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For quality of life

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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. Efficient utilization of these nutrients is, however, limited by crop demand for nitrogen (N) and phosphorus (P). Moreover, goals as implied by the EU Nitrates Directive impose constraints on the use of manure and mineral fertilizer N. The present study explores the limits to the use of manure and mineral fertilizer on dairy farms in the Netherlands via calculations based on experimental data from various sources. The study concludes that cut grassland can utilize cattle manure up to average rates of 330-340 kg N (120 kg P_2O_5) per ha per year without exceeding a target value of 11.3 mg N per litre in the upper groundwater (nitrate-N on sandy soils) and in drain or ditch water (total N on clay and peat soils) or accumulating P in the soil, provided that *i)* sufficient supplementary mineral fertilizer N is applied, and *ii)* growing conditions are good and the grassland is well-managed. When grassland has the common mixed use of cutting and grazing, manure rates have to be reduced by 60 kg N per ha per year (20 kg P_2O_5) to achieve environmental targets. Similarly, not more than 170-200 kg manure N (60-70 kg P_2O_5) per ha per year should be applied to silage maize. When grown on sandy soils with a groundwater level deeper than 0.80 meter, manure rates on maize land need a further reduction to 155 kg N per ha per year (= 55 kg P_2O_5). 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 share of grass and maize, and *iii)* the hydrological situation. The study also indicates that reductions of 45-60 kg and 30-35 kg manure-N per ha per year are required in grassland and maize, respectively, if growing conditions or cropping management are suboptimal.

Keywords: environment, dairy farming, leaching, management, manure, nitrogen, phosphorus.

1. Introduction

Sustainable crop production requires nitrogen (N) and phosphorus (P) inputs to compensate for the N and P removed from the system via either exported produce and losses (Jarvis, 1998; Carton & Jarvis, 2001). Losses pertain to the volatilization of ammonia N, mainly from animal excrements, and elementary N and N oxides resulting from (de)nitrification predominantly occurring in soils, and to the run-off and leaching of mineral and organic N and P compounds into groundwater and surface water. Losses into water bodies compromise the quality of drinking water and ecosystems (e.g. Tunney *et al.*, 1997; Rabalais, 2002). Consequently, N and P concentrations in water are considered indicators of environmental quality, as reflected in legislation of the European Union (EU) (Anonymous, 1991a; -, 2000; -, 2003).

Concerning N, pedoclimatic conditions together with the area-weighted land use at watershed level, determine the magnitude of losses and the eventual N concentration in groundwater and surface water at a regional scale (e.g. Schröder *et al.*, 2004). As farming represents the most important form of land use in the Netherlands, the management of N within farms is a major determinant of losses and the consequential environmental quality. As for fields and crops, farm N management refers to crop choice, husbandry and rotation and to the nature, the rate, the timing and the placement of manures and fertilizers.

In view of the negative relationship between animal density and water quality on a European scale, a precautionary application threshold of 170 kg manure-N per ha per year has been established in the Nitrates Directive of the EU (Anonymous, 1991a). Manure application rates, however, are to a certain extent at most indicative of environmental quality and not necessarily effective as environmental effects are determined by all inputs and outputs together rather than just by the input of manure (Schröder *et al.*, 2004). Undeniably, farming in general and livestock farming in particular, is associated with the risk of nutrient losses not in the least because manures are inherently difficult to manage (Schröder, 2005). Dairy farming dominates the land use in the Netherlands, in particular in regions where the soil type is conducive to losses. Therefore, the contribution of dairy farming to water quality needs special scrutiny.

The aim of this paper is *i)* to give an overview of N management of forage crops on dairy farms in the Netherlands in view of institutional and pedoclimatic conditions, *ii)* to establish relationships between inputs i.e. combinations of fertilizer N and cattle manure N, N outputs and the soil N surplus, and *iii)* to estimate the N concentration of groundwater and surface water resulting from the soil N surplus.

2. Some features of dairy farming in the Netherlands

In the Netherlands about 70% (2.0 million ha) of the land area is used by agriculture. This area gradually decreases over time, due to the conversion of agricultural land to land for infrastructure, housing and nature reserves. Agricultural land prices increased to an average level of 35,000 Euro per ha, much higher than in most other European countries.
Sand, clay and peat soils cover about 50, 40 and 10% of the land area, respectively. At present, about 50% of the agricultural area is grassland, 10% silage maize, 32% arable land and 6% is used for horticulture. Grass and maize are mainly used to feed dairy cattle. Of the grassland about 90% is permanent, the remaining 10% is temporary grassland. Other forage crops are hardly grown in the Netherlands (Aarts, 2003).
The total number of farms in 2002 was approximately 89,600. The number declines by an annual 3% and the remaining farms expand in size. About 90% of farms comprise specialised farms. The dominant farm type is the grazing livestock farm in which approximately half of the grassland production is harvested via grazing, with 44,400 farms accounting for 50% of the total farm number (Table 1). About 25,000 livestock farms are specialised in dairy farming, with a size that permits them to gain an acceptable income from dairying only. Pig and poultry farms, often with little land (on average 7.5 ha), are concentrated on sandy soils in the southern and eastern provinces, whereas most arable farms are located on clay soils in the north and west. The clustering of pig and poultry farming in the south and east and that of arable farming in the north and west results in a poor spatial match between supply of and demand for manure. Consequently, large quantities of manure have to be transported from the south and east to the north and west, to be applied in the arable sector. Dairy farming, however, is relatively evenly distributed over the country.

Table 1. Numbers of farm types in 2002 (CBS 2004; source: www.statline.nl).

Farm type	Number	% of total
Grazing livestock	44,376	50
Arable	12,756	14
Horticulture	16,554	18
Pigs and poultry	7,198	8
Mixed	8,696	10
Total	89,580	100

Table 2. Characteristics of specialized commercial dairy farms (average 1998-2002; after Aarts et al., 2005).

	All	Clay	Peat	Sand (wet*)	Sand (dry*)_
Cultivated area (ha per farm)	44	46	49	41	37
grassland (ha per farm)	36	39	44	32	27
maize (ha per farm)	8	7	5	9	10
Milk production/ha (kg per ha)	13053	12901	12293	13082	14197
Dairy cows (number per farm)	74	79	82	69	68
Milk production/cow (kg per cow)	7470	7461	7411	7495	7502
Young stock (number per farm)	57	60	59	54	56
Facility to irrigate (% of farms)	37	36	12	38	62
concentrates (kg per cow)	2236	2233	2358	2193	2197
Mineral-N on grassland (kg per ha)	210	221	192	205	219
Slurry-N on grassland (kg per ha)	207	199	194	207	238
Mineral-N on maize (kg per ha)	47	61	32	43	45
Slurry-N on maize (kg per ha)	199	180	123	233	249
Net dry matter yield grassland (kg per ha)	10419	10322	9569	10443	11503
Net dry matter yield maize (kg per ha)	12561	12878	12892	12243	12605

* *in wet sandy soils preponderance of soils with MHG levels above 0.40 to 0.80 meter below the soil surface, in dry sandy soils preponderance of soils with MHG levels below 0.40 to 0.80 meter below the soil surface*

The great majority of the dairy farms are highly specialised, accounting for 93% of dairy cows held in the Netherlands. More than half (53%) are located on sandy soils, 24% on clay soils and 22% on peat soils. Dairy farms on sandy soils, i.e. in the south and east of the country, are generally more intensive (in terms of litres milk output/ha and fertilizer and feedstuff input/ha) and smaller than farms on clay or peat soils in the north and west (Figure 1). The main reasons for this regional difference are to' days land prices and the historic origin of farms.

The main characteristics of dairy farms are given in Table 2. The size of an average dairy farm in the Netherlands is about 40-50 ha, the size of farms on peat and clay soils generally being slightly larger than that of farms on sandy soils. The main inputs are concentrates and mineral fertilizer. Milk production per ha feed crops in the peat areas is approximately 1-2 tons lower than in sand regions. The dominant land use on all dairy farms is grassland. If possible, a part of the land is used to grow silage maize, particularly on sandy soils where approximately 25% (range: 20-50%) of the agricultural land area is devoted to maize production. Over the last decade the share of maize has changed little (Figure 2).

In general, farmers in the Netherlands are well educated and professional skills are high. Besides, feed supplying, health care and milk processing are well organized, mainly in a cooperative way. Most of the field labour (slurry application, growing and harvesting of maize, silage making) is carried out by contractors permitting farmers to focus their attention on farm management in general and herd performance in particular.

Dairy farming in the Netherlands is undeniably intensive in terms of both nutrient inputs and produce outputs per unit area. However, nutrient losses to the environment are determined by the product of inputs and their use efficiency rather than by inputs alone (e.g. Schröder *et al.*, 2003). It is definitively very demanding for entrepreneurs to reconcile production intensity and environmental quality, but not impossible, provided that growing conditions are favourable. Considerable improvements have been recently made in terms environmental impact, as illustrated by the decreasing N surpluses of dairy farms on sandy soils (Figure 3).

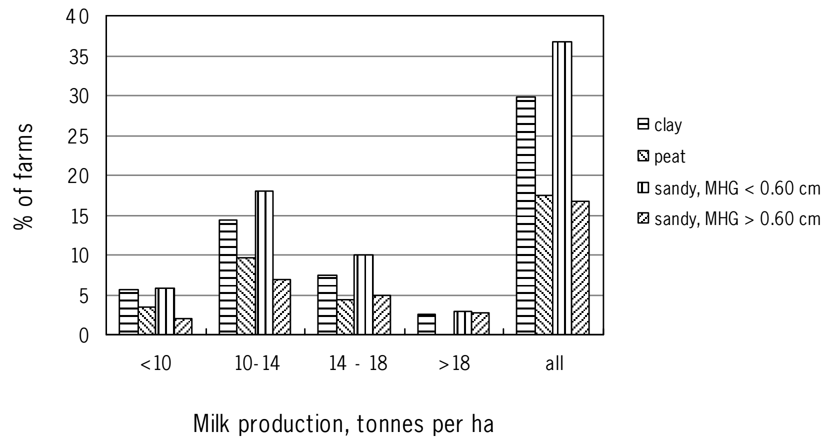

Figure 1. *The distribution of specialised commercial dairy farms over soil types and milk production classes in 1998-2002 (after Aarts et al., 2005).*

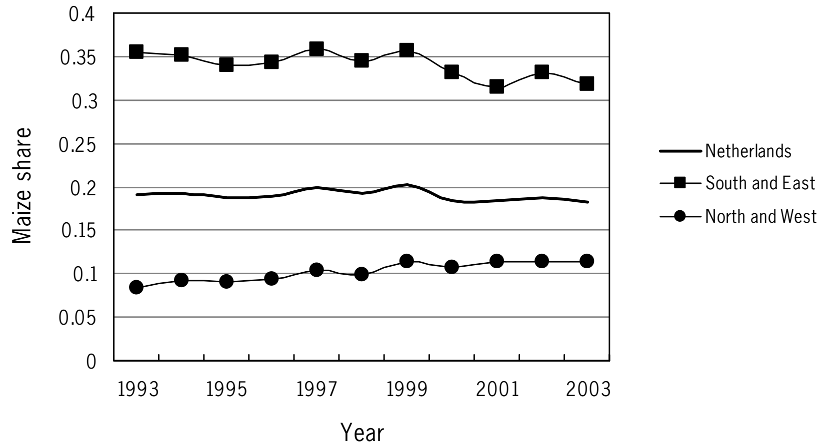

Figure 2. *The share of maize in the total area of grassland and silage maize during the last decade (Source: www.statline.nl, CBS, The Hague).*

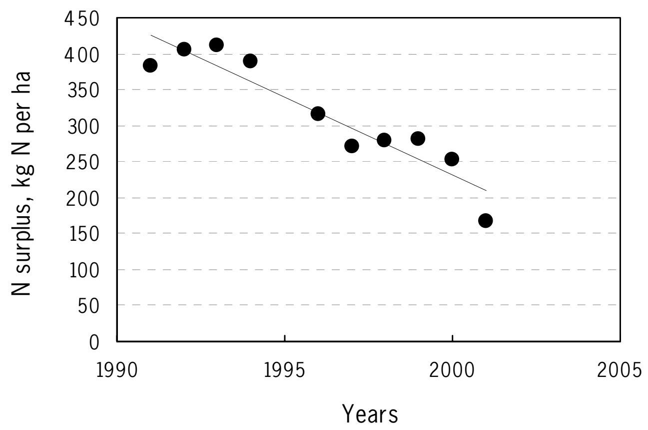

Figure 3. *Temporal trend of the N surplus of dairy farms on sandy soils (Source: LEI/RIVM The Hague/Bilthoven).*

3. Soil and climate

3.1 Climate

The average temperature in the Netherlands is 9.6 °C, with average monthly temperatures ranging from 2.8 °C in January to 17.2 °C in July (Figure 4). However, there are considerable differences between years. The average total annual precipitation in the Netherlands was 770 mm in the period 1971–2000. There are slight differences between months in average precipitation, but extremely high (> 150 mm) or low (< 5 mm) precipitation can occur during the whole year, as illustrated in Figure 5 for the Bilt (centrally located in the Netherlands).

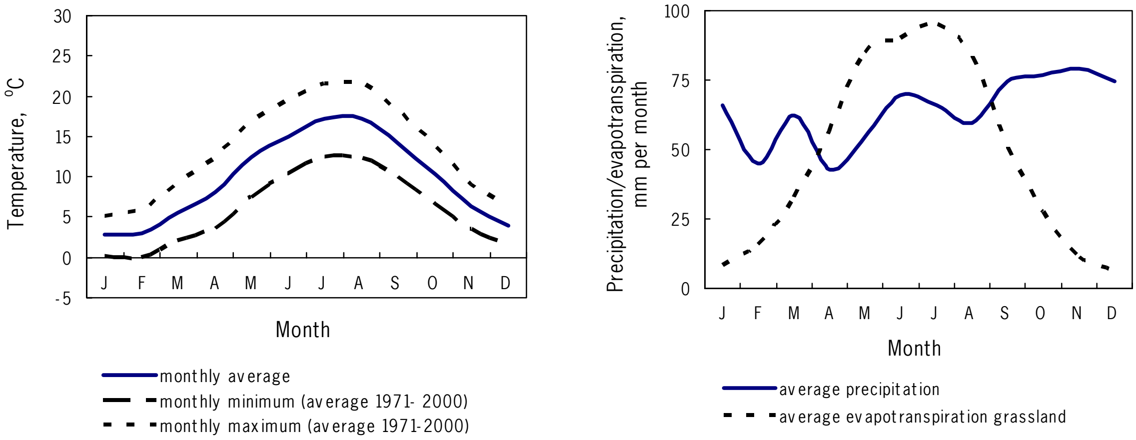

Figure 4. Average monthly temperature, precipitation and potential evapotranspiration (Makkink) in the Netherlands in the period 1971 – 2000 at 15 weather stations (Source: KNMI).

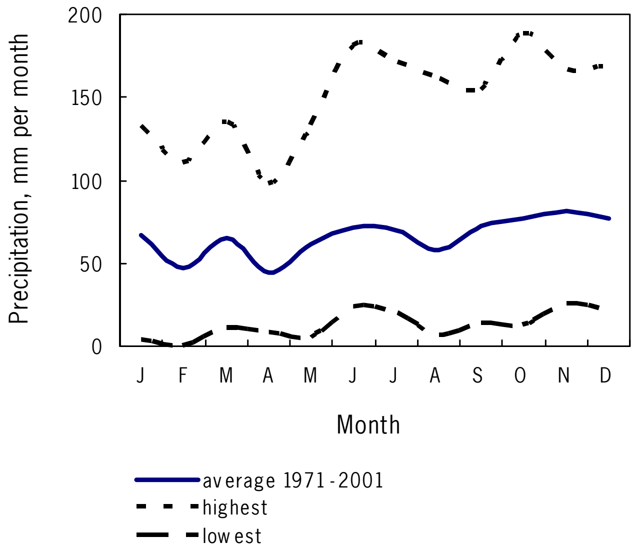

Figure 5. Average monthly precipitation and extreme values recorded at weather station De Bilt in the period 1971 – 2000 (Source: KNMI).

3.2 Soils and hydrology

Grassland covers approximately 50% of the total agricultural area in the Netherlands and is evenly distributed (Figure 6) on the different soil types that are present in the Netherlands (Figure 7). Approximately 44% of the total grassland area is situated on sandy soils, mainly in the south and east. Another 39% can be found on clay and loam soils. The remaining 17% is found on the peat soils in the North- and Midwest. Maize land is mainly found on sandy soils in the south and east.

Groundwater in the Netherlands usually occurs within 1 to 2 m from the surface. Groundwater depth and regime (average fluctuation within a year) in the clay and peat regions in the north and west are mainly determined by the polder water level in combination with soil physical characteristics. In the clay regions soils are usually drained with tile drains (depth about 1 m) and ditches. Peat soils are usually drained by gullies and ditches. Groundwater depth in clay and peat soil usually fluctuates within 1 m below surface level.

Sandy soils are mostly free draining (natural drainage) in combination with ditches. In the sand regions in the east and south, there are large differences in groundwater depth. On average, groundwater level is at 1.5 m below the soil surface level, but in the loess region and some elevated areas in the eastern sandy part of the Netherlands groundwater levels well below 5 m below the surface level occur. These elevated sandy areas are mainly occupied by forests and other (semi) natural vegetations.

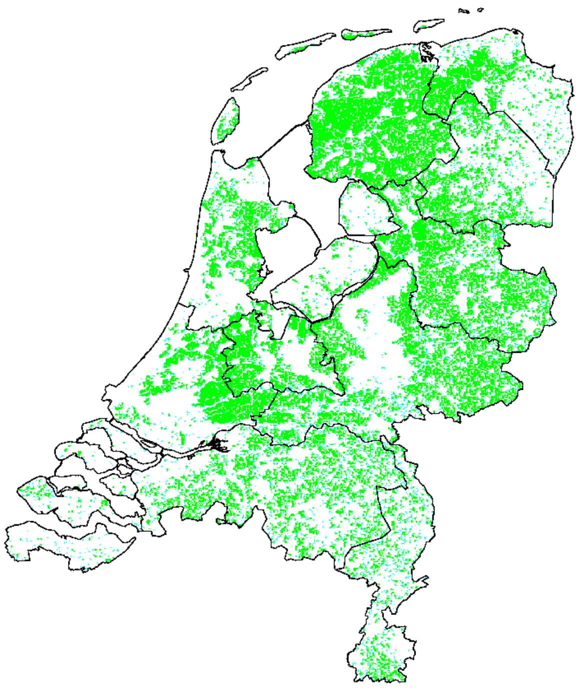

Figure 6. Grassland areas (in green) in the Netherlands (Source: Alterra).

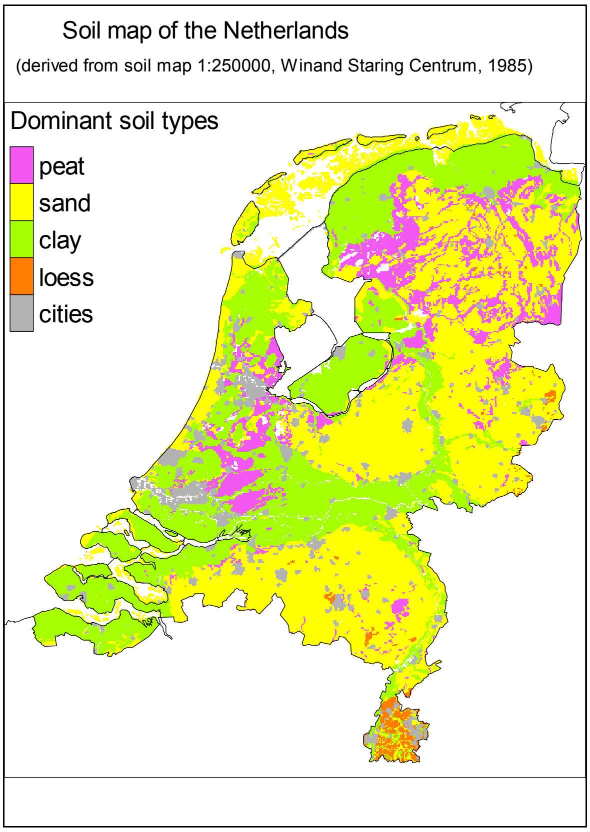

Figure 7. Soil map of the Netherlands with dominant soil types (Source: Alterra, Wageningen).

3.3 Effects on yields and N processes

In the period from about April to September, evapotranspiration generally exceeds precipitation, but this may differ between years and between crops. In this period, the water supply of a crop is strongly dependent on soil properties and irrigation. The depth of groundwater and rooting zone, and the soil structure are important factors controlling water supply during periods with low precipitation. A limited water supply during the growing season can reduce yields on sandy soils with a relatively deep groundwater level. Without irrigation, the N off take will be limited on these soils, which would increase risk of N losses if manure and fertilizer N applications are not adjusted to the reduced growth rate. In these sandy areas, however, most farmers irrigate grassland. Equipment for irrigation is available on 37% of all dairy farms and on 62% of the dairy farms on dry sandy soils (Aarts *et al.*, 2005). In peat and clay soils and wet sandy soils, problems with the water supply of the crop are uncommon.

In the period from October to March, the precipitation surplus causes a downwards transport of water through the soil, which may result in N leaching when there is an excess of mineral nitrogen (mainly nitrate) in the soil. Soil moisture content increases in this period and the groundwater level rises. The soil becomes wetter, which promotes denitrification.

In the Netherlands a special national Monitoring Program for the effectiveness of the Minerals Policy is carried out. It aims at sampling on-farm waters e.g. the upper meter of groundwater (sand), tile drain water (clay) and ditch water (peat). The results of the Monitoring Program for the 2000-2002 period show that the average nitrate-N concentration in the upper groundwater under agricultural land on sandy soils is still too high. Concentrations decrease in the order sandy soils (average 17 mg nitrate-N/l) > clay soils (average 9 mg/l) > peat soils (1 mg/l) (Anonymous, 2004a, b; Fraters *et al.*, 2004). Peat and clay soils are wet in winter which promotes losses via denitrification. The high organic matter contents in peat soils also stimulate denitrification (see also the discussion section of this paper).

4. Interactions between inputs, outputs, surpluses and water quality

4.1 Introduction

Water quality under and along agricultural land is, among other factors, determined by the discrepancy between N and P inputs into and outputs 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 hence crucial how inputs and outputs are defined and assessed.

Figure 8 shows that i) without N applications, crops remove considerable amounts of N ('the intercept'), ii) applied N is generally not fully converted into harvestable N ('the slope < 45°'), and iii) from a certain N rate the N uptake ceases to increase ('the plateau'). The size of the intercept is determined by the indigenous soil N supply consisting of soil mineral N at the onset of the season (SMN_{spring}), biologically fixed N in mixed stands, N from crop residues, residual N mineralized from manures applied in previous years, N mineralized from peat and N contributions from the atmosphere, and by the use efficiency of these sources, i.e. the (initial) slope of the response curve. The slope is determined by the fertilizer equivalency of the various N sources resulting from the nature of the N source and the time and place of their application, by the extent to which the root system of a crop can absorb soil mineral N and by the extent to which absorbed N is invested in harvested plant parts. The plateau is determined by the factors that limit yield potential such as radiation, temperature, moisture, genotype and harvesting regime.

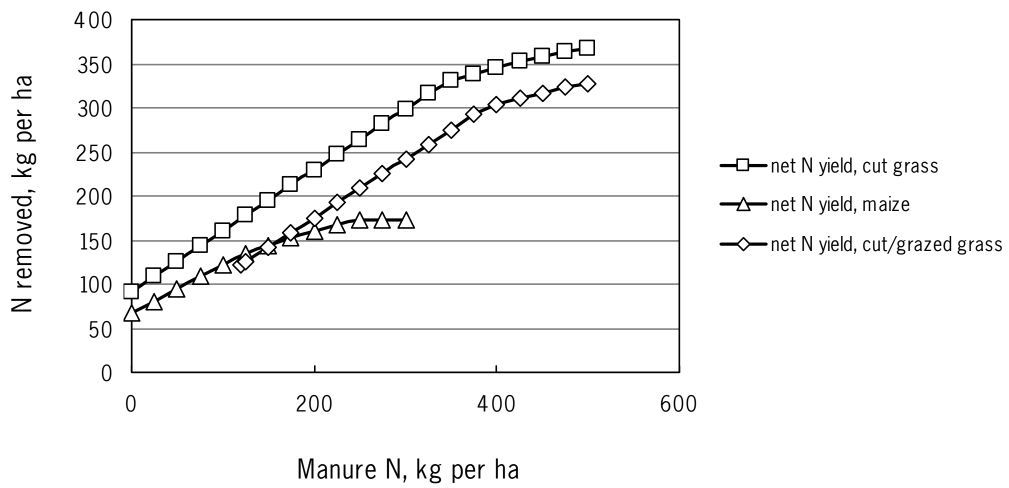

Figure 8. Annual net N removal of cut grassland, grassland with a mixed use of grazing and cutting and silage maize on sandy soils, in combination with good growing conditions and management, as related to the total N input of manure (see text for assumptions).

4.2 Input

We define the N input here as the sum of manure-N (so, after subtraction of the ammonia losses from housing and storage), mineral fertilizer N, SMN_{spring}, deposited atmospheric N and N mineralized from soil organic matter. Sources of mineralization are peat, crop residues (including roots, stubbles, harvest losses and winter cover crops), and manure applied in previous years. Biologically fixed N is omitted as an input term here as clover is hardly present in Dutch grassland. In the discussion section we address situations in which it is.

As for the inputs we assume here a SMN_{spring} of 30 kg N per ha (Schröder *et al.*, 1998), annual atmospheric depositions of 31 kg N per ha (Anonymous, 2004c), and a N mineralization exceeding N immobilization on peat soils by 160 kg N per ha per year, due to the inevitable oxidation of soil organic matter once these soils are drained (Van Kekem, 2004). We estimate that on an annual basis 75 kg N per ha mineralize from grass roots and stubbles (Velthof & Oenema, 2001), 25 kg N per ha from maize roots and stubbles (Schröder, 1991) and 40 kg N per 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, we account for the N mineralization from manure inputs in previous years. Our calculations are restricted to slurry which is by far the dominant manure type in the Netherlands (Anonymous, 1995). The long term residual N mineralization from cattle slurry (i.e. beyond the first 12 months after its application) amounts to 25% of the total manure input (Lammers, 1983). We balance this mineralization via a similar annual investment into the soil organic N pool.

4.3 Output

To assess the soil N surplus several outputs must be subtracted from the inputs. We define the N output as the sum of the crop N which is either grazed or removed via harvests (see below), N investments in (new) crop residues, and N stored in the organic N fraction of manure in as much as this organic N is not yet 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 yearly N input from mineralizing crop residues and formerly applied manure, more or less equals the N output into these pools. In other words: to sustain a system several inputs need annual renewal. This may not hold on an annual basis in regularly renovated grassland or in each individual field when crops are rotated. In the discussion section we address this situation in more detail.
Ammonia losses, too, need to 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; -, 1994; Huijsmans, 1999). 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 & Oenema, 2001).
The crop N output is determined by *i)* the fertilizer equivalency i.e. the availability of N from various sources relative to mineral fertilizer N (Table 3, 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 4, 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 5, including references).
Throughout this study we assume that manure ex storage is applied in spring (grassland, maize) and early summer (grassland) and injected. Grassland is irrigated whenever necessary and maize is followed by a frost resistant cover crop.

Table 3. 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*	64%	60%	Lammers, 1983; Huijsmans, 1999; Schröder, 2005; Van Dijk *et al.*, 2004
Excreted urine and dung during grazing*	16%**	-	Vellinga *et al.*, 2001; Van Dijk *et al.*, 2004
Atmospheric deposition	75%	75%	After Schröder & Van Keulen, 1997
Mineralization of soil N, including crop residues and the resistant organic N fraction of manure	75%	60%	Lammers, 1983; after Schröder & Van Keulen, 1997

* *in the first 12 months after application/excretion*
** *i.e. 25% of the fertilizer-N equivalency of mechanically applied cattle slurry*

Table 4. N uptake efficiency of fertilizer N equivalents for grass and maize (derived from: Alberda (1968); Prins (1980); Sibma & Ennik (1988); Middelkoop & Aarts (1991); Schröder et al. (1998); Vellinga & André (1999); Ten Berge et al. (2002); Nevens & Reheul (2002); Nevens & Reheul (2003a); Nevens (2003); Schils & Kok (2003)).

	Grass	Maize
Initial efficiency at low input rates	85%	75%
Indicative mineral fertilizer N rate (kg N per 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 per ha) at which marginal N efficiency becomes 0%	510 and 460 for fully cut swards and swards with mixed use, respectively	330, 190 and 130 for clay, sandy soils with MHG < 0.80 and sandy soils with MHG > 0.80 m, respectively

Table 5. Fraction (%) of crop not taken in or exported from the field due to mechanical damage to sward by animals and due to lost crop material during wilting and mechanical harvesting (Beuving et al., 1989; Corporaal, 1993).

Regime	Soil type	Grass	Maize
Cutting only	All	4%	5%
Mixed use of cutting and grazing	Peat	20%	-
	Clay	15%	-
	Sand with MHG < 0.80 m	15%	-
	Sand with MHG > 0.80 m	10%	-

4.4 Fate of the soil N surplus

In the previous sections it was explained how the N soil surplus in a steady state situation is calculated. In equilibrium situations where soil organic N pools are depleted nor augmented, the soil N surplus is either denitrified or leached. We calculated the effect of the N soil surplus on N concentrations in groundwater and surface water by multiplying the surplus by crop and soil specific factors i.e. leaching fractions (Table 6). These factors were derived from a current national Monitoring Program established in the early nineties of the last century. In this network, soil N surpluses of farms are linked to corresponding nitrate-N concentrations in the upper 1.00 meter groundwater under sandy soils or to corresponding total N concentrations drained to surface water into ditches along the fields on clay and peat soils.

Table 6. Net leaching fractions (kg N leached per kg soil N surplus; s.d.'s based on yearly variation in brackets) and the and soil N surplus (kg N per ha; in brackets the 95% probability interval as derived from yearly variation of the leaching fractions) associated with a N concentration of 11.3 mg nitrate-N (sandy soils) or total N (clay and peat soils) per litre, as affected by land use, soil type and mean highest groundwater level (MHG) (National Monitoring Program 1992-2001; Sources: Van Drecht & Scheper, 1998; Meinardi & Schotten, 1999; Fraters et al., 1998; -, 2001; -, 2002; -, 2004).

Land use	Soil type	Net leaching fraction (kg/kg)		Precipitation surplus (mm)	Allowable soil N surplus (kg N/ha)	
Arable land	Clay	0.31	(0.06)	387	141	(102-230)
	Sand, MHG < 0.40 meter	0.50	(0.08)	387	88	(66-132)
	Sand, 0.80 < MHG < 0.40 meter	0.75	(0.09)	434	65	(52-87)
	Sand, MHG > 0.80 meter	1.06	(0.08)	453	48	(42-57)
Grassland	Peat	0.04	(0.01)	242	>300	
	Clay	0.11	(0.05)	266	273	(143- >300)
	Sand, MHG < 0.40 meter	0.18	(0.04)	268	165	(115-290)
	Sand, 0.80 < MHG < 0.40 meter	0.28	(0.05)	329	134	(98-212)
	Sand, MHG > 0.80 meter	0.39	(0.06)	355	103	(79-148)

In view of ecological targets in surface water (mainly to be found in areas with clay and peat soils) the total N concentration is considered more relevant an indicator than just nitrate N. Moreover, the share of non-nitrate-N (i.e. ammonium-N, dissolved organic N) in water increases from 12-16% on sandy soils to 15-19% on clay soils and 80-100% on peat soils (Fraters *et al.*, 1998; -, 2001; -, 2002; -, 2004).
Groundwater quality of farms on sandy soils is measured once every year in the period March-September. On most farms 16 samples are taken, whereas on semi-experimental pilot farms (e.g. the ones on which Figure 10 is based) 48 samples are taken. The whole farm area is sampled, taking account of the grassland : maize ratio. On clay farms the drain water from 16 drains per farm is sampled with a maximum of four times per drainage season (October-April). Water discharge from the sampled drains is measured at the same time. Water quality of farms on peat soils refers to ditches which are sampled in the winter season. The number of samples depends on drainage density. Sampling is restricted to ditches which originate from fields belonging to the farm itself in order to minimize influence of adjacent farms and other nutrient sources such as waste water.
In the 1992-2002 period, the national network consisted of 24 (0-35) arable farms, 40 (12-75) dairy farms and 10 (0-24) mixed farms and/or pig/poultry farms each year. Of the monitored arable farms approximately 55% and 45% are located on clay and sandy soils, respectively. Of the dairy farms approximately 12%, 24% and 64% are located on peat, clay and sandy soils, respectively. Other farms are almost all (about 90%) located on sandy soils.

Table 7. N-surplus and nitrate-N concentration in groundwater (sandy soils), drain water (clay soils) and ditch water (peat soils) of dairy and arable farms, as observed on farms participating in the National Monitoring Program, sampled in the 1992-2002 period (yearly average (aver.), minimum (min), maximum (max) and standard deviation (s.d.).

	N-surplus (kg/ha)	Nitrate-N (mg/l)*
Dairy farms – sand	aver. 315 (min-max: 168-411; s.d. 80)	aver. 26 (min-max: 10-45; s.d. 13)
Dairy farms – clay	aver. 332 (min-max: 273-423; s.d. 50)	aver. 12 (min-max: 3-27; s.d. 9)
Dairy farms – peat	aver. 316 (min-max: 267-407; s.d. 62)	aver. 4.5 (min-max:3.5-5.7;s.d. 0.9)
Arable farms – sand	aver. 173 (min-max: 139-201; s.d. 23)	aver. 21 (min-max: 9-33; s.d. 8)
Arable farms – clay	aver. 167 (min-max: 107-201; s.d. 34)	aver. 13 (min-max: 8-19; s.d. 4)

* *for dairy farms on peat soil total-N instead of nitrate N is given*

The yearly average N-surplus and nitrate-N concentration of dairy and arable farms in the period 1992-2002 is given in Table 7. The N-surpluses of dairy farms and of arable farms were 315-332 and 167-173 kg per ha, respectively. Maximum surpluses of dairy farms were twice the value of arable farms. The nitrate-N concentration in groundwater of farms on sandy soils is generally higher than in drain water of farms on clay soils (mean values of 21-26 and 12-13 mg per litre on sandy soils and clay soils, respectively. The yearly average nitrate-N concentration shows a minor difference between dairy farms and arable farms. For dairy farms on peat soil the total N concentration in surface water was on average 4.5 mg per litre. Nitrate-N concentrations averaging 0.8 mg per litre are low on these peat soils.

According to the measurements collected in the Monitoring Program 11% (number of years = 6, s.d. 5%) and 31% (number of years = 6, s.d. 6%) of the soil N surplus on clay soils leaches from drains into ditches from grassland and from arable land, respectively. For grassland on peat soils this leaching fraction amounts to 4% (number of years = 4, s.d. 1%). The leaching fractions on sandy soils with a deep ground water (Mean Highest Groundwater level (MHG) deeper than 0.80 meter) are higher. On grassland 39% leaches to the groundwater (number of years = 10, s.d. 6%). This is in agreement with results of Wachendorf *et al.* (2004) who found a leaching fraction of 30-40% on a similar soil type. 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%).

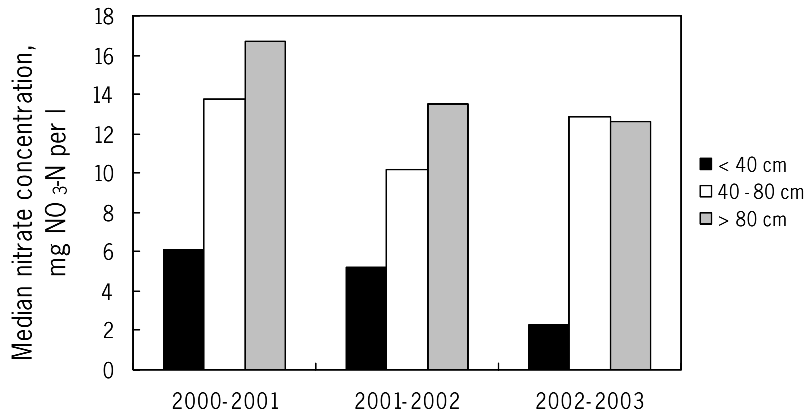

Figure 9. Median nitrate-N concentrations in the upper 1 meter groundwater measured on farms on sandy soils in the period 2000-2003 at three classes of groundwater table, i.e. shallow groundwater level (MHG < 0.40 m, n=70-72), groundwater at intermediate level (0.80 < MHG < 0.40 m, n=183-185), and deep groundwater level (MHG > 0.80 m, n= 212-217). (Source: Velthof, 2004)

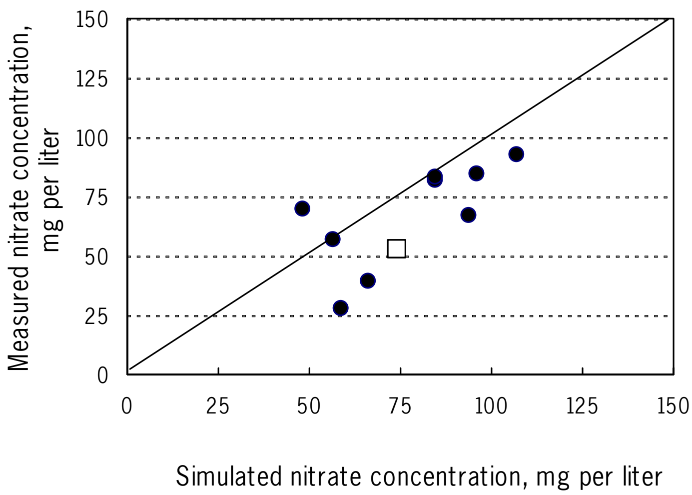

Figure 10. Observed (average 2000-2002) vs. simulated (cf. Table 6) nitrate concentrations in the upper 1 meter groundwater of dairy farms on sandy soils participating in the Cow & Opportunities project (De Marke indicated by □).

In sandy soils where the upper 1 meter of the groundwater is sampled, the observed nitrate-N concentrations are lower, the higher the groundwater level. This reduction of concentration ranges from 0% on sandy soils with 'deep' groundwater (MHG > 0.80 meter deep) to 57% (s.d. 7%) on sandy soils with shallow groundwater (MHG < 0.40 meter deep). These figures are based on research carried out in the years 1982-1991 on field level (Van der Meer, 1991) on farm level (Boumans *et al.*, 1989; Breeuwsma *et al.*, 1991), as well as on experiments with lysimeters (Steenvoorden, 1988). The quantitative implications of these studies showed a great similarity (Willems *et al.*, 2000) and all lead to the conclusion that the amount of nitrate recovered in the shallow groundwater was negatively related to groundwater levels (see Figure 9 for recent monitoring results). This phenomenon has been attributed to denitrification in the layer between the root zone and 1 meter 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 reactive carbon, by which denitrification is enhanced. In the discussion section we address this matter in greater detail.
As for sandy soils, the coefficients of Table 6 could be tested against a recent independent dataset of pilot farms on sandy soils participating in the Cows & Opportunities project (Oenema *et al.*, 2001). These farms achieved lower N surpluses than the farms used to derive the coefficients presented in Table 6. Application of the coefficients to these pilot farms resulted in a slight overestimation of nitrate-N concentrations compared to the observed concentrations (Figure 10). It is beyond the scope of this paper to further examine this moderate discrepancy in detail but it is possible that rainfall in the specific locations and years was slightly higher than the (30 year mean) default rainfall numbers in the model. On balance, we conclude that the validation shows a reasonable fit of simulated and observed nitrate-N concentrations on a whole farm level.

4.5 Phosphorus balance

An integrated approach towards nutrient emission from agriculture also requires attention to the phosphorus balance, as manure P is added with manure N. For that purpose we have adopted a simple soil surface P-balance. The P input is the sum of atmospheric P (2 kg P_2O_5 per ha per year according to Anonymous (2004c) but set here at 1 kg P_2O_5 (= 0.43 kg P) as recent, methodologically more adequate measurements point at values < 1 kg) and excreted or applied manure-P which is deduced from the manure-N input. For manure, we have used a fixed P_2O_5-N ratio in cattle 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; Schröder & Ehlert, 1998 Tamminga *et al.*, 2004).

We did not make an attempt to calculate P concentrations in water because that would require 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. Barraclough *et al.*, 1983; Deenen, 1994, Garret *et al.*, 1992; Aarts *et al.*, 2001), the situation for P is more complicated. The P status of the soil and the aeration of the soils are the dominant factors determining the P concentration in water that drains to the surface water (Schoumans & Groenendijk, 2000; Schoumans & Chardon, 2003). So, the relation between the inputs, surpluses and the concentration in water is less direct for P than for N.

4.6 Case study: grassland on a sandy soil

We have included a numerical example of the relationships between inputs, outputs, the soil surplus and N concentration in groundwater, to show how the previous reasoning works. This illustration refers to well managed grassland with a mixed use of cutting and grazing on a sandy soil with a MHG at depths between 0.80 and 0.40 meter (Table 8).

Table 8. Calculation of the N concentration in groundwater under grassland with a mixed use of cutting and grazing on sandy soils with a mean highest groundwater depth between 0.40 and 0.80 meter in combination with good growing conditions and management (see text for explanations and assumptions), as related to the inputs and outputs at the field level.

				Total N	Fertilizer N equivalents ***	Total P_2O_5
Inputs	Manure		kg/ha	275	118	99
	Fertilizer		kg/ha	171	171	
	Clover		kg/ha	0		
	Deposition		kg/ha	31	23	1
	SMN_{spring}		kg/ha	30	30	
	Mineralization	Roots	kg/ha	75	56	
		Harvest losses	kg/ha	46	35	
		Manure*	kg/ha	69	52	
		Cover crop	kg/ha	0		
	TOTAL		kg/ha	*697*	*485*	*100*
Outputs	Crop		kg/ha	324		100
	Ammonia		kg/ha	19		
	Investments	SMN_{spring}	kg/ha	30		
		Roots	kg/ha	75		
		Harvest losses	kg/ha	46		
		Manure**	kg/ha	69		
		Cover crop	kg/ha	0		
	TOTAL		kg/ha	*563*		*100*
Soil surplus			kg/ha	134		0
Leaching fraction			kg/kg	0.28		
Precipitation surplus			mm	329		
Nitrate-N concentration			mg/l	11.3		

* *N mineralized from manure applied in previous years (residual N)*
** *residual manure N invested (residual N)*
*** *see Table 3*

5. Exploration of environmentally safe fertilizer-manure combinations

5.1 Configurations

Following the above methodology, we first evaluated which combinations of manure-N and mineral fertilizer N would generate a N concentration of 11.3 mg nitrate-N (sandy soils) or 11.3 total N (clay and peat soils) per litre (or lower rates when no further yield increase was brought about by additional N inputs). Explorations were made using the MS Excel ™ Solver Tool. The evaluations are presented in the form of isoquants. An isoquant is a line that mirrors that a similar effect (in this case a N concentration of at most 11.3 mg per litre) can be brought about by different combinations of more than one factor (in this case manure-N and mineral fertilizer-N).

We established these isoquants for grassland with a 'cutting only' regime, for grassland with a mixed use (i.e. in which half the production is harvested via grazing), and for silage maize. As maize is rarely grown on peat soils (Table 2), calculations for maize were restricted to clay and sandy soils. As for the mixed use of grassland, we assumed that, on average, 120 kg N per ha per year is excreted outdoors as urine and dung (Tamminga *et al.*, 2004). This excretion can vary from farm to farm, being a function of milk quota, the N content of diets, the maize-grass ratio and the grazing regime. However, effects (e.g. high milk production per ha, but limited number of hours daily grazing) are assumed to balance each other (Aarts *et al.*, 2005).

First, we tabulated which manure rate would achieve a N concentration of 11.3 mg N per litre, regardless the great potential P accumulation associated with this strategy. This kind of scenarios yield such large annual application rates of manure that the likely sward damage would make our surmised uptake efficiencies questionable. The results should hence be seen as a theoretical approximation, demonstrating that manure application rates are limited by allowable P surpluses rather than by just N concentrations in water. Therefore, we subsequently tabulated which combinations of manure and fertilizer N would achieve a N concentration of 11.3 mg N per litre without accumulating any P.

Both previous scenarios were run with the coefficients presented in Tables 3-5. These coefficients are derived from field experiments and, as such, are relevant to good growing conditions (in particular irrigation and drainage, soil fertility status, exclusion of field borders) and good management (in particular the timing of operations). Evaluations based on these data are not necessarily representative for all practical forage production systems in which, for instance, the timing of operations cannot be optimized on each individual field. In addition, the utilization of N and P on a whole field basis may be somewhat lower than on those parts of the field where experiments are normally situated. To mimic these effects we have also run the model for a situation in which we reduced the assumed crop uptake efficiency by 10% (so, for instance, 67.5% instead of 75%) and simultaneously reduced the harvest efficiency by 5% points (so, for instance, 9% losses instead of 4%). Consequential manure-fertilizer N combinations of this third scenario were tabulated also. In the discussion section we confront these assumed reductions with a recent survey of commercial farms.

As the leaching fractions derived from the Monitoring Program vary from year to year (Table 6), the allowable rates vary too. Therefore we added 95% probability intervals to our calculated manure and mineral fertilizer N rates. The upper and lower boundaries of these intervals were calculated in a similar way, i.e. through the MS Excel ™ Solver Tool we have explored which combination of manure and mineral fertilizer N complied with targeted P surplus and nitrate-N concentration as reflected in the allowable N surpluses according to Table 6. If crop demand and not the N concentration in water (and thus leaching fractions) limited the application rates, intervals narrowed to zero in some cases.

On farms where both grass and maize are present, it is the combination of crops that achieves goals in terms of N concentration in water and P soil surplus. Therefore, we have also explored the consequences on farms where both crops are present. This exploration was restricted to the average leaching fractions, a targeted P surplus of 0 kg per ha and to clay and sandy soils.

5.2 Results

Figures 11-16 show isoquants of combinations of manure and mineral fertilizer N generating N concentrations of (at most) 11.3 mg N per litre. In each isoquant it is indicated where exactly the substitution of mineral fertilizer by manure becomes associated with a positive P surplus. The utmost right point of each isoquant represents the situation where no mineral N is being used. In the figures for cut grassland, the utmost left point of each isoquant represents the situation where no manure at all is applied. The figures referring to the mixed use of grassland lack this 'full mineral N' point, as the grassland receives at a minimum the N excreted via urine and dung during grazing.
If N concentrations in groundwater and surface water are the sole aim and management is optimal, manure rates of 450-610 kg N per ha (160-220 kg P_2O_5 per ha) would be possible on cut grassland. However, with such rates, amounting to more than 100 m^3 slurry per ha, much more P_2O_5 would be applied than the annual off take of grassland. When grassland is used for both cutting and grazing it could still utilize 410-610 kg N per ha (150-220 kg P_2O_5 per ha), again under the condition that growing conditions and management are good, and P accumulation is fully ignored. Silage maize, however, should not receive more than 180-320 kg manure-N per ha (60-110 kg P_2O_5 per ha) to stay below the targeted N concentrations, even when growing conditions and management are good (Table 9).
Much lower inputs of manure are likely to be needed in practice as all rates indicated in the previous scenario are associated with a large P accumulation. If avoidance of P accumulation is included as a constraint, manure rates of 330-340 kg N per ha (120 kg P_2O_5 per ha) would be possible on cut grassland, provided that the growing conditions and management are good. Similarly, rates of 270-280 kg manure-N per ha (100 kg P_2O_5 per ha) would be possible on grassland used for both cutting and grazing, and rates of 155-200 kg manure N per ha (55-70 kg P_2O_5 per ha) on silage maize (Table 10).
When correcting for 'suboptimal growing conditions and management', input rates need a further reduction to comply with targeted N concentrations in groundwater and surface water and to attain a balance between P inputs and outputs. Manure rates of 270-290 kg N per ha (100 kg P_2O_5 per ha) would be possible on cut grassland then. Similarly, rates of 210-240 kg manure-N per ha (70-90 kg P_2O_5 per ha) would be possible on grassland used for both cutting and grazing, and rates of 135-165 kg manure N per ha (50-60 kg P_2O_5 per ha) on silage maize on most soils. On sandy soils with a MHG < 0.80 m, especially, not more than 125 kg manure N per ha could be applied to maize land (Table 11).
The indicated manure rates (Tables 9-11) 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 by definition if P surpluses are to be avoided (Schröder, 2005). Mineral fertilizer rates associated with the indicated manure rates are hence included in Tables 9-11.
Our explorations of the effect of the maize share on the limits to the use of manure and mineral fertilizer N, show that allowable inputs are negatively related to the maize share (Table 12).

Table 9. Allowable manure rate (kg N per ha per year, 95% probability interval in brackets), the associated mineral fertilizer N rate (kg N per ha per year, 95% probability interval in brackets) and resulting net N and P_2O_5 yields in crops (kg per ha per year, ranges associated with the probability interval of application rates in brackets) on dairy farms targeting at nitrate-N or total N < 11.3 mg/litre (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 but without concerning 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	Peat	465 (465-465)*	0	375 (375-375)	124 (124-124)
		Clay	611 (505-611)*	0	375 (368-375)	124 (121-124)
		Sandy, MHG < 0.40 meter	532 (470-610)	0	371 (362-375)	122 (119-124)
		Sandy, 0.80 < MHG < 0.40 meter	494 (447-585)	0	366 (357-375)	121 (118-124)
		Sandy, MHG > 0.8 meter	453 (420-512)	0	359 (351-369)	118 (116-122)
	Mixed use	Peat	478 (478-478)*	0	313 (313-313)	96 (96-96)
		Clay	608 (448-641)	0	331 (310-332)	102 (96-103)
		Sandy, MHG < 0.40 meter	478 (408-627)	0	316 (299-332)	98 (92-102)
		Sandy, 0.80 < MHG < 0.40 meter	445 (390-548)	0	316 (300-335)	98 (93-104)
		Sandy, MHG > 0.8 meter	407 (340-475)	0	311 (271-330)	96 (84-102)
Maize		Peat	-	-	-	-
		Clay	315 (261-423)	0	189 (178-203)	70 (66-75)
		Sandy, MHG < 0.40 meter	241 (208-244)	0	172 (163-173)	64 (60-64)
		Sandy, 0.80 < MHG < 0.40 meter	207 (187-240)	0	163 (156-172)	60 (58-64)
		Sandy, MHG > 0.8 meter	180 (170-181)	0	154 (150-154)	57 (56-57)

* *if crop demand instead of nitrate concentration limited N inputs, calculated rates become insensitive for the imposed variation of leaching fractions and, so, intervals may narrow to even zero*

Table 10. Allowable manure rate (kg N per ha per year, 95% probability interval in brackets), the associated mineral fertilizer N rate (kg N per ha per year, 95% probability interval in brackets) and resulting net N and P_2O_5 yields in crops (kg per ha per year, ranges associated with the probability interval of application rates in brackets) on dairy farms targeting at nitrate-N or total N < 11.3 mg/litre (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	Peat	341	(341-341)*	103	(103-103)	375	(375-375)	124	(124-124)
		Clay	341	(338-341)*	225	(164-225)	375	(371-375)	124	(123-124)
		Sandy, MHG < 0.40 meter	340	(333-341)	187	(136-224)	374	(366-375)	123	(121-124)
		Sandy, 0.80 < MHG < 0.40 meter	336	(329-341)	155	(118-224)	370	(362-375)	122	(120-124)
		Sandy, MHG > 0.8 meter	330	(323-338)	123	(97-170)	363	(356-372)	120	(117-123)
	Mixed use	Peat	265	(265-265)*	175	(175-175)	313	(313-313)	96	(96-96)
		Clay	282	(269-282)*	299	(179-299)	332	(317-332)	103	(98-103)
		Sandy, MHG < 0.40 meter	274	(261-282)	203	(149-299)	322	(308-332)	100	(95-103)
		Sandy, 0.80 < MHG < 0.40 meter	275	(263-289)	171	(132-253)	324	(309-340)	100	(96-105)
		Sandy, MHG > 0.8 meter	273	(261-287)	139	(112-188)	321	(308-338)	99	(95-104)
Maize		Peat	-		-		-		-	
		Clay	199	(187-208)	119	(78-198)	196	(184-205)	72	(68-76)
		Sandy, MHG < 0.40 meter	175	(170-175)	53	(42-53)	173	(168-173)	64	(62-64)
		Sandy, 0.80 < MHG < 0.40 meter	169	(162-175)	41	(28-52)	163	(160-173)	62	(59-64)
		Sandy, MHG > 0.8 meter	155	(154-155)	19	(17-19)	154	(153-154)	57	(57-57)

* *if crop demand instead of nitrate concentration limited N inputs, calculated rates become insensitive for the imposed variation of leaching fractions and, so, intervals may narrow to even zero*

Table 11. Allowable manure rate (kg N per ha per year, 95% probability interval in brackets), the associated mineral fertilizer N rate (kg N per ha per year, 95% probability interval in brackets) and resulting net N and P2O5 yields in crops (kg per ha per year, ranges associated with the probability interval of application rates in brackets) on dairy farms targeting at nitrate-N or total N < 11.3 mg/litre (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	Peat	291	(291-291)*	133	(133-133)	320	(320-320)	106	(106-106)
		Clay	291	(282-291)*	255	(156-255)	320	(311-320)	106	(103-106)
		Sandy, MHG < 0.40 meter	286	(276-291)	180	(128-255)	315	(304-320)	104	(100-106)
		Sandy, 0.80 < MHG < 0.40 meter	281	(271-290)	147	(110-228)	309	(299-319)	102	(99-105)
		Sandy, MHG > 0.8 meter	272	(264-283)	114	(89-162)	300	(291-312)	99	(96-103)
	Mixed use	Peat	223	(223-223)*	205	(205-205)	264	(264-264)	81	(81-81)
		Clay	238	(219-238)*	304	(168-327)	281	(259-281)	87	(80-87)
		Sandy, MHG < 0.40 meter	225	(209-238)	192	(138-322)	266	(247-281)	82	(76-87)
		Sandy, 0.80 < MHG < 0.40 meter	223	(196-241)	160	(118-243)	264	(231-284)	81	(71-88)
		Sandy, MHG > 0.8 meter	211	(170-235)	126	(92-177)	249	(202-277)	77	(62-86)
Maize		Peat	-		-		-		-	
		Clay	165	(153-177)	118	(78-208)	163	(151-175)	60	(56-65)
		Sandy, MHG < 0.40 meter	147	(137-149)	64	(41-68)	146	(136-148)	54	(50-55)
		Sandy, 0.80 < MHG < 0.40 meter	136	(129-147)	40	(27-63)	135	(128-146)	50	(47-55)
		Sandy, MHG > 0.8 meter	127	(123-132)	23	(17-32)	126	(122-131)	47	(45-48)

* *if crop demand instead of nitrate concentration limited N inputs, calculated rates become insensitive for the imposed variation of leaching fractions and, so, intervals may narrow to even zero*

Table 12. Allowable average manure rates (kg N per ha per year) and associated mineral fertilizer N rates (kg N per ha per year) on dairy farms targeting at nitrate-N or total N < 11.3 mg/litre (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).

Growing conditions / management quality		Good				Suboptimal			
Harvest regime of grass		Cutting only		Mixed use		Cutting only		Mixed use	
Soil type*	Maize share	Manure-N	Mineral fertilizer-N	Manure-N	Mineral fertilizer-N	Manure-N	Mineral fertilizer-N	Manure-N	Mineral fertilizer-N
Clay	15	320	209	270	272	272	234	227	276
	30	298	193	257	245	253	214	216	248
	45	277	177	245	218	234	193	205	220
Sandy, MHG < 0.40 meter	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 meter	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 > 0.80 meter	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

* *peat soils are not included here as silage maize is hardly grown on that soil type*

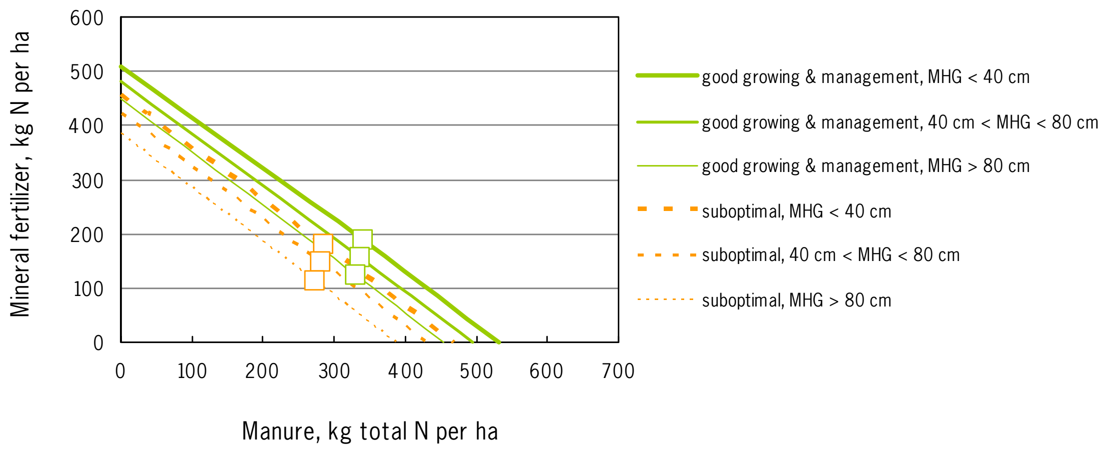

Figure 11. *Isoquant of mineral fertilizer N and manure N combinations for cut grassland on sandy soils, generating a nitrate-N concentration in groundwater of at most 11.3 mg per litre (the combination associated with an annual P soil surplus of 0 kg per ha is indicated by □).*

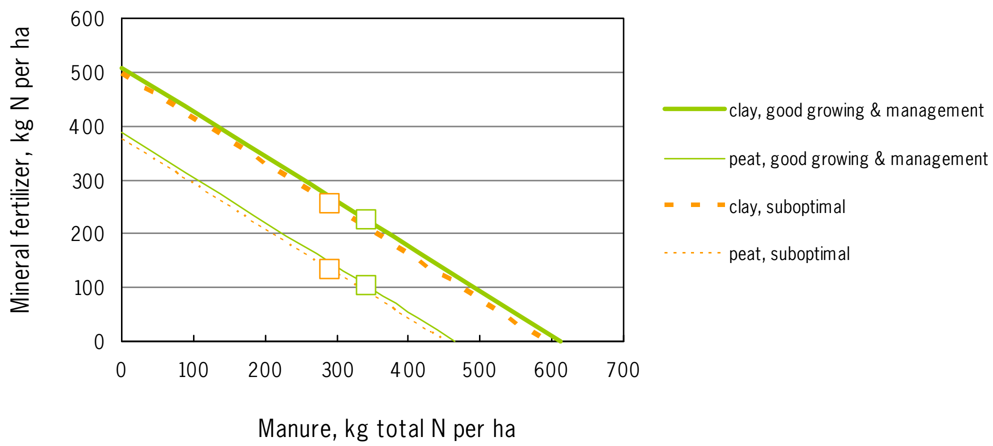

Figure 12. *Isoquant of mineral fertilizer N and manure N combinations for cut grassland on clay and peat soils, generating a total N concentration in drain (clay) or ditch (peat) water of at most 11.3 mg per litre (the combination associated with an annual P soil surplus of 0 kg per ha is indicated by □).*

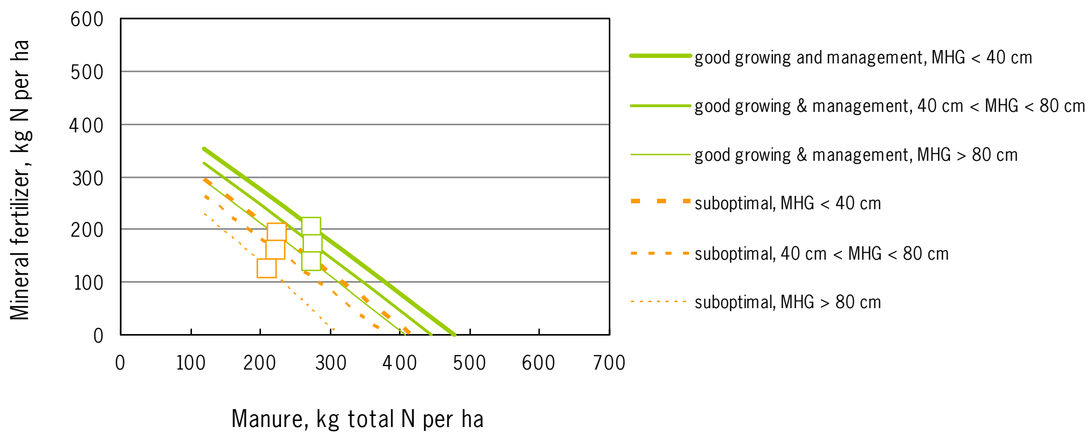

Figure 13. *Isoquant of mineral fertilizer N and manure N combinations for grassland with a mixed use of cutting and grazing on sandy soils, generating a nitrate-N concentration in groundwater of at most 11.3 mg per litre (the combination associated with an annual P soil surplus of 0 kg per ha is indicated by □).*

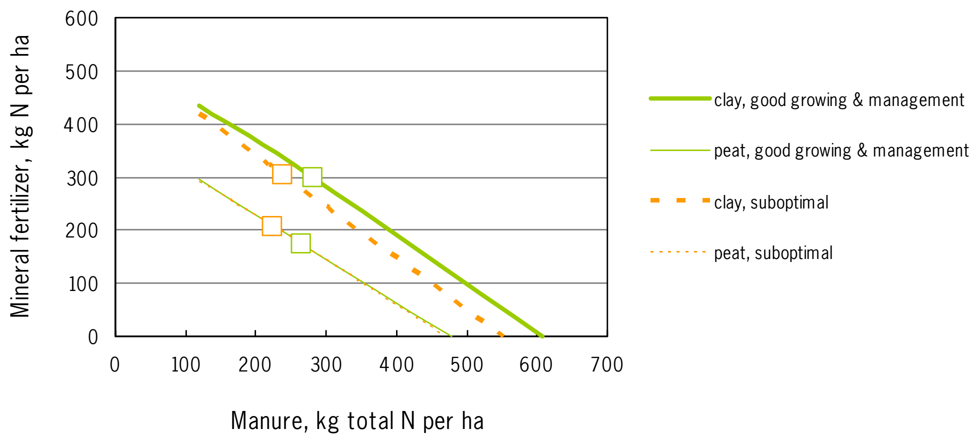

Figure 14. *Isoquant of mineral fertilizer N and manure N combinations for grassland with a mixed use of cutting and grazing on clay and peat soils, generating a total N concentration in drain (clay) or ditch (peat) water of at most 11.3 mg per litre (the combination associated with an annual P soil surplus of 0 kg per ha is indicated by □).*

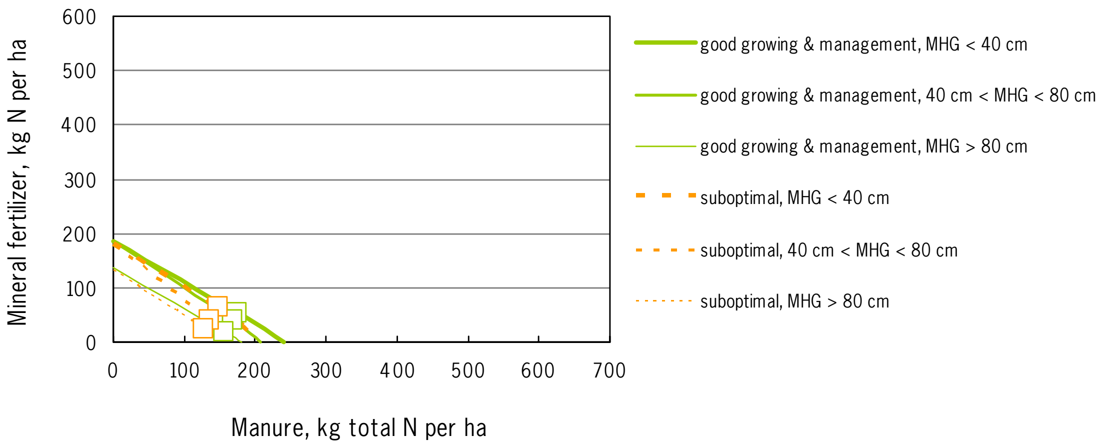

Figure 15. *Isoquant of mineral fertilizer N and manure N combinations for silage maize on sandy soils, generating a nitrate-N concentration in groundwater of at most 11.3 mg per litre (the combination associated with an annual P soil surplus of 0 kg per ha is indicated by □).*

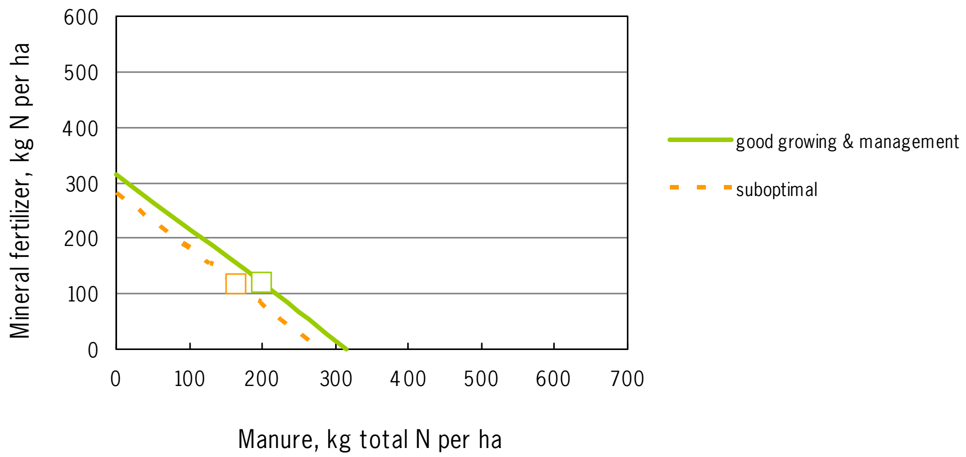

Figure 16. *Isoquant of mineral fertilizer N and manure N combinations for silage maize on clay soils, generating a total N concentration in drain water of at most 11.3 mg per litre (the combination associated with an annual P soil surplus of 0 kg per ha is indicated by □).*

6. Discussion

6.1 Should 170 kg manure-N per ha be the limit?

The present study indicates that N concentrations in groundwater and surface water are not just determined by the inputs of manure but rather by the combination of manure and mineral fertilizer N. Moreover, especially on grassland with a 'cutting only regime', up to 330-340 kg manure N per ha can be applied annually without exceeding a concentration of 11.3 mg nitrate-N (sandy soils) or 11.3 mg total-N (clay and peat soils) per litre, and without applying more P than the amount removed via crops. Similar conclusions were drawn by Aarts *et al.* (1999a, b), Willems *et al.* (2000) and 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 & Kopec, 1996), as well as to the empirical evidence that only a fraction of the soil N surplus is recovered in the upper 1 meter of the groundwater in situations with shallow groundwater. Theoretically, the latter can be explained by either temporary accumulation of the soil N surplus or by denitrification. We do not think that accumulation is a likely explanation. First, because land use has not recently changed towards grassland, which would favour accumulation (Figure 2), and second, because soil N surpluses have decreased during the last decade (Figure 3). This leaves denitrification as the most probable explanation why only a fraction of the soil N surpluses ends up in groundwater. This apparent denitrification is larger than the estimates based on measurements using the acetylene inhibition technique (Dolfing *et al.*, 2004; Van Beek *et al.*, 2004; Zwart *et al.*, 2004). However, the acetylene inhibition technique underestimates denitrification, especially in wet soils where gas diffusion is hampered. Moreover, in most studies, denitrification is only measured in the top soil (Barton *et al.*, 1999). The combination of shallow groundwater levels 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 (Figure 9). 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; Dolfing *et al.*, 2004; Plette *et al.*, 2004). However, outputs such as gaseous losses from crop canopies or leaching of dissolved organic N (Macdonald & Jones, 2004), may also be underestimated.

The present study also shows that environmentally acceptable manure inputs at the farm level depend on growing conditions and management in general, the harvest regime of grassland in particular and the share of crops other than grassland (i.e. silage maize) in the rotation of dairy farms. Even with mixed use of grassland (i.e. half of the production harvested via grazing) and a maize share up to 30% of the farm area, the annual use of approximately 240 kg manure N per ha (sandy soils with deep groundwater) to 260 kg manure N per ha (clay soils) can be reconciled with a N concentration of 11.3 mg per litre and a P-surplus of 0 kg per ha when growing conditions are good and crops are well managed. Corresponding numbers for poorer growing conditions and management would be 190 and 220 kg manure N per ha (Table 12).

From the point of view of P accumulation and N concentration in groundwater and surface water, limiting cattle slurry application rates to 170 kg N per ha per year as embedded in the Nitrates Directive (Anonymous, 1991a) appears to be an unnecessary restriction for many dairy farms given their crop choice, crop production levels and the soil and climate conditions of the Netherlands, provided that cattle slurry inputs larger than these 170 kg N per ha are sufficiently balanced with reduced mineral fertilizer N inputs. This is even more so if manure rates are to be founded on N related effects only.

Our calculations show that there is a trade-off though between the extent to which manure can be used safely and the room for grazing, because manure excreted during grazing results in greater leaching losses than manure from animal housing that is mechanically applied (Vellinga *et al.*, 2001; Nevens & Reheul, 2003b; Wachendorf *et al.*, 2004). So, if one takes the position that grazing contributes positively to landscape quality and animal welfare, there are also trade-offs between these features and the extent to which one can apply manure within targeted nitrate-N concentrations. One could generally say that any farm, deliberately aiming at more functions than just milk production within N and P constraints (e.g. landscape, animal welfare, nature), will have to reduce application rates to levels below the ones calculated in the present paper. However, Corré *et al.* (2003) concluded that such multifunctional extensive farms are not necessarily efficient in terms of resource utilization.

All numbers presented in this section refer to calculations based on average leaching fractions (Table 6). Our calculations indicate that slurry rates should be approximately 10 kg N per ha less if rates are to be based on the premise that N concentrations are not to be exceeded in 95 out of 100 years. Mineral fertilizer N rates would in that case need a reduction of, on average, 30 kg N per ha (Table 10-11).

6.2 How to avoid incorrect estimates of inputs?

6.2.1 Actual inputs of manure N and P

The applicability of our calculations to the general practice strongly relies on a precise determination and account for all relevant N inputs. This refers, for instance, to a correct assessment of the N and P excretion per animal category present on the farm, per production level and per type of diet (e.g. Grubber & Steinwidder, 1996; Paul *et al.*, 1998; Kebreab *et al.*, 2001). Estimates of the 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).
In addition to a correct assessment of the amount of N excreted, accurate estimates of the gaseous N losses from housing and manure storages are needed to assess how much manure-N is eventually applied to the fields (e.g. Bussink & Oenema, 1998; Monteny & Erisman, 1998). Moreover, reliable accounts are needed for manure imported to or exported from the farm. Special attention is also needed for the P-N ratio of manures, as pig and poultry slurries as well as solid manures from cattle, for instance, 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 be somewhat larger without accumulating P, if the P to N ratios in crops would become narrower as a consequence of a more restricted N use. Crop composition, including that of crops used for concentrate production, hence deserves constant monitoring. Our assumption that 25% of the total N input (i.e. 50% of the organic N input) of slurry mineralizes after the first 12 months after application (Lammers, 1983) can be questioned because more recent insights point at 30-40% of the total N input (i.e. 66-75% of the organic N input) of slurry (Schröder *et al.*, 2005). However, this does not affect our conclusions because we based our calculations on an equilibrium situation in which the long term fertilizer N equivalency of manures including residual N effects is fully accounted.
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 the legal obligations to inject or immediately incorporate manure. If this would not be the case, more mineral fertilizer N should have been 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 must emphasize that we adopted fixed values for several crop residue related characteristics (resulting in negative soil N surpluses at low input levels i.e. depletion of soil N pools), whereas reduced input levels will generally have a negative feedback on the quantity of N invested in crop residues and SMN_{spring} and thus on the assumed contribution to annual N inputs (and outputs for that matter, see Table 8). Reduced inputs will 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.

6.2.2 N Input via biological fixation

N inputs via biological fixation need to be taken into account, too. However, at present mixed stands of grass and white clover are relatively rare in the Netherlands. Based on seed sales, Corré & Pinxterhuis (2000) estimated the area of grassland with white clover between 50 and 100 thousand hectares, i.e. 5%-10% of the grassland area in the Netherlands.
If one assumes that 54 kg N per ha are fixed per harvested ton dry matter of clover (Van der Meer & Baan Hofman, 1989; Elgersma & Hassink, 1997), an input of 130-160 kg N per ha should be accounted for in a sward with a visually assessed cover of, for instance, 30% (Schils, 2002; Schils *et al.*, 1999). If the area of mixed stands increases, this input would deserve more attention, as N losses would be higher than expected on the basis of just manure and fertilizer inputs. However, one could reason that excess N application to mixed swards will eventually increase the yield of the grass component at the expense of the clover content, its yield and its N fixation.

The increased N uptake from fertilizer application is completely countered then by a decreased clover derived N uptake (e.g. Schils & Snijders, 2004). Due to this self-regulation it may seem unnecessary to adjust the application standards for N in Action Programmes for grass/clover mixtures. Accounting of biologically fixed N may yet be necessary to prevent unintended circumvention by farmers considering a switch to a 'two sward' strategy. In such a strategy, a certain proportion of the grassland area consists of grass/clover mixtures receiving no or only little fertilizer N. The amount of N 'saved' in these plots is transferred and used on intensively managed, grass-only swards at rates exceeding those required to generate a N concentration of 11.3 mg per litre or less.

6.2.3 N input from ploughed swards

General considerations

Ploughing of grassland leads to an enhanced mineralization of accumulated plant material and soil organic matter. The quantity and fate of mineralised 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. Total N mineralization from soil organic matter and the ploughed sward in the first year after ploughing-up temporary grasslands ranges from approximately 125 to 400 kg N ha^{-1} (Aarts *et al.*, 2001; Johnston *et al.*, 1994; Van Dijk *et al.*, 1996; Vertès *et al.*, 2002; Whitehead *et al.*, 1990; Zwart *et al.*, 1999). 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 (< 10 years) can temporarily immobilize 20-130 kg N ha^{-1} $year^{-1}$, thus reducing immediate losses to the environment (Cuttle & Scholefield, 1995; Hassink, 1994; Hoogerkamp, 1984; Tyson *et al.*, 1990; Whitehead *et al.*, 1990; Velthof & Oenema, 2001; Velthof & Hoving, 2003).
In the Netherlands approximately 90% of the total grassland area is permanent grassland, i.e. lasting more than five years irrespective of renovation. On average, grassland on sandy, clay and peat soils is renovated every 5, 10 and 30 years, respectively (Schils *et al.*, 2002). In these grasslands, accumulation of organic N does not continue for as long as it would in non-renovated grassland, due to the enhanced mineralization after each renovation. The remaining 10% of the total grassland area in the Netherlands is temporary grassland (Schils *et al.*, 2002) which is usually rotated with maize but also with potatoes or flower bulbs. In grass-arable rotations, periods of grassland of 2-5 years are followed by periods of arable land of 2-5 years. Therefore, soil organic N content fluctuates more strongly than in renovated permanent grasslands (e.g. Vertès *et al.*, 2002).
To reduce the risk of N leaching after ploughing, several measures can be taken. The longer a soil remains fallow after ploughing the higher the risk of N losses (Adams & Jan, 1999; Davies *et al.*, 2001; Lloyd, 1992). Postponing cultivation of temporary grassland from autumn to spring reduces N leaching, provided that a crop is grown (Francis, 1995; Djurhuus & Olsen, 1997). For permanent grassland, the risk of N leaching decreases if grassland is cultivated and reseeded early in the season, so that the period with a high mineralization coincides with the growing season. Hence, sward destruction should be postponed from autumn to spring, especially in lightly textured soils that are prone to N leaching. The risk of N leaching can also be decreased by growing crops with a high N uptake capacity of which grass is an outstanding example. If followed by a perfectly managed winter cover crop, the uptake dynamics and capacity of silage maize, too, may correspond with the dynamics and magnitude of mineralization. Perfect management includes a timely establishment of the cover crop, the use of a winter hardy species, choosing destruction dates that do not jeopardize the available N or water of a subsequent crop, and taking full account of the N that the cover crop provides to the subsequent crop (Francis, 1995; Schröder *et al.*, 1996). Several arable crops also have a N uptake capacity that is comparable with maize followed by a cover crop, but are rarely grown in ruminant production systems and therefore not further considered in the present paper.
Adjustment of the applied rates of manure and fertilizer N to the N mineralization from the ploughed grassland, is also a prerequisite to minimize N leaching. Studies of Aarts *et al.* (2001) and Nevens *et al.* (2002) showed that N application can be largely omitted in the first year of maize growing after ploughing grassland. Nitrogen fertilizer application can be regulated on basis of soil mineral N measurements in the period after ploughing of grassland.
A literature review of Velthof *et al.* (2002) indicates that the method and depth of grassland cultivation is not a clear option to decrease the risk on N leaching after grassland renovation. The positive correlation between grassland age and organic matter content suggests that reducing the age at which grassland is cultivated may be an option to decrease the risk of N losses after grassland cultivation. However, studies of Johnston *et al.* (1994) and Shepherd *et al.* (2001) indicate minor effects of grassland age on N loss in temporary grassland (< 5 years). Because of the

relatively small effect of the length of the grass period on N mineralization and N losses, the frequency of ploughing appears to be more decisive for total N losses on the long run. Risk of N losses depends on weather conditions and N management in the year of ploughing the grassland. Increasing the age of temporary grassland from, for instance, 3 to 5 years implies that the number of periods during the rotation with an increased risk of N loss decreases.

Table 13. 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 built-up associated with a rotation are ignored (I) (consult text for further assumptions).

Scenario	Scale	N (kg per ha per year)					Nitrate-N (mg per litre water)	P_2O_5 surplus, kg per 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*

An example

Despite the previous considerations, we explored the limits to the use of manure and mineral fertilizer N by treating grass and silage maize separately in our calculations. This approach ignores that crops are often grown in a rotation and that silage maize is then preceded by ploughed-down grassland from which large amounts of N will mineralize (Nevens & Reheul, 2002). This mineralization can contribute to both plant nutrition and potential N leaching. So far, we reasoned that the reduction of N inputs needed to keep the leaching from maize below acceptable levels, would be balanced by a comparable built-up of organic N under the newly established grassland grown after maize. This built-up could, in turn, allow additional N inputs on grassland without increased risks of leaching.
In order to yet give an impression of the consequential adjustments in individual crops, we have run the model once more whilst taking a account of the N dynamics in a rotation. We restricted our calculations to a sandy soil with 0.80 < MHG < 0.40 meter, a mixed use of grassland, and the average soil-specific leaching fractions (Table 6). Further, we assumed that growing conditions and management are good and P accumulation at the whole farm level should be avoided. We adopted a rotation comprising four years of temporary grassland followed by two years maize ('66,6% grassland and 33.3% maize'). We surmised an additional annual N mineralization of 150 kg N per ha from the ploughed down grassland (so, 2 years x 150 = 300 kg N per ha) and an additional annual N built up of 75 kg N per ha under grassland (so, 4 years x 75 = 300 kg N per ha). This is a simplification because both mineralization and built-up may vary strongly from one year to another as earlier indicated. Besides, mineralization and built-up are generally larger during the first year(s) of each crop phase than during the final year(s). It is unclear at which spatial scale the nitrate targets in the Nitrates Directive should be achieved, so we distinguished two situations: one in which nitrate should stay below the target under each individual crop (C) and another in which nitrate should stay 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).

The N yields of individual crops are reasonably similar in all three cases (Table 13). However, considerable adjustments of N inputs are needed in individual crops, in particular when nitrate targets are to be achieved under each individual crop (C). The allowable application rates of manure on maize in our initial approach (169 kg N per ha per year) should be reduced to nil, in agreement with Aarts *et al.* (2001) and Nevens & Reheul (2002). On the contrary, the allowable application rate of manure on grassland could be increased from an initial annual rate of 275 kg manure N per ha to approximately 365 kg N per ha. Under the given assumptions, maximization of the average whole-farm N yield was slightly favoured by transferring some mineral fertilizer N from grassland to maize. This would only be an option if nitrate targets were to be achieved at the whole farm level (F) instead of the level of individual crops (C). 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 (plain grassland renovation).

6.2.4 N input from mineralizing peat

Peat soils are commonly used for dairy farming and approximately 17% of the total grassland area of the Netherlands is located on peat soil. In the western part of the Netherlands a large part of the grasslands on peat soils is located in polders, i.e. at an elevation below mean sea level. The fields are separated by ditches to improve drainage. When evapotranspiration exceeds precipitation, water is supplied to the polder from the surrounding lakes and channels (mostly during summer). When precipitation exceeds evapotranspiration (mostly during winter) water is pumped out of the polder. In winter the soil water storage capacity is frequently exceeded and then fields become waterlogged. The N mineralization in drained peat soils is enhanced because oxygen enters the soil and stimulates decomposition of organic matter. Mineralization and denitrification in peat soils are closely related. Measured N losses via denitrification in intensively managed grassland on peat soils (0-0.20 meter layer) range from about 50-79 kg N ha^{-1} yr^{-1} (Koops *et al.*, 1996 & 1997; Van Beek *et al.*, 2004) and are higher than measured denitrification rates in sandy soils. A large part of the denitrification in peat soils originates from soil layers deeper than 0.20 meter below soil surface. Based on a recent study (Van Kekem, 2004), we estimate that the additional net mineralization (gross mineralization minus (gross immobilisation + denitrification)) on peat soils amounts to approximately 160 kg N per ha per year.

6.3 How to avoid incorrect estimates of outputs?

The applicability of our calculations does not only depend on a correct assessment of inputs (see section 2.1), 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, indicators based on just fertilizer and manure application standards do not contain explicit incentives to reduce nutrient inputs via feed imports and, hence, do not automatically stimulate the production of home-grown crop outputs through optimized crop management. We have anticipated this shortcoming by running our calculations also for sub optimal conditions (Table 11, Figures 11-16). Such suboptimal conditions can 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 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.
Parallel to the present study, Aarts *et al.* (2005) made an estimate of the net N yield of grassland in the Netherlands through an analysis of the records from four recent years (1998, 1999, 2001 and 2002) of 255 commercial dairy farms. Their point of departure was the energy requirement of the herd on each individual farm and estimated energy and N contents of forages and concentrates. By combining these data with registered purchases of feed stuffs and estimates of the on-farm N yield of silage maize and the N losses from forages during conservation and feeding, they made an estimate of the apparent N yield from grassland. Their data base 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 make a comparison between the use efficiency of N (NUE) as achieved on these farms, and the NUE of comparable

inputs when using the present modelling approach. This comparison revealed that the estimated 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 clay and wet sandy soils, however, was in closer agreement with our scenarios that assume 'suboptimal growing conditions and management' (Table 14). The NUE at the whole farm level, reflecting the weighted contributions from both crops in terms of their hectares and amounts of 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 was above the long term average in all four years (Anonymous, 1998-2002). 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. All in all, the observed NUE of commercial farms as based on the approximated N yields of grassland, appears to support the assumptions in our model concerning the integral effect of the fertilizer equivalency, the uptake efficiency and the harvest efficiency.

Table 14. 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 255 commercial dairy farms (Aarts et al., 2005 (in prep.)), 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
		'good growing conditions and management'	'suboptimal growing conditions and management'	'preliminary estimates from practice 1998-2002'
Peat	Grassland	0.67	0.57	0.64
Clay	Grassland	0.66	0.56	0.65
	Maize	0.76	0.65	0.69
	Whole farm	*0.66*	*0.56*	*0.65*
Sandy, MHG < 0.40 meter	Grassland	0.67	0.57	0.67
	Maize	0.65	0.56	0.57
	Whole farm	*0.67*	*0.57*	*0.66*
Sandy, MHG > 0.80 meter	Grassland	0.65	0.56	0.67
	Maize	0.54	0.46	0.55
	Whole farm	*0.64*	*0.54*	*0.65*

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

First, we must note that the coefficients (Table 6) for sandy soils used to translate soil N surpluses to nitrate-N concentrations pertain to the upper 1.00 meter groundwater.
The 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 change gradually 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 reactive carbon, the leached fraction of the soil N surplus may not necessarily be the same at reduced input levels in the future, changing the coefficients towards higher nitrate-N concentrations. Conversely, precipitation surpluses will increase, as N will become a growth limiting factor reducing the transpiration (De Wit, 1958), thus changing coefficients towards lower

nitrate-N concentrations due to dilution. These uncertainties, together with gradual but hard to measure changes of the amount of N in soil organic matter, may affect future coefficients and thus leaching. Still, the validation of the relationship against an independent recent data set representing farms with reduced inputs rates (Figure 10), gives some confidence.
The EU Nitrates Directive (Anonymous, 1991) is not very specific about the spatial and temporal scale at which EU members states should achieve the target of 11.3 mg nitrate-N per litre or less. The smaller the scale, the more relevant it becomes to get an accurate picture of variability in space and time. Unaware of the eventual decisions on the required precaution and resolution, we analyzed the uncertainties by distinguishing groundwater levels, soil types and crop types (spatial variation) and year to year variation of the leaching fraction (temporal variation). The latter type of variation encompasses weather-induced effects on the surplus and its fate, in as far as defaults had to be used in the balances provided by the Monitoring Program. This kind of effects is reflected in the s.d.'s of the leaching fractions (Table 6) and the 95% probability intervals of permitted application rates (Tables 9-11).

6.5 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 total N (clay and peat soils) or nitrate-N (sandy soils) per litre, the latter in agreement with the Nitrates Directive. However, according to the Water Framework Directive (WFD; Anonymous, 2000), practices must be directed at ecological targets in surface waters which, depending on their eventual definition, may require lower total N concentrations than 11.3 mg per litre, let alone nitrate-N per litre (Plette *et al.*, 2002; Van Liere & Jonkers, 2002). In addition, N (and P) originating from the sub soil may contribute to the eutrophication of surface water in peat regions in particular, in addition to the emissions directly linked to the use of manure and fertilizers, as 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 ground- and surface water. This deserves attention as denitrification is associated with nitrous oxide production, which is a very potent greenhouse gas (IPCC, 1996). When designing policies and measures directed at global effects, however, it is sensible to evaluate impacts on a per litre milk basis rather than on a per hectare basis. From that perspective, extensification as such does not necessarily reduce the emission at the global scale (e.g. Schröder *et al.*, 2003; 2004). So far, it is uncertain which measures dairy farmers must take for the benefit of the global climate (e.g. Velthof & 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 water hardness (Ca, Mg), and the increase of heavy metal and sulphate concentrations in deeper groundwater, especially in calcareous soils rich in sulphides (i.e. through pyrite oxidation; Willems *et al.*, 2002). 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 Groundwater Directive (Anonymous, 2003).

7. Conclusions

N and P demand of cut grassland in the Netherlands can be satisfied by cattle manure at rates of 330-340 kg manure N (120 kg P_2O_5) per ha per year without exceeding a target value of 11.3 mg N per litre or accumulating P in the soil, provided that i) sufficient supplementary mineral fertilizer N is applied, ii) growing conditions are good and the grassland is well managed, iii) the hydrological situation is conducive to denitrification. On silage maize rates should not exceed 160-200 kg manure N (= 50-70 kg P_2O_5) per ha per year, the lower values referring to sandy soils with groundwater levels deeper than 0.80 meter. Suboptimal growing conditions or crop management and substitution of cutting by grazing reduce the scope for manure applications considerably. Manure rates should therefore be determined by the share of both crops, the hydrological situation, the harvest regime, growing conditions, and management quality.

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