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# Nitrous oxide emission from dairy farming systems in the Netherlands

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## Abstract

A large part of the nitrogen (N) input in dairy farming systems in the Netherlands is lost from the system via N leaching and volatilization of gaseous N compounds, including the greenhouse gas nitrous oxide ($N_2O$). The aim of the present study was to quantify $N_2O$ emission from dairy farming systems in the Netherlands, using a whole-farm approach. A total of 14 $N_2O$ sources was identified and emission factors were derived for each of these using literature. Figures are presented for the amounts of $N_2O$ produced per kg herbage N produced (ranging from 4 to 89 g $N_2O$-N $kg^{-1}$ herbage N), depending on soil type and grassland management. Using Monte Carlo simulations, variations in mean total $N_2O$ emissions from the different sources were calculated for three model dairy farming systems differing in nutrient management. These different farming systems were chosen to assess the effect of improved nutrient management on total $N_2O$ emission. The total direct $N_2O$ emissions ranged from 15.4 ± 9.4 kg $N_2O$-N $ha^{-1}$ $yr^{-1}$ for the average dairy farming system in the eighties to 5.3 ± 2.6 kg $N_2O$-N $ha^{-1}$ $yr^{-1}$ for a prototype of an economically feasible farming system with acceptable nutrient emissions. Leaching-derived, grazing-derived and fertilizer-derived $N_2O$ emissions were the major $N_2O$ sources on dairy farming systems. The total direct $N_2O$ emissions accounted for 3.2 to 4.6% of the N surplus on the dairy farming systems, suggesting that only a small amount of N was lost as $N_2O$. Total $N_2O$ emissions from dairy farming systems in the Netherlands were estimated at 13.7 ± 5.1 Gg N $yr^{-1}$, which is about 35% of the estimated total $N_2O$ emission in the Netherlands. It is concluded that improvement of nutrient management of dairy farming systems will significantly decrease the $N_2O$ emissions from these systems, and thus the total $N_2O$ emission in the Netherlands.

*Keywords:* budget, dairy farming systems, emission factors, grassland, nitrous oxide, nutrient management, Monte Carlo simulations, The Netherlands

## Introduction

Dairy farming is the dominant land use system in the Netherlands. A total of $1.02 \times 10^6$ ha of land is under permanent grassland and $0.23 \times 10^6$ ha under forage maize, which together amounts to more than 30 percent of the total area of the Netherlands

(Anonymous, 1995a). The permanent grasslands are used for grazing and forage production. Both grassland and maize are intensively managed to allow high yields of good quality forage. These high yields are obtained among others by the application of plant nutrients via animal slurry and fertilizers.

The reverse side of the intensification of the dairy farming systems shows up in the large surplus of nitrogen (N) on budgets of these farming systems (Korevaar, 1992). On average, about 80% of the N input on dairy farming systems in the Netherlands is not recovered in animal products. Part of this N may be incorporated in soil organic matter, but the major part of this excess N is lost from the system via nitrate ($NO_3^-$) leaching, and volatilization of ammonia ($NH_3$), dinitrogen ($N_2$), nitrogen oxides ($NO_x$) and nitrous oxide ($N_2O$). Nitrous oxide is a trace gas involved in both the enhanced greenhouse effect and the destruction of stratospheric ozone and is produced during oxidation-reduction reactions of nitrogenous compounds (Houghton *et al.*, 1996). Major $N_2O$ sources are denitrification and nitrification in soil and a few studies have been carried out to quantify the $N_2O$ emission from grassland soils (e.g. Egginton & Smith, 1986; McTaggart *et al.*, 1994; Velthof *et al.*, 1996a). Other possible sources of $N_2O$ on dairy farming systems are generally assumed to be unimportant. There are many interactions within the complex N-cycle of dairy farming systems. It is well known that changing the nutrient management may affect the N flows at many different places (e.g. Aarts *et al.*, 1992) and also the $N_2O$ leakages from the N cycle. This complicates a straightforward quantification of the effect of nutrient management on $N_2O$ emission from dairy farming systems.

The aim of the present study was i) to identify the major sources of $N_2O$ production in dairy farming systems, and ii) to quantify the $N_2O$ emission from dairy farming systems in the Netherlands, using a whole-farm approach. We chose for a whole-farm approach to be able to quantify all possible sources of $N_2O$ on dairy farming systems. Emission factors were derived from literature for the different $N_2O$ sources and the possible variations in total mean $N_2O$ emission were calculated using Monte Carlo simulations for three model dairy farming systems, largely differing in nutrient management. These different farming systems were chosen to assess the effects of nutrient management on total $N_2O$ emission.

## Materials and methods

### *Description of farms*

Emissions of $N_2O$ were calculated for three model dairy farming systems on sandy soil in the Netherlands, namely i) the average farm in the eighties, mentioned as Farm '80, ii) farm Kloosterboer, and iii) the experimental model farm De Marke. These dairy systems were chosen because they strongly differed in nutrient management and because the major N flows of these systems are well described, based on measurements and modeling. Data on land use, milk production and the major N flows for the three dairy farming systems are presented in Table 1.

Data for Farm '80 were based on nutrient budgets of groups of specialist dairy

Table 1. Some key properties of the three dairy farming systems.

| Property | | Farm '80 | Kloosterboer | De Marke | Relative standard deviation[2] |
|---|---|---|---|---|---|
| Area | grassland, ha | 22 | 20 | 31 | |
| | maize, ha | 3 | 12 | 18 | |
| | fodder-beet, ha | 0 | 0 | 6 | |
| Milk production, kg $ha^{-1}$ $yr^{-1}$ | | 13195 | 12760 | 11724 | |
| N flows and pools[1], kg N $ha^{-1}$ $yr^{-1}$ | | | | | |
| | Purchased N fertilizer | 330 | 156 | 53 | 5% |
| | Purchased concentrates | 136 | 90 | 82 | 5% |
| | Purchased roughage | 42 | 0 | 2 | 5% |
| | Produced cattle slurry | 198 | 232 | 209 | 25% |
| | Soil applied cattle slurry | 120 | 206 | 185 | 15% |
| | N excreted during grazing | 191 | 167 | 52 | 25% |
| | Biological N fixation | 4 | 4 | 12 | 15% |
| | Nitrate leaching | 200 | 130 | 50 | 100% |
| | Ammonia volatilization | 109 | 38 | 24 | 50% |
| | Silage-nitrate | 8 | 5 | 5 | 15% |
| | N surplus | 477 | 249 | 141 | |

[1] Assumptions
- Farm '80 and farm Kloosterboer: Biological N fixation: 4 kg N $ha^{-1}$
- Farm '80: Aarts *et al.* (1992) presented total N emission by leaching and denitrification. It is assumed that 70% of this N was leached and 30% was denitrified, based on leaching-denitrification ratios for a moderately drained sandy loam given by Scholefield *et al.* (1991).
- Farm Kloosterboer: Amount of N leached was calculated from: N surplus = $NH_3$ emission + denitrification + leaching + other loss, assuming that 70% of the total amount of N lost by leaching + denitrification was lost by leaching (Scholefield *et al.*, 1991) and that 'other loss' accounted for 10% of the N surplus. Emission of $NH_3$ was derived from Den Boer *et al.* (1990).

[2] Relative standard deviations used in the Monte Carlo calculations

farms on sandy soil during 1983–1986 (Aarts *et al.*, 1992). The farm Kloosterboer has introduced a package of measures in 1988 to reduce nutrient emissions to the environment. These measures included expanded storage capacity of slurry, injection of slurry on grassland in spring and summer, no autumn and winter applications of slurry, application of N fertilizer and slurry strictly according to current recommendations, and restricted grazing. We calculated $N_2O$ budgets for 1991/1992 using data derived from Den Boer *et al.* (1990), Den Boer (1993) and Nutrient Management Institute (NMI, unpublished results). The experimental farm De Marke started in 1992, with the aim to develop a prototype of an economically feasible farming system with acceptable nutrient emissions. Measures include those taken at Kloosterboer, but are more stringent. Much attention is given to the nutrition of the cattle, to obtain a high production of milk per cow and a high efficiency of utilization of ingested N. The $N_2O$ budget was calculated for 1993/1994 using data presented by Aarts *et al.* (1994).

*Flows of N and sources of $N_2O$*

The major N flows and pools in dairy farming systems are presented in Figure 1. There is a rapid cycling of N in this system. Input of N is via N fertilizer, purchased roughage and concentrates, biological N fixation, and atmospheric deposition. Output is via milk, meat and N emissions. Accumulation of N may (temporarily) occur in the soil, slurry storage basins and in roughage and silage.

The magnitudes of the major N flows on the three dairy farms were derived from the pertinent literature (Table 1). Standard deviations were assigned to the mean magnitude of the N flows based on additional literature and best guess values: the larger the variability or uncertainty in the N flow the larger the standard deviation (Table 1).

Emissions of $N_2O$ from the dairy farming system occur from the top soil, the sub soil, cattle, slurry storage units, and sites where fuel is burned. When the site of $N_2O$ production is inside the dairy farming system sources are mentioned as direct sources of $N_2O$. Sources of $N_2O$ outside the dairy farming system are mentioned as indirect sources, e.g. $N_2O$ that is emitted during the production of the purchased N fertilizer, roughage and concentrates. We included these three indirect $N_2O$ sources in the calculations, because the management of the farming system strongly affects the magnitude of these sources. Other possible indirect $N_2O$ sources were not considered.

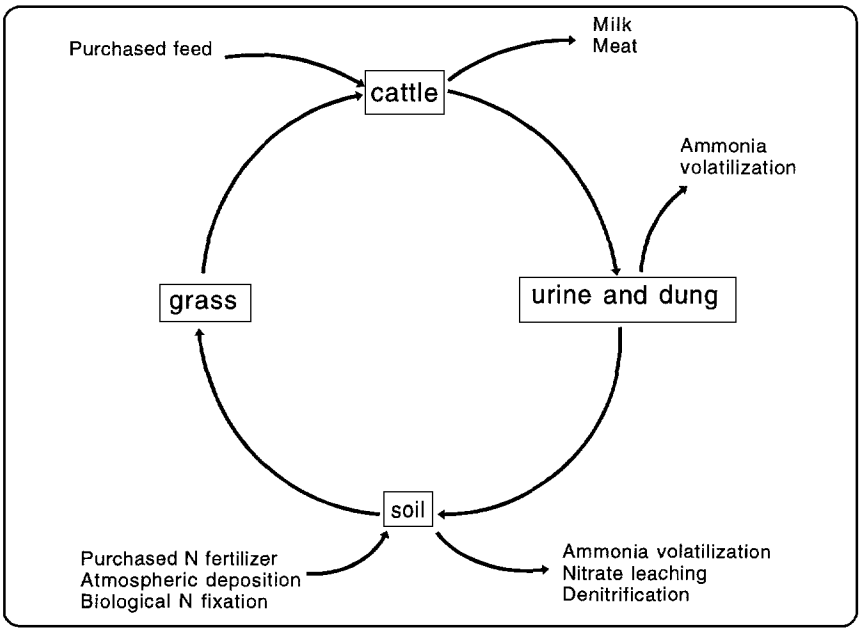


Figure 1. Schematic presentation of the major N flows in the N cycle in dairy farming systems. Essential all pools and flows are capable of releasing $N_2O$.

*Emission factors of $N_2O$*

A mean $N_2O$ emission factor was assigned to each source, based on literature data or a best guess if literature data were not available. The emissions factors are expressed in g $N_2O$-N per kg N, assuming a linear relationship between the N flow and $N_2O$ production. Emission factors are generally applied in $N_2O$ budget studies (Kroeze, 1994; Houghton *et al.*, 1996). Because the $N_2O$ production is highly stochastic by nature, the variability in $N_2O$ emissions is large both in time and space (e.g. Velthof *et al.*, 1996a, b). Consequently, the mean emission factors have a large standard deviation or in case of best guess values there is a large uncertainty. We assigned a 'standard deviation' to all mean emission factors based on literature data and best guess values. The direct and indirect sources are shortly described and $N_2O$ emission factors are given for both mineral and peat soils. A summary of the emission factors is given in Table 2.

*Background $N_2O$ emission from soils*

Unfertilized and mown-only grasslands produce $N_2O$ during nitrification and/or denitrification of N released during mineralization of soil organic N and from atmospheric N deposition. In the calculations we estimate the background $N_2O$ flux for sandy and clayey soils at 900 ± 300 g $N_2O$-N $ha^{-1}$ $yr^{-1}$ and for peat soils at 5300 ± 5200 g $N_2O$-N $ha^{-1}$ $yr^{-1}$, based on the results of a $N_2O$ monitoring study on grasslands of two years on sand, clay and peat soils (Velthof *et al.*, 1996a).

Table 2. Emission factors used in the calculations: mean ± standard deviation.

| Source of $N_2O$ | Mineral soils | Peat soils |
|---|---|---|
| Direct | | |
| Soil, background, g $N_2O$-N $ha^{-1}$ $yr^{-1}$ | 900 ± 300 | 5300 ± 5200 |
| Soil, N fertilizer, g $N_2O$-N $kg^{-1}$ fertilizer N | 10 ± 5 | 30 ± 13 |
| Soil, cattle slurry, g $N_2O$-N $kg^{-1}$ slurry N | | |
| Surface-applied | 3 ± 3 | 6 ± 6 |
| Application with low $NH_3$ emission | 5 ± 5 | 10 ± 10 |
| Grazing, g $N_2O$-N $kg^{-1}$ excreted N | 25 ± 15 | 60 ± 46 |
| Biological N fixation, g $N_2O$-N $kg^{-1}$ fixed N | 5 ± 5 | 5 ± 5 |
| Leaching, g $N_2O$-N $kg^{-1}$ leached N | 25 ± 25 | 25 ± 25 |
| Housing and slurry storage, g $N_2O$-N $kg^{-1}$ slurry N | 0.05 ± 0.05 | 0.05 ± 0.05 |
| Ammonia volatilization, g $N_2O$-N $kg^{-1}$ $NH_3$-N | 5 ± 5 | 5 ± 5 |
| Silage, g $N_2O$-N $kg^{-1}$ $NO_3^-$-N | 15 ± 10 | 15 ± 10 |
| Rumen, g $N_2O$-N $kg^{-1}$ consumed N | 0.05 ± 0.05 | 0.05 ± 0.05 |
| Energy use, g $N_2O$-N $GJ^{-1}$ | 1 ± 1 | 1 ± 1 |
| Indirect | | |
| Purchased N fertilizer, g $N_2O$-N $kg^{-1}$ fertilizer N | 5 ± 5 | 5 ± 5 |
| Purchased roughage, g $N_2O$-N $kg^{-1}$ roughage N | 20 ± 10 | 20 ± 10 |
| Purchased concentrates, g $N_2O$-N $kg^{-1}$ concentrate N | 10 ± 5 | 10 ± 5 |

*Fertilizer-derived $N_2O$ emission from soils*

The fertilizer-derived $N_2O$ emissions (the amount of applied N fertilizer that is lost as $N_2O$) for grassland fertilized with calcium ammonium nitrate (CAN) were set at $10 \pm 5$ g $N_2O$-N $kg^{-1}$ N for mineral soils and at $30 \pm 13$ g $N_2O$-N $kg^{-1}$ N for peat soils, based on the study of Velthof *et al.* (1996a). For arable land, we also use $10 \pm 5$ g $N_2O$-N $kg^{-1}$ N as emission factor for fertilizer-derived $N_2O$ emission on mineral soils. There is evidence that $N_2O$ emissions are larger from $NO_3^-$ containing fertilizers than from fertilizers only containing $NH_4^+$, especially during wet conditions (McTaggart *et al.*, 1994; Velthof *et al.*, 1997). We will not distinguish between the different N fertilizers, because CAN is by far the major N fertilizer on grassland in the Netherlands (Anonymous, 1995b).

*Slurry-derived $N_2O$ emissions from soils*

In studies of Egginton & Smith (1986), Velthof & Oenema (1993), and Velthof *et al.* (1997) $N_2O$ emissions from grassland were much smaller after application of cattle slurry than of CAN or ammonium nitrate (AN). The difference ranges from a factor of 5 up to more than 1000. Slurry application technique may also affect $N_2O$ emissions from grassland, because it affects $NH_3$ emissions and the site of $N_2O$ production in the soil. For mineral soils, we estimate the average slurry-derived $N_2O$ emission (the amount of the total slurry N applied lost as $N_2O$) at $3 \pm 3$ g $N_2O$-N $kg^{-1}$ N for surface-applied slurry and $5 \pm 5$ g $N_2O$-N $kg^{-1}$ N for slurry applied with a technique that minimizes $NH_3$ emissions. Emission factors for peat soils were set at twice those of mineral soils (Table 2).

*Grazing-derived $N_2O$ emission from soils*

Emissions of $N_2O$ were much larger from N fertilized and grazed grasslands than from N fertilized and mown grasslands (Velthof *et al.*, 1996a). On average, $25 \pm 15$ g $N_2O$-N $kg^{-1}$ N excreted as urine and dung during grazing was lost as $N_2O$ on the mineral soils and $60 \pm 46$ g $N_2O$-N $kg^{-1}$ N on peat soils.

*Biological nitrogen fixation*

Studies of Ruz-Jerez *et al.* (1994) in New Zealand and Garret *et al.* (1992) in Northern Ireland suggest smaller denitrification and $N_2O$ emissions from grass-clover than from N fertilized grassland. We assume the $N_2O$ emission derived from biological nitrogen fixation is $5 \pm 5$ g $N_2O$-N $kg^{-1}$ N fixed.

*Leaching of N*

Considerable amounts of N may be lost from intensively managed grasslands via $NO_3^-$ leaching (Ryden *et al.*, 1984). This leached $NO_3^-$ may be denitrified in the subsoil or may be drained to surface water, where it will be denitrified for the greater

part in the long term. The amount of $N_2O$ that is produced from leached $NO_3^-$ is unknown and is difficult to predict (Mosier, 1994). We adopt the $N_2O$ emission factor for N leaching derived by the Intergovernmental Panel on Climate Change (IPCC), i.e. 25 g $N_2O$-N $kg^{-1}$ leached N, which includes $N_2O$ that is produced from leached $NO_3^-$ and directly leached $N_2O$ (Anonymous, 1997). Due to the many uncertainties we use a large relative standard deviation of 100%: 25 ± 25 g $N_2O$-N $kg^{-1}$ leached N.

*Emission of $N_2O$ in housing and during the storage of slurry*

Measurements in housing indicated that $N_2O$ fluxes from fresh cow urine applied to a stable floor were less than 5 μg N $m^{-2}$ $hr^{-1}$, during two hours after application (Velthof, unpublished results). Therefore, we assume that $N_2O$ emissions directly from the stable floor are negligible. Cattle slurry produced in cubicle houses in autumn and winter is stored below the stable floor or in a separate slurry pit. Emissions of $N_2O$ from cattle slurry stored for up to 6 months were less than 0.05 mg N $kg^{-1}$ slurry N $day^{-1}$ (Oenema & Velthof, 1993 and Oenema *et al.*, 1993). The small $N_2O$ emissions were attributed to the absence of $NO_3^-$ in the anoxic slurry. Total emissions of $N_2O$ in housing and during the slurry storage were set at 0.05 ± 0.05 g $N_2O$-N $kg^{-1}$ slurry N.

*Ammonia volatilization*

Nitrogen lost from dairy farming systems via $NH_3$ volatilization may ultimately return to the atmosphere as $N_2$ and $N_2O$ after nitrification and denitrification. The greater part of the volatilized $NH_3$ will deposit elsewhere. The impact of this $NH_3$ on $N_2O$ emissions has not been addressed explicitly yet (Mosier, 1994). A fraction of the $NH_3$ will be deposited on grassland and than will contribute to the background $N_2O$ emissions from soils. We assume that the amount of $N_2O$ produced from $NH_3$ is lower than the fertilizer-derived $N_2O$ emission: 5 ± 5 g $N_2O$-N $kg^{-1}$ $NH_3$-N.

*Silage production*

With N application rates of less than 400 kg N $ha^{-1}$ $yr^{-1}$, $NO_3^-$ contents of grass cut at silage stage (2500–4000 kg dry matter $ha^{-1}$) are typically less than 8 g $kg^{-1}$ herbage dry matter (Prins, 1983). If total annual N application is higher and/or grass is cut in a younger stage, $NO_3^-$ contents may be in the range of 8 to 15 g N $kg^{-1}$. Ensiled grass is stored under anoxic conditions and under these conditions $NO_3^-$ in the ensiled herbage is reduced. Within a few hours after ensiling, the reduction of $NO_3^-$ starts, with $N_2O$ as one of the possible end products (Spoelstra, 1985). In a study of Ataku (1982), referred by Spoelstra (1985), 0.9–2% of $^{15}N$-$NO_3^-$ added to grass was recovered as $N_2O$. We used as emission factor 15 ± 10 g $N_2O$-N $kg^{-1}$ $NO_3^-$-N in silage.

*Rumen of cattle*

Kaspar and Tiedje (1981) showed in a study under controlled conditions that trace

amounts of $N_2O$ were produced during dissimilatory reduction of nitrite ($NO_2^-$) to $NH_4^+$ in the rumen. We assume that the amounts of $N_2O$ emitted directly by the cattle are small: 0.05 ± 0.05 g $N_2O$-N $kg^{-1}$ consumed N.

*Energy use*

The $N_2O$ emission from gas fired power plants in the Netherlands was estimated at 0.1–0.4 mg $N_2O$-N $MJ^{-1}$ (Spoelstra, 1995). Emission of $N_2O$ from personal cars with engine type diesel was estimated at 6.4 mg N $km^{-1}$ (Baas, personal communication). Assuming that this factor is also applicable for agricultural machinery and assuming a usage of 0.1 liter diesel $km^{-1}$ we calculate an emission factor of 64 mg $N_2O$-N $l^{-1}$ diesel or 1.4 mg $N_2O$-N $MJ^{-1}$, taking 44.5 MJ $l^{-1}$ as average energetic value for diesel (Van Dasselaar & Pothoven, 1994). We used one uniform emission factor for energy use on dairy farms: 1.0 ± 1.0 mg $N_2O$-N $MJ^{-1}$, from both the uses of electricity and diesel. The energy use was set at 6 ± 0.5 MJ $kg^{-1}$ milk produced for dairy farms in the eighties and 5 ± 0.5 MJ $kg^{-1}$ milk produced for dairy farms with improved nutrient management (Van Dasselaar & Pothoven, 1994).

*Indirect source: production of mineral fertilizer*

The catalytic oxidation of $NH_3$ to nitric oxide (NO) is a key step in the production of $NO_3^-$ containing mineral fertilizers (France & Thompson, 1993). During this process $N_2O$ may be formed. Estimates of $N_2O$ emission factors for nitric acid production range from 4–27 g $N_2O$-N per kg $HNO_3$-N produced, with the lower values for modern fertilizer plants (Granli & Bøckman, 1994; De Soete, 1993; France & Thompson, 1993). We use an emission factor of 5 ± 5 g $N_2O$-N $kg^{-1}$ N produced as CAN, that contains $NO_3^-$-N and $NH_4^+$-N in equal amounts.

*Indirect source: purchased feeds*

The production of purchased concentrates and feeds is accompanied by emissions of $N_2O$, directly from the soils and, indirectly at the fertilizer plant if N fertilizer is used. As pointed out by Granli & Bøckman (1994), almost no figures are presented in literature about the $N_2O$ emission expressed as the amount of $N_2O$-N emitted per unit crop production. We calculated the $N_2O$ emission per kg herbage dry matter and herbage N for different soil types and management types, using the data on $N_2O$ emission of Velthof *et al.* (1996a) and those on herbage dry matter and N yields of Vellinga *et al.* (1996). These studies were carried out on the same experimental plots and during the same period. Emissions of $N_2O$ per kg herbage dry matter and herbage N increased by application of N fertilizer and by grazing and were larger for the peat soils than for the mineral soils (Table 3). We estimated $N_2O$ emission during the production of roughage and concentrates using these results.

Purchased roughage mainly consists of grass and maize products. Based on the $N_2O$ emissions from N fertilized and mown grasslands on mineral and peat soils (Table 3), we assume an emission factor of 20 ± 10 g $N_2O$-N $kg^{-1}$ roughage N, which

Table 3. Application rate of CAN fertilizer and $N_2O$ emission in g N $kg^{-1}$ dry matter (DM) produced and in g N $kg^{-1}$ herbage N, for four soils and three types of grassland management. Averages of March 1992 - March 1994. Data of DM and N yields are from Vellinga *et al.* (1996). Data of $N_2O$ emissions are from Velthof *et al.* (1996a).

| Soil | Treatment | Application rate kg N $ha^{-1}$ | $N_2O$ emission* | | | |
|---|---|---|---|---|---|---|
| | | | g N $kg^{-1}$ DM | | g N $kg^{-1}$ herbage N | |
| Sand | Unfertilized-mown | 0 | 0.18 | (0.18) | 7 | (7) |
| | N fertilized-mown | 370 | 0.35 | (0.49) | 12 | (16) |
| | N fertilized-grazed | 370 | 0.73 | (0.87) | 21 | (25) |
| Clay | Unfertilized-mown | 0 | 0.10 | (0.10) | 4 | (4) |
| | N fertilized-mown | 357 | 0.27 | (0.39) | 8 | (11) |
| | N fertilized-grazed | 357 | 0.94 | (1.07) | 27 | (31) |
| Peat I | Unfertilized-mown | 0 | 0.33 | (0.33) | 11 | (11) |
| | N fertilized-mown | 365 | 0.73 | (0.88) | 20 | (24) |
| | N fertilized-grazed | 365 | 1.22 | (1.37) | 33 | (37) |
| Peat II | Unfertilized-mown | 0 | 0.97 | (0.97) | 32 | (32) |
| | N fertilized-mown | 242 | 1.53 | (1.63) | 44 | (47) |
| | N fertilized-grazed | 242 | 3.22 | (3.31) | 89 | (91) |

* in parentheses $N_2O$ emissions including the $N_2O$ emissions during the production of the used N fertilizer, assuming that 5 g $N_2O$-N is lost per kg produced CAN-N

includes both the $N_2O$ emission directly from the soil and the $N_2O$ emission during the production of the required N fertilizer. Purchased concentrates may consist of many products, partly grown in the Netherlands and partly grown abroad, e.g. in the tropics. On average, the N input is smaller to crops from which concentrates are produced than to crops from which roughage is produced. We use an emission factor of $10 \pm 5$ g $N_2O$-N $kg^{-1}$ concentrate N.

*Calculations*

Calculations of $N_2O$ emissions were carried out using a simple spreadsheet model. Monte Carlo simulations were used to calculate the effects of variations and uncertainties in N flows and pools (Table 1) and $N_2O$ emissions factors (Table 2) on the total $N_2O$ emissions at farm level. Monte Carlo simulations (2000 iterations) were carried out with the computer program @RISK (Anonymous, 1995c). It was assumed that all N sources on the dairy farming systems were normally distributed. Because of the high variability of $N_2O$ emissions in time and space it was assumed that $N_2O$ emissions were lognormally distributed.

## Results and discussion

### *Emissions of $N_2O$ from the three dairy farming systems*

There were large differences in direct and indirect $N_2O$ emissions among the three farming systems on sandy soils (Table 4). Direct $N_2O$ emissions ranged from 15.4 ± 9.4 kg $N_2O$-N $ha^{-1}$ $yr^{-1}$ on Farm '80 to 5.3 ± 2.6 kg $N_2O$-N $ha^{-1}$ $yr^{-1}$ on De Marke. The large standard deviations show that there is a considerable uncertainty in the estimated total mean $N_2O$ emissions. The direct $N_2O$ emissions accounted for 3.2, 4.6 and 3.7% of the N surplus on Farm '80, Kloosterboer and De Marke, respectively. Clearly, emissions of $N_2O$ are only a minor N loss from dairy farming systems.

On all farms, N leaching accounted for about 25% of the total $N_2O$ emission (i.e., direct + indirect emission). The relative standard deviation of the estimated leaching-derived $N_2O$ emission was very large (170–180%), due to the large uncertainties in both the emission factors for leaching-derived $N_2O$ emission and the amounts of leached N. Grazing was also an important $N_2O$ source, accounting for 25% of the total $N_2O$ emission. A third major source of $N_2O$ was N fertilizer use. The sum of the direct and indirect $N_2O$ emission from N fertilizer amounted to 13 to 26%. All other $N_2O$ sources were relatively small in comparison to the leaching-, grazing- and N fertilizer-derived $N_2O$ emissions.

Table 4. Direct and indirect emissions of $N_2O$ in kg N $ha^{-1}$ $yr^{-1}$ (mean ± standard deviation) for the three farming systems.

| Source | Farm '80 | Kloosterboer | De Marke |
|---|---|---|---|
| Direct | | | |
| Soil-background | 0.9 ± 0.3 | 0.9 ± 0.3 | 0.9 ± 0.3 |
| Soil-N fertilizer | 3.3 ± 1.7 | 1.6 ± 0.8 | 0.5 ± 0.3 |
| Soil-cattle slurry | 0.6 ± 0.7 | 1.0 ± 1.1 | 1.0 ± 1.0 |
| Grazing | 4.8 ± 3.2 | 4.2 ± 2.8 | 1.3 ± 0.9 |
| Biological N fixation | 0.0 ± 0.0 | 0.0 ± 0.0 | 0.0 ± 0.0 |
| Leaching | 5.2 ± 8.6 | 3.4 ± 6.1 | 1.3 ± 2.2 |
| Housing and slurry storage | 0.0 ± 0.0 | 0.0 ± 0.0 | 0.0 ± 0.0 |
| Ammonia volatilization | 0.3 ± 0.4 | 0.2 ± 0.2 | 0.1 ± 0.1 |
| Silage | 0.1 ± 0.1 | 0.1 ± 0.0 | 0.1 ± 0.1 |
| Rumen | 0.0 ± 0.0 | 0.0 ± 0.0 | 0.0 ± 0.0 |
| Energy use | 0.1 ± 0.1 | 0.1 ± 0.1 | 0.1 ± 0.1 |
| Total direct sources | 15.4 ± 9.4 | 11.5 ± 6.8 | 5.3 ± 2.6 |
| Indirect | | | |
| Purchased N fertilizer | 1.6 ± 1.5 | 0.8 ± 0.7 | 0.3 ± 0.2 |
| Purchased roughage | 0.8 ± 0.4 | 0.0 ± 0.0 | 0.0 ± 0.0 |
| Purchased concentrates | 1.4 ± 0.7 | 0.9 ± 0.4 | 0.8 ± 0.4 |
| Total indirect sources | 3.8 ± 1.7 | 1.7 ± 0.9 | 1.1 ± 0.5 |
| Total direct + indirect sources | 19.2 ± 9.6 | 13.1 ± 6.9 | 6.4 ± 2.6 |

*Effects of management measures*

The differences in $N_2O$ emission among the three farming systems were due to the differences in nutrient management (Table 4). Largest differences were shown for leaching-derived, grazing-derived and N fertilizer-derived $N_2O$ emissions. Nitrate leaching strongly decreased in the order Farm '80 > Kloosterboer > De Marke (Table 1). Nitrate leaching is affected by (a combination of) many management measures, including restricted grazing, proper slurry application, adjusted N application and a lower N content in urine due to changes in the nutrition of the cattle. The results of Table 4 show that measures taken to reduce $NO_3^-$ leaching may also considerably reduce $N_2O$ emission. The improved nutrient management implied smaller amounts of required N fertilizer (Table 1) and, by that, also smaller N fertilizer-derived $N_2O$ emission. Restricted grazing will decrease the amount of N deposited to the soil as urine and dung and increase the amount of slurry N collected in housing. The slurry is subsequently applied to the soil and when applied properly, the emission factor for slurry N is much lower than that for grazing-derived N (Table 2).

On Kloosterboer and De Marke measures were taken to reduce $NH_3$ volatilization from housing, slurry storage units and grassland. Reduction of $NH_3$ volatilization leaves more N in the slurry, so that less N fertilizer has to be purchased. Taking the differences in emission factors between N fertilizer and slurry into account (Table 2) and the difference in effectivity of the N from slurry and N fertilizer, we calculate that a reduction of the $NH_3$ volatilization from dairy farming systems with 1 kg N $ha^{-1}$ $yr^{-1}$ results in a reduction of the $N_2O$ emission with 7.5 g N $ha^{-1}$ $yr^{-1}$ from these systems.

Effects of soil cultivation and changes in land use on $N_2O$ emissions were not accounted for in this study. On De Marke 56% of the total area was used for grassland and 44% for fodder beets and maize (Table 1). Only 29% of the total area was permanent grassland. The other 27% was in rotation with fodder beet and maize. After ploughing out grassland, large amounts of N are mineralized (e.g. Whitehead *et al.*, 1990). These amounts frequently exceed the capacity of the subsequent crop to take up. Consequently, significant quantities may be lost via leaching and via $N_2O$ emission. Quantitative information on the fate on short-term ley's is lacking, however.

*Total $N_2O$ emissions from dairy farming systems in the Netherlands*

Total direct $N_2O$ emissions from dairy farming systems in the Netherlands were calculated for 1994 using the $N_2O$-emission factors derived in the present study and data of dairy farming systems in the Netherlands (see footnote of Table 5). Total $N_2O$ emissions from dairy farming systems in the Netherlands were 13.7 ± 5.1 Gg $N_2O$-N $yr^{-1}$, from which 8.0 ± 4.3 Gg $N_2O$-N $yr^{-1}$ was derived from mineral soils and 5.7 ± 3.5 Gg $N_2O$-N $yr^{-1}$ from peat soils (Table 5). This indicates the importance of dairy farming systems on peat soils as $N_2O$ source in the Netherlands. Largest $N_2O$ sources are grazing-derived, fertilizer-derived and, in case of mineral soils, leaching-derived $N_2O$ emissions. Background $N_2O$ emissions from grasslands on peat soils

Table 5. Total direct emissions of $N_2O$ from dairy farming systems in the Netherlands in 1994, in Gg $N_2O$-N $yr^{-1}$. For each source the mean ± standard deviation is presented, based on Monte Carlo simulation (2000 iterations) using the emission factors ± standard deviations presented in Table 2 and data derived from literature (see footnotes 1–8). Standard deviations for data derived from literature were based on best guess and are given as footnote 9.

| Source | Minerals soils | Peat soils | Total |
|---|---|---|---|
| Soil-background[1] | 0.8 ± 0.5 | 1.3 ± 0.3 | 2.1 ± 1.8 |
| Soil-N fertilizer[2] | 2.4 ± 1.2 | 2.0 ± 0.8 | 4.4 ± 1.5 |
| Soil-cattle slurry[3] | 0.7 ± 0.7 | 0.4 ± 1.1 | 1.0 ± 0.8 |
| Grazing[3] | 2.2 ± 2.9 | 1.8 ± 2.8 | 3.9 ± 4.1 |
| Biological N fixation[4] | 0.0 ± 0.0 | 0.0 ± 0.0 | 0.0 ± 0.0 |
| Leaching[5] | 1.6 ± 2.7 | 0.1 ± 0.7 | 1.7 ± 2.8 |
| Housing and slurry storage[3] | 0.0 ± 0.0 | 0.0 ± 0.0 | 0.0 ± 0.0 |
| Ammonia volatilization[6] | 0.3 ± 0.4 | 0.1 ± 0.1 | 0.4 ± 0.4 |
| Silage[7] | 0.1 ± 0.1 | 0.0 ± 0.0 | 0.1 ± 0.0 |
| Rumen | 0.0 ± 0.0 | 0.0 ± 0.0 | 0.0 ± 0.0 |
| Energy use[8] | 0.0 ± 0.0 | 0.0 ± 0.0 | 0.0 ± 0.0 |
| Total | 8.0 ± 4.3 | 5.7 ± 3.5 | 13.7 ± 5.1 |

[1] Anonymous (1995a) and Steur *et al.* (1985). Assumes that all maize and fodder crops are grown on mineral soils.
[2] Anonymous (1995b). Mean N fertilizer application rate was 280 kg N $ha^{-1}$, for all soils.
[3] Anonymous (1995a). Assuming a total N content of 5 g N $kg^{-1}$ slurry (= dung + urine).
[4] Assuming 4 kg fixed N $ha^{-1}$ on mineral soils and 0 kg N $ha^{-1}$ on peat soils.
[5] Assuming 75 kg N $ha^{-1}$ $yr^{-1}$ leached $NO_3^-$ on mineral soils and 10 kg N $ha^{-1}$ $yr^{-1}$ on peat soils
[6] Meeuwissen (1993)
[7] Anonymous (1995a). Assuming a dry matter content of grass silage of 33% and of maize silage of 60% and a $NO_3^-$ content in grass silage of 2 g N $kg^{-1}$ and in maize silage of 5 in g N $kg^{-1}$ dry matter.
[8] Anonymous (1995a).
[9] Relative standard deviations: 5% for grassland area, 10% for N fertilizer, 25% for cattle slurry N, 25% for N excreted during grazing, 15% for fixed N, 25% for slurry N in housing and storage unit, 100% for leached N, 15% for silage N, 50% for volatilized N and 15% for energy use.

are also a considerable source of $N_2O$, accounting for about 10% of the total $N_2O$ emissions from dairy farming systems in the Netherlands.

Kroeze (1994) estimated the total $N_2O$ emission towards the atmosphere in the Netherlands from agriculture, energy generation, industry and traffic at 37.1 Gg N $yr^{-1}$. Total $N_2O$ emission from agriculture was estimated at 16.9 Gg N $yr^{-1}$. Our data thus demonstrate that dairy farming systems are a major source of $N_2O$ in the Netherlands. About 35% of the total amount of $N_2O$ emitted into the atmosphere originates from dairy farming systems. Clearly, a significant reduction in $N_2O$ emission from dairy farming systems due to improved nutrient management may not only reduce $N_2O$ emission from these systems but will also contribute to a significant reduction of the total $N_2O$ emissions in the Netherlands.

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