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Permissible manure and fertilizer use in dairy farming systems on sandy soils in The Netherlands to comply with the Nitrates Directive target

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Abstract

Properly managed manures have a high fertilizer equivalency and are thus a valuable source of nutrients in forage production systems. An efficient utilization of these nutrients, however, is limited by the crop's demand for nitrogen (N) and phosphorus (P). Moreover, environmental goals implied by the EU Nitrates Directive impose constraints on the use of manure and mineral fertilizer. Through calculations based on experimental data from various sources, the present study explores the limits on the use of cattle slurry and mineral fertilizer in grass and silage maize production on sandy soils in the Netherlands. The study concludes that cut grasslands can utilize cattle slurry up to average rates of 330–340 kg N/ha (120 kg P_2O_5) per year without exceeding a target value of 11.3 mg nitrate-N/l in the upper groundwater or accumulating P in the soil, provided that (i) appropriate amounts of mineral fertilizer N are supplemented, and (ii) growing conditions are good and the grassland is well-managed. When grassland is commonly used for both cutting and grazing, slurry rates have to be reduced by 60 kg N/ha per year (20 kg P_2O_5) to achieve these targets. Similarly, not more than 170 kg slurry-N/ha (60 kg P_2O_5) per year should be applied to silage maize. When grown on dry soils susceptible to leaching, slurry rates on maize land need a further reduction to 155 kg N/ha per year (=55 kg P_2O_5). When grass and maize are grown in rotation, cattle slurry and fertilizer applications to maize should be reduced even more drastically, whereas application rates to grassland can be extended to compensate for the temporal investment in the new sod. Consequently, from the point of view of N leaching and P accumulation, manure rates should be determined by (i) the harvest regime of the grass, (ii) the proportions of grass and maize in the farm area and the way they are positioned in a rotation, and (iii) the susceptibility to leaching (i.e. the hydrological situation). The study also indicates that reductions of 50–60 kg and 25–30 kg slurry-N/ha per year are required in grassland and maize, respectively, if growing conditions or cropping management are suboptimal.

Keywords: Dairy farming; Leaching; Management; Manure; Modelling; Nitrates Directive; Nitrogen; Phosphorus; Sandy soils

1. Introduction

Sustainable crop production requires nitrogen (N) and phosphorus (P) inputs to compensate for the N and P removed from the system by exported produce and losses (Carton and Jarvis, 2001). Losses of N and P are positively related to input levels and negatively related to their use efficiency, i.e. output–input ratios. Agricultural losses should be minimized as they largely determine the quality of water bodies (Tunney et al., 1997; Rabalais, 2002). In view of the negative relationship between animal density and water quality on a European scale, the European Union (EU) Nitrates Directive (Anonymous, 1991) has set 170 kg manure-N/ha per year as a precautionary application threshold for regions that are vulnerable to N leaching. Manure application rates however, are at most indicative of environmental quality and not necessarily effective as environmental effects are determined by all inputs and outputs together instead of the manure input only. Farming in general and livestock production in particular, is undeniably associated with the risk of exceeding the EU target of 11.3 mg nitrate-N/l water, especially because manures are inherently difficult to manage (Schröder, 2005). Manure management on grassland and maize deserves special

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scrutiny in the Netherlands; these two crops occupy almost 80% of the agricultural area in regions with sandy soils, the soil type most sensitive to N leaching (http://statline.cbs.nl). Due to the favourable climate in The Netherlands and the presence of ample irrigation equipment on farms growing grass and maize (Schröder et al., 2005a), forage yields and thus N outputs belong to the highest in Europe (Peeters and Kopec, 1996). Moreover, similar differences between inputs and outputs, i.e. N surpluses, may result in completely different N losses to water bodies due to differences in climate, hydrology, management and land use (Schröder et al., 2004). P also poses a threat to water quality, whenever input rates exceed removal via crops, as accumulated P may eventually leach as well.

The aim of this paper is (i) to estimate which soil N surplus can be permitted on sandy soils without exceeding the target of 11.3 mg nitrate-N/l groundwater, (ii) to explore via modelling which rate combinations of fertilizer N and cattle manure N applied to grassland and maize achieve this nitrate target without depleting or accumulating soil-P pools, and (iii) to check to what extent these rates support the specifications of the EU Nitrates Directive.

2. Materials and methods

2.1. General

Water quality under and on agricultural land is determined, among other factors, by the discrepancy between N and P inputs and outputs to and from that land (i.e. the surplus per unit area) and the loss pathways of this surplus. In order to relate (allowable) inputs to (required) water quality and vice versa, it is crucial to assess and define inputs and outputs and the fate of their difference in any model.

2.2. Input

In our model we define N input as the sum of manure-N (so, minus the gaseous N losses from housing and storage), mineral fertilizer N, soil mineral N at the onset of the growing season (SMN_{spring}), deposition of atmospheric N, biologically fixed N and N mineralized from soil organic matter. Sources of this mineralization are crop residues (including roots, stubbles, harvest losses and winter cover crops), and manure applied in previous years. Of these inputs manure-N, mineral fertilizer N and atmospheric N are the external inputs, the others represent internal fluxes (Fig. 1).

We assume a SMN_{spring} input of 30 kg N/ha (Schröder et al., 1998) and an annual atmospheric deposition of 31 kg N/ha (Anonymous, 2004). We estimate that on an annual basis 75 kg N/ha is mineralized from grass roots and stubbles (Velthof and Oenema, 2001), 25 kg N/ha from maize roots and stubbles (Schröder, 1991) and 40 kg N/ha from winter cover crops grown after maize (Schröder et al., 1996). These contributions to mineralization can only be sustained through similar annual inputs into the soil organic N pool. Likewise, the N mineralization from manure inputs in previous years is also accounted for. Our calculations are restricted to cattle slurry which is by far the dominant manure type on farms growing grass and maize in the Netherlands (Menzi, 2002). The long term residual N mineralization from cattle slurry (i.e. beyond the first 12 months after

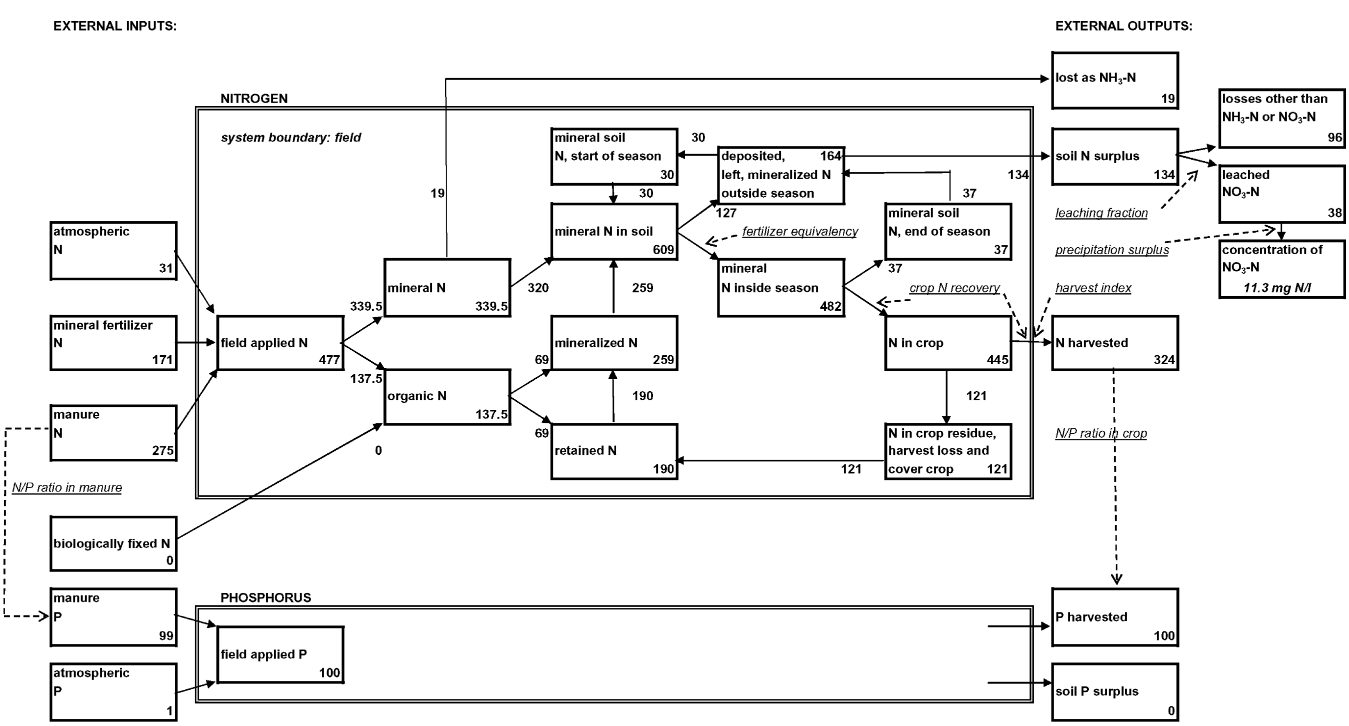

Fig. 1. Flow diagram of external N inputs and outputs and internal N fluxes in the present model, including a numerical example referring to grassland under good growing conditions and management with a mixed use of cutting and grazing, grown on a sandy soil with a mean highest groundwater table between 0.40 and 0.80 m below the soil surface.

Table 1
Fertilizer-N equivalency of various N sources for grass and maize

Source	Grass (%)	Maize (%)	Reference
Soil mineral N in spring	100	100	By definition
Applied cattle slurry N[a]	64	60	Huijsmans (2003), Schröder (2005) and Van Dijk et al. (2004)
Excreted urine and dung during grazing[a]	16[b]	–	Vellinga et al. (2001) and Van Dijk et al. (2004)
Atmospheric deposition	75	75	After Schröder and van Keulen (1997)
Mineralization of soil N, including crop residues and the resistant organic N fraction of manure	75	60	After Schröder and van Keulen (1997) and Van Dijk et al. (2004)

[a] In the first 12 months after application/excretion.
[b] i.e. 25% of the fertilizer-N equivalency of mechanically applied cattle slurry.

its application) amounts to 25% of the total manure input (Van Dijk et al., 2004). We balance this mineralization via a similar annual investment into the soil organic N pool. Biologically fixed N is assumed to be zero as clover is hardly present in Dutch grasslands but in the discussion section we address situations in which it is present.

2.3. Output

In order to assess the soil N surplus, all other outputs must be subtracted from the inputs. We define these outputs as the sum of crop N which is removed by either grazing or harvests (see below), N investments in (new) crop residues, and N stored in the organic N fraction of manure if it is not mineralized in the first 12 months after application (Schröder, 2005). In an equilibrium situation, defined here as a situation in which there is no change in total N content of the soil, the annual N input from mineralizing crop residues and formerly applied manure, equals the N invested in these pools. In other words: to sustain a system several inputs require annual renewal. This may not hold on an annual basis in regularly renovated grassland or in all phases of a crop rotation. To explore the consequences we address this situation in more detail in our scenarios (see below).

Ammonia losses, too, must be subtracted for a correct assessment of the soil N surplus. We estimate these losses to be 5% of the total manure N input when injected, and 8% of the total manure N when excreted during grazing (Jarvis et al., 1989; Bussink, 1992; Bussink, 1994; Huijsmans, 2003). Ammonia volatilization from fertilizers is set at 1% of the mineral fertilizer N input, bearing in mind that calcium ammonium nitrate and not urea is by far the dominant fertilizer type in the Netherlands (Velthof et al., 1990). Of these outputs harvested N, ammonia losses and the soil N surplus are the outputs crossing the system boundary, the other outputs represent internal fluxes becoming inputs again (Fig. 1).

The harvested N output is determined by (i) the fertilizer equivalency, i.e. the availability to plants of N from various sources relative to mineral fertilizer N (Table 1, including references), (ii) the uptake efficiency, i.e. the fraction of the available N taken up by the crop whilst accounting for the reduction in uptake efficiency at higher input levels (Table 2, including references) and (iii) the harvest efficiency, i.e. 1 minus the fraction of crop N which is lost before it is either eaten or removed via harvests (Table 3, including references).

In accordance with present legislation in The Netherlands, we assume that manure ex storage is applied in spring (grassland and maize) and early summer (grassland) that manure is injected into the sod or bare soil and that maize is followed by a cover crop. Grassland is irrigated whenever necessary, simulating situations with good agricultural practice.

2.4. Fate of the soil N surplus

In the previous sections the calculation of N soil surplus in a steady state situation was explained. In equilibrium situations for soil organic N pools, the soil N surplus (i.e. corrected for ammonia volatilization) is either denitrified or leached. We calculated the effect of the N soil surplus on N concentrations in groundwater (surface water is hardly present in the Dutch sandy soil regions) by multiplying the N surplus by crop and soil specific

Table 2
N uptake efficiency of fertilizer N equivalents for grass and maize

	Grass	Maize
Initial efficiency at low input rates	85%	75%
Indicative mineral fertilizer N rate (kg N/ha) at which efficiency commences to diminish (‘deflection point’)	270	80
Efficiency reduction (% (absolute) per 100 kg additional mineral N per ha) beyond deflection point	10%	10%
N uptake plateau (kg N/ha) at which marginal N efficiency becomes 0%	510 and 460 for fully cut swards and swards with mixed use, respectively	190 and 130 for sandy soils with a MHG[a] shallower than 0.80 m and a MHG deeper than 0.80 m, respectively
References	Alberda (1968), Prins (1980), Sibma and Ennik (1988), Vellinga and André (1999), Ten Berge et al. (2002) and Schils and Kok (2003)	Schröder et al. (1998), Nevens and Reheul (2002) and Nevens (2003)

[a] MHG = mean highest groundwater table.

Table 3
Fraction (%) of crop not taken in by animals or exported from the field due to mechanical damage to sward, and due to lost crop material during grazing, wilting and mechanical harvesting, as affected by land use and mean highest groundwater table (MHG) (Beuving et al., 1989; Corporaal, 1993)

Regime	Soil type	Grass (%)	Maize (%)
Cutting only	All	4	5
Mixed use of cutting and grazing	Sand with MHG above 0.80 m	15	–
	Sand with MHG below 0.80 m	10	–

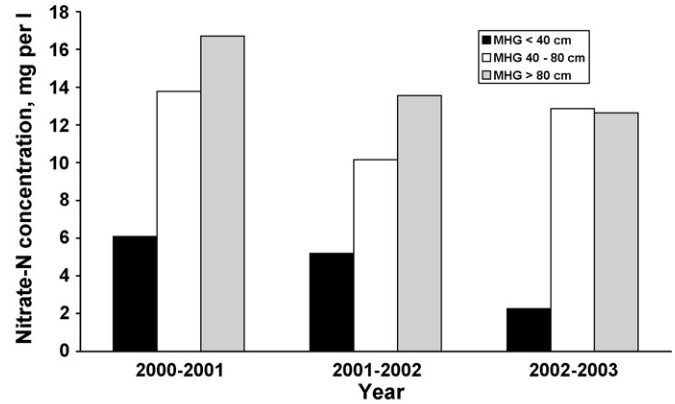

Fig. 2. Median nitrate-N concentrations in the upper 1 m groundwater measured on farms with comparable intensities on sandy soils in the period 2000–2003 at three classes of groundwater table, i.e. shallow groundwater level (mean highest groundwater table (MHG) above 0.40 m, $n = 70–72$), groundwater at intermediate level ($0.80 < MHG < 0.40$ m, $n = 183–185$), and deep groundwater level (MHG below 0.80 m, $n = 212–217$) (*Source*: Velthof, 2004).

leaching fractions and dividing the product by crop and soil specific precipitation surpluses (Table 4). The combined effects of these two factors were derived from a current national Monitoring Program established in the early nineties. In this network, soil N surpluses of farms are linked to corresponding nitrate-N concentrations in the upper 1.00 m groundwater on sandy soils.

Groundwater quality on sandy soils is annually measured once during March–September. On most farms only 16 samples are taken, compared to 48 on semi-experimental pilot farms. The whole farm area is sampled, taking the grassland: maize ratio into account. The national network consisted, on average, of 40 (12–75) dairy farms on average from 1992 to 2002, 24 (0–35) arable farms, and 10 (0–24) mixed farms and/or pig/poultry farms each year. Approximately 64% of the dairy farms are located on sandy soils. Approximately 45% of the monitored arable farms, also providing data to assess the leaching fractions for arable crops such as maize, are located on sandy soils.

According to measurements collected in the Monitoring Program 39% of the N surplus from grasslands on dry sandy soils leaches to the groundwater (number of years = 10, S.D. 6%). Much higher leaching fractions are indicated by the Monitoring Program for arable land. On sandy soils with deep groundwater, the calculated leaching fraction averages 106% (number of years = 7, S.D. 7%).

The observed nitrate-N concentrations become lower with higher groundwater levels. This reduction of concentration ranges from 0% on sandy soils with 'deep' groundwater (mean highest groundwater table during winter (MHG) below a depth of 0.80 m) to 57% (S.D. 7%) on sandy soils with shallow groundwater (MHG above a depth of 0.40 m). These figures are based on research carried out from 1982 to 1991 on field level (Van der Meer, 1991), farm level (Boumans et al., 1989; Breeuwsma et al., 1991), as well as in experiments with lysimeters (Steenvoorden, 1988). The quantitative implications of these studies showed a great similarity and all lead to the conclusion that the amount of nitrate recovered in shallow groundwater was negatively related to groundwater levels (see Fig. 2 for recent monitoring results). This phenomenon has been attributed to denitrification in the layer between the root zone and 1 m below the groundwater table in situations with shallow groundwater, although denitrification was not explicitly measured. Under those conditions groundwater is in direct contact with soil layers containing degradable carbon, by which denitrification is enhanced (Munch and Velthof, 2006). In the discussion section we address this matter in greater detail.

2.5. Phosphorus balance

An integrated approach towards nutrient emission from agriculture requires attention to P, as manure P is added with manure N. For that purpose we adopted a simple soil surface P-balance (Fig. 1). The P input is the sum of atmospheric P (1 kg P_2O_5 (=0.43 P)/ha per year (Ellerman et al., 2003)) and excreted or applied manure-P, which is deduced from the manure-N input. We have used a fixed P_2O_5–N ratio in cattle

Table 4
Net leaching fractions (kg N leached/kg soil N surplus; S.D.'s based on yearly variation in parentheses), precipitation surplus (mm), and the soil N surplus (kg N/ha; in parentheses the 95% confidence interval around the mean values derived from the yearly variation of the leaching fractions) associated with a N concentration of 11.3 mg nitrate-N/l in the upper 1.00 m of groundwater, as affected by land use and mean highest groundwater table (MHG) (National Monitoring Program 1992–2001; *Sources*: Fraters et al., 1998, 2002)

Land use	Soil type	Net leaching fraction (kg/kg)	Precipitation surplus (mm)	Allowable soil N surplus (kg N/ha)
Arable land	Sand, MHG above 0.40 m	0.50 (0.08)	387	88 (66–132)
	Sand, 0.80 < MHG < 0.40 m	0.75 (0.09)	434	65 (52–87)
	Sand, MHG below 0.80 m	1.06 (0.08)	453	48 (42–57)
Grassland	Sand, MHG above 0.40 m	0.18 (0.04)	268	165 (115–290)
	Sand, 0.80 < MHG < 0.40 m	0.28 (0.05)	329	134 (98–212)
	Sand, MHG below 0.80 m	0.39 (0.06)	355	103 (79–148)

manure (applied slurry and field excreted urine and dung) of 0.36 (Van Dijk, 2003; Tamminga et al., 2004), as we limit our calculations to cattle manure. The P output comprises the P removed by either grazing animals or harvests. Outputs are calculated as the product of N outputs and a P_2O_5–N ratio of 0.33 for cut grassland, 0.29 for grazed grassland and 0.37 for silage maize (Beukeboom, 1996; Tamminga et al., 2004). To explore the sensitivity of the previous assumptions we varied the P_2O_5–N ratio of the manure and the crop in a specific case.

We did not attempt to calculate P concentrations in water because it requires site-specific information on the P status of the soil, the hydrology and chemical and biological transformations of inorganic and organic P. Contrary to N, where experiments show that N leaching rapidly changes when the N-input changes (e.g. Garrett et al., 1992; Aarts et al., 2001), the situation for P is more complicated. The P status of the soil and its aeration are the dominant factors determining the P concentration in leachate (Schoumans and Chardon, 2003). Thus, the relation between the inputs, surpluses and concentration in water is less direct for P than for N.

2.6. Validation of the model

Before applying the model we tested its assumptions. Parallel to the present study, Aarts et al. (2005) estimated the net N yield of grasslands in the Netherlands by analysing records of 127 commercial dairy farms on sandy soils from four recent years (1998, 1999, 2001 and 2002). Their point of departure was the energy requirement of the herd on each individual farm and the estimated energy and N contents of forages and concentrates. By combining these data with registered purchases of feed and estimates of the on-farm N yield of silage maize (other crops were not grown) and the N losses from forages during conservation and feeding, they made an estimate of the apparent N yield from grassland. Their database also comprised data on the allocation of manure N and mineral fertilizer N to either grassland or maize land on each farm. This allowed us to compare the use efficiency of N (NUE, i.e. the product of fertilizer equivalency, uptake efficiency and harvest efficiency; Tables 1–3) realized on these farms, to the NUE of comparable inputs with the present model. The coefficients in Table 4 were tested against a recent independent dataset of soil N surpluses and nitrate N concentrations in the upper groundwater from pilot farms on sandy soils participating in the Cows and Opportunities project (Oenema et al., 2001).

2.7. Exploration of fertilizer-manure combinations

We subsequently used the MS Excel™ Solver Tool to determine which combinations of manure N and fertilizer N ('variable cells') would maximize the harvestable N yield ('target cell'). This was done under the constraints that (i) the N concentration in groundwater is 11.3 mg nitrate-N/l (or less when no further yield increase was brought about by additional N inputs), and (ii) the P surplus is 0 kg/ha, i.e. soil P pools are depleted nor augmented ('constraints cells'). The algorithms are given in Appendix A. The Solver Tool has the unfortunate property that it prematurely stops optimizations whenever too low starting values of the variables have been chosen, regardless the defined maximum number of iterations. It then suffices with a message that a solution was not found or that variable cell values have converged to certain values instead of arriving at a real optimum. This phenomenon could be successfully avoided by choosing larger starting values, vizually 200 kg slurry N and 200 kg mineral fertilizer N/ha. In addition we tested various rate combinations around the recommended optimum for each scenario, to check whether the optimum in terms of target and constraints had really been achieved.

Separate calculations were made for grasslands with a 'cutting only' regime, for grasslands with the common mixed use, i.e. in which about half the production is harvested via grazing, and for silage maize. For mixed grassland use we assumed that an average of 120 kg N/ha per year is excreted outdoors in the form of urine and dung (Tamminga et al., 2004). This excretion is a function of milk quota, the N content of diets, the maize–grass ratio and the grazing regime and hence varies from farm to farm. However, effects (like high milk production per ha, but limited number of hours daily grazing) are assumed to balance each other (Aarts et al., 2005).

The previous scenarios were run with the coefficients presented in Tables 1–3. These coefficients were derived from field experiments and thus relevant to good growing conditions (i.e. irrigation and drainage, soil fertility status, exclusion of field borders) and good management (i.e. the timing of operations). Evaluations based on these data are not necessarily representative for all practical forage production systems. For instance, the timing of operations cannot always be optimized on each individual field, or the utilization of N and P on a whole field basis may be somewhat lower than on experimental plots in the same field. To mimic these effects we also ran the model for a situation in which we reduced the assumed crop uptake efficiency by 10% (e.g. 67.5% instead of 75%) and simultaneously reduced the harvest efficiency by 5% (e.g. 9% losses instead of 4%).

As the leaching fractions derived from the Monitoring Program vary from year to year (Table 4), the allowable rates vary as well. Therefore we added 95% confidence intervals around our calculated mean manure and mineral fertilizer N rates. The lower and upper boundaries of this interval were approximated by running the MS Excel™ Solver Tool again with imposed leaching fractions equal to the average value plus 2 × S.D. and the average value minus 2 × S.D., respectively, instead of the average value.

Most dairy farms on sandy soils in The Netherlands grow both grass and maize. With this combination of crops the goals in terms of N concentration in water and P soil surplus must be realized. Therefore, we have also explored the consequences of a gradual substitution of grass by maize. This exploration was restricted to the average leaching fractions.

When both crops are present on a farm, they are often rotated implying that grassland is regularly ploughed. Ploughing stimulates mineralization, contribution approximately 125–400 kg N/ha (Velthof and Oenema, 2001; Velthof et al., 2002). If ignored as an input of mineral N to subsequent crops, most of the mineralized N will be lost to the environment.

Conversely, young grassland can temporarily immobilize 20–130 kg N/ha per year, thus reducing immediate losses to the environment (Velthof and Oenema, 2001; Velthof et al., 2002). We accounted for these turnover processes by running the model once more while taking the N dynamics in a rotation into consideration. We restricted our calculations to a sandy soil with 0.80 < MHG < 0.40 m, a mixed use of grassland, and the average soil-specific leaching fractions. Further, we assumed good growing conditions and management and avoidance of P accumulation at the whole farm level. We adopted a rotation comprising 4 years of temporary grassland followed by 2 years maize (‘66.6% grassland and 33.3% maize’). We surmised an additional annual 150 kg N mineralization/ha from the ploughed grassland (2 years × 150 = 300 kg N/ha) and an additional annual N build up of 75 kg N/ha under grassland (4 years × 75 = 300 kg N/ha). In reality, however, mineralization and immobilization are generally larger during the first year(s) of each crop phase. It is unclear at which spatial scale the nitrate target in the Nitrates Directive should be achieved, so we distinguished two situations: one in which nitrate stays below the target under each individual crop (C) and another in which nitrate stays below the target at the whole farm level (F). The outcomes were compared with our initial approach where we ignored the N dynamics of the rotation (I).

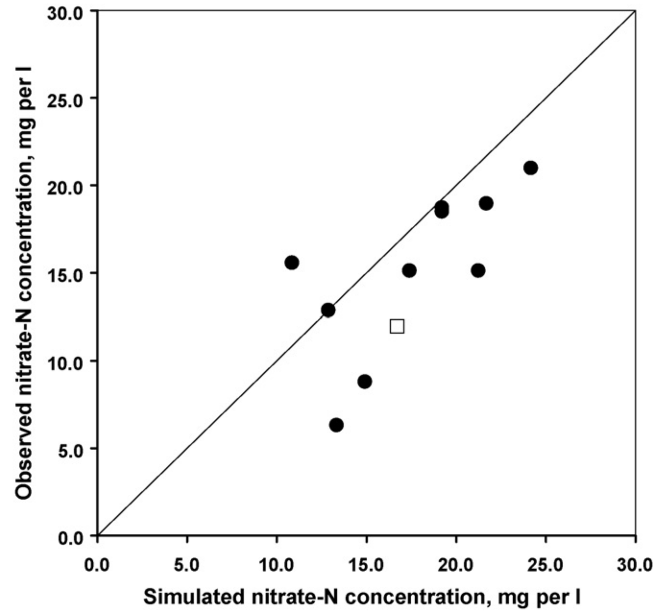

Fig. 3. Observed (average 2000–2002) vs. simulated (following Table 4) nitrate-N concentrations in the upper 1 m of groundwater of dairy farms on sandy soils participating in the Cows and Opportunities project (experimental farm De Marke indicated by □).

3. Results

3.1. Validation

The NUE of grassland on dairy farms in the Netherlands according to Aarts et al. (2005), was very close to the NUE in our scenarios that assume ‘good growing conditions and management’. The NUE of maize on wet sandy soils, however, was more similar to our scenarios that assume ‘suboptimal growing conditions and management’ (Table 5). The NUE at the whole farm level, reflecting weighted contributions from both crops in terms of hectares and N involved, was very close to ‘good growing conditions and management’. The average daily temperature during the growing season in the years investigated by Aarts et al. (2005) was slightly higher than the average of the last 30 years, a common phenomenon in the last decade. Accumulated rainfall in all 4 years was also above the long term average. The NUE on dry sandy soils has probably been favoured by these weather conditions, whereas the NUE on wetter soil types may have suffered due to increased trampling damage and harvest losses. The observed NUE of commercial farms based on the approximated grassland N yields, generally appears to support the assumptions in our model concerning the integral effect of the fertilizer equivalency, uptake efficiency and harvest efficiency.

Applying our leaching fractions and precipitation surpluses (Table 4) to the observed soil N surpluses of pilot farms slightly overestimated the nitrate-N concentrations compared to the observed concentrations (Fig. 3). It is, however, beyond the scope of this paper to examine this moderate discrepancy in detail. In general, we conclude that the validation shows a reasonable fit between simulated and observed nitrate-N concentrations on a whole farm level.

Table 5
Comparison of the N use efficiency (net N yield of crops/(manure-N applied or excreted during grazing + mineral fertilizer N)) of registered N inputs and estimated N yields on 127 commercial dairy farms on sandy soils (Aarts et al., 2005), and the calculated N use efficiency of similar N inputs by the model (‘mixed use of grassland’) used in the present study

Soil type	Crop	Model		Farm data ‘estimates from practice 1998–2002’
		‘Good growing conditions and management’	‘Suboptimal growing conditions and management’	
Sandy, MHG[a] above 0.40 m	Grassland	0.67	0.57	0.67
	Maize	0.65	0.56	0.57
	Whole farm	0.67	0.57	0.66
Sandy, MHG below 0.80 m	Grassland	0.65	0.56	0.67
	Maize	0.54	0.46	0.55
	Whole farm	0.64	0.54	0.65

[a] MHG = mean highest groundwater table.

Table 6
Allowable manure rate (kg N/ha per year, 95% confidence interval in parentheses), the associated mineral fertilizer N rate (kg N/ha per year, 95% confidence interval in parentheses) and resulting net N and P_2O_5 yields in crops (kg/ha per year, ranges associated with the confidence interval of application rates in parentheses) on dairy farms targeting at nitrate-N < 11.3 mg/l (or less when no further yield increase is brought about), as affected by crop type, the grass harvesting regime and the soil type, in combination with *good growing conditions and management* whilst avoiding P-accumulation (see text for explanations and assumptions)

Crop type	Harvest regime	Soil type	Manure-N	Mineral fertilizer-N	N yield	P_2O_5 yield
Grass	Cutting only	Sand, MHG[a] above 0.40 m	340 (333–341)	187 (136–224)	374 (366–375)	123 (121–124)
		Sand, 0.80 < MHG < 0.40 m	336 (329–341)	155 (118–224)	370 (362–375)	122 (120–124)
		Sand, MHG below 0.80 m	330 (323–338)	123 (97–170)	363 (356–372)	120 (117–123)
	Mixed use	Sand, MHG above 0.40 m	274 (261–282)	203 (149–299)	322 (308–332)	100 (95–103)
		Sand, 0.80 < MHG < 0.40 m	275 (263–289)	171 (132–253)	324 (309–340)	100 (96–105)
		Sand, MHG below 0.80 m	273 (261–287)	139 (112–188)	321 (308–338)	99 (95–104)
Maize		Sand, MHG above 0.40 m	175 (170–175)	53 (42–53)	173 (168–173)	64 (62–64)
		Sand, 0.80 < MHG < 0.40 m	169 (162–175)	41 (28–52)	163 (160–173)	62 (59–64)
		Sand, MHG below 0.80 m	155 (154–155)	19 (17–19)	154 (153–154)	57 (57–57)

[a] MHG = mean highest groundwater table.

3.2. Allowable rates at the crop and farm level

If avoidance of P accumulation is included as a constraint, manure rates of 330–340 kg N/ha (120 kg P_2O_5/ha) would be possible on cut grasslands, provided that the growing conditions and management are good. Similarly, rates of 275 kg manure-N/ha (100 kg P_2O_5/ha) would be possible on grassland used for both cutting and grazing, and rates of 155–175 kg manure N/ha (55–65 kg P_2O_5/ha) on silage maize (Table 6).

A sensitivity analysis indicated that a 10% increase of the P_2O_5–N ratio of cattle manure, e.g. resulting from an unsuccessful reduction of gaseous N losses from collected manure, would reduce the allowable manure-N rate by approximately 25 kg N/ha. A similar reduction would be necessary if the P_2O_5–N ratio of cut grass would be 10% lower. Thus, fertilizer N inputs could be slightly higher under these circumstances without affecting final N yields.

When correcting for ‘suboptimal growing conditions and management’, input rates require a reduction in order to comply with the targeted N concentration in groundwater and to attain a balance between P inputs and outputs. Manure rates of 270–290 kg N/ha (100 kg P_2O_5/ha) would then be possible on cut grassland. Similarly, rates of 210–225 kg manure-N/ha (70–80 kg P_2O_5/ha) would be possible on grassland used for both cutting and grazing, and rates of 135–150 kg manure N/ha (50 kg P_2O_5/ha) on silage maize on most soils. Especially on sandy soils with a MHG below 0.80 m not more than 125 kg manure N/ha could be applied to maize land (Table 7).

The indicated manure rates (Tables 6 and 7) can only be considered environmentally safe when mineral fertilizer rates are sufficiently reduced. Too little mineral N supplements, however, will generally reduce yields, P removal and, thus, room for manure application, as any manured crop is short of N due to the larger N/P ratios in crops than in manures (Schröder, 2005). Mineral fertilizer rates associated with the indicated manure rates are hence included in Tables 6 and 7. Our calculations also indicate that slurry rates should be approximately 10 kg N/ha less if rates are to be based on the assumption that N concentrations are not to be exceeded in 95 out of 100 years. In that case, mineral fertilizer N rates would require a mean reduction by 30 kg N/ha (Tables 6 and 7). Our explorations of the effect of the proportions of maize and grass in the total farm area show that allowable inputs are negatively related to the presence of maize (Table 8).

Table 7
Allowable manure rate (kg N/ha per year, 95% confidence interval in parentheses), the associated mineral fertilizer N rate (kg N/ha per year, 95% confidence interval in parentheses) and resulting net N and P_2O_5 yields in crops (kg/ha per year, ranges associated with the confidence interval of application rates in parentheses) on dairy farms targeting at nitrate-N < 11.3 mg/l (or less when no further yield increase is brought about), as affected by crop type, the grass harvesting regime and the soil type, in combination with *suboptimal growing conditions and management* whilst avoiding P-accumulation (see text for explanations and assumptions)

Crop type	Harvest regime	Soil type	Manure-N	Mineral fertilizer-N	N yield	P_2O_5 yield
Grass	Cutting only	Sand, MHG[a] above 0.40 m	286 (276–291)	180 (128–255)	315 (304–320)	104 (100–106)
		Sand, 0.80 < MHG < 0.40 m	281 (271–290)	147 (110–228)	309 (299–319)	102 (99–105)
		Sand, MHG below 0.80 m	272 (264–283)	114 (89–162)	300 (291–312)	99 (96–103)
	Mixed use	Sand, MHG above 0.40 m	225 (209–238)	192 (138–322)	266 (247–281)	82 (76–87)
		Sand, 0.80 < MHG < 0.40 m	223 (196–241)	160 (118–243)	264 (231–284)	81 (71–88)
		Sand, MHG below 0.80 m	211 (170–235)	126 (92–177)	249 (202–277)	77 (62–86)
Maize		Sand, MHG above 0.40 m	147 (137–149)	64 (41–68)	146 (136–148)	54 (50–55)
		Sand, 0.80 < MHG < 0.40 m	136 (129–147)	40 (27–63)	135 (128–146)	50 (47–55)
		Sand, MHG below 0.80 m	127 (123–132)	23 (17–32)	126 (122–131)	47 (45–48)

[a] MHG = mean highest groundwater table.

Table 8
Allowable average manure rates (kg N/ha per year) and associated mineral fertilizer N rates (kg N/ha per year) on dairy farms targeting at nitrate-N < 11.3 mg/l (or less when no further yield increase is brought about), as affected by growing conditions and/or management quality, by the grass harvesting regime, by the soil type, and by the share of silage maize in the total area of grass and maize (%), whilst avoiding P-accumulation (see text for explanations and assumptions)

Soil type		Good[a]				Suboptimal[a]			
		Cutting only[b]		Mixed use[b]		Cutting only[b]		Mixed use[b]	
	Maize share	Manure-N	Mineral fertilizer-N	Manure-N	Mineral fertilizer-N	Manure-N	Mineral fertilizer-N	Manure-N	Mineral fertilizer-N
Sandy, MHG[c] above 0.40 m	15	315	167	259	181	265	163	213	173
	30	291	147	244	158	244	145	202	154
	45	266	127	229	136	223	128	190	134
Sandy, 0.80 < MHG < 0.40 m	15	311	138	259	152	259	131	210	142
	30	286	121	243	132	238	115	197	124
	45	261	104	227	113	216	99	184	106
Sandy, MHG below 0.80 m	15	304	107	255	121	250	100	198	111
	30	278	92	238	103	229	87	186	95
	45	251	76	220	85	207	73	173	80

[a] Growing conditions/management quality.
[b] Harvest regime of grass.
[c] MHG = mean highest groundwater table.

Table 9
Manure and mineral fertilizer N inputs permitted in individual crops to achieve nitrate targets at the level of individual crops (C) or at the farm level (F) when grassland (mixed use) and silage maize are grown in a 66.6%/33.3% rotation, as compared with the results of calculations where the N dynamics of mineralization and build-up associated with a rotation are ignored (I) (consult text for further assumptions)

Scenario	Scale	N (kg/ha per year)					Nitrate-N (mg/l water)	P_2O_5 surplus (kg/ha per year)
		Mineralized N	Manure-N	Mineral fertilizer-N	Immobilized N	N yield		
I	Grassland	0	275	171	0	324	11.3	0
	Maize	0	169	41	0	168	11.3	0
	Whole farm	0	240	129	0	272	11.3	0
C	Grassland	0	361	179	75	338	11.3	26
	Maize	150	0	32	0	148	11.3	−54
	Whole farm	50	242	131	50	275	11.3	0
F	Grassland	0	368	129	75	332	8.3	31
	Maize	150	0	94	0	173	17.5	−63
	Whole farm	50	247	117	50	280	11.3	0

3.3. *Adjustments of rates in a crop rotation*

Considerable adjustments of N inputs are necessary in individual crops, particularly when the nitrate target is to be achieved for each individual crop (case C). The allowable application rates of manure on maize in our initial approach (169 kg N/ha per year) should be reduced to nil. Conversely, the allowable application rate of manure on grassland could be increased from an initial annual rate of 275 kg manure N/ha to approximately 365 kg N/ha (Table 9).

4. Discussion

4.1. *How much manure from a nitrate perspective?*

The present study confirms that N concentrations in groundwater are not only determined by the inputs of manure but rather by the combination of manure and mineral fertilizer N. Moreover, on grassland with a 'cutting only regime', up to 330–340 kg manure N/ha can be applied annually without exceeding a concentration of 11.3 mg nitrate-N per litre, and without applying more P than the amount removed via crops. This is considerably more than 170 kg manure N/ha stipulated by the Nitrates Directive. A similar conclusion was drawn by Ten Berge et al. (2002). This phenomenon can be attributed to the large N uptake capacity of grass, to favourable growing conditions and to the long growing season (Peeters and Kopec, 1996), as well as to the empirical evidence that only a fraction of the soil N surplus is recovered in the upper 1 m of the groundwater, especially in situations with shallow groundwater. Theoretically, the latter can be explained by either denitrification or by temporary accumulation of organic N in the soil. Gradual changes of the amount of N in soil organic matter are hard to measure. Still we do not think that accumulation is a likely explanation because land use has not been recently changed towards grasslands, which favours accumulation, and because soil N surpluses have decreased during the last decade (Schröder et al., 2005a). This leaves denitrification as the most probable explanation of why only a fraction of the soil N surpluses ends up in groundwater. This apparent denitrification is much larger than what is commonly measured when using the acetylene inhibition technique. However, this technique underestimates denitrification, especially in wet soils where gas diffusion is hampered (Bollman and Conrad, 1997; Seitzinger et al., 1993). Moreover, in most studies denitrification is only measured in the top soil (Barton et al., 1999). The combination of shallow groundwater tables and the presence of fresh organic matter in the upper soil layers may create conditions favouring denitrification, resulting in relatively low nitrate-N concentrations in the upper groundwater (Fig. 2). Additional indications for the possible underestimation of denitrification is provided by farm balance calculations showing a considerable 'not accounted for' term, especially in grassland (Van der Meer, 1991; Garrett et al., 1992; Jarvis, 2000; Van der Salm et al., 2007). In line with these findings, Wachendorf et al. (2004) found a leaching fraction of 30–40% on grassland, which is comparable with our observation (39%) on a similar soil type.

The present study also shows that growing conditions and management, the harvest regime of grasslands and the share of crops other than grasslands (i.e. silage maize) in the rotation of dairy farms, all determine the permissible rates from the perspective of nitrate leaching and P accumulation. Even with mixed grassland use (i.e. half of the production harvested via grazing) and a maize share up to the common 30% of the farm area (http://statline.cbs.nl), the annual use of approximately 240 kg manure N/ha can be reconciled with a N concentration of 11.3 mg per litre and a P-surplus of 0 kg/ha when growing conditions are good and crops are well managed. Corresponding numbers for poorer growing conditions and management would be 190–220 kg manure N/ha (Table 8).

Our calculations, however, show a trade-off between the extent to which manure can be safely used and the room for grazing, because manure excreted during grazing results in greater leaching losses than mechanically applied manure ex storage (Vellinga et al., 2001; Nevens and Reheul, 2003; Wachendorf et al., 2004). Thus, if one is of the opinion that grazing positively contributes to landscape quality and animal welfare, trade-offs also exist between these features and the extent to which one can apply manure within targeted nitrate-N concentration.

4.2. How to avoid incorrect estimates of inputs?

4.2.1. Actual inputs of manure N and P

The applicability of our calculations to the general practice strongly relies on a precise determination of all relevant N inputs. This refers to, e.g. a correct assessment of the N and P excretion per animal category present on the farm, per production level and per type of diet (Kebreab et al., 2001). Estimates of excretion should be consistent with estimates and observations of the amounts of N and P removed in crops, milk and meat and inputs from feed (Tamminga et al., 2004). Subsequently, accurate estimates of the gaseous N losses from housing and manure storages are needed to assess how much manure-N will eventually be applied to the fields (e.g. Bussink and Oenema, 1998). Moreover, reliable accounts of the manure imported to or exported from the farm are necessary. Special attention is also required for the P–N ratio of manures, because for instance pig and poultry slurries as well as solid manures from cattle contain much more P per kg N than the cattle slurry used in the present study. Consequently, less manure N can be applied if P inputs into the soil have to be balanced with P outputs (Schröder, 2005). Conversely, the relative substitution of mineral fertilizer N by manure could increase without accumulating P, if the P to N ratios in crops became narrower through restricted N use, as illustrated in our sensitivity analysis. Hence, the composition of manures and crops, including concentrates, deserves constant monitoring.

We assumed that 25% of the total N application (i.e. 50% of the organic N input) in cattle slurry mineralizes after the first 12 months (Van Dijk et al., 2004) although recent insights point to 30–40% of the total N input (i.e. 66–75% of the organic N input) in slurry (Schröder et al., 2005b). However, this does not affect our conclusions because they were based on an equilibrium situation in which the long term fertilizer N equivalency of manures including their residual N effects is fully accounted for.

Note that our calculations are based on the assumption that the application of cattle slurry is associated with low ammonia volatilization losses as a result of Dutch law demanding injection or the immediate incorporation of manure. If this were not the case, more mineral fertilizer N would have to be applied to either maximize the yield or could have been applied without exceeding the permitted soil N surplus, be it at the expense of the air quality.

We emphasize that we adopted fixed values for several crop residue related characteristics, which result in negative soil N surpluses at low input levels. However, reduced input levels can have a negative feedback on the quantity of N invested in crop residues and SMN_{spring} and thus on the assumed contribution to available N. Reduced inputs can also affect the fertilizer equivalency of these organic N sources and, on a regional scale, will sooner or later indirectly affect the amounts of N deposited via the atmosphere.

4.3. N Input via biological fixation

N inputs via biological fixation need to be taken into account as well even though, at present, mixed stands of grass and white clover are relatively rare in the Netherlands. The area of grassland with white clover ranges between 50 and 100 thousand hectares, i.e. 5%–10% of the grassland area in the Netherlands, as suggested by seed sales (Corré, personal communication). If one assumes that approximately 50 kg N/ha are fixed per harvested ton dry matter of clover (Elgersma and Hassink, 1997), an input of 130–160 kg N/ha should be accounted for in a sward with a visual cover of, for instance, 30% (Schils, 2002). If the area of mixed stands increases, this input will deserve more attention as N losses could be higher than expected on the basis of manure and fertilizer inputs only.

4.4. N input from ploughed swards

Ploughing grassland leads to an enhanced mineralization of accumulated plant material and soil organic matter. The quantity and fate of mineralized N is related to the history of the old sward, to the time of ploughing, to the type of subsequent crop and to weather conditions. Adjustment of the applied rates of manure and fertilizer N to the N mineralization from ploughed grassland is a prerequisite to minimize N leaching. The present study suggested that application rates of manure on maize following grass should be reduced to nil, in agreement with the experimental results of Nevens and Reheul (2002). Conversely, the allowable application rate of manure on grassland following maize could be increased by almost 90 kg N/ha. Clearly, crop rotations involving the regular ploughing of grassland and the associated re-establishment of new leys, require considerable adjustments of N inputs. *Mutatis mutandis*, similar implications apply to situations where grassland is ploughed down and followed by new grassland (i.e. plain grassland renovation).

4.5. How to avoid incorrect estimates of outputs?

The applicability of our calculations does not only depend on a correct assessment of inputs (see Section 4.2), but also on the anticipated level of outputs. Output levels are determined by assumptions concerning the extent to which inputs are properly utilized by crops and net production potentials are exploited as much as possible. Unlike indicators based on a farm balance approach which were used in The Netherlands until recently, indicators based on just fertilizer and manure application standards do not contain explicit incentives to reduce nutrient inputs via feed imports. Hence, they do not automatically stimulate the production of home-grown crop outputs through optimized crop management. We have anticipated this shortcoming by running our calculations for sub optimal conditions as well (Table 7). Such suboptimal conditions may pertain to many aspects such as an incorrect timing of tillage, manuring, the establishment or destruction of swards and cover crops, and harvests including those via grazing. Proper attention should also be paid to growth factors other than N and P such as soil supplies of Ca, Mg and K, the physical soil fertility, crop protection, and appropriate drainage and irrigation strategies. If such 'best practices' are not enforced by law via incentives and fees, the message to farmers should at least be that high inputs can only be justified by high crop outputs.

4.6. Annotations to the relationships between soil N surpluses and nitrate-N concentrations

The leaching coefficients are derived from a national Monitoring Program and not based on a full mechanistic understanding of the underlying processes. The network consists of a population of farms that may gradually change in terms of hydrology, land use and input levels of N and P. Nevertheless, we have applied the coefficients to our supposedly steady state situations. As denitrification is promoted by the presence of nitrate and degradable carbon (Munch and Velthof, 2006), the leached fraction of the soil N surplus may not necessarily be the same at reduced input levels, changing future coefficients towards higher nitrate-N concentrations. This was confirmed by data in Schröder and van Keulen (1997). However, Van Beek et al. (2003) found that the leaching fraction is positively related to the soil N surplus which would reduce the fraction at lower input levels. These uncertainties added to the complex effects of climate change on crop performance and soil processes, may affect coefficients and thus leaching. Still, the validation of the relationship against an independent, recent data set representing farms with reduced inputs rates (Fig. 3), does provide us with some confidence.

4.7. Ecological targets may demand more

In our study we evaluated the room for manure and fertilizer use in view of a N concentration of 11.3 mg nitrate-N per litre, in agreement with the Nitrates Directive. Our study was limited to sandy soils and, hence, to the impact on groundwater. Sooner or later, groundwater may become surface water in downstream regions. In view of the ecological targets of these surface waters, the total N concentration is considered to be a more relevant indicator than just nitrate N. The share of non-nitrate-N (i.e. ammonium-N, dissolved organic N) in groundwater on sandy soils amounts to only 12–16% (Fraters et al., 1998; Fraters et al., 2002). However, ecological targets, to be defined in accordance with the EU Water Framework Directive (WFD; Anonymous, 2000), may require lower total N concentrations than 11.3 mg/l, let alone nitrate-N per litre (Camargo and Alonso, 2006). Moreover, N (and P) originating from the sub soil may contribute to the eutrophication of surface waters, as well as to emissions directly linked to the use of manure and fertilizers, as has been accounted for in our calculations. Therefore, our conclusions will not necessarily mean that nitrate leaching will comply with the WFD when fully implemented.

The present study suggests that denitrification plays an important role in the relationship between N input and N concentrations in groundwater. This deserves attention as denitrification is associated with nitrous oxide production, which is a very potent greenhouse gas. However, when designing policies and measures directed at global effects, it is sensible to evaluate impacts per litre milk rather than per hectare. From that perspective, extensification as such does not necessarily reduce the emission at the global scale (e.g. Schröder et al., 2004). So far, it is uncertain which measures dairy farmers should take for the benefit of the global climate (e.g. Velthof and Oenema, 1997). Final decisions on this issue may also affect the conclusions of the present study. An aspect of denitrification which is to be evaluated and addressed locally, however, pertains to the negative effect of denitrification on heavy metal and sulphate concentrations in deeper groundwater (Cremer et al., 2003), especially in calcareous soils rich in sulphides (i.e. through pyrite oxidation). Moreover, denitrification resulting from the oxidation of pyrite supplies and organic matter in deeper soil layers, has a finite character. Hence, extensive monitoring is required for timely adjustments, if only because the aforementioned chemical compounds need to be addressed to achieve compliance with the EU Groundwater Directive (Anonymous, 2006).

4.8. Conclusions

Cut grasslands grown on sandy soils in the Netherlands can utilize cattle manure up to rates of 330–340 kg manure N/ha (120 kg P_2O_5) per year without exceeding a target value of 11.3 mg nitrate-N per litre or accumulating P in the soil. It can be realized provided that (i) appropriate amounts of mineral fertilizer N are supplemented, and (ii) growing conditions are good and the grassland is well managed. Under similar conditions, cattle manure rates on grasslands with a mixed use of cutting and grazing, should be reduced to 275 kg manure-N/ha (100 kg P_2O_5) per year, including the N excreted during grazing. Rates on silage maize should not exceed 160–175 kg manure N/ha (=50–65 kg P_2O_5) per year, the lower values referring to sandy soils with groundwater tables deeper than 0.80 m. Suboptimal growing conditions and crop management reduce the scope for manure applications considerably. When grass and maize are grown in rotation, manure and fertilizer applications to maize should be reduced even more drastically, whereas application rates to grassland can be extended to compensate for the temporal N investments in the new sod.

Manure rates should therefore be determined by the share of both crops and the way they are positioned in a rotation, the hydrological situation, the harvest regime, growing conditions, and management quality.

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Appendix A

The MS Excel Solver Tool^{TM} is instructed to find the exact combination of manure-N (and associated manure-P) and mineral fertilizer-N, that maximizes the harvestable crop N yield (and associated P yield), while achieving a P soil surplus of exactly 0 kg and (a soil type and crop type specific) N surplus leading to a nitrate-N concentration in groundwater of 11.3 mg/l or less:

- Maximize:

$$\mathrm{NY} = \mathrm{MIN}\,(\mathrm{NY}_{max}; (\mathrm{Rna} \times (1 - L) \times \mathrm{NA}))$$

with NY is the harvestable N yield (kg N/ha), NY_{max} a constant defining the highest possible harvestable N yield (kg N/ha, Table 2), NA the plant available soil N (kg N/ha) $= \sum_1^i E_i N_i$ (with E_i is the fertilizer equivalency of input source i (kg N/kg N, Table 1) and N_i is the rate of input source i (kg N/ha)), Rna the recovery fraction of plant available soil N being a function of the amount of NA (kg N/kg N, following from Table 2) and L is a constant reflecting the fraction of recovered N which is not harvested (kg N/kg N, Table 3);

- Under the constraints:

$$\sum_1^i (N_i) - \mathrm{NY} < \text{allowable soil N surplus}$$

(kg N/ha, following from Table 4), and

$$N_{\mathrm{manure}} \frac{P_2O_5}{N_{\mathrm{manure}}} - \mathrm{NY}\frac{P_2O_5}{N_{\mathrm{crop}}} + P_2O_{5\,\mathrm{dep}} = 0\,(\mathrm{kg}\,P_2O_5/\mathrm{ha})$$

with N_{manure} is the rate of manure N (kg N/ha), $P_2O_5/N_{\mathrm{manure}}$ a constant reflecting the kg P_2O_5/kg manure-N, P_2O_5/N_{crop} a constant reflecting the kg P_2O_5/kg crop-N and $P_2O_{5\,\mathrm{dep}}$ is a constant reflecting the atmospheric P_2O_5 deposition (kg P_2O_5/ha).

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