



# How do emission rates and emission factors for nitrous oxide and ammonia vary with manure type and time of application in a Scottish farmland?



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## ABSTRACT

The use of livestock manure as an organic fertiliser on agricultural land is an attractive alternative to synthetic fertiliser. The type of manure and the timing and method of application can however be crucial factors in reducing the extent of nitrogen lost from the system. This is important not only to enhance crop production, but in controlling gaseous emissions, including nitrous oxide (N<sub>2</sub>O) and ammonia (NH<sub>3</sub>). Emissions of N<sub>2</sub>O and NH<sub>3</sub> were measured for 12 months from two experiments at an arable site in Scotland, to determine the effect of manure type and the timing (season) of application. Emission factors (EFs) were calculated for each manure applied in each season, and compared to IPCC standard EFs of 1% for N<sub>2</sub>O and 20% for NH<sub>3</sub>. Cattle farmyard manure, broiler litter, layer manure, and cattle slurry by surface broadcast and trailing hose application were applied to one experiment in October 2012 (autumn applications) and one in April 2013 (spring applications). Experimental areas were sown with winter wheat (*Triticum aestivum*) and manures were applied at typical rates. Crop yield was recorded to allow calculation of N<sub>2</sub>O and NH<sub>3</sub> emission intensities. Mean annual N<sub>2</sub>O emissions across all manure treatments were greater from autumn (2 kg N<sub>2</sub>O–N ha<sup>-1</sup>) than spring (0.35 kg N<sub>2</sub>O–N ha<sup>-1</sup>) applications, and in the spring experiment were significantly lower from cattle slurry than other treatments. Ammonia emissions were generally greater (though not significantly) from spring than autumn applications. Significantly greater NH<sub>3</sub> emissions were measured from layer manure than all other manures at both times of application. N<sub>2</sub>O and NH<sub>3</sub> EFs were highly variable depending on the season of application and manure type. The mean autumn and spring N<sub>2</sub>O EFs across all manure treatments were 1.72% and –0.33% respectively, and mean NH<sub>3</sub> EFs across all treatments were 8.2% and 15.0% from autumn and spring applications, respectively. These results demonstrate large deviation from the IPCC default values for N<sub>2</sub>O and NH<sub>3</sub> EFs, and the considerable effect that manure type and time of application have on N<sub>2</sub>O and NH<sub>3</sub> emissions.

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## 1. Introduction

Manures and slurries provide a significant nitrogen (N) input to agricultural land. In 2011 the total N excreted by livestock in the EU was 9.2 Tg which is only 15% less than the N added by synthetic fertilisers (Velthof et al., 2015). The large quantity of nutrients present in manures means that they are commonly applied to agricultural land to recycle N, phosphorus and potassium for plant growth (Defra, 2010). However, considerable amounts of the applied manure N will not be utilised by crops as a result of nitrification and denitrification, and the subsequent emissions of nitrous oxide (N<sub>2</sub>O) (Chadwick et al., 2011), dinitrogen (N<sub>2</sub>) (Cardenas et al., 2007), and ammonia (NH<sub>3</sub>) (Misselbrook et al.,

2005a). Leaching of nitrate (NO<sub>3</sub><sup>-</sup>) into groundwater and surface waters leads to further N loss from the soil (Rodhe et al., 2006) and other environmental impacts including eutrophication and soil acidification.

Globally, agricultural soil is responsible for 65% of N<sub>2</sub>O emissions (Reay et al., 2012), a greenhouse gas (GHG) approximately 300 times more powerful than CO<sub>2</sub>, that is also responsible for stratospheric ozone layer depletion (Stocker et al., 2013). In the UK it is estimated that 73% of anthropogenic N<sub>2</sub>O emissions and 92% of NH<sub>3</sub> emissions are from agricultural sources, including direct emissions from soils, animal wastes and manure stores (Dore et al., 2008; Skiba et al., 2012). Indirect N<sub>2</sub>O emissions also result from deposition of volatilised NH<sub>3</sub> and NO<sub>3</sub><sup>-</sup> leaching and transport in aquatic and terrestrial environments.

The potential for N<sub>2</sub>O and NH<sub>3</sub> emission after manure applications to agricultural soil is dependent on a combination of manure properties and environmental conditions. High temperatures, high wind speed

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and low rainfall immediately following manure application promote  $\text{NH}_3$  emissions from manures containing a high amount of readily available N (Meisinger and Jokela, 2000; Misselbrook et al., 2005a), meaning that the timing of application can be critical if significant losses of N from the soil are to be avoided. Conversely, loss of N via  $\text{N}_2\text{O}$  emissions is higher when manure is applied in wet conditions as  $\text{N}_2\text{O}$  production via denitrification will occur before the crop is able to utilise the available N. Nitrate leaching will also occur if excess rainfall and drainage take place between manure application and crop N uptake (Defra, 2010; Shepherd and Newell-Price, 2013). It is generally recommended therefore to apply manures when crops are actively growing and removing N from the soil (Granli and Bøckman, 1994; Meisinger and Jokela, 2000; Defra, 2010).

In the UK, manure application in autumn and winter is restricted by Nitrate Vulnerable Zone (NVZ) regulations to decrease  $\text{NO}_3^-$  pollution of aquatic environments. Expansion of these measures to other areas could assist in decreasing indirect  $\text{N}_2\text{O}$  emissions from  $\text{NO}_3^-$  leaching and direct  $\text{N}_2\text{O}$  emissions from denitrification if application in wet conditions is avoided. The time of application should aim to provide a balance between the need to apply manure during the period of maximum crop N requirement, and the need to reduce seasonal climate effects on emissions (Meisinger and Jokela, 2000). Reducing losses of N from the soil is also beneficial for crop growth as more N is available for use by the growing crop (Rodhe et al., 2006; Shepherd, 2009).

The magnitude of  $\text{N}_2\text{O}$  and  $\text{NH}_3$  emissions generated from manures is also dependent on their total-N content and the proportion present as readily available N (ammonium-N and uric acid-N), which varies with manure type (Defra, 2010; Shepherd and Newell-Price, 2013). Large quantities of readily available N (35–70% of total N) are typically found in slurries and poultry manures, compared to only 10–25% of total N in farmyard manure (FYM) (Defra, 2010). Manures containing large amounts of readily available N have a higher probability of losing N via  $\text{NH}_3$  volatilization (Misselbrook et al., 2005a),  $\text{N}_2\text{O}$  production (Chadwick et al., 2011), or as a result of  $\text{NO}_3^-$  leaching (Chambers et al., 2000; Dampney et al., 2000; Shepherd, 2009). Manure moisture content can also affect  $\text{N}_2\text{O}$  emissions, as an increase in soil moisture can enhance the production of  $\text{N}_2\text{O}$ , with greatest  $\text{N}_2\text{O}$  emissions most likely to occur between 50 and 70% WFPS (Flecharde et al., 2007). Slurry typically has a moisture content of >90%, increasing the risk of high  $\text{N}_2\text{O}$  emissions after application (Jørgensen et al., 1998). The moisture content of manures can also affect  $\text{NH}_3$  emission rate, and slurries with higher moisture contents are generally associated with lower  $\text{NH}_3$  emissions as they rapidly infiltrate into the soil, with the majority of the emission typically occurring in the 12 h post-application (Sommer and Hutchings, 2001). Poultry litter, in contrast, has a much lower moisture content and a lower initial loss of  $\text{NH}_3$ , but emissions occur over a longer timescale as uric acid is broken down and urea hydrolysed to  $\text{NH}_4^+$  (Meisinger and Jokela, 2000; Jones et al., 2007). It has also been suggested that the C:N ratio of organic manures may affect N losses from soil. Akiyama et al. (2004) argue that higher C:N ratios in manure compared with inorganic chemical fertilisers provide optimum conditions for denitrification. The high C contents of organic manures (typically 35% organic C), can also stimulate microbial activity, thereby creating anaerobic zones in the soil that allow denitrification and  $\text{N}_2\text{O}$  production to occur at a lower % WFPS than for chemical fertilisers (Akiyama et al., 2004). Incorporation of manures into the soil immediately after application, and the method of slurry application can also influence the extent of  $\text{N}_2\text{O}$  and  $\text{NH}_3$  emissions (Webb et al., 2010). However, the use of these methods and their degree of success will depend on the presence/stage of crop growth.

The amount of  $\text{N}_2\text{O}$  or  $\text{NH}_3$  emitted from N sources applied to soils is often calculated using an emission factor (EF), which defines the quantity of  $\text{N}_2\text{O}$  or  $\text{NH}_3$  emitted as a proportion of the total N applied. The UK currently uses the IPCC's Tier 1 EF in its national  $\text{N}_2\text{O}$  inventory, where  $\text{N}_2\text{O}$  emissions from soils receiving organic amendments are equal to 1% of the total N applied (IPCC, 2006), with no accounting for locally variable factors such as soil type or climate, variations in manure type, or

the time of application. The IPCC default EF for  $\text{NH}_3$  emission following manure application to land is 20% of the applied N. However, the EF used to estimate  $\text{NH}_3$  emissions from manure application in the UK  $\text{NH}_3$  emissions inventory is derived from an empirical model taking account of manure type and some soil and climatic factors (Nicholson et al., 2013).

The variety of conditions affecting N loss from soils amended with livestock manures mean it is imperative that applications are carefully managed to avoid significant environmental pollution. It is vital to understand how the form and time of application may affect environmental impacts. The results of the research presented in this paper which forms part of a nationwide project, will contribute to reducing uncertainty in the UK's agricultural GHG inventory, and will enhance the sustainability and GHG mitigation potential of farming systems (GHG, 2013). This study aimed to compare soil  $\text{N}_2\text{O}$  and  $\text{NH}_3$  emissions and EFs following autumn and spring manure applications to arable land in Scotland. Nitrous oxide and  $\text{NH}_3$  emissions were measured for all manure types following application in both seasons, and the suitability of the IPCC Tier 1 EFs to represent  $\text{N}_2\text{O}$  and  $\text{NH}_3$  emissions from different manure types and seasons of application was assessed. Effects of the timing and form of manure application on crop yield and crop N uptake were also investigated, to assess the impact of the type and time of manure application on crop production.

## 2. Materials and methods

### 2.1. Site description and experimental design

Two 12 month field experiments were undertaken at Boghall farm (NT 248653, 190 m elevation), in East-central Scotland in 2012/2013. Both experiments were located in the same field, on a sandy loam soil (pH 6, 6% OM), with a 30 year (1980–2009) site mean annual precipitation of 979 mm and mean daily temperature in July and January of 14.3 °C and 3.3 °C, respectively. Spring barley (*Hordeum vulgare*) had been grown in the field for the previous four years. The site was one of a network of UK sites measuring emissions as part of the UK GHG research platform, and was selected following a geographical assessment of UK arable land under a range of soil/climatic zones, and a 'gap analysis' to identify zones lacking in current/planned experimental data. The entire field, covering both experimental areas was sown with winter wheat (*Triticum aestivum*), a typical crop for the area, on 25th October 2012 at a seed rate of 400  $\text{m}^{-2}$ . In the first experiment manures were applied on 3rd October 2012 to assess  $\text{N}_2\text{O}$  and  $\text{NH}_3$  emissions following autumn applications. In the second experiment the crop was left untreated until 10th April 2013, when manures were applied to assess emissions from spring applications. The crop was harvested from both experimental areas on 15th September 2013, with all stubble left in the field and no new crop planted until after the experiment ended. Both experiments ran for a total of 12 months following the date of manure application.

Manure treatments (cattle farmyard manure (autumn only), CFYM; broiler litter, BL; layer manure, LM; cattle slurry by trailing hose application, CSH; cattle slurry by surface broadcast application, CSSB) and a control (to which no manure was added) were applied to a fully replicated randomised block design with three blocks, in each experiment respectively. Treatment plots were orientated in a NNE direction and measured 12 m × 6 m. Manures were applied at rates commonly practised for the specific manure type, with target application rates for the solid manures of 180 kg total N  $\text{ha}^{-1}$ , and for slurry of 40  $\text{m}^3 \text{ha}^{-1}$ . In the autumn experiment all manures apart from CFYM were incorporated into the bare soil 24 h after application, and in the spring experiment they were top dressed on the growing crop. The plots were orientated at 20° to the vertical, and 90° to the prevailing wind to minimise the carry-over of volatilised  $\text{NH}_3$  from one plot to another when making  $\text{NH}_3$  emission measurements using wind tunnels. Target N application rates and the results of manure chemical analysis were used to calculate total manure application rates. Actual N application rates varied between treatments (Table 1) as a result of changes in the N concentration

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