A Whole-Farm Strategy to Reduce Environmental Impacts of Nitrogen

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Dutch regulations for ammonia emission require farmers to inject slurry into the soil (shallow) or to apply it in narrow bands at the surface. For one commercial dairy farm in the Netherlands it was hypothesized that its alternative farming strategy, including low-protein feeding and surface spreading, could be an equally effective tool for ammonia emission abatement. The overall objective of the research was to investigate how management at this farm is related to nitrogen (N) losses to the environment, including groundwater and surface water. Gaseous emission of ammonia and greenhouse gasses from the naturally ventilated stables were 8.1 and 3.1 kg yr^{-1} AU^{-1} on average using the internal tracer (SF_6)-ratio method. Measurements on volatilization of ammonia from slurry application to the field using an integrated horizontal flux method and the micrometeorological mass balance method yielded relatively low values of ammonia emissions per ha (3.5–10.9 kg NH_3–N ha^{-1}). The mean nitrate concentration in the upper ground water was 6.7 mg L^{-1} for 2004 and 3.0 mg L^{-1} for 2005, and the half-year summer means of N in surface water were 2.3 mg N L^{-1} and 3.4 mg N L^{-1} for 2004 and 2005, respectively. Using a nutrient budget model for this farm, partly based on these findings, it was found that the calculated ammonia loss per ton milk (range 5.3–7.5 kg N Mg^{-1}) is comparable with the estimated ammonia loss of a conventional farm that applies animal slurry using prescribed technologies.

Volatilization of ammonia from liquid animal manure significantly contributes to soil acidification (Van Breemen et al., 1982) and the eutrophication of ecosystems (Galloway et al., 2002). As a response to this problem, environmental policies have been developed in many European countries to minimize ammonia emission from manure. In The Netherlands, regulations were formulated in the 1990s for the housing of animals (mainly pigs and poultry), the coverage of stored manure, the period during which application of animal manure is allowed, and methods by which slurry must be applied to the land (Neeteson et al., 2001). The contribution of Dutch agriculture to the emission of ammonia has dropped from 237 Gg in 1990 to 117 Gg in 2003 (MNP, 2005), suggesting that enforced regulations did indeed have an impact. Enforced strategies to combat nutrient losses from agriculture must find a balance between environmental objectives, farm management, and regulatory requirements for the government. Farming conditions differ among soils, whereas regulations have to consider possibilities for inspection, associated administrative burdens, and a certain need for harmonization (Schröder et al., 2004). According to Driessen (2003), this balance has strongly shifted toward the governmental side. Environmental policies were developed from a central governmental perspective, emphasizing input by technology and enforcement aspects of regulations.

Low-emission techniques have formally been prescribed to regulate the application of liquid manure on land surfaces. Liquid manure, in particular slurry, is by far the most common type of manure in The Netherlands. Dutch regulations for ammonia emission require farmers to inject slurry (depth about 5 cm) into the soil or to apply it in narrow bands at the surface. Shallow injection slits at the soil surface and narrow bands of slurry at the surface are clearly visible, allowing inspectors to conclude that regulations have been followed. These techniques are supported by scientific data showing emission reductions under experimental conditions (e.g., Mulder and Huijsmans, 1994). Adverse impacts on soil and

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Abbreviations: EU, European Union; GW, groundwater observation well.

water quality or biodiversity have hardly been investigated, although desktop studies (e.g., Korevaar et al., 1991) have indicated that negative impacts on, for example, meadow-bird populations and soil structure could be severe in particular years.

Since the introduction of low-emission techniques, several farmers who categorically refused to follow governmental regulations when applying slurry to their land have been fined and taken to court. Instead, they used surface spreading of animal slurry to avoid damage to soil structure and biological activity. They also claimed that their low-N manure caused low ammonia emission.

The Ministry of Agriculture is reluctant to allow exemptions to the established regulations. Exemptions can be granted only for research purposes with limits on the duration and the overall set-up of the project. Moreover, the research should fulfill the requirements that (i) it will be able to demonstrate in quantitative terms the favorable environmental effects of alternative procedures, (ii) it will not negatively affect soil quality, and (iii) it will allow enforceable alternative regulations.

The Spruit dairy farm, located in the central peat district in the Netherlands, has received considerable public and media attention because of the discrepancy between its apparent successful nature-oriented management and its public violation of official regulations. Unfortunately, management results were not scientifically documented, which made discussions inconclusive and ideological. A team of scientists offered to perform research on this farm to document the environmental quality without any superimposed experimental design. Following this suggestion, the Ministry of Agriculture, Nature and Food Quality granted this farm a temporary exemption in 2004 and 2005 to deviate from the regulations. In the ensuing research project, it was hypothesized that its alternative farm strategy, including a low-protein feeding strategy, the use of bedding material, the application of slurry by surface spreading during rainy weather, or adding water after slurry application, could be equally effective tools for ammonia emission abatement as compared with the technology-oriented, low-emission techniques required by law. Because of extended legislation on other environmental issues, such as ground and surface water quality and emission of nitrous oxide, the research was broadened to cover these topics as well. Nutrient dynamics at the farm level should be studied by a comprehensive systems analysis including all fluxes and should not be restricted to air or water quality alone (Goss et al., 1995; Sonneveld and Bouma, 2003).

Therefore, the overall objective of this research was to investigate how management at this farm is related to nitrogen (N) losses to the environment. This paper describes the results of the research at this farm in the 2-yr period of the granted exemption. In addition, possible implications and pitfalls for future regulations and for on-farm research in general are discussed.

Materials and Methods

Farm Characteristics

The Spruit dairy farm covered 37.1 ha of grassland on average over the years 2003 to 2005 with 79 milking cows. The farm also holds almost 80 young animals and some bulls and sheep. More than 517,000 kg milk on average has been produced annually. The urea content of this milk, an indicator for the nitrogen-use efficiency for animals, was around 0.16 g L^{-1} milk on average, compared with the national reported mean of 0.25 g L^{-1} for 2005 (MNP, 2006). The dairy farm is mainly located on drained peat soils that have been artificially raised in the past with "toemaak," a combination of sand, sludge, and city waste. The subsoil largely consists of nondecomposed remains of reed and trees. Mineralization of available N stocks in the soil contributes to uptake of N by crops and emission to the environment (Van Beek et al., 2004b). All fields have subsurface drains to enhance subsurface infiltration of water in dry periods from the surrounding ditches.

Nitrogen concentrations in the infiltrating surface water are relatively low, and the infiltrating N has no substantial influence on N uptake by the crops. However, the wetting of the top soil by infiltrating water enhances water and nutrient uptake by the crop through better water and nutrient availability for the grass roots under wetter conditions. The subsurface drains enhance drainage in periods with a precipitation excess and subirrigation with surface water during dry periods. For the Spruit farm, the subsurface drains (drain spacing 12–20 m at about 60 cm depth) and the controlled surface water level of about 40 to 50 cm below the soil surface are supposed to have mainly an effect on the water balance of the soil and a much lesser effect on denitrification or N and P losses to the surface water.

The major part of the manure is collected in a tiestall and in a cubicle housing system. The cubicle housing system holds 67 positions for animals. The animals can freely roam in this stable. Slurry—a mixture of feces and urine—is collected from this stable. The bedded tiestall holds mostly dry cows and young animals. Feces and urine are collected twice a day from the stall and stored as solid manure in an outside open concrete structure. Other materials, such as horse manure, sludge, liquid manure, and carbon-rich additives, are added to immobilize ammonium nitrogen (NH_4–N). Disturbances of the manure heap are common because of regular turning and mixing. Liquid manure draining from the bottom of the manure heap is added to the slurry storage. Application of slurry to grassland is done by surface spreading. Often this precedes anticipated rainfall or, in the absence of rainfall, is followed by applying pumped-up surface water. In spring and summer, sludge from the ditches is pumped up and applied to the fields.

Grazing takes place only during daytime and occurs on 160 d over the year. Direct inputs of urine and manure from grazing cattle into water-filled ditches are prevented by wire fencing around the fields.

Atmospheric Emission

Standards

Only national emission standards are available for ammonia, imposed by European Union (EU) Directive 2001/81/EC (EC, 2001), which specifies a national target of 128 Gg of ammonia in 2010. Instead of emission standards at the farm level, only national substandards for the different compartments and processes exist. Calculated as the percentage of ammonium N that volatilizes to the atmosphere (Van der Hoek, 2002), ammonia emissions after slurry application are supposed to vary from 12%

Table 1. Conditions during the measurements performed with the integrated horizontal flux method.

Experiment	Height of measurements	Start of measurements	Hours after application	Wind speed (at 2 m)	Temperature	Rainfall
	m			m s^{-1}	°C	mm
1†	1, 2, 4, 8, 12	17 Mar. 2004; 16:45	–	–	–	–
2	1, 2, 4, 8, 12	18 June 2004; 08:15	0–5	3.4	18.8	0.0
			5–12	4.4	17.4	0.0
			accumulated	–	–	0.0
3	1, 2, 4, 8, 12	12 Aug. 2004; 15:00	0–4	2.4	20.3	0.2
			4–7	1.7	17.3	0.0
			7–18	3.4	16.0	19.0
			18–29	5.3	17.7	8.8
			accumulated	–	–	28.0
4	0.4, 1, 2, 4, 8, 12	3 Apr. 2005; 21:15	0–13	2.0	10.6‡	0.0‡
			13–19	5.3	17.4‡	0.0‡
			19–36	2.9	10.0‡	5.0‡
			36–47	4.7	10.2‡	0.0‡
			accumulated	–	–	

† In Experiment 1 and in phases 3 and 4 in Experiment 2, wind was from the wrong direction, so no data are available.

‡ Data from a weather station in Wageningen.

(shallow injection) and 20 to 29% (trailing shoe) to 68% (surface spreading). For the existing animal housing, calculated standards are 4.3 kg NH_3 yr^{-1} per animal unit (AU) for the tiestall and 9.5 kg NH_3 yr^{-1} AU^{-1} for the cubicle housing system, respectively. In the future, attention should be paid to greenhouse gases, which are not considered in current regulations.

Methods

Emission measurements were performed for both types of stables, the outdoor storage facility, and during and after slurry application. Measurements on gaseous emission of ammonia and greenhouse gasses from the naturally ventilated stables were done using the internal tracer (SF_6)-ratio method (Mosquera et al., 2002, 2005c). This was done for two periods in 2004 in the cubicle housing system and during two periods in 2005 in the cubicle housing system and the tiestall. For these measurements, a sample line with at least one sample point per 10 m^2 was placed close to the ridge of the barn to obtain a representative sample of the air leaving the animal house. This air sample was sent to a NO_x analyzer (coupled with NH_3 to NO converters) to be analyzed for NH_3 concentrations or collected over a 24-hr period in 1-L evacuated canisters, which were later analyzed for N_2O and CH_4 concentrations in a gas chromatograph.

Gaseous emissions from the open manure storage facility outside the barn were measured using the gradient method. Measurements of ammonia concentrations were done using the open-path tunable diode laser (Gasfinder 2.0; Boreal Laser Inc., Spruce Grove, Alberta, Canada) and a photoacoustic monitor (Innova 1312; ENMO Services, The Netherlands). These were performed for 1 d in 2004 before and after mixing the manure and during 2 d in 2005, without any treatment. Further details of the monitoring methods used are given in Mosquera et al. (2005a). Manure samples were also taken for chemical analysis at three different dates and analyzed for total-N, mineral-N, pH, dry matter, and ash content.

Two different methods were used to assess the volatilization of ammonia from slurry application to the field. The integrated horizontal flux method was used to measure the emission from the whole field (4 ha), and the micrometeorological mass balance method was used to measure the emission from small plots of about 0.2 ha (Huijsmans, 2003). The integrated horizontal flux method is based on the integration of the product of ammonia concentration and wind speed over a pole height of 12 m perpendicular to the wind direction. The ammonia concentrations were determined in passive samplers located on the pole (Mosquera et al., 2002, 2005c). The total ammonia emission was determined over a period of 36 h in different phases: a first phase during spreading, a second phase until the morning of the next day, and a third phase until the end of the next day. When application of the slurry took place, the amount, composition, time span, and fertilized area were determined as were meteorological conditions (wind direction, wind speed, temperature, relative humidity, and precipitation).

Measurements of ammonia emission using the integrated horizontal flux method were determined three times in 2004 and once in 2005. The conditions under which these experiments took place are given in Table 1.

The first experiment in 2004 failed because of a change in wind direction. The same also held for Phases 3 and 4 of the second experiment in 2004. The timing of slurry application was different for these experiments. Experiment 2 started early in the morning, Experiment 3 started in the afternoon, and Experiment 4 started late in the evening. Associated differences in climate conditions are likely to influence ammonia emission (Huijsmans, 2003). Details on the application of the slurry in the experiments are given in Table 2.

Table 2. Details of the slurry application during the experiments.

Experiment	Area	Application	Method of application	Manure composition: N-total	N-mineral	Dry matter
	ha	m^3 ha^{-1}		g kg^{-1}		
1	–	–	–	–	–	–
2	3.85	15.6	water†, sludge‡	3.9	1.4	108
3	3.85	15.6	water†, rain§	3.4	1.3	88
4	3.70	11.4	water†	3.7	1.5	95

† Water was applied to the field directly after slurry application.

‡ Sludge originating from ditches surrounding the Spruit farm was applied to the field approximately 2 h after application. About 75% of the total area was covered with sludge.

§ Rainfall within 12 h after application.

Two experiments were performed with the micrometeorological mass balance method. In 2004, the slurry from the Spruit farm was applied on two grassland fields using broadcast spreading. In 2005, the slurry from the Spruit farm was applied using broadcast spreading on one field, whereas the slurry from the nearby *Zegveld* experimental farm was applied using band-application with a trailing shoe machine.

Ground- and Surface Water

Standards

Threshold values for N and phosphorus (P) for surface water are defined in the context of the EU Water Framework Directive (EC, 2000). For Dutch conditions, preliminary values are used expressing the maximum allowable risks as defined by the National Environmental Plan NMP4 (NMP4, 2001) for stagnant waters as 2.2 mg L^{-1} for total N and 0.15 mg L^{-1} for total P. These values have been formulated in terms of summer half-year (1 April–30 September) means and are applied to the upper 50 cm of the surface water in this study. A threshold value for nitrate in the upper ground water is defined in the EU Nitrates Directive (EC, 1991) as a maximum nitrate concentration of 50 mg L^{-1}. The nutrient concentrations in the ground and surface water are partly presented as the compound form (NO_3^-, NH_4^+) because the norms are often given in this form.

Methods

Regular measurements were made on N and P concentrations in the upper ground water and surface water. Samples of the upper 50 cm of ground water were taken from installed ground water observation wells (GWs) at nine locations, distributed over four different grassland fields. Samples from three pairs of ground water observation wells were mixed (sample codes GW4, GW5, and GW6), yielding a total of six measurements per sampling date. Frequency of sampling was every month or after 50 mm of precipitation excess. Water samples were collected by pumping out the water of the tubes of the ground water observation well to a depth of 50 cm under the actual ground water level. The tubes filled again with ("fresh") ground water, and a sample was taken from this water in the top 50 cm of the tube. Samples were cooled, stored, and analyzed for NO_3, NH_4, total-N, PO_4, and total-P within 24 h after sampling.

Surface water was sampled from perforated plastic tubes at 10 locations in the vicinity of the ground water observation wells. Measurements were done at the same time as the ground water samples were taken, and samples from four locations were mixed (sample code S6) and analyzed as one sample. The procedure of taking water samples is similar to the ground water sampling. Samples were again cooled, stored, and analyzed for NO_3, NH_4, total-N, PO_4, and total-P.

The elements NO_3, NH_4, total-N, and PO_4 in all water samples were analyzed using a segmented flow analyzer in a $CaCl_2$ system (Anonymous, 2004). Nitrate was reduced to NO_2, and in an acid environment the NO_2 concentration was measured using spectrophotometry at a wavelength of 540 nm. Ammonium was measured after a Berthelot reaction and using spectrophotometry at a wavelength of 660 nm. Total-N was measured after the destruction of organic N using acids at pH 4 and UV destruction; then the sample was dialyzed and converted to NO_3, which was measured following the procedure described previously. Phosphate was measured in an acid environment after forming a complex with molybdate ions using spectrophotometry at a wavelength of 880 nm. Total P was analyzed using ICP–AES analysis, based on the measurement of the emitted light of the P element after ionization in an argon plasma.

The mixing of water samples was based on a reduction in costs for analysis. Measured values for the mixed samples were given a higher weight in calculating the overall arithmetic mean for N and P. The calculation of an arithmetic mean was used because this method is prescribed by the Dutch Committee on Integral Water Management (CIW, 2000). Individual data points are given attention in the Results section, and the way of calculating a mean is discussed.

Whole-Farm Nitrogen Cycle

Methods

A N and P budget model was constructed for this farm on the basis of Schröder (2000) and Schröder et al. (2003). It is based on the assumption that the gap between input and output (i.e., the surplus at farm level) reflects losses to the environment. The model intends to allocate all ingoing and outgoing nutrients to fluxes within the farm and simulate the conversions from feed-N and fertilizers via manure and soil to crop-N. The starting point is the feed conversion.

Kebreab et al. (2001) reported that the conversion of feed-N into economic products (milk and meat) is more efficient at lower protein contents in the diet. Their relationships were used to estimate the required feed-N intake from the N exported in recorded economic products and the protein content, the latter being estimated from the relative shares of registered diet constituents. The difference between the feed-N intake and the N exported in milk and meat equals the N excreted as manure, either indoors or outdoors.

Because the farm import of N for cattle feed is known, the remainder must be the result of feed-N that is produced on the farm. In the case of the Spruit dairy farm, this N originates from mineralized peat and toemaak, atmospheric deposition, biologically fixed N, and manure-N.

The crude protein content of the ration was estimated to be 134 g kg^{-1}, which is based on chemical analyses of the stored roughage (N content 2.4%), the assumption that the N content in field-grass approximates 1.45 times the value of the stored roughage and assuming a daily intake of 22 kg DM AU^{-1} from imported feed, home-grown grass silage, and grazed grass together. Although milking cows use N in feed with a crude protein content of 134 g kg^{-1} at an efficiency of approximately 29% (Kebreab et al., 2001), the overall farm efficiency was estimated to be 22%, taking into account that the feed-N conversion of the associated young dairy, bulls, and sheep (producing meat instead of milk) was about 12%. Registered feed imports at farm level amounted to 168 kg N ha^{-1} on average. The percentage of clover in the dry matter production was estimated at 10%, correspond-

Table 3. Conditions and results of measurements at the stables.

	Period 1 (May 2004)	Period 2 (Sept. 2004)	Period 1 (Feb. 2005)		Period 2 (Apr. 2005)	
	CHS†	CHS	CHS	Tiestall	CHS	Tiestall
Conditions						
No. of animals	71	71	73	45‡	72	45‡
Milk urea (g L^{-1})	0.130	0.220	0.130	–	0.190	–
Urine composition						
N_{total} (g L^{-1})	4.4	6.9	7.8	8.6		
Urea-N (g L^{-1})	2.1	4.1	6.2	6.7		
pH	8.5	8.6	8.5	8.4		
Manure composition						
N_{total} (g kg^{-1})	–	3.5	3.9	6.9	3.7	5.7
NH_4 (g kg^{-1})	–	1.4	1.6	1.4	1.5	0.8
pH	–	6.6	6.8	8.7	6.7	7.5
Outdoor temp. (°C)	14	15	4	4	10	10
Indoor temp (°C)	16	17	8	11	13	15
Airflow (m^3 h^{-1} AU^{-1}§)	1096	792	717	558	568	653
Gaseous emissions						
NH_3 (kg yr^{-1} AU^{-1})	9.5	7.3	7.7	2.9‡	9.9	3.2‡
CH_4 (kg d^{-1} AU^{-1})	0.52	0.12	0.73	0.98	0.78	0.76
N_2O (g d^{-1} AU^{-1})	1.40	2.85	0.97	1.11	1.04	0.86

† CHS, cubicle housing system.

‡ On the basis of 215 d in the stable and 150 d of no use.

§ AU, animal unit.

ing with a net contribution of 32 kg N ha^{-1} to the produced feed N at farm level. This implies that the amount of nonleguminous feed-N supplied via soil N must have been around 289 kg ha^{-1}.

As a first step (Scenario I), the model is used to calculate how much background N mineralization was required to generate the required amount feed-N in home-grown grass. Measured N losses from manure were used as input in this scenario, together with a surmised mineral N uptake efficiency (0.80 kg N kg^{-1} N applied), which was derived from literature data referring to comparable soils and a few small in situ trials on the recovery of mineral fertilizer N. The outcome of scenario I ("background N mineralization") was contrasted with the net N mineralization, as deduced from the crop N harvested from small unfertilized ungrazed plots corrected for the N derived from other sources, including the residual manure N applied in previous years (Schröder et al., 2005), which were installed in a number of fields. In a second series of model runs, this observed background mineralization was used as input to explore the extent to which the measured N loss from stables and manure storages (Scenario II), the measured N loss from surface spreading (Scenario III), or the N uptake efficiency of crops (Scenario IV) could have been under- or overestimated. All runs yielded numbers of estimated ammonia-N loss per Mg milk, the indicator that initially triggered this project. These numbers were compared with the estimated ammonia loss of a conventional farm with conventional concentrations of crude protein (173 g kg^{-1}) in the roughage, no use of bedding material, and slurry application by trailing shoes, one of the legally permitted "low-emission" techniques (Scenario V). We surmised that the use of heavy machinery like these reduced the N uptake efficiency of crops from 80 to 70%.

Results

Atmospheric Emissions

Results for the ammonia, methane, and nitrous oxide emissions from the stables are given in Table 3. On average, the weighed emissions of ammonia from the cubicle housing system and the tiestall were 8.1 and 3.1 kg yr^{-1} AU^{-1}, respectively. Both values are lower than the official standards of 9.5 (yearly emission based on a grazing season of 175 d) and 4.3 kg yr^{-1} AU^{-1} (yearly emission based on an occupancy of 215 d yr^{-1}). The relatively low ammonia emission from the cubicle housing system in September 2004 can be explained by the fact that most of the herd grazed by day. About half of the herd stayed inside the stables in May 2004, resulting in higher emission values.

Methane emissions were between 120 and 780 g d^{-1} AU^{-1} for the cubicle housing system and between 760 and 980 g d^{-1} AU^{-1} for the tiestall. The highest values were recorded in February and April 2005 when most animals were kept inside. Nitrous oxide emissions were low, in agreement with the values found in other studies (Husted, 1994; Mosquera et al., 2005b; Sommer et al., 1998).

Three types of solid manure could be distinguished in the open manure storage facility: fresh manure, intermediate products, and (assumed) end products. Compositions of these types of manure for different dates are presented in Table 4. These values must be considered preliminary because it was difficult to take representative samples. We observed higher total-N values for all types of product at later sampling dates (Table 4). Dry matter contents varied for the different dates from 200 to 225 g kg^{-1} (fresh product), 251 to 466 g kg^{-1} (intermediate product), and 243 to 438 g kg^{-1} (assumed end product). The end products did not show consistently higher dry matter contents compared with the intermediate products. It is likely that the observed spatial heterogeneity of the manure heap results in different dry matter contents for similar types of product. From the data, we deduced that ammonium N to total-N ratios showed a decreasing trend from fresh

Table 4. Composition of solid manure from the manure heap at the storage facility.

Sampled heap	Sampling date	pH	Total-N	NH_4–N	Dry matter	Mineral content
			g kg^{-1}			
Fresh product	21 Feb. 2005	8.7	6.5	1.4	225	–
	5 Apr. 2005	7.5	5.7	0.8	200	47.1
	13 July 2005	8.2	7.0	1.2	202	45.6
Intermediate product	21 Feb. 2005	8.5	5.7	0.7	251	–
	5 Apr. 2005	8.4	6.8	0.5	279	65.7
	13 July 2005	8.2	14.4	1.6	466	106.7
Assumed end product	21 Feb. 2005	8.4	5.2	0.5	243	–
	5 Apr. 2005	8.5	6.9	0.4	277	81.7
	13 July 2005	8.7	11.8	1.1	438	127.7

Table 5. Ammonia emissions after slurry application measured with the integrated horizontal flux method.

Experiment	Emission		
	kg NH_3–N ha^{-1}	% applied NH_4–N	% applied total-N
2 (two phases)	3.7	18.5	6.5
3 (all phases)	3.5	17.8	6.9
4 (all phases)	10.9	68.4	27.3

products to assumed end products, which is likely to be the result of the emission of ammonia.

The ammonia emission from the manure heap outside was calculated to be 8.9 kg yr^{-1} AU^{-1}. Thus, it is comparable with the loss of ammonia from the cubicle housing system, although the measurements were limited in duration and spatial coverage of the heap.

Table 5 presents the outcome of the field measurements using the flux window method at the Spruit farm. The relatively low levels of NH_4–N (Table 2) resulted in relatively low values of ammonia emissions per hectare. Experiments 2 and 3 also showed relatively low values of ammonia emissions (~18% of the applied NH_4–N, 3.5–3.7 kg NH_3–N ha^{-1}), whereas the fourth experiment showed a significantly higher value (68%). Experiment 1 did not yield satisfactory results because of a change in wind direction. In contrast with other studies (e.g., Huijsmans et al., 2004) that measured over a period of 96 h, the measuring periods for these experiments were relatively short, suggesting that measurements represent lower limits of ammonia emissions. Actual values are not much higher; other studies indicated that ammonia emission does not increase much after 40 h after application. Slurry application for Experiment 4 was performed later than desired by the farmer because wind direction was not suitable for earlier measurements. Expected rainfall did not occur after Experiment 4.

Experiments with the micrometeorological mass balance method showed that NH_3 emission was not significantly different between the two manure application techniques (Table 6). Ammonia emission ranged from 26 to 36% of the applied NH_4–N (2.9–7.0 kg NH_3–N ha^{-1}).

Ground and Surface Water Quality

Measurements of ground and surface water quality were made on 19 dates. Results for ground water concentrations of nitrate, ammonium, and total N are given in Fig. 1, 2, and 3, respectively. Presented values for locations GW4, GW5, and GW6 are for samples combined over two locations. The mean nitrate concentration in the upper ground water was 6.7 mg L^{-1} for 2004 and 3.0 mg L^{-1} for 2005, indicating that the EU nitrate standard of 50 mg L^{-1} was not exceeded (Fig. 1).

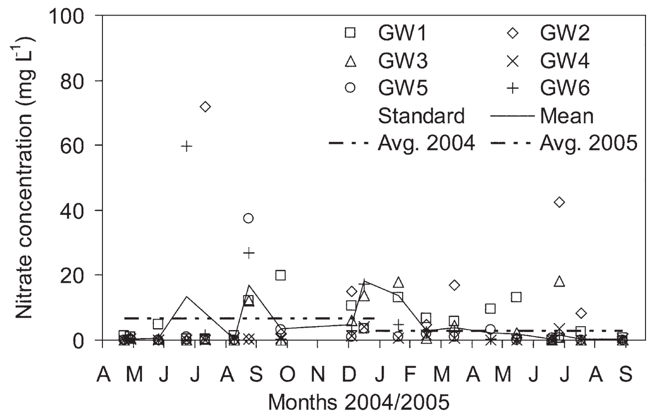

Fig. 1. Nitrate concentrations in the groundwater measured at different locations on the Spruit farm. Average values and the European Union nitrate standard are indicated. GW, groundwater observation wells.

The mean ammonium concentration for 2004 and 2005 was around 4.6 mg L^{-1}. Except for some outlier values in summer, measured values were mostly within the range of 0 to 10 mg L^{-1} (Fig. 2). The mean concentration of total N in the ground water for 2004 and 2005 was around 10 mg N L^{-1}. For both years, an increase in concentration was found for the summer period (Fig. 3).

Results of total N and total P in surface water are given in Fig. 4 and 5, respectively. The half-year summer mean in 2004 was 2.3 mg N L^{-1} and thus slightly exceeded the standard of 2.2 mg N L^{-1}. The half-year summer mean for 2005 was 3.4 mg N L^{-1}. Measured values in 2005 exhibited a higher variation for which no particular cause could be identified. Mean P concentration in the summer of 2004 was 0.14 mg L^{-1}, just below the standard of 0.15 mg L^{-1}. Mean P concentration in 2005 was much higher, with a value of 0.35 mg L^{-1}. This contrasts the trend of the phosphate surplus at farm level, which decreased from 23 kg ha^{-1} in 2003 to 4 kg ha^{-1} in 2005.

Whole-Farm Nitrogen Cycle

Collected data on farm N and phosphate (P_2O_5) flows, together with an estimation of mineralization and deposition, suggested that N surplus levels were 297 kg N ha^{-1}, 269 kg N ha^{-1}, and 254 kg N ha^{-1} in 2003, 2004, and 2005, respectively. These years also showed a surplus for P_2O_5, although the amount of P_2O_5 decreased from 23 kg ha^{-1} in 2003 to approximately 4 kg ha^{-1} in 2005 (Table 7).

Nitrogen flows at the Spruit dairy farm following Scenario I are presented as a flow diagram in Fig. 6. Calculated losses of N direct-

Table 6. Ammonia emission using the micrometeorological mass balance method.

Experiment (field)	Origin of slurry and application method	Starting date	Emission		
			NH_3–N ha^{-1}	Applied NH_4–N	Applied total-N
			kg	%	
1 (1)	'Spruit'; surface spreading	16 Sept. 2004; 11:37	5.4	36.0	17.0
1 (2)	'Spruit'; surface spreading	16 Sept. 2004; 11:58	4.5	31.1	14.7
2 (1)	'Spruit'; surface spreading	22 July 2005; 12:00	2.9	26.2	12.8
2 (2)	'Zegveld'; trailing shoe	22 July 2005; 12:15	7.0	29.5	12.8

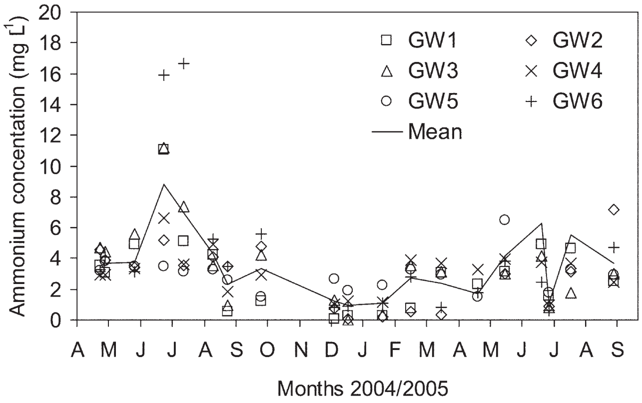

Fig. 2. Ammonium concentrations in the groundwater measured at different locations on the Spruit farm. GW, groundwater observation wells.

ly to the air (74 kg N ha^{-1}) and to deeper soil layers and/or water (166 kg N ha^{-1}) result directly from the imposed input parameters. To balance all inputs and outputs, an annual background mineralization of 108 kg N ha^{-1} was calculated. However, measurements hinted at a mineralization of around 142 kg ha^{-1}. Although further research is needed to validate the latter estimate, this discrepancy may point at underestimated losses to air and/or water because all other fluxes may be afflicted with errors as the background mineralization. Using the estimated background mineralization of 142 kg ha^{-1} as input, either the relative N losses from stables and storages had to be increased from the measured 30 to 73% (Scenario II), losses from surface spreading had to be increased from the measured 35 to 75% (Scenario III), or the estimated N uptake efficiency (i.e., 100% minus the relative losses to deeper soil layers and/or water) had to be lowered from 80 to 74% (Scenario IV). It is difficult to validate the latter with water quality measurements based on increases in concentrations alone. Denitrification in peat soils can be extremely high, ranging from 100 to 200 kg N ha^{-1} yr^{-1} (Van Beek et al., 2004a). This high denitrification is in line with the low nitrate concentrations under all Dutch peat soils summarized by the Netherlands Environmental Assessment Agency (MNP, 2006) and the overview of data on denitrification in The Netherlands to evaluate denitrification models (Heinen, 2006). Water quality measurements indicate a small nitrate leaching, but

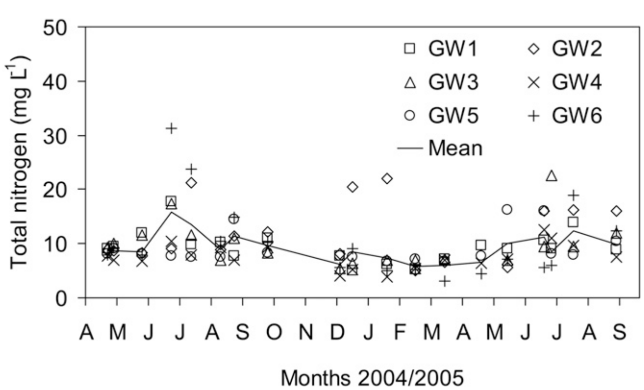

Fig. 3. Total nitrogen concentrations in the groundwater measured at different locations on the Spruit farm. GW, groundwater observation wells.

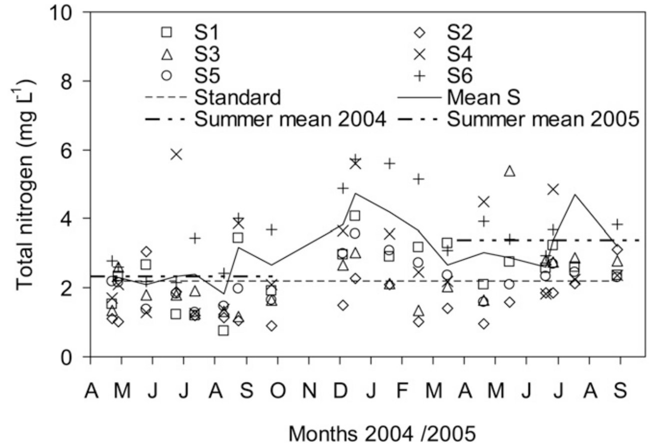

Fig. 4. Total nitrogen concentrations in surface water at different locations (symbols). The surface water quality standard and summer averages are indicated.

N leaching to surface water is difficult to quantify at the field- and farm scale due to the technical difficulties in measuring water and nutrient fluxes of incoming and outgoing water in ditches and canals. Regardless of whether Scenario I, II, III, or IV reflects the real situation at the Spruit farm, the calculated ammonia loss per ton of milk (range, 5.3–7.5 kg N per Mg) was fully comparable with the estimated ammonia loss (6.5 kg N ton^{-1}) of the conventional farm (Scenario V) (Table 8).

The regular disturbance of the manure heap and the above-ground spreading of the slurry at the Spruit farm contribute to a higher ammonia loss per kilogram of mineral N in manure. At the whole farm level, this is fully compensated by the lower mineral N in manure production due to the low-protein diet of the cows combined with the use of C-rich bedding material.

Discussion

Evaluating the Environmental Effects of the Farm Strategy

In general, a whole-farm strategy of low-protein feeding and use of bedding material can reduce ammonia losses substantially. Emissions are comparable to a conventional farm, even though the Spruit farm broadcasts manure on the land surface, because it is applied before rain or followed by adding water or sludge.

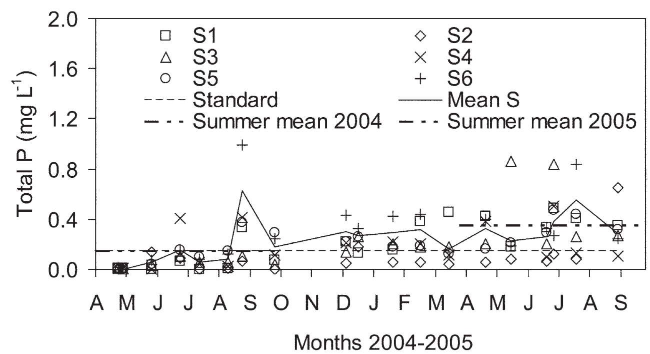

Fig. 5. Total phosphorous (P) concentrations in surface water at different locations (symbols). The surface water quality standard and summer half-year averages are indicated.

Thus, at least three components can be distinguished that contribute to the observed low ammonia emissions from this farm, which relate directly to requirements defined in current regulations. First, the low-protein feeding strategy results in a lower total N content in the manure and a lower proportion of ammonium N (Misselbrook et al., 2005). Second, the surface spreading of slurry takes place under cloudy, rainy weather conditions. The farmer tries to apply his manure under these conditions to reduce the emission of ammonia, a strategy supported by earlier findings (Klarenbeek and Bruin, 1990). He sometimes mimics the effects of rain by also applying water to the manure after it has been deposited. Third, timing of slurry application is related to grass growth (sufficiently high temperature) and grass length, which should be short enough to permit the slurry to reach the soil surface between the leaves.

A substantial amount of the calculated farm ammonia loss (74 kg N ha^{-1}) is due to losses from farm buildings, animal housing, and the manure heap. Ammonia emission from the manure heap is likely to be a significant pathway by which nitrogen is lost. This source is not included in the regulations.

Results for surface water quality were different between the years. We used the arithmetic mean of all samples and all times during the summer half-year to obtain average nutrient concentrations in ground and surface water, following the procedure of CIW (CIW, 2000). It is questionable whether this simple statistical analysis is adequate because calculating the geometrical mean results in lower means because the weight of the few high concentrations ("outliers") is smaller. However, the individual data points and the means support the general conclusion on nutrient concentrations in ground- and surface waters. In the summer half-year of 2004, water quality in terms of N and P concentration was below the environmental threshold values (Fig. 4 and 5). However, in 2005, concentrations were just above the thresholds. This cannot as yet be explained. Perhaps some increased surface runoff into the ditches and outflow from drains occurred in the rather wet summer of 2005. Surface water pollution resulting from cows trampling along the ditches could be excluded as an explanation because all ditches were fenced. Official threshold values for N and P in these surface waters have not been defined, so the results are inconclusive. However, we used rather strict thresholds for the water quality of the Spruit farm, and, compared with historical water quality data of other peat districts in The Netherlands, the N and P concentrations of the surface water of Spruit were smaller or close to these national data. In the (dry) summer half-year, the surface water of ditches and canals in peat areas often consists of large quantities of inlet water. The quality of the inlet water has a large impact on the water quality in the ditches. The contribution of inlet water and drainage water from agricultural land has to be better quantified to allow us to discriminate between inlet water of agriculture as a source of the nutrient concentrations.

Table 7. Estimated N and P_2O_5 surplus (kg $ha^{-1}yr^{-1}$) and efficiencies (kg kg^{-1}) for 2003, 2004, and 2005 at the Spruit dairy farm.

	N			P_2O_5		
	2003	2004	2005	2003	2004	2005
Input						
Concentrates replacers	89	69	57	45	31	26
By-products, brewery's spent grains, maize pellets	89	83	75	21	18	16
Straw	19	14	10	5	4	4
Solid manure	0	6	6	0	4	4
Fertilizer	0	0	0	0	0	0
N_2–fixation (clover)	40	40	40	–	–	–
Mineralization†	142	142	142	–	–	–
Deposition	29	29	29	1	1	1
TOTAL	408	383	358	72	58	50
Output						
Milk	89	83	78	35	29	31
Meat	19	23	18	12	15	13
Manure	0	6	6	0	3	3
TOTAL	109	112	102	48	47	46
Surplus	297	269	254	23	11	4
Efficiency	0.27	0.29	0.28	0.67	0.81	0.92

† Estimated, partly on the basis of N-yields from in situ trials.

Regulatory Implications

Having demonstrated with measurements that low ammonia emissions after surface spreading can be realized at this farm, it seems that current regulations based on prescribed low-emission techniques could be relaxed, but regulations should not be changed on the basis of data of one farm. As a follow-up, the Ministry of Agriculture, Nature and Food Quality provided 29 farmers in Friesland in the north of The Netherlands with an experimental exemption permit to further investigate on-farm nutrient emissions from manure. Additional measurements will be made from 2006 to 2008. This may result in a change from regulations that define a means to reach a certain environmental objective to regulations that define the environmental objectives but that allow farmers

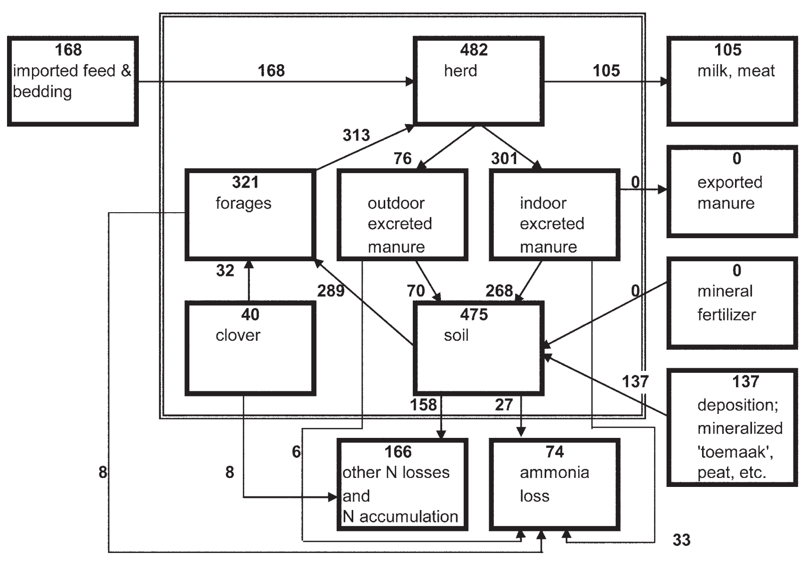

Fig. 6. Nitrogen fluxes at the Spruit dairy farm according to Scenario I (low-protein diet, use of bedding material, and aboveground spreading).

Table 8. Farm characteristics and ammonia losses at the Spruit farm following different scenarios (I–IV). Scenario V presents a similar farm without low-protein feeding, no use of bedding material but with the application of the trailing shoe machine. Model outcomes are indicated.†

Scenario	I	II	III	IV	V
'protein feeding'	Low	Low	Low	Low	Conventional
Use of allowed application technique	no	No	No	No	Yes
Bedding material	yes	Yes	Yes	Yes	No
Crude protein content, (g kg^{-1})	13.4	13.4	13.4	13.4	17.3
Self-sufficiency roughage (%)	67	67	67	67	67
Bedding material (kg animal^{-1} d^{-1})	1.9	1.9	1.9	1.9	0
Mineralization‡ (kg N ha^{-1})	108†	142	142	142	142
Fertilizer (kg N ha^{-1})	0	0	0	0	52†
Efficiency of soil N uptake by grass (kg kg $^{-1}$)	0.80	0.80	0.80	0.74†	0.70
Efficiency of feed N utilization by cattle (kg kg^{-1})	0.22	0.22	0.22	0.22	0.20
Produced manure (kg N ha^{-1})	377†	377†	377†	377†	418†
Applied manure§ (kg total N total ha^{-1})	344†	297†	344†	344†	381†
Nm/Ntot at excretion¶	0.47†	0.47†	0.47†	0.47†	0.56†
Nm/Ntot at application	0.29†	0.14†	0.29†	0.29†	0.50†
NH_3–N loss from stables and manure heap (kg kg Nm^{-1})	0.30	0.73†	0.30	0.30	0.20
NH_3–N loss urine-N during grazing [kg (kg Nm^{-1})]	0.15	0.15	0.15	0.15	0.15
NH_3–N loss feces-N during grazing (kg [kg Nm^{-1}])	0.01	0.01	0.01	0.01	0.01
NH_3–N loss from slurry [kg (kg Nm^{-1})]	0.35	0.35	0.75†	0.35	0.25
N-surplus (kg ha^{-1})	241†	275†	275†	275†	331†
lost to soil (kg ha^{-1})	166†	170†	170†	201†	240†
lost to air as NH_3–N (kg ha^{-1})	74†	105†	105†	74†	91†
NH_3–N loss (kg Mg milk^{-1})	5.3†	7.5†	7.5†	5.3†	6.5†
Farm N use efficiency (kg kg^{-1})	0.30†	0.28†	0.28†	0.28†	0.24†

† Model results.

‡ From peat and anthropogenic topsoil.

§ Including grazing manure.

¶ Nm, mineral N; Ntot, total N.

greater flexibility to mobilize their innovative capabilities to reach those objectives. However, to arrive at realistic enforceable proposals, a comprehensive risk analysis needs to be made that includes all the implications of the various procedures. For example, when comparing injection or surface application of conventional manure, possible adverse soil structure effects of using low-emission technology need to be quantified.

Identified characteristics of farm management that contribute to a low environmental impact should be translated into indicators that are cost-effective and robust. For example, having manure with a lower ammonium content is no guarantee of a lower emission. Most likely, a combination of standardized measurements on other farm characteristics will have to be implemented, such as the urea content in the milk and N content in the slurry. For the latter, improved procedures are needed to deal with the heterogeneity in composition.

The nitrate content in the ground water was consistently below the 50 mg L^{-1} standard on the Spruit farm. This is mostly due to denitrification in these peat soils because most of the nitrates in the soil are denitrified and escape as nitrogen gas or as N_2O, a prominent greenhouse gas. The ground water nitrate standard has been "translated" into a N input standard of 170 kg manure-N ha^{-1} for most European countries and a N input standard of 250 kg manure-N ha^{-1} for the Netherlands, providing that the grassland area is more than 70% of the total farmed land area. These inputs can be translated to cattle densities by estimating an average N excretion per animal based on milk production per cow and urea content in the milk. Because the whole of The Netherlands has been defined as a Nitrate Vulnerable Zone by the Dutch government, farms on peat soils are restricted in the permitted cattle stocking rate. From a ground water quality perspective, this is irrelevant for these peat soils. The preferable stratification of the regulations by soil type (because of the different processes involved in different soils) is counter-balanced by the desire to minimize the administrative burden for the government and to create a level playing field for all Dutch farmers. Between the desire to incorporate local variability and the desire to maximize harmonization, science and policy have to work together with farmers to find the optimal policy constructions.

Conclusions

A whole-farm analysis of N fluxes at a commercial dairy farm that aims to combine broadcast slurry spreading under favorable weather conditions with a low-protein diet revealed that the total ammonia emission at farm level is comparable to more conventional farms in the Netherlands, where, following current regulations, manure is directly applied at the soil surface. For a possible future relaxation of current regulations, additional research at field, farm, and landscape level is needed.

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