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AMMONIA AND NITROUS OXIDE EMISSIONS FOLLOWING FIELD-APPLICATION OF MANURE: STATE OF THE ART MEASUREMENTS IN THE NETHERLANDS

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by

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ABSTRACT.

Manure application to grassland and arable land is an important source of ammonia and nitrous oxide losses. For both gasses, national and international policies have been developed with the objective of reducing the emissions. Since the early 1990s, measurements have been carried out in The Netherlands to assess the gaseous losses from manure application, especially for ammonia. Measurements of nitrous oxide emissions are relatively scarce. This paper presents the results of these measurements with the objective of providing an updated quantification of the effect of techniques for application and incorporation of manure, and to assess influencing factors. The manure application techniques differ in their spreading or placement of the manure onto the grass or soil surface or into the soil. The following techniques are treated in this paper: surface spreading, narrow-band application, shallow injection with open slots on grassland, and surface spreading, surface incorporation, deep placement on arable land. Low emission techniques such as narrow band application, shallow injection, incorporation or injection on arable land show a significant reduction of ammonia emission compared to surface spreading. On grassland, average emission factors (% of total ammonium nitrogen) were 74% for surface spreading and 16% for shallow injection. On arable land, the emission factors were 69% for surface spreading and 2% for deep placement. However, the nitrous oxide emission factor from manure applied with low ammonia emissions techniques is higher than the emission factor for surface applied manures. In a whole farm context, the higher nitrous oxide emission with shallow injection is partly offset directly by reduced emissions from fertiliser savings and indirectly from lower ammonia losses.

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Keywords: ammonia, nitrous oxide, manure application techniques, grassland, arable land.

1. INTRODUCTION.

1.1. General background.

For centuries, livestock manure has been utilised on farms as an important, if not the only, source of nutrients for the growing of crops. The storage, handling and use of these manures are associated with the volatilisation of ammonia and other gasses, among which is nitrous oxide.

Ammonia (NH_3) and nitrous oxide (N_2O) emissions have in various ways a negative impact on the environment. Gaseous ammonia contributes importantly to the problem of acidification (Van Breemen *et al.*, 1982). The deposition of ammonia-nitrogen also contributes to the eutrophication of forests and other natural ecosystems. An increased availability of nitrogen (N), in combination with soil acidification, may cause a disturbed nutrient balance and nutrient deficiencies.

Nitrous oxide (N_2O) is a greenhouse gas with a Global Warming Potential approximately 296 times higher than carbon dioxide (CO_2), and an atmospheric lifetime of about 120 years (IPCC, 2007). Nitrous oxide also contributes to the depletion of stratospheric ozone (IPCC, 1995). Atmospheric concentrations of N_2O have increased from approximately 270 ppb during the pre-industrial era to 319 ppb in 2005.

1.1.1. Ammonia.

Manure application on farmland is an important source of ammonia (NH_3) volatilisation. In the 1980s air contamination by ammonia demanded a proper quantification of ammonia volatilisation from field-applied manure and a search started for tools to reduce the NH_3 volatilisation.

In the Netherlands livestock manure is by far the most important source of ammonia losses (Buijsman *et al.*, 1987; Heij and Schneider, 1995). The annual ammonia volatilisation from animal manure was estimated to be more than 200 million kg in 1980 and was about 150 million kg in 1998 and 1999. In 1980 37% of the total ammonia volatilisation from agriculture originated from animal housing, 56% from field application of manure and 7% from grazing cattle. In 1999, the corresponding figures were 50%, 41% and 9% (Anon., 2000).

In the 1980s and early 1990s, the environmental problems associated with the use of livestock manure became a major issue of the Dutch government's environmental policy (Anon., 1984, Anon., 1993). The need arose for an efficient recycling of nutrients to create a sustainable agricultural use of manure. The reduction of ammonia volatilisation after manure application to farmland received much attention, among other factors. The contribution from this source was the largest, and measures to reduce ammonia volatilisation after manure application seemed to be easy to introduce at relatively low costs. Furthermore, the effects of measures to reduce ammonia volatilisation in animal housing and during storage would be relatively small, if no measures to reduce volatilisation after field application were taken. Ammonia saved in housing or storage would, after all, volatilise when applying the

manure to farmland without volatilisation-reducing measures. Injection of liquid manure into grassland and incorporation of manure into arable land were the first measures considered for the reduction of ammonia volatilisation. However, Wadman (1988) estimated that only 33% of the grassland in the Netherlands is suitable for deep injection. The conventional deep injector worked at a depth of *circa* 25-30 cm and had lateral wings that cut the soil horizontally underneath the sward. Unsuitability for deep injection is caused by the required draught force and crop damage along the slit on various soil types, and the remains of tree trunks in the soil. Therefore, new application techniques for grassland had to be developed. The effect of new techniques on ammonia volatilisation and the suitability of new techniques on different soil types under varying soil conditions had to be evaluated. On arable land, the effect of the choice of implement to incorporate the manure was insufficiently known and the effect of time delay between application and incorporation needed to be assessed, because ammonia volatilisation peaks directly after surface spreading.

The Dutch ministry of agriculture, farmers' organisations, research institutes and manufacturers jointly worked on solutions to reduce ammonia volatilisation. In other countries, ammonia volatilisation has attracted similar attention in recent years, partly due to the national targets set by the EC directive (EC, 2001). Thus the need emerged for knowledge about the volatilisation of ammonia, and for practical measures to reduce ammonia volatilisation from manure applied to farmland. In the Netherlands research was initiated for liquid manures (slurry) from dairy and pigs, being the dominant types of manure applied to farmland. Low emission application techniques have been a requirement in the Netherlands since 1995, and broadcast surface spreading is not allowed.

1.1.2. Nitrous oxide.

The agricultural sector contributes to global warming potential through the emission of methane (CH_4) and nitrous oxide. The major sources of methane are enteric fermentation in ruminants and manure storages. Nitrous oxide is emitted directly from soils and manure storage, and indirectly (i.e. downstream and downwind) through nitrate leaching and ammonia losses. The soil emissions are related to different N inputs such as fertiliser, applied animal wastes, animal excreta during grazing, mineralisation of organic matter, and crop residues. In this paper, the focus is on emissions from applied animal manures.

Animal wastes applied to soils are an important source of direct N_2O emission from agricultural soils (Mosier *et al.*, 1998). In the European Union (EU-15), approximately 20% of the direct soil emission is attributed to manure application (EEA, 2006). To put this into perspective: some 40% of the emission is attributed to fertiliser application. In the Netherlands, the contribution from applied animal manures is around 50% of the direct soil emissions. Fertiliser, grazing excreta and cultivation of peat soils are the other main sources.

Application of manure increases the contents of NH_4^+ and mineralisable N and carbon (C) in the topsoil, and thereby activates microbial activity in the soil. This in turn may increase nitrification and subsequently denitrification locally. The organic compounds in slurry and manure provide readily available substrate for denitrifiers.

The default IPCC emission factor, i.e. the percentage of applied N emitted as N_2O , is 1.00%, irrespective of the type of fertiliser or manure (IPCC, 2006). Also application time, application method and application level are not considered. The default emission factor is based on an average of published measurements on a wide range of soil types, crops, fertiliser types, N sources, N levels and application times and methods (Bouwman, 1996; Bouwman *et al.*, 2002; Stehfest and Bouwman, 2006). Individual countries can use a different emission factor that corresponds to country-specific soil types or farm management. The data used to derive the IPCC default emission factor originate predominantly from fertiliser experiments. The lack of experiments with organic N sources is in contrast with the farming practice of intensively managed grasslands in Europe, where manures are an important source of N for crops.

In the Netherlands, the emission factor for applied manure depends on the application technique (www.greenhousegases.nl). For surface spreading, the IPCC default of 1% is used, but for low NH_3 emission application techniques a factor of 2% is used. This distinction between application methods was based on a review by Kroeze (1994). This is of significant importance, because in 1990, the reference year for the Kyoto protocol, all manure was surface applied to the soil. However, from the 1990's onwards an increasing proportion of manure was applied with low NH_3 emission techniques. According to the Dutch monitoring protocol, this change in manure application techniques resulted in higher direct N_2O emissions from agricultural soils. In a second literature review, Kuikman *et al.* (2006) were unable to derive new emission factors for these techniques. Their review included, amongst others, results from Chadwick (1997), Sommer *et al.* (1996) and Clemens *et al.* (1997). Where Chadwick (1997) reported increased emissions from injection, the others did not find any significant effect of application method.

1.2. Processes.

1.2.1. Ammonia.

Ammonia volatilisation is often determined by a combination of factors and by interactions between these factors. The rate of ammonia volatilisation decreases with time, when the ammonia source (the manure) becomes exhausted by volatilisation and take-up in the soil. Ammonia fluxes sometimes show a day and night cycle. Due to a low wind speed, low temperature and high relative humidity, ammonia volatilisation may decrease during the night, and may rise again in the daytime. However, in the long term, the ammonia source becomes exhausted, the ammonia volatilisation rate decreases to the background level, and the effect of weather conditions disappears.

Table 1: *Main factors affecting the volatilisation of ammonia from manure (Huijsmans, 2003).*

Factor		Direction of change
Manure properties	pH	+
	TAN ^a content	+
	Water content	-
	Dry matter content	+
	Crusting ^b	-
Meteorological factors	Air temperature	+
	Solar radiation	+
	Wind speed	+
	Rainfall	-
	Relative humidity	-
Crop and soil properties	Presence of crop residues ^b	+
	Soil moisture content	+/-
	Infiltration rate	-
	CEC	-
	Soil pH	+
Application technique	Band application ^b	-
	Injection ^b	-
	Direct incorporation ^b	-

+ denotes that an increase of the magnitude of the factor increases ammonia volatilisation.

- denotes that an increase of the magnitude of the factor decreases ammonia volatilisation.

^a TAN, total ammoniacal nitrogen ($\text{NH}_4^+ + \text{NH}_3$).

^b The factors marked with a (^b) are either present or absent; the effect of the presence of the indicated factor is denoted as + or -, when the presence of the factor increases or decreases ammonia volatilisation, respectively

A higher application rate increases the volatilisation rate (Horlacher and Marschner, 1990; Thompson *et al.*, 1990) due to a larger source of ammonia. The higher application rate also slows down the infiltration of manure into the soil. However, ammonia volatilisation, expressed as percentage of the total ammoniacal nitrogen ($\text{NH}_4^+ + \text{NH}_3$) applied to the field, may vary between high and low application rates (Horlacher and Marschner, 1990; Pain and Klarenbeek, 1988; Thompson *et al.*, 1990). The volatilisation percentage may be even higher at low application rates, because a thin layer of manure may dry faster, causing an increase of the total ammoniacal nitrogen (TAN) concentration in the manure (Brunke *et al.*, 1988). Crust formation, however,

may hamper the diffusion of gaseous ammonia from the manure ($\text{NH}_{3,g,m}$) into the air ($\text{NH}_{3,g,a}$) (Thompson *et al.*, 1990). Jarvis and Pain (1990) mention the concentration of ammoniacal nitrogen, pH and dry matter content as the main manure characteristics that determine ammonia volatilisation. Volatilisation that occurs after field application of the manure will be influenced by factors such as weather conditions, soil type and soil condition, presence of a crop, and application rate. Furthermore, the application technique, dilution of the manure, or supply of additives to the manure may affect ammonia volatilisation. Interactions between these factors may be complex. Brunke *et al.* (1988) suggested that ammonia volatilisation is more influenced by the manure composition and application technique, than by any other factor. The influence of application technique, manure characteristics, weather and soil conditions is shown from statistical analysis of large number of field experiments and model studies of the emission process (Bussink *et al.*, 1994; Génereumont and Cellier, 1997; Huijsmans and De Mol, 1999; Huijsmans *et al.*, 2001, 2003; Misselbrook *et al.*, 2002, 2005; Smith *et al.*, 2000; Sommer and Hutchings, 2001; Sommer *et al.*, 2003; Søgaard *et al.*, 2002). The main factors influencing ammonia volatilisation and their expected effects are summarised in Table 1 (previous page).

1.2.2. Nitrous oxide.

Nitrous oxide is produced through different microbial soil processes. The most important are thought to be nitrification, nitrifier denitrification and denitrification. A short description, mainly taken from Wrage *et al.* (2001) is given below. Furthermore these pathways are shown in Figure 1.

- Nitrification is the stepwise oxidation of ammonium (NH_4) or ammonia (NH_3) to nitrate (NO_3) via nitrite (NO_2). These reactions are carried out by two groups of micro-organisms, together addressed as Nitrobacteriaceae. These are aerobes and many are obligate autotrophs. N_2O is formed during NH_3 oxidation through chemical decomposition of intermediates between NH_4 and NO_2 such as NH_2OH or NO_2 itself.
- Denitrification is the stepwise reduction of NO_3 to N_2 . Several intermediates are developed which can be emitted. The reactions are carried out by denitrifiers, which are widely distributed across the bacterial taxa, including *Pseudomonas*, *Bacillus*, *Thiobacillus*, and *Propionibacterium*. These predominantly heterotrophic micro-organisms are facultative anaerobes that are able to use NO_3^- instead of oxygen in respiration under low-oxygen or anaerobic conditions. In contrast to nitrification, N_2O is a regular intermediate of denitrification. The portion of the intermediate N_2O that is released is higher if the pH is low. The ratio $\text{N}_2\text{O} / \text{N}$ also rises if NO_3^- is abundant in the soil. If some oxygen is present, the ratio also increases. But at high O_2 concentrations, the aerobic metabolism of denitrifiers is promoted so that the reduction of NO_3^- does not take place. In summary, N_2O is an intermediate of denitrification, which can be released in high quantities in low-oxygen environments with sufficient NO_3^- and metabolisable organic C.

- In nitrifier denitrification, the oxidation of NH_3 to NO_2 is followed by the reduction of NO_2 to N_2O and N_2 (Figure 1). This sequence of reactions is carried out by only one group of micro-organisms, namely autotrophic NH_3 -oxidisers. Nitrifier denitrification should not be confused with coupled nitrification–denitrification, where different groups of coexisting micro-organisms can together transform NH_3 to N_2 . The term is used to stress that NO_2 or NO_3 produced during nitrification can be utilised by denitrifiers. This coupling between nitrification and denitrification can take place in soils where favourable conditions for both nitrification and denitrification are present in neighbouring microhabitats

Apart from these main processes, other pathways have also been identified, but according to current knowledge, they are less important. Besides autotrophic nitrifiers using nitrification as an energy source for fixing carbon dioxide (CO_2), heterotrophic nitrifiers are also known. These nitrifiers use organic carbon (C) as a source of C and energy. Although heterotrophic nitrification is generally considered to be only a minor source of N_2O , it might produce significant amounts of N_2O under certain sets of circumstances such as low pH, high oxygen amounts and availability of organic material. Finally, chemo-denitrification is the chemical decomposition of intermediates from the oxidation of NH_4 to NO_2 or of NO_2 itself with organic or inorganic compounds. It is a non-biological reaction usually taking place at a low pH.

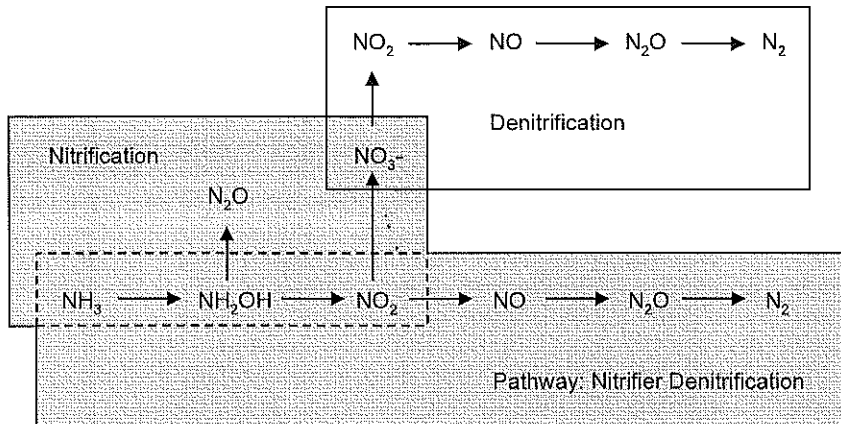


Figure 1: Transformations of inorganic nitrogen in soil (Wrage et al., 2001).

The environmental conditions favouring the different pathways of the N cycle lead to complicated distribution of nitrification, denitrification and nitrifier denitrification in soils. However, taking all pathways into account, the main factors determining the amount of N_2O production are temperature, pH, moisture content and the availability of inorganic N and decomposable C (Figure 2, overleaf). It has to be stressed that the relationships are not as clear as Figure 2 might suggest. For instance, it is known that there is an optimal range for the pH, but the exact values are uncertain. The N_2O losses increase

with increasing soil moisture content up to an optimum of around 60 to 80% water-filled pore space. Short-lived events of high rainfall can give sudden increases in N_2O production. Recently, Vermeulen and Mosquera (2009) showed that soil compaction increases N_2O emission in arable systems.

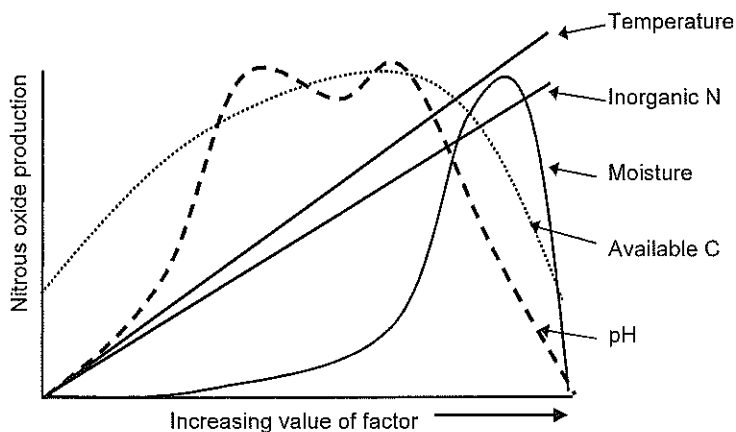


Figure 2: *Factors determining the production of nitrous oxide (Van Groenigen, 2007).*

1.3. Scope.

Since the early 1990s a large number of measurements have been carried out in The Netherlands to assess the ammonia volatilisation from manure applied by different manure application techniques to grassland and to arable land. In the beginning these measurements were carried out specifically to assess the difference in cumulative ammonia volatilisation following respectively broadcast surface application of manure and manure application by various low-emission techniques. Later on, when low emission techniques were regulated by law (since 1995), measurements were carried out under various conditions to reveal effects of characteristics of the manure, of the weather and of the soil, on the ammonia flux. The results up to 1998 were reported by Huijsmans *et al.* (2001). Additional measurements on grassland, improvement of statistical techniques and the need for solid general emission factors for national emissions registrations including the detection of trends, required an update of the analysis of data on ammonia emissions in the Netherlands. Furthermore a good estimation of the nitrous oxide emissions when applying manure with different manure application techniques was lacking.

This paper deals with the ammonia and nitrous oxide emissions following manure application and incorporation on farmland in the Netherlands. The objective of this paper is to provide an updated quantification of the effect of techniques for application and incorporation of manure on these emissions and to assess influencing factors, based on all available data for the Netherlands.

2. MANURE APPLICATION.

Manure application techniques differ in their placement of the manure onto the grass or soil surface or into the soil. In the experiments to assess the ammonia or nitrous oxide emission, commercially available application implements were used in all cases. Figure 3 shows the placement of the manure by the various application techniques for grassland and arable land.

Manure placement - grassland



Broadcast surface spreading. Manure spread on top of the grass.



Narrow band application by trailing shoe (or foot). Manure in bands on top of the soil between the grass leaves



Shallow injection open slot. Manure in slots in the grass sod.

Manure placement - arable land



Broadcast surface spreading. Manure spread on top of the soil.



Incorporation of surface applied manure or injection of manure.

Figure 3: *Placement of the manure when applying manure with different application and incorporation methods.*

On grassland and arable land *surface spreading* was carried out by a tanker fitted with a splash-plate. The manure was pumped through an orifice onto a splash-plate from where it was spread onto the soil and the grass. The net working width was about 8 m.

The techniques for the application of manure in narrow bands and for manure injection on grassland have been described by Huijsmans *et al.* (1998).

- *Narrow-band application* on grassland was carried out by trailing narrow sliding feet (also called 'shoes') over the soil surface, pushing aside the grass cover but not cutting the sward. Each foot was 0.37 m long and 0.02 m wide and was kept horizontal by a parallelogram construction. Manure was released at the back of the feet leaving narrow bands of manure on the soil surface. The bands had a width of about 0.03 m and were spaced 0.20 m apart. Contamination of the grass with manure was negligible. A tanker was equipped with 25 trailing feet with a total working width of 5 m.
- *Shallow injection* (open slot) was carried out with injection coulters. Coulters and discs were used to cut vertical slots into the grass sward. The

vertical slot could be cut by an angled-disc coultter or a double-disc opener, a thick-disc coultter, a disc coultter followed by a vertical injection coultter, or a knife coultter followed by a vertical injection coultter. Manure was released into the slots, which were left open. The slots were up to 0.05 m deep and were spaced 0.20 m apart. The total working width of the implements used was 4.0 to 5.6 m. Depending on the application rate, the slots were more or less filled with manure. Unlike the conventional deep injector, the shallow injectors used had no lateral wings and did not cut the soil horizontally underneath the sward.

On arable land different application techniques or incorporation techniques were used in the experiments. Next to surface spreading the techniques could be suitably grouped into two application methods, based on their positioning of the manure on or into the soil: surface incorporation and deep placement. Surface incorporation was defined as the treatment by which manure was surface spread and, subsequently, incorporated into the soil. Conventional tillage implements (cultivators with rigid tines, spring tines, discs, or harrows) were used to incorporate the surface-applied manure into the topsoil directly following the surface spreading. Deep placement was defined as the treatment by which the manure was buried in the soil, either directly by an injector or indirectly by ploughing with a mouldboard plough directly after surface spreading. The arable land injector was equipped with spring tines, which placed the manure directly underneath the soil surface at a depth of 15 to 20 cm. At the same time the injector carried out a tilling operation by covering the manure with soil.

3. AMMONIA EMISSIONS FOLLOWING MANURE APPLICATION.

3.1. Measurements.

In the ammonia emissions analyses the only experiments taken into account are those in which the volatilisation of NH_3 following manure application was determined per plot using the micrometeorological mass balance method (Denmead, 1983; Ryden and McNeill, 1984), as applied by Huijsmans *et al.* (2001 and 2003). Measurements continued for 96 h after manure was applied. During the first 12 h – when the rate of NH_3 volatilisation was highest – NH_3 traps were replaced 4 to 5 times. Further replacement took place every morning for the following 4 days. The amount of NH_3 volatilised during each interval was calculated from the amount of NH_3 trapped and the airflow data.

Table 2: *Number of ammonia observations (n) on grassland and arable land, according to the various application techniques.*

Grassland	n	Arable land	n
Surface spreading	81	Surface spreading	26
Narrow-band application	29	Surface incorporation	25
Shallow injection	89	Deep placement	7

The Dutch data of experiments on grassland from 1989 – 1993 and on arable land from 1990 – 1998 are reported by Huijsmans *et al.* (2001) and Huijsmans *et al.* (2003). Additional measurements were made on grassland from 1995–2003. The total number of observations was 199 on grassland and 58 on arable land, relating to various application techniques (Table 2).

Table 3: *Means and ranges (in parentheses) of various characteristics of the conditions during the experiments on grassland.*

Variable	Surface spreading	Narrow-band application	Shallow injection
TAN ^a content (g/kg)	2.4 (1.5 - 6.4)	2.8 (1.8 – 6.4)	2.1 (1.6 – 6.3)
Application rate (m ³ /ha)	15 (8 – 25)	14 (7 – 28)	22 (10 – 45)
Wind speed (m/s)	3.3 (0.4 -8.7)	3.4 (0.4 – 7.2)	3.0 (0.4 – 7.3)
Radiation (J/cm ² /h)	103 (0 – 318)	100 (0 – 300)	105 (0 – 375)
Air temperature (°C)	15 (2 – 32)	15 (3 – 32)	16 (0 – 32)
Relative humidity (%)	73 (16 – 100)	72 (34 – 100)	73 (24 – 100)
Grass height (cm)	7 (4 – 12)	7 (5 – 12)	8 (5 – 14)
Soil moisture content (%)	34 (13 – 67)	34 (24 – 67)	30 (13 – 61)
pH	7.2 (6.8 – 8.0)	7.2 (6.9 – 8.0)	7.3 (6.7 – 8.0)
Dry matter content of manure (g/kg)	80 (46 – 119)	83 (56 – 113)	80 (52 – 113)

^a TAN, total ammoniacal nitrogen ($\text{NH}_4^+ + \text{NH}_3$) at time of application.

Table 4: *Means and ranges (in parentheses) of various characteristics of the conditions during the experiments on arable land.*

Variable	Surface spreading	Surface incorporation	Deep placement
TAN ^a content (g/kg)	4.4 (2.4 - 6.1)	4.6 (2.8 - 6.1)	4.9 (2.8 - 6.1)
Application rate (m ³ /ha)	22 (14 - 39)	25 (12 - 43)	27 (19 - 38)
Wind speed (m/s)	4.1 (0.5 - 8.6)	3.9 (0.5 - 8.7)	3.5 (0.5 - .6)
Radiation (J/cm ² /h)	88 (0 - 298)	73 (0 - 270)	56 (0 - 230)
Air temperature (°C)	14 (0 - 34)	14 (5 - 25)	13 (7 - 26)
Relative humidity (%)	78 (22 - 100)	81 (55 - 100)	82 (58 - 98)
Stubble height (cm)	5 (0 - 16)	7 (0 - 16)	6 (0 - 12)
Soil moisture content (%)	18 (9 - 26)	18 (9 - 29)	15 (10 - 21)
Dry matter content of manure (g/kg)	86 (55 - 153)	83 (55 - 136)	89 (64 - 107)

^a TAN, total ammoniacal nitrogen ($\text{NH}_4^+ + \text{NH}_3$) at time of application.

A number of characteristics of the manure, the weather conditions and the field were recorded. Cow manure and pig manure were used on grassland and pig manure only on arable land. Wind speed, air temperature, relative humidity and global radiation were recorded every 10 minutes, but were averaged over the duration of each interval that NH_3 volatilisation was measured. The means and ranges of various data are presented in Table 3 (grassland) and Table 4 (arable land) on page 13.

3.2. Data analysis.

3.2.1. Application method comparisons.

The original scope of the research was to reveal the reduction of the cumulative ammonia volatilisation following various low-emission manure application and incorporation methods, compared with traditional surface spreading. For this purpose, the cumulative volatilisation (expressed as % of TAN applied with the manure) was statistically analysed. The basis for modelling was a saturation curve (Figure 4), describing the NH_3 volatilisation in the period following application by the following equation (Huijsmans *et al.*, 2001; Huijsmans *et al.*, 2003):

$$\mu_i(t) = t / (\beta_{0i} + \beta_{1i}t) \quad \dots \dots \dots \text{Equation 3.1.}$$

where $\mu_i(t)$ is the expected value of the cumulative volatilisation at time t for treatment i , t the time lapsed since the manure was applied, β_{0i} the inverse of the slope of the curve for treatment i at the start of the experiment, β_{1i} the inverse of the intercept of the asymptote on the ordinate of the curve for treatment i .

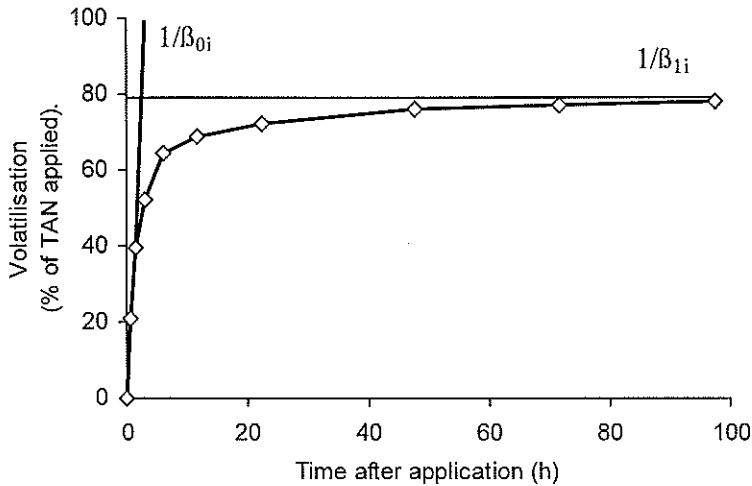


Figure 4: Ammonia volatilisation as the calculated cumulative emissions with the initial volatilisation rate ($1/\beta_{0i}$) and the total emission (intercept, $1/\beta_{1i}$).

After linearisation of the equation, the REML procedure (Payne *et al.*, 1993), with $1/t$ and treatment factor as fixed terms was used to estimate the treatment parameters (β_{0i} and β_{1i}) and the random effects (with 'experiment' as random term in both slope and intercept) in a linear mixed model (LMM). The total cumulative volatilisation of treatment i , when t approaches infinity, was calculated as $\mu_i = 1/\beta_{1i}$ and was used as the basis for testing the differences between methods (Huijsmans *et al.*, 2001; Huijsmans *et al.*, 2003). In this paper the low-emission methods were analysed pair-wise using sets with data from only those experiments in which the application or incorporation method was simultaneously compared with surface spreading.

3.2.2. Effects of characteristics of manure, soil and weather.

The analysis of the effect of characteristics of the manure, of the weather and of the soil on the ammonia flux focused on modelling the declining volatilisation rate (expressed as kg NH₃-N/ha/h) after application depending on time itself, characteristics of the manure and the soil, and weather characteristics during a 96 h time span after application. As the change in TAN content of the manure (depletion of the source due to volatilisation) could be calculated from the data, the adjusted total ammoniacal nitrogen content (ATAN) was used in the analysis. Other data on changes in characteristics of the manure or the soil, such as TAN lost from the available pool by infiltration, adsorption or biochemical changes, were not available. The analysis was carried out on all available data for arable land and grassland, separately for each method of application. The basic equation used to describe the logarithm of the volatilisation rate z at time t after application is:

$$\ln^1 z(t) = \alpha_0 + \alpha_t \ln(t) + \sum \alpha_m x_{mt} \quad \dots \quad \text{Equation 3.2.}$$

where α_0 is a constant, t the time lapsed since the manure was applied, α_t the coefficient for the term $\ln(t)$, α_m the coefficient for the term x_{mt} , x_{mt} the model term for explanatory variable m at time t .

The logarithm of time after manure application was included in the model because the decline of the volatilisation rate with time could not be fully explained from the change in NH₃ content (ATAN). Other factors analysed for inclusion into the equation were weather conditions (wind speed, air temperature, relative humidity and radiation), soil type (sand, sandy loam, clay and peat), soil moisture content, manure characteristics (ATAN content and dry matter content), application rate, and stubble/grass height.

The REML procedure, with factors and terms as described in Equation 3.2, and 'experiment' as random term, was used to estimate the constant and the coefficients of models with various explanatory variables. Wald tests (Payne *et al.*, 1993) were used for model selection to identify influencing variables that contribute significantly to the explanation of the experimentally-found volatilisation rates ($P < 0.05$).

¹ \ln = natural logarithm.

3.2.3. Emission factors per method.

The earlier estimates of the cumulative emission per method were taken from the tests on difference between application methods (Huijsmans *et al.*, 2001; Huijsmans *et al.*, 2003). These estimates were calculated as the reciprocal of the mean of β_1 . However national estimates and balances of emissions preferably use mean values of cumulative emissions calculated as the average of the cumulative emission per event. Moreover, the estimates were based on a restricted dataset with the purpose of pair-wise comparisons between methods. The best estimate on a national level should be based on all available data per method. Therefore, for each application method, relevant cumulative emission figures were derived by fitting the saturation curve through the measured cumulative emissions per observation, estimating the total emission per observation by the reciprocal of β_1 . The emission factor was calculated as the average of the total emission values per observation for each application method, based on all available data.

Particularly for the shallow injection method on grassland, a trend seemed to be visible in the data that the total emissions increased since the observations of the total emission started, in 1989. For all application methods for grassland an analysis was carried out to determine whether or not the ammonia fluxes to the atmosphere increased since the start of the experiments. For this purpose, a new variable y , being the number of years since 1988 was calculated for each observation. Variable y was added as a factor in the final model (Equation 3.2) for each application method, thus preventing possible changes in application conditions being responsible for the observed trend.

3.3. Results.

3.3.1. Application method comparisons.

Table 5: *Estimated reciprocal of the cumulative ammonia volatilisation (β_1), number of measurements (n) per application method using the data sets for pair-wise comparison of low emission methods with surface spreading, and least significant difference ($P < 0.05$) between the methods.*

	Low emission method		Surface spreading		lsd
	β_1	n	β_1	n	
Grassland					
Narrow band application	0.0417	19	0.0145	15	0.0109
Shallow injection	0.1437	25	0.0179	30	0.0440
Arable					
Surface incorporation	0.0415	22	0.0147	14	0.0033
Deep placement	0.7742	7	0.0287	7	0.2606

Table 5 shows the estimated reciprocal of the total ammonia volatilisation, β_1 , for each method when comparing the various low emission methods directly

with surface spreading. In all cases, NH_3 volatilisation is significantly reduced by using the low-emission method.

3.3.2. Effects of characteristics of manure, soil and weather.

The analysis of the factors that influence the volatilisation rate (Table 6) revealed that the decrease of the volatilisation rate with time (t) after application was significant for all methods. This indicates that the source of ammonia for volatilisation depletes not only by volatilisation itself, which is accounted for by the term ATAN, but also by other factors that were not measured, for instance take-up of ammonia in the soil. The volatilisation rate was affected by temperature, wind speed and the manure application rate for all methods except deep placement of manure on arable land. No factor except $\ln(t)$ affected the volatilisation rate when the manure was deeply placed into the soil on arable land. ATAN, relative humidity, incoming radiation, grass stubble height and soil type affected the volatilisation rate for some methods only. Dry matter (DM) content of the manure showed not to be a significant factor.

Table 6: Results of the analysis by method of factors that influence the volatilisation rate.

Factor	Grassland			Arable land		
	surface spreading	narrow band	shallow injection	surface spreading	surface incorporation	deep placement
$\ln(t)$	-	-	-	-	-	-
Adjusted TAN (ATAN)	ns	+	ns	+	+	ns
Application rate	+	+	+	+	+	ns
Wind speed	+	+	+	+	+	ns
Air temperature	+	+	+	+	+	ns
Relative humidity	-	-	-	ns	ns	ns
Incoming radiation	+	ns	ns	+	+	ns
Grass/stubble height	-	-	ns	ns	ns	ns
DM content of manure	ns	ns	ns	ns	ns	ns
Soil type: sand	a	.	a	a	a	a
clay	a	a	b	b	a	a
sandy loam /loam	.	.	.	b	a	a
peat	a	a	ab	.	.	.
% of variation accounted for	84	78	69	89	80	57

ns = not significant ($P < 0.05$);

a, b = different letters indicate significant difference ($P < 0.05$);

- = negative correlation with volatilisation rate;

+ = positive correlation with volatilisation rate;

. = no data available.

3.3.3. Emission factors per method for grassland and arable.

The emission factors, defined as the average total emission per method in % of TAN applied with the manure, based on all available data (Tables 1 and 2) are presented in Table 7, including the ranges in total emission per method.

Table 7: *Emission factors (% total ammoniacal nitrogen [TAN] applied) and range in the data per application method.*

Method	Total emission %	Minimum %	Maximum %
Grassland			
surface spreading	74	28	100
narrow band	26	9	52
shallow injection (average)	16	1	63
shallow injection (trend)	19	-	-
Arable land			
surface spreading	69	30	100
surface incorporation	22	3	45
deep placement	2	1	3

The trend analysis revealed that measured volatilisation rates after shallow injection on grassland had increased significantly since 1989 when the experiments started (Figure 5). As a possible explanation for the increased volatilisation, it is suggested that the furrows made during shallow injection became less deep over the years, thereby gradually shifting towards the method of manure placement using the narrow band application method. Because the year 1999 was the last year with many observations of the total volatilisation for shallow injection, the estimated total volatilisation in this year, being 19%, was taken as the most representative emission factor (Table 7, shallow injection, trend). No effect of time since 1989 on the total ammonia volatilisation was found for surface spreading and narrow band application on grassland.

4. NITROUS OXIDE EMISSIONS FOLLOWING MANURE APPLICATION.

Measurements of nitrous oxide emissions of applied manure are surprisingly scarce in the Netherlands, given the amounts of animal manure applied to grassland and arable land. On an annual basis, approximately similar amounts, nearly 300 mio t N, are applied from fertiliser and animal manure. A series of laboratory and field experiments have been carried out between the years 2000 and 2002. Recently, new experiments have started in 2007, and at present these are still running.

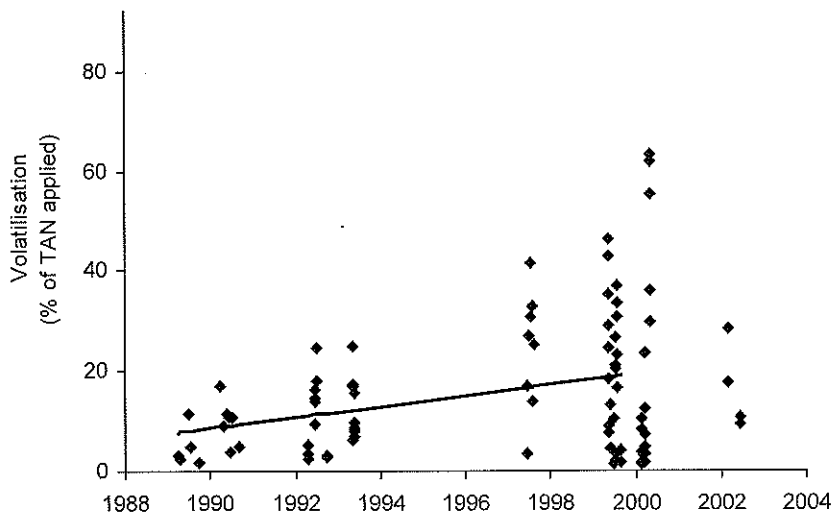


Figure 5: *Ammonia volatilisation from field-applied manure by shallow injection on grassland.*

4.1. Field experiments.

Field experiments were carried out to establish the year-round N_2O emissions of current fertilising and manuring practices in the Netherlands.

4.1.1. Methods.

The experiments were mainly carried out on grassland sites on sandy soils in the vicinity of Wageningen (Schils *et al.*, 2008). One experiment was located at an arable site on a clay soil in the northern parts of the Netherlands (Van Groenigen *et al.*, 2004). The experiment consisted of a comparison of treatments receiving either nothing, only fertiliser, only manure, or a combination of fertiliser and manure. The fertiliser used was calcium ammonium nitrate (CAN, 13.5% $\text{NH}_3\text{-N}$ and 13.5% $\text{NO}_3\text{-N}$), which is the most commonly used fertiliser type in the Netherlands. The cattle manures were liquid slurries with an average total N content of 4 to 5 g/kg, of which 45 to 50% was $\text{NH}_4\text{-N}$. Fertiliser was broadcast applied by hand. The cattle slurry was applied, according to common practice on mineral soils in the Netherlands, with low emission techniques. On grassland a shallow injection technique was used, applying the cattle slurry in open equidistant slits (20 cm), approximately 2 cm wide and with a depth of 5 to 10 cm. On arable land, the slurry was surface spread in rows, and directly incorporated.

Nitrous oxide fluxes were measured throughout the year. The measuring frequency was up to three times a week after fertiliser and cattle slurry application and two to four times a month in the remainder of the growing season. To prevent effects of diurnal variation, all emission measurements were carried out between 09.00 and 12.00 hours.

The N₂O concentrations were measured in the headspace of vented closed PVC flux chambers, with a diameter of 16 cm and a height of 15 cm, using a Brüel and Kjær photo-acoustic spectroscopic infrared gas analyser (Velthof and Oenema, 1995a). The flux chambers were placed onto semi-permanent rings, inserted into the soil to a depth of 2 cm. The rings remained in the soil during each harvest cycle, and were moved to new positions after each fertiliser and cattle slurry application. On grassland, the placement of the rings on the plots receiving cattle slurry received special attention in order to achieve the correct proportion between applied and non-applied surface within the flux chambers. Therefore the centre of the rings was placed at a distance of 6 cm from the application slit.

The N₂O concentration was measured before the chambers were closed and approximately 45 to 60 minutes after closing. The analyser was attached directly to the flux chambers by two Teflon tubes. A soda lime trap was installed at the inlet of the analyser to prevent undesired accumulation of carbon dioxide (CO₂). The analyser was fitted with optical filters for selective measurement of N₂O, CO₂ and water vapour. Any remaining CO₂ was automatically corrected for. The accuracy of the analyser was approximately 5% in the range of 300 to 5,000 ppb.

The N₂O flux was calculated as the difference between ambient concentration and the concentration in the closed chamber, assuming a linear relationship between concentration and time (Velthof and Oenema, 1995b), which was occasionally checked during the experiment. The accumulated fluxes were calculated by linear interpolation between measurement days. Emission factors (EF) are calculated from the N₂O emission of fertilised and/or manured plots (N₂O fert), the N₂O emission of control plots (N₂O zero) and the amount of total N applied, all expressed in kg N/ha:

$$EF [\%] = (N_2O \text{ fert} - N_2O \text{ zero}) * 100 / N \text{ applied}$$

4.1.2. Results.

Overall, the emission factors in this experimental series ranged between 0.10 and 1.21% of total applied N (Figure 6). On grassland, nitrous oxide emissions from manure were generally equal or slightly higher than those from calcium ammonium nitrate fertiliser. The emissions on arable land were higher than those on grassland, but this effect was linked with the effect of soil type. The grassland experiments were all carried out on sandy soils, while the arable experiment had one location on sand and another location on clay. On sandy soils the average level of emission was lower, but manure emitted more nitrous oxide per kg N than calcium ammonium nitrate. On clay soils, the average level of emission was high, but there was no difference between manure and calcium ammonium nitrate.

On sandy soils, the observed N₂O emission levels were rather low compared to the IPCC default emission factor of 1.0%. It is suspected that the low values of water-filled pore space (WFPS) during the experiment is the driving factor for the low N₂O emissions, in line with findings in literature (Del Prado *et al.*,

2006; van Groenigen *et al.*, 2005), that suggest maximal N_2O losses at 70 to 80% WFPS. The sparse WFPS measurements in the experiment are not conclusive, but the observed range between 18 and 55% suggests poor conditions for N_2O production.

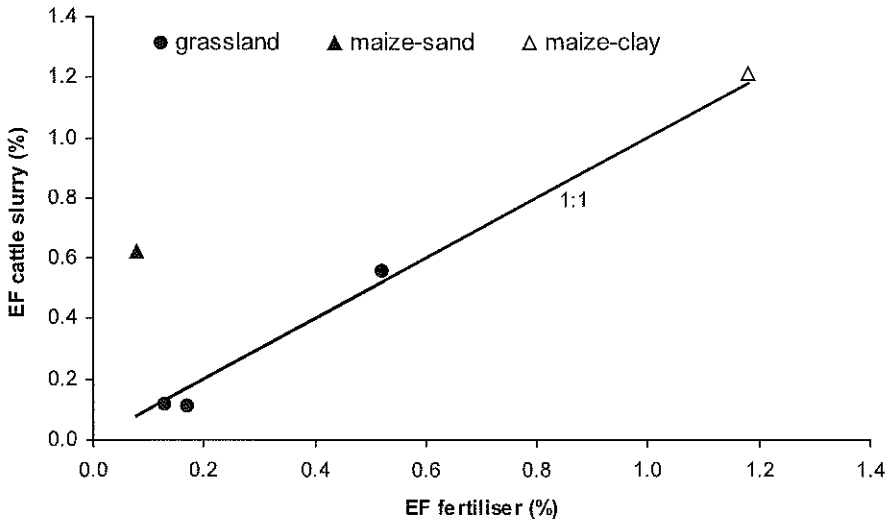


Figure 6: *Emission factor (EF, %) for application of cattle slurry to grassland and arable land (forage maize), compared to emission from calcium ammonium nitrate (CAN) fertiliser.*

The emissions showed a temporal variation that was mainly driven by fertiliser and cattle slurry application events. Emission peaks generally occurred within the first week following application of fertiliser and cattle slurry (Figure 7, overleaf). This is in agreement with other studies on intensively managed grasslands (Velthof and Oenema, 1995b). Grasslands can rapidly absorb the applied N, and soil mineral N contents usually decrease quickly within two weeks of application. Therefore the period of highest N_2O emission risk on grasslands are found within the first two weeks after application.

The emission factors were calculated from the treatments receiving only fertiliser or only manure. This is in contrast with the farming practice of intensively managed grasslands in Europe, where both fertiliser and cattle slurry are usually applied shortly after one another. In those cases, interactions between those different N sources should be considered. Higher N_2O losses may be expected due to enhanced denitrification through the simultaneous availability of fertiliser derived nitrate (NO_3) and cattle slurry derived easily degradable carbon (C). Therefore these experiments also included treatments with combined application of fertiliser and manure. However, this expected effect was not apparent in these experiments. Only in

one case (grassland, 2001) were the observed emissions from fertiliser higher due to the addition of cattle slurry. This is in contrast to the results of other researchers (Clayton *et al.*, 1997; Dittert *et al.*, 2005; Stevens and Laughlin, 2001, 2002). They reported higher N₂O emissions when fertiliser and cattle slurry were applied simultaneously than when they were applied separately or with a larger interval between the applications. However, these other experiments were all carried out with surface application of manure, which involves a more direct contact between fertiliser and slurry.

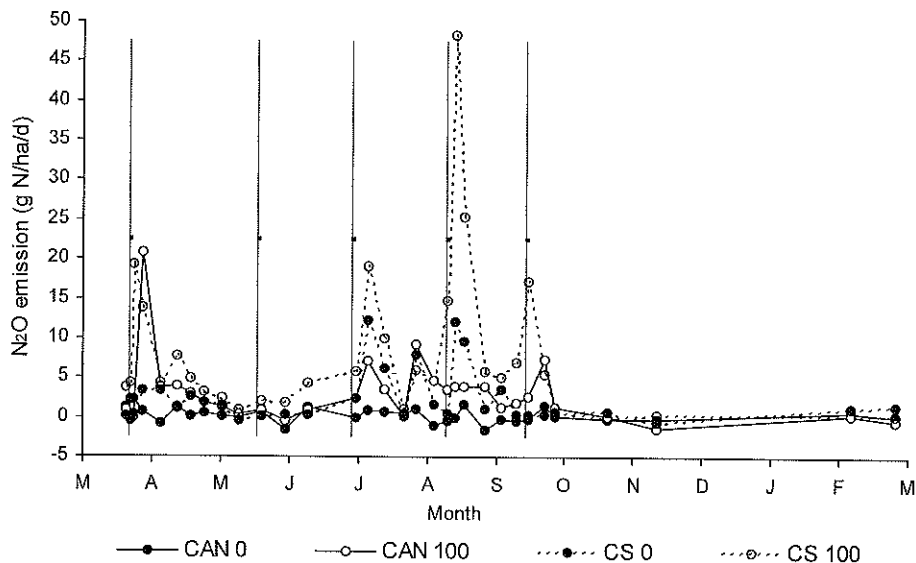


Figure 7: Nitrous oxide emissions during a grassland experiment of four selected treatments. (Control: CAN 0, fertiliser only: CAN 100, cattle slurry only: CS 0, fertiliser and cattle slurry: CS 100). Vertical lines represent the application dates of fertiliser and cattle slurry.

4.1.3. Current experiments.

In 2007, new field experiments were started to verify the difference in nitrous oxide emissions between surface application and low emission application of manures (Mosquera *et al.*, 2009). The experiments were carried out on a sandy soil near Wageningen on a grassland site and an arable site, cultivated with forage maize. The experiments continued until the end of 2009. The treatments were: no fertiliser, calcium ammonium nitrate, surface applied cattle slurry and low emission applied cattle slurry. On grassland the low emission technique was shallow injection, while on arable land the slurry was injected into the soil. The N application level of the fertiliser treatments was approximately 55% of the N application level of the manure treatments. This is to account for differences in the effectiveness of fertiliser and manure N.

The nitrous oxide measurements were carried out in the same way as in the previous experiments.

The application of CAN fertiliser on grassland resulted in higher N_2O emissions compared to emissions from cattle slurry (Figure 8). On maize, the opposite was observed, with higher N_2O emissions from cattle slurry. Compared to surface spreading, the use of low ammonia emission manure application techniques resulted in higher N_2O emissions, both for grassland and maize.

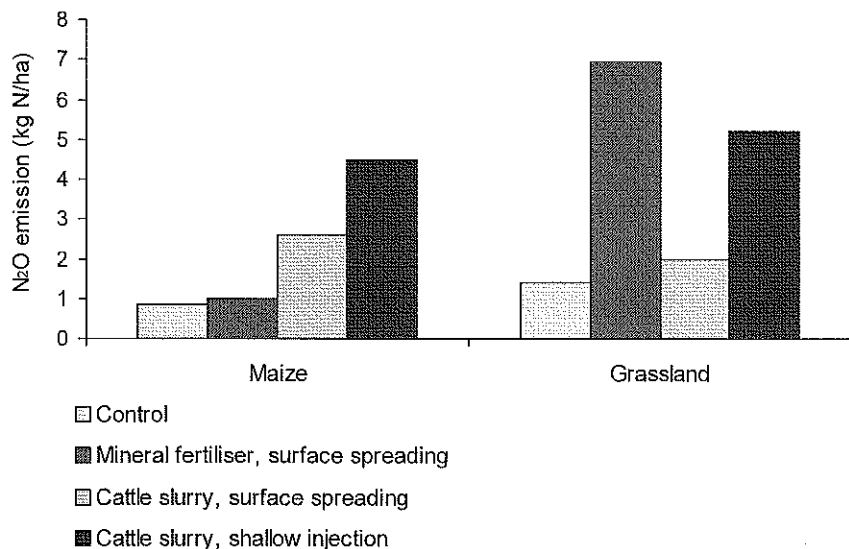


Figure 8: Total N_2O emissions measured during the period April 2007 to March 2009.

4.2. Laboratory experiments.

Experiments under controlled conditions were carried out to determine the effects of manure type and composition on N_2O emissions.

4.2.1. Methods.

Incubation experiments (Velthof *et al.*, 2003) were carried out with 500 g moist sandy soil in 1 litre jars at 15°C. Nine manures were selected which included liquid and solid manures with both low and high C and N contents (Table 8). The manures were collected from farms in the Netherlands and analysed for dry matter, pH, total and mineral N content, and total C contents. Fluxes of N_2O were measured 30 times over a 98-day period. The concentration of N_2O in the headspace was measured 1 h after closing the jar. The N_2O flux was calculated assuming a linear relationship between the N_2O concentration in the headspace and time which was checked several times (Velthof and Oenema, 1995a).

Table 8: *Chemical composition and application rate (100 mg N/kg) of the manures.*

Manure	DM	Total N	NH ₄ -N	NO ₃ -N	Total C	Mineral N % of total N	C/N ratio organic matter	pH	Application rate g/kg soil
	g/kg product								
Cattle slurry; traditional farming	114	5.31	1.66	0	51	31	14	6.67	19
Cattle slurry; organic farming	106	3.74	1.33	0	44	36	18	6.75	27
Young cattle slurry	106	5.93	2.55	0	44	43	13	7.71	17
Liquid pig manure; traditional farming	19	2.46	1.47	0	7	60	7	7.23	41
Liquid pig manure; organic farming	144	5.12	1.48	0	58	30	16	6.26	20
Liquid sow manure	20	1.9	0.98	0	7	52	7	7.16	53
Layer manure	644	32.72	2.91	0	256	9	9	5.73	3
Broiler-breeder manure	699	51.6	3.98	0	255	8	5	6.22	2
Duck manure	255	8.44	1.8	0	103	21	15	7.73	12

4.2.2. Results.

The total N₂O emission varied considerably between the fertilisers and manures (Table 9).

The composition of the animal manure had a large effect on the emission of N₂O. The N₂O emissions were higher from the three pig manures (7.3 to 13.9%), than from the cattle slurries (1.8 to 3.0%) and poultry manures (0.5 to 1.9%). Regression analysis indicated that both mineral N and total C contents of the applied manures were major factors controlling N₂O emission.

In these incubations there was no removal of N by plant uptake or leaching, so that the mineral N content in the soil was relatively high throughout the whole experimental period. This may have resulted in higher N₂O emissions than under field conditions, especially for grassland. However, in arable farming systems, manures are often applied several weeks or months before the crop is grown.

Table 9: *Total N₂O emission from soil for all fertilisers and manures. Different letters indicate statistically significant differences (=0.05) in log-transformed N₂O emission between treatments.*

Treatment	N ₂ O emission		
	mg N/kg soil		% of N applied
Control	0.6	a	-
Ammonium sulphate - (NH ₄) ₂ SO ₄	4.6	cd	4.0
Ammonium nitrate - NH ₄ NO ₃	2.7	bc	2.1
Liquid pig manure; traditional farming	7.9	de	7.3
Liquid pig manure; organic farming	8.1	bde	7.5
Liquid sow manure	14.5	e	13.9
Cattle slurry, traditional farming	3.6	cd	3.0
Cattle slurry, organic farming	2.4	bc	1.8
Young cattle slurry	2.5	bc	1.9
Layer manure	2.5	bc	1.9
Broiler manure	1.1	ab	0.5
Duck manure	1.2	ab	0.6

A separate study under controlled conditions has been undertaken to determine the effect of nitric-acid-treated cattle slurry (Velthof and Oenema, 1993). Acidification of slurries was tried in the early 1990s to reduce NH₃ emissions, but is no longer permitted. Untreated slurry was compared with nitric-acid-treated slurries with pHs of 6.0 and 4.5, respectively, and with CAN fertiliser. The N₂O losses from acidified slurries were more than tenfold higher than with the untreated slurries. However, the differences between treated slurry and fertiliser were only very small. So apparently, the amount of nitrate was the determining factor for N₂O emission.

5. DISCUSSION AND CONCLUSIONS.

Narrow band application and shallow injection on grassland, and incorporation and injection on arable land were shown to reduce ammonia emissions significantly. Compared to surface spreading the placement of the manure is different and in some cases deeper in the soil. In theory this could increase the risk that roots may not reach the nutrients in time. The savings on mineral N fertiliser could be lost and extra leaching might occur. However, experiments with deep injection (20 cm) did not show higher leaching losses on sandy soil, as long as the increased N utilisation was compensated by a lower fertiliser application rate (Jansen, 1990).

Recent experiments on grassland show that the N utilisation is higher after shallow injection than after surface spreading (Figure 9). The Nitrogen Fertiliser Replacement Value (NFRV) (=apparent N recovery of manure/apparent N recovery of fertiliser) after shallow injection was *circa* 20% (range 15%-28%) higher than after surface spreading. This confirms results of earlier research, taking into account that the earlier research was more focused on deep injection (Wouters *et al.*, 1994). Little research has been carried out on the N utilisation of narrow band application. Wouters *et al.* (1994) showed a N utilisation of 12% and 49% after surface spreading and narrow band application, respectively. The difference in N utilisation is mainly caused by the lower ammonia emission after shallow injection or narrow band application. However, other effects like sward damage may affect the N utilisation as well. The level of ammonia emission, however, may also depend on the weather conditions shortly after manure application and the TAN content of the manure, in addition to the application technique (Huijsmans *et al.*, 2001). As for phosphorus (P), placement of manure too deep may have a negative effect on the availability of P at early growth stages where P is most needed when demand per unit root length is high (Neeteson *et al.*, 2006).

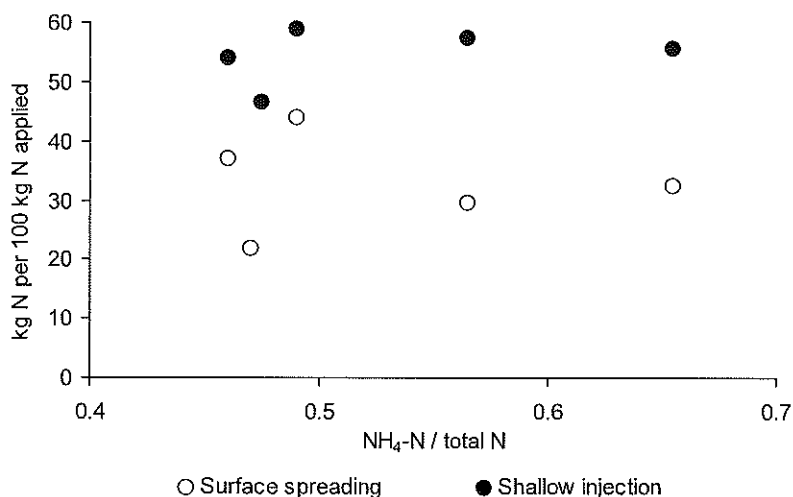


Figure 9: Nitrogen fertiliser replacement value of manure applied to grassland (i.e. kg mineral fertiliser N saved per 100 kg manure N applied) on sandy soils as a function of the application technique (surface spreading and shallow injection) and the TAN fraction ($\text{NH}_4\text{-N}/\text{total N}$) in the manure (Schils and Kok, 2003; Geurink and Van der Meer, 1995; Schröder *et al.*, 2007).

Nitrogen utilisation on arable land is also improved by the low emissions application techniques. Furthermore the P utilisation may be improved by these techniques due to better placement of the manure close to the rows. (Schröder, 2005).

The present study shows that reduction of NH_3 volatilisation can be achieved by incorporation of the manure into the soil and that the degree of reduction depends on the method of incorporation. Direct burying with a mouldboard plough (deep placement) yielded more reduction of NH_3 volatilisation than incorporation by a rigid tine cultivator (surface incorporation). Huijsmans and De Mol (1999) showed that incorporation by a mouldboard plough does not always result in lower NH_3 volatilisation than incorporation by a rigid tine cultivator. The present study does not account for the effect of a time-lag between surface spreading and incorporation on the NH_3 volatilisation. On the experimental plots the manure was immediately incorporated and the time-lag was minimal. In practice on a whole field scale, immediate incorporation is not always achievable. There will always be some time between surface spreading and incorporation and during this time volatilisation of NH_3 from the surface-applied manure takes place. The study of Huijsmans and De Mol (1999) showed that the time-lag between spreading and incorporation should be considered when assessing NH_3 volatilisation from manure applied and incorporated on arable land. In case of deep placement by injection the time-lag is zero and low volatilisation rates can be achieved, as shown in the present study.

Low-emission manure application techniques have been compulsory in The Netherlands since the mid 1990s. Broadcast surface spreading of manure has been banned since then. In a recent evaluation the side-effects of low emission application were studied (Huijsmans *et al.*, 2008). The study showed that low emission application, in general, does not harm soil structure, soil life nor the crop compared to surface spreading. The breeding season of meadow birds and periods most suitable for manure application may coincide in some years. In that situation low-emission application causes the loss of more clutches. However, the strong decline of meadow bird numbers in The Netherlands is mainly caused by other agricultural interventions. The well-drained farm land, moderate precipitation, flat land and an effective network of contractors to apply manure are part of the wide acceptance of low-emission applications in the Netherlands.

Low emission application techniques are a part of the abatement of ammonia emissions from agriculture. To assess and control ammonia emissions the whole farm system should be considered. The feeding strategy, housing, manure handling and storage, grazing management and fertiliser use also affect the ammonia emissions on farm and national scale (Hutchings *et al.*, 2001; Misselbrook *et al.*, 2000; Reidy and Menzi, 2007; Velthof *et al.*, 2009; Webb and Misselbrook, 2004; Webb *et al.*, 2005). Abatement techniques in these areas affect the available TAN applied to farmland and therefore affect the emissions after field application. Furthermore weather conditions during and after manure application significantly affect ammonia emissions.

Low emission manure application affects the availability of mineral N and may affect the amount of oxygen around the applied manure. Lack of oxygen in deeper soil layers and/or in wet conditions may stimulate denitrification. The results of recent ongoing experiments seem to confirm the higher N_2O

emission with shallow injection on grassland and injection on arable land. The analysis focused on emissions from manure application. However, manure application is only one of many sources of ammonia and nitrous oxide emissions. When considering options to reduce NH_3 or N_2O emissions, they should always be viewed in a whole farm context. This is necessary to prevent any negative trade-offs in the N cycle via fertiliser, soil, crop, feed, animal and manure. This can be illustrated with an analysis of nitrous oxide emissions from manure application.

Table 10: *Comparison of nitrous oxide (N_2O) emissions from manure applied to grassland (30 t/ha), either with shallow injection or surface application.*

	Shallow injection	Surface spreading
Application rate (t/ha)	30	30
Total N in slurry (kg/t)	4.2	4.2
N application (kg N/ha)	126	126
Manure NFRV (%)	50	25
Effective N from manure (kg/ha)	63	32
NH_3 emission factor (%)	8	37
NH_3 emission (kg N/ha)	10	47
Fertiliser (kg N/ha)	187	219
Emission factor (%)		
Fertiliser application	1	1
Fertiliser production	0.5	0.5
Manure application	2	1
NH_3 deposition	1	1
Nitrous oxide (kg N/ha)		
Fertiliser application	1.9	2.2
Fertiliser production	0.9	1.1
Manure application	2.5	1.3
NH_3 deposition	0.1	0.5
Total	5.4	5.0

Table 10 compares an application of 30 tonnes of cattle slurry to grassland with shallow injection or with surface application. In both cases, the total N application is 126 kg N/ha. The ammonia-N losses are 10 kg/ha for shallow injection and 47 kg/ha for surface application. The higher NFRV of shallow injection reduces the fertiliser requirement. In this example, a total annual N application of 250 kg/ha is assumed. Shallow injection requires a fertiliser

supplementation of 187 kg/ha, while surface application requires a supplementation of 219 kg N/ha.

As shown in this paper, the direct N_2O emissions are higher with shallow injection. In this case the losses are twice as high as with surface application. However, the N_2O emissions from the use and production of fertiliser are higher for surface application. Furthermore the indirect N_2O emissions, arising from the ammonia losses are also higher for surface application. In a whole farm context, there is still an advantage for surface application, but smaller than the difference of manure application alone.

5.1. Conclusions.

Low emission techniques like narrow band application and shallow injection on grassland and incorporation or injection on arable land show a significant reduction of ammonia emission compared to surface spreading. These techniques also show a better N utilisation than surface spreading, which is at least partly caused by the reduction of ammonia emission.

The nitrous oxide emission factor from manure applied with low ammonia emissions techniques is higher than the emission factor for surface applied manures. The N_2O emission of manures increases with increasing contents of nitrogen and easily degradable carbon. Therefore, the observed emission of pig slurries is higher than the emission of cattle slurries.

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