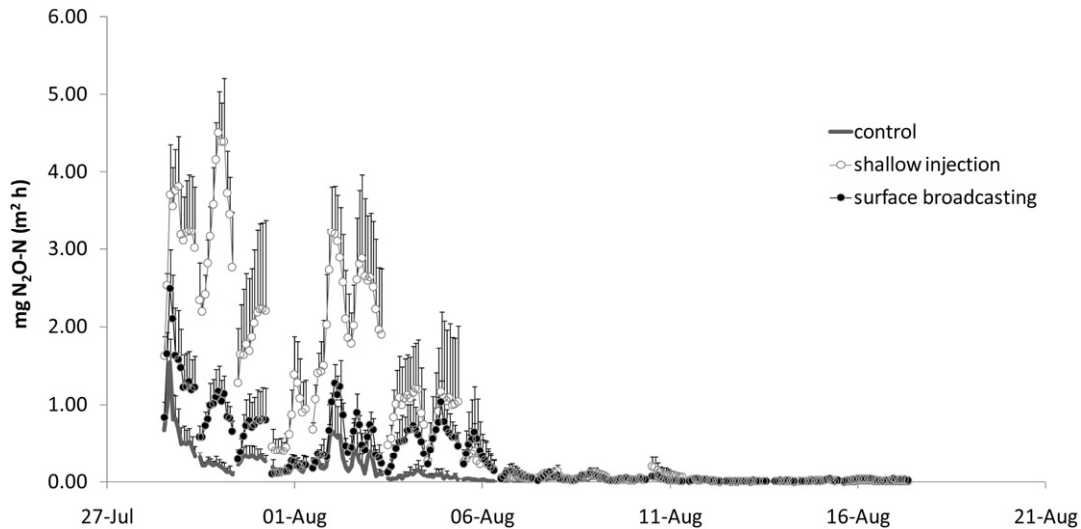
Fig. 3. N_2O emissions following slurry application by slurry injection and surface broadcasting using field-scale slurry applicators. Measurements made using a mobile automated gas chromatography laboratory in the field (Chadwick et al., 2000a).

2.3.4. Effect of manure application rate

Whilst there are few published studies which have explicitly investigated effects of manure application rate on N_2O emissions, one could expect a non-linear relationship with disproportional losses of N_2O at higher rates of manure N application, as observed by van Groenigen et al. (2004) after application of dairy slurry to a clay soil. Similar findings have also been reported with increasing rates of N fertiliser application (Cardenas et al., 2010). van Groenigen et al. (2004) suggested that at high application rates, O_2 depletion of the soil increases denitrification rates and N_2O production. van der Meer (2008) concluded that, by optimising rate, timing and technique of manure application to crops and grassland, more effective utilisation of manure N could be achieved with reduced losses of both direct and indirect N_2O. Responses may also differ with soil type, as a linear relationship between pig slurry application rate and N_2O emissions was recorded for a sandy soil (Velthof et al., 2003).

2.3.5. Effect of application method

Livestock slurries can be applied to the surface of the soil either through broadcasting across the surface, or in discrete surface bands through hoses. Slurries can also be injected into soil of grassland or during tillage. Solid manures can only be broadcast on the surface of grassland but, as with tillage, there is opportunity to incorporate slurries and solid manures into the soil. Application procedures that minimise contact of manure with air tend to reduce NH_3 emissions (Misselbrook et al., 2002; Webb et al., 2010) and odour.

One might expect that since these application methods retain more slurry N in the soil (Chadwick et al., 2001), that there is the potential for higher N_2O emissions. Whilst this has been shown to occur in some studies (Flessa and Beese, 2000; Wulf et al., 2002; Velthof et al., 2003), this is not always the case (Sommer et al., 1996; Vallejo et al., 2005). The example in Fig. 3 demonstrates that although shallow injection reduced NH_3 emissions from 72 to 11% of the NH_4^+-N applied in the slurry (Misselbrook et al., 2002), N_2O emissions were increased markedly with cumulative fluxes representing 10.2% (shallow injection) and 2.9% (surface broadcast) of the NH_4^+-N applied or 6.09% and 1.69% of the total slurry N applied, respectively.

Some research has shown that in different years of the same study (*e.g.*, Rubaek et al., 1996; Sistani et al., 2010) injection of slurry can increase, decrease or have no effect on N_2O emission compared to surface broadcasting. Misselbrook et al. (1996) showed shallow injection to grassland in March increased denitrification losses, but this response did not occur with October or June applications. It would appear that when conditions are favourable for denitrification, slurry injection can result in increased N_2O emissions compared with surface broadcasting. But under other conditions slurry injection has the potential to reduce NH_3 emissions without increasing N_2O emissions. It is important that researchers continue to integrate measurements of NH_3 and N_2O to generate an understanding of potential tradeoffs between one form of N emission and another.

Fewer studies have compared N_2O emissions from different low trajectory slurry applications. Rodhe et al. (2006) showed higher N_2O emissions from closed shallow injection slots than from band spread cattle slurry application (Table 4). Whilst, Thomsen et al. (2010) showed higher N_2O emissions following slurry injection with straight and winged tines compared to slurry application by trailing hose (Table 4). The same study demonstrated that the soil moisture content was consistently higher from within the injection slots than between slot positions, indicating wet conditions maintained in restricted zones where inorganic N and C were present could have resulted in increased N_2O emissions, probably via denitrification (Venterea, 2007).

Solid manure incorporation decreases N_2O emissions compared to leaving manure on the surface (Webb et al., 2004), perhaps due to increasing the residence time of N_2O that is generated from the manure N source that has been incorporated, such that more of it is converted to N_2. Thorman et al. (2007a) found no consistent effect of incorporation of pig or cattle FYM on N_2O emissions within 4 h of application. Effects of rapid incorporation of solid manure on N_2O loss was inconsistent (Thorman et al., 2008), who concluded that in conditions likely to induce intense denitrification, incorporation is likely to reduce N_2O emissions and may be used as a 'win–win' technique to mitigate NH_3 and N_2O losses. However, in conditions where denitrification is unlikely to be intense, incorporation may increase N_2O emissions.

2.3.6. Effects of slurry treatment on emissions from spreading

Anaerobic digestion or separation removes organic matter and affects infiltration of manure slurry and the content of volatile solids (VS) in the soil slurry mixture. Reducing VS in the soil–slurry mixture reduces risk of N_2O emissions, as the lower VS content decreases microbial demand for O_2 and consequently heterotrophic denitrification (Petersen et al., 1996). Some researchers have reported lower N_2O emissions from soils amended with digested slurries than from untreated slurries (Petersen, 1999; Bhandral et al., 2009), but this response has not been consistent (Amon et al., 2006; Clemens et al., 2006; Thomsen et al., 2010), suggesting that application conditions and soils properties may influence effects of digested slurries on N_2O emissions (Oenema et al., 2005).

As mechanical separation is likely to reduce the solid content of livestock slurries to a lesser extent than anaerobic digestion, it may be less effective at reducing N_2O emissions when the effluent is applied to land (Bertora et al., 2008). However, combining mechanical and chemical approaches to enhance solids removal may be more successful in reducing potential N_2O emissions once the effluent is land applied. A laboratory study (Fangueiro et al., 2008b) and a field study on grassland (Fangueiro et al., 2008c) also showed that slurry separation by screw press or a combination of screw press separation and chemical settling had no effect on overall N_2O emissions after land application as compared to untreated slurries. Bhandral et al. (2009) also found no effect of removal of slurry solids by settling-decanting or mechanical aeration on N_2O emissions after land application of the liquid fraction, although NH_3 emissions were consistently reduced by mechanical aeration.

Recent studies have shown conflicting effects of slurry pre-treatment on N_2O emissions, as increasing DM content of the slurry may promote N_2O emissions (Sommer et al., 2004b; van Groenigen et al., 2004) on loamy sand and clay soils. This finding has been contradicted by a recent study which showed that reducing the slurry DM content by mechanical separation did not affect N_2O emissions from slurry injected into a loamy sand soil (Thomsen et al., 2010). Thomsen et al. (2010) presented a conceptual model of the relationship between N_2O/N_2 emissions and the supply and demand of O_2 in the soil to explain their results, which suggested that when injecting a high DM slurry into a very 'reductive' soil environment, that nitrification – denitrification processes proceed to N_2 if large amounts of DM promote a reduction in the redox potential. Alternatively, if the soil environment is less reductive, then adding a slurry with a high DM content may change the soil redox potential and, in this situation, it may enhance denitrification but it does not proceed to formation of N_2 and N_2O is produced.

Acidification of slurry or slurry fractions delays nitrification and consequently the N_2O emissions in soils amended with acidified slurry or slurry fractions occur later relative to soil amended with non-acidified materials. Furthermore, lower amounts of N_2O were released from soils amended with acidified slurry fractions, but no effect occurred with the whole slurry (Fangueiro et al., 2010).

In a plot scale study, Dittert et al. (2001) found that N_2O emissions were reduced from shallow injection slots by >30% when 3,4-dimethylpyrazole phosphate (DMPP) was added to the slurry prior to application. Vallejo et al. (2005) showed that addition of dicyandiamide reduced N_2O emissions from 2.95% to 0.50% of applied N when injected into soil in a Mediterranean climate. Other studies have demonstrated the efficacy of nitrification inhibitors in land-applied slurries (Merino et al., 2002; Hatch et al., 2005), although inhibition appears to be more pronounced in laboratory studies than in field studies. However in some studies, when a nitrification inhibitor has been added to slurry, there has been no effect on N_2O emissions after application to soil (Mkhabela et al., 2006a,b).

3. Methane emissions from manure management

3.1. Methane emissions from animal housing and collection yards

Manure deposited in animal houses emits CH_4. However, the majority of measurements of CH_4 emissions from animal houses have been conducted to quantify enteric emissions and have not attempted to apportion manure derived CH_4 from enteric CH_4. Methane emissions occur from slurry stored below livestock buildings, and frequent removal of it can reduce emissions from the livestock house (Sommer et al., 2009). Collection yards have also been identified as minor sources of emitted CH_4 (Ellis et al., 2001; Misselbrook et al., 2001).

3.2. Methane emissions from manure storage and treatment

Solid manure stores have been shown to be sources of CH_4 emissions (Sommer and Møller, 2000; Amon et al., 2001; Chadwick, 2005; Yamulki, 2006; Hansen et al., 2006; Szanto et al., 2007) with losses from cattle FYM heaps being 0.4–9.7% of

Table 6
Livestock manure N_2O emission factors (±SE) from studies in England.

Manure type (number measurements)	% total N applied	% total N applied remaining after NH_3 loss	% readily available N applied	% readily available N applied remaining after NH_3 loss
Slurry[a] (51)	0.57 (0.13)	0.67 (0.15)	1.06 (0.24)	1.76 (0.41)
FYM[a] (27)	0.28 (0.08)	0.30 (0.08)	1.29 (0.27)	1.97 (0.46)
Poultry manure (14)	0.75 (0.14)	0.79 (0.14)	2.05 (0.41)	2.70 (0.54)
Mean (92)	0.51 (0.08)	0.58 (0.09)	1.27 (0.16)	1.96 (0.27)

[a] Cattle and pig.

the total C content of the heap (Chadwick, 2005). Sommer (2001) measured smaller losses than these using static chambers, whereas Chadwick (2005) quantified CH_4 from the entire heap using a tunnel system. The study of Sommer et al. (2004a) showed that measuring gas emissions from composting animal manure heaps with static chamber technology may greatly underestimate emissions because the static chambers may not be able to capture the convective flow of gases from the heap as the chambers were developed to measure diffusive flow of gases from soil.

Chadwick (2005) showed evidence that heap management affects the magnitude of CH_4 emissions, in that covering and compacting FYM heaps could either increase or decrease CH_4 emissions, presumably as a function of heap anaerobicity and temperature. Yamulki (2006) demonstrated that addition of straw to solid manure heaps could reduce CH_4 emissions as the mixing of 50% (v/v) straw with cattle manure at the start of storage reduced CH_4 emissions by 45%. Addition of phosphogypsum (PG) to cattle feedlot manure has also been shown to reduce CH_4 emissions during storage (Hao et al., 2005), perhaps because of effects of higher S and NH_4^+ concentrations on methanogens and impacts of lower pH on rate of CH_4 oxidation.

Methane emissions from stored solid manure can be reduced by two completely different strategies aiming at either promoting or preventing anaerobic conditions. An air-tight cover may be used to cover the heap, thereby inhibiting activity of aerobic microorganisms and the associated increase in temperature that stimulates CH_4 emissions from anaerobic microenvironments. For example, efficient covering reduced CH_4 emissions from a heap of a DM rich separated slurry fraction from 1.6 to 0.2 kg C/t, or from 1.3 to 0.17% of the initial C content (Hansen et al., 2006). Chadwick (2005) estimated that 1.8 and 4.4% of initial C was emitted as CH_4 from three storage periods of conventionally stored cattle FYM, and that covering and compaction of the heaps had no consistent effect on CH_4 emissions. Alternatively, frequent turning can be used to reduce anaerobic zones in the heap. This technique reduced CH_4 emissions to about 0.5% of initial C content (Amon et al., 2001, 2006).

Slurry stores are sources of CH_4 emissions as the anaerobic environment favours methanogenesis. Mild agitation of slurry has been to shown to increase CH_4 emissions, as dissolved gas and bubbles are released (VanderZaag et al., 2010a,b), but losses by this route are thought to be small and short-lived. Indeed, allowing formation of a slurry crust can produce a CH_4 sink as a result of CH_4 oxidation (Petersen et al., 2005). Covering slurry stores with porous surfaces of straw, expanded clay pebbles or recycled polyethylene may reduce CH_4 emissions due to oxidation to CO_2 (Sommer et al., 2000; VanderZaag et al., 2009). Amending slurry with straw may enhance methanogenic activity (Berg et al., 2006; Amon et al., 2007), thus a surface crust of slowly digestible manure material or inert leca pebbles may be more efficient at reducing CH_4 emission than this approach. VanderZaag et al. (2010b) found that a permeable synthetic floating cover, although successful at reducing NH_3, N_2O and CO_2 emissions from stored liquid manure, did not reduce CH_4 emissions. High concentration of cellulose and lignin may limit the rate of CH_4 production due to the reduced hydrolysis of the lignified structures in the biomass (Ianotti et al., 1979).

Frequent removal of slurry from the store or channel reduces the pool of methanogenic bacteria within this environment. Thus in pig houses where slurry was removed from channels after each fattening period, emissions were 40% lower then in houses where channels were not cleared as frequently (Haeussermann et al., 2006).

A positive correlation between CH_4 emissions during storage and the temperature of manure or slurry has been observed (Massé et al., 2003; Møller et al., 2004b; Pattey et al., 2005). Methane production is low at temperatures <15 °C but increases exponentially as temperature rises above 15 °C (Clemens et al., 2006; Husted, 1994; Khan et al., 1997; Sommer et al., 2007). Massé et al. (2008) measured higher CH_4 emissions from slurry at 20 °C compared to slurry at 10 °C, and VanderZaag et al. (2010a) found that the CH_4 flux was positively correlated with slurry temperature. Emission of CH_4 from slurry in livestock houses can be efficiently mitigated by frequent slurry removal to an outside store provided the outside temperature is lower (Massé et al., 2008). The effect of a cool climate on CH_4 emission is also reflected in the IPCC (2007) algorithm that is related to climatic zones. It is proposed that algorithms should be developed that take into account the temperature in animal houses and outside stores to generate regionally specific CH_4 emission data.

Fangueiro et al. (2008b) showed that, compared to whole slurry, separation of cattle slurry into liquid and solid fractions using a screw press reduced CH_4 emissions by >35% and, when combined with chemical settling, by up to 50%. Nevertheless, Dinuccio et al. (2008) reported a small 3–4% increase (or 8–9% decrease) in CH_4 emissions during storage of separated slurry depending on temperature and slurry type (*i.e.*, pig or cattle). Hence, it is difficult to say if slurry separation increases or decreases CH_4 emissions since it depends mainly on the storage conditions and the characteristics of the slurry fractions

Table 7
Potential mitigation methods for N_2O and CH_4 from the manure management continuum.

	Nitrous oxide	Methane
Animal house	• Modify feeding strategy • Adopt a slurry based system compared to a straw or deep litter based system	• Modify feeding strategy • Removal of slurry from beneath the house • Cooling slurry, *e.g.*, below the slatted floor
Manure stores	• Modify feeding strategy • Keep anaerobic (*e.g.*, cover and compact) • Adopt a slurry based system compared to a straw of deep litter based system • Add additional straw to immobilise ammonium-N	• Modify feeding strategy • Removal of slurry from the slurry store • Minimising slurry volume stored in summer months • Cooling slurry • Aerate solid manure heaps – composting • Anaerobic digestion • Enhancing crust formation
Land spreading	• Modify feeding strategy • Nitrification inhibition • Spring application of slurry • Integrate manure N with fertiliser N • Slurry separation? • Solid manure incorporation?	• Modify feeding strategy

obtained. However CH_4 in slurry can be emitted during separation. Anaerobic digestion of slurry also enhances CH_4 production, but as the CH_4 can be captured and used as a substitute for fossil fuels, this approach reduces the potential for CH_4 emissions during subsequent storage (Sommer et al., 2000).

Reducing the organic matter content of slurry through separation or fermentation of slurry in a biogas digester may prove to be the most efficient way of reducing CH_4 emissions during outside storage. Emissions from digested slurry during storage were 30 to 66% lower than from untreated slurry (Clemens et al., 2006; Amon et al., 2006). However, Sommer et al. (2000) showed that digested slurry should be cooled to ambient temperatures in post-treatment storage tanks to reduce CH_4 emissions. Also, acidification of slurry for the purpose of reducing NH_3 emissions from storage has been observed to reduce CH_4 emissions (Berg et al., 2006).

3.3. Methane emissions from manure spreading

Emissions of CH_4 generally occur immediately after manure application to land (Chadwick and Pain, 1997; Chadwick et al., 2000a). These emissions are usually short-lived, as methanogenesis is sensitive to O_2 and diffusion of O_2 into the manure on the soil surface inhibits CH_4 formation. Kirchmann and Lundvall (1993) and Sommer et al. (1996) have shown that volatile fatty acids decrease in manure within a few days of application and the amount of CH_4 emitted is negligible (Chadwick and Pain, 1997; Yamulki et al., 1999; Sherlock et al., 2002; Rodhe et al., 2006). Several authors indicate that most CH_4 emitted following slurry application originates from a pool of dissolved CH_4 (Chadwick and Pain, 1997; Sherlock et al., 2002; Clemens et al., 2006). As might be expected, when slurry is applied via shallow injection, the anaerobic nature of the slot environment results in higher CH_4 emissions compared to surface broadcasting (Flessa and Beese, 2000).

4. Potential mitigations

A number of methods to mitigate N_2O and CH_4 emissions from manure are summarised in Table 7. However, it is important to consider that indirect N_2O losses (*i.e.*, derived from NO_3^- leaching and NH_3 volatilisation) can at times be larger (Table 2) than direct N_2O losses (Mkhabela et al., 2009). As a result, reducing NO_3^- leaching and NH_3 volatilisation should also contribute to reductions in total N_2O emissions.

A mitigation method that will potentially affect all phases of the manure management continuum is optimisation of the N content of the diet of the animal, as this reduces N excretion/unit product produced. Diet formulation might also be used to reduce CH_4 emissions from the rumen and the manure store. Several studies have shown that reducing the crude protein content of diets can reduce N excretion and ammonia emissions (Powell et al., 2008). Nitrous oxide emissions during slurry storage have also been reported to be lower following a reduction in the crude protein content of forage (Kulling et al., 2002). Furthermore, Kulling et al. (2003) observed differences in N_2O emissions during storage of liquid manure, slurry or FYM from dairy cows fed forage in the form of fresh grass or hay. Velthof et al. (2005) demonstrated effective reductions in CH_4 emissions during storage of pig slurry from pigs fed a modified diet, whilst studies have shown that altering diets of livestock can reduce N_2O emissions following spreading of resultant manures (Oenema et al., 2005; Velthof et al., 2005; Cardenas et al., 2007). Although, Misselbrook et al. (1998) showed a reduced crude protein diet for pigs resulted in lower total denitrification losses after land-application of manure, no difference in N_2O emissions occurred.

More efficient use of manures as sources of N, P and K reduces reliance on inorganic fertilisers, and thus reduces N_2O emissions associated with manufacture and use of inorganic fertilisers. It also reduces fossil fuel use and associated CO_2 emissions from the manufacturing and transportation of inorganic fertilisers. Land application of livestock manure can also increase the C content of soils. Any increase in soil C content can offset some of the GHG emissions associated with manure management. However, the potential for C sequestration by manure application is much higher in tilled soils than in grasslands (Smith et al., 2001).

5. Conclusions

It is clear that manure management impacts quantities of direct and indirect N_2O emissions and CH_4 emissions at each stage of the manure management continuum. Since production of these gases is of microbial origin, the DM content and temperature of manure and soil are key factors in on farm manure management decisions that influence the magnitude of GHG losses. There remains a degree of uncertainty in emission rates of GHG gases from different stages of manure management, as indicated in the ranges described in this review, and researchers continue to investigate interactions of the management and environmental factors which control emissions.

Some specific approaches to reducing GHG emissions include optimal timing of manure application within the growing season to reduce N_2O emissions from soil, and anaerobic digestion of livestock manure to reduce CH_4 emissions during storage. More holistic opportunities may yet be exploited to reduce both CH_4 and N_2O emissions, such as optimising diet formulation, although the extent and sustainability of these reductions need to be assessed in practice throughout entire farming systems to validate modelling approaches (Schils et al., 2007).

Some legislation may result in 'win–win' scenarios, such as the Nitrates Directive (91/676/EEC) which has led to development of Nitrate Vulnerable Zone action plans to prevent application of (high available N content manures slurry and poultry manure) in autumn, a practice which reduces direct and indirect N_2O losses. Whereas, other legislation may result in potential 'pollution swapping', as is sometimes the case with use of slurry injection to reduce NH_3 emissions at the expense of an increase in N_2O emissions. However, in this latter example there is no clear understanding of why this pollution swapping only occurs on some occasions.

The nature of the N cycle and its interaction with the C cycle demands a holistic approach to addressing GHG emissions and mitigation research at a process level of understanding. There are a number of farm scale C accounting tools (*e.g.*, the CLA Carbon Accounting for Land Managers (CALM) tool), which include GHG emissions from manure management which may be useful in spite of their current limitations to reflect targeted mitigation. Systems based modelling must play a key role in integrating the complexity of management and environmental controls on emissions. Progress has been made to this end (Sommer et al., 2009), with some studies producing whole farm models encompassing livestock production (del Prado et al., 2010). An evidence based database is required to validate and test such models to determine the scope to which management practices can be used to reduce GHG from livestock manure.

Conflict of interest statement

None.

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