Nutr Cycl Agroecosyst (2007) 77:155–167
DOI 10.1007/s10705-006-9051-9

ORIGINAL PAPER

Nitrate in upper groundwater on farms under tillage as affected by fertilizer use, soil type and groundwater table

F. J. de Ruijter · L. J. M. Boumans ·
A. L. Smit · M. van den Berg

Received: 27 February 2006 / Accepted: 24 August 2006 / Published online: 13 October 2006

Abstract Indicators are needed to check whether policies on protection of groundwater are effective and if regulations are complied with. We evaluated various indicators at different scales, both in space and in time, and at different degrees of complexity. Groundwater was sampled on 34 arable farms for 3 years. Nitrate concentration in upper groundwater was low on clay soil. On sandy soil, peat layers reduced the nitrate concentration with about 80 mg/l on average. Sandy soils with high groundwater tables had nitrate concentrations that were less than half of those at sandy soils with low groundwater tables. The relationship between different fertilization variables and nitrate in groundwater was investigated for sandy soils without peat layers. N surplus poorly correlated with nitrate concentrations in groundwater when individual sampling points were studied, but clearly increased when data were averaged at the farm level. Soil mineral nitrogen correlated best with nitrate concentrations in groundwater. The relationships show that especially on well drained soil drastic measures will be inevitable to reach good water quality.

Keywords Arable farming · Bulb cultivation · Element balance · Horticulture · Nitrogen · Nutrient balance · Nursery stock production

Introduction

Intensification of agriculture since the 1950's has increased emissions of nitrogen (N) to the detriment of the environment (Matson et al. 1997; Smith 2003). To protect groundwater and reduce or prevent eutrophication of surface waters, the EU has adopted the Nitrates Directive (EC 1991) and the Water Framework Directive (EC 2000). These directives oblige Member States to establish national action plans to reduce nutrient emissions and deliver 'good water quality for all purposes'. For groundwater, for instance, measures have to be taken to keep nitrate concentrations below 50 mg/l. Indicators are needed to check whether policies are effective and whether regulations are complied with. As for the emission of N to water bodies, various indicators have been proposed, ranging from a direct measure-

---

F. J. de Ruijter (✉) · A. L. Smit
Plant Research International, Wageningen University
and Research Centre, P.O. Box 16,
6700 AA Wageningen, The Netherlands
e-mail: frank.deruijter@wur.nl

L. J. M. Boumans
National Institute For Public Health
and the Environment, Bilthoven,
The Netherlands

M. van den Berg
South African Sugarcane Research Institute (SASRI),
Mount Edgecombe, South Africa

ment of the nitrate concentration in water or soil to N inputs or N surpluses (Oenema et al. 2003; Schröder et al. 2004; Van Beek et al. 2003). Schröder et al. (2004) argued that no single indicator is at the same time effective, attributable, responsive, efficient and integrative. Nitrate concentration in water, for instance, is by definition an effective ('goal-oriented') indicator but not always attributable and responsive to the management of the farmer involved, nor is it easily assessed. Conversely, N-inputs represent a very attributable ('behavior-oriented') indicator and can be relatively simply derived from book keepings. However, N-inputs have a limited predictive power as far as their ultimate impact on nitrate concentrations is concerned. N-balances (i.e. N inputs versus N outputs) hold an intermediate position between these goal-oriented and behavior-oriented indicators. N-balances can be assessed at different degrees of complexity (Oenema et al. 2003; Watson and Atkinson 1999). The farm-gate balance records the amount of N in all kinds of products that enter and leave the farm via the farm-gate. The soil surface balance records all N that enters the soil via the surface (input) and that leaves the soil via removed crops. Farm-gate balances and soil surface balances are equal for crop production systems without storage of products at the farm ('pool changes on the yard'), provided that corrections are made for the gaseous N losses associated with the application of imported manures and the N contributed to the soil via biological N fixation. Contrary to the farm-gate and soil surface balances, a full soil system balance takes also account of processes that occur in the soil including all N sources and N sinks other than leaching to groundwater.

Theoretically, the N load to groundwater is a function of total-N surplus (N-input minus N-harvested), denitrification and changes of the soil N pool (Oenema et al. 1998; Schröder et al 2003). The N concentration in groundwater is, obviously, also determined by the precipitation surplus (Fraters et al. 1998; Boumans et al. 2001), which in turn is a function of the water balance.

Although N in groundwater of sandy soils in The Netherlands is mainly present in the form of nitrate (Fraters et al. 2004), it remains quite complicated to link the effects of N-management at the soil surface to the N concentration in groundwater. In addition to the variable effect of the precipitation surplus on dilution, relationships between N management and N concentrations are also influenced by the soil type and groundwater table, which affect denitrification and travel time (D'Haene et al. 2003; Elmi et al. 2002).

Indicators of N loss to groundwater can be calculated at different scales, both in space and in time. In space, indicators can be determined at the level of an individual sampling point within fields, field, farm or region. As for time, indicators are usually determined at the scale of individual years or series of consecutive years.

In the Netherlands there is a renewed interest among policy makers to compare indicators. This interest originates from the recent decision of the EU Commission to force The Netherlands to define crop-specific limits to N inputs, implicitly suggesting that limiting N inputs represents a more robust guarantee for groundwater protection than the formerly imposed limitation of N surpluses.

To get more insight in the relation between the various indicators and nitrate concentration we used data from 2000 to 2004 of 34 arable farms. The farms cultivated various crops on different soil types and groundwater tables. The relation was analyzed at different levels of aggregation: sampling points versus farm averages and individual years versus multiple year averages.

Materials and methods

Methodological approach

Groundwater was sampled on 34 arable farms for 3 years. Nitrogen balances were calculated per field and crop, and soil mineral nitrogen after harvest was determined at each individual field. Additionally, an index was calculated using a model and weather data to correct for dilution and duration of percolation. These indicators were tested as explanatory variables to account for the observed nitrate concentration in groundwater.

Agriculture in the Netherlands

Of the Netherlands, almost 70% of the total land area is agricultural land (CBS 2006; LEI 2006). The northern and western half consists mainly of clay (800,000 ha) and peat soils (300,000 ha), with shallow groundwater levels. The southern and eastern half is characterized by sand (800,000 ha) and loess soils (90,000 ha), with relatively deep groundwater levels (Oenema et al. 2004). Half of the area of agricultural land is covered with grassland. Arable crops (mainly cereals, potato, sugarbeet and maize) cover 43%, vegetables 2.2%, flower bulbs 1.2% and nursery stock (ornamental trees) 0.7% (LEI, 2006).

Origin of data

Data for this study were collected from 2000 to 2004 from the Dutch on-farm project 'Farming with a Future'. This project was carried out by farmers, extension services and researchers together and mainly directed at a rapid adoption of methods to reduce N-losses to acceptable levels (Langeveld et al. 2005). The project included 30 commercial arable farmers, horticulture farmers, bulb growers and producers of nursery stock (ornamental trees). This population was extended with four experimental farms (Fig. 1). All farms but five on clay soils in the south west of the Netherlands, were located on sandy soils. These sandy soils differed, however, in terms of the presence of peat layers, organic matter contents and the depth of the groundwater table (Table 1). The sandy soils associated with bulb cultivation are coarse dune sand with a low organic matter content and high groundwater table. The farms of the northeast of the Netherlands are on reclaimed peat soil with relatively high organic matter content.

Sampling of groundwater

Groundwater samples were taken on each farm in the years 2002 through 2004 between April and October. Sixteen individual samples were taken on farms with clay soil, and 48 on farms with sandy soil, except in 2004 when the number of samples was reduced to 24 on farms with bulb cultivation and on two other farms on sandy soils with relatively little variation in nitrate concentrations. The sampling points were evenly distributed over the farms, according to a stratified randomized sampling scheme.

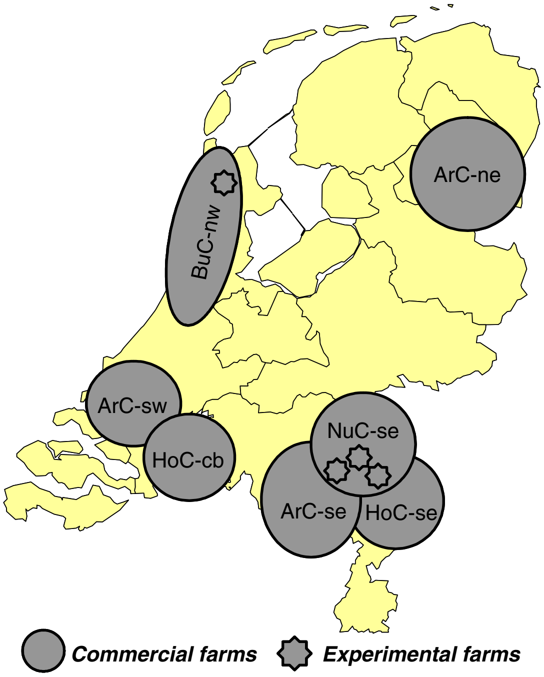

Fig. 1 Location of the farms. See Table 1 for explanation of the code

To sample upper groundwater, holes were made with an auger up to 0.8 m below the groundwater table (Fraters et al. 1998). Subsequently, the groundwater was sampled with a perforated PVC-tube connected to a pump and filter. Nitrate concentrations were determined instantaneously on the spot with test strips and a Nitracheck reflectometer. Two or three strips were used, depending on the variation. More details about monitoring groundwater nitrate are given by Boumans et al. (2001). At each sampling point the actual depth of the groundwater table was determined and presence of peat layers recorded. A peat layer was defined as a layer of at least five cm thickness and over 35% of organic matter. Peat layers were found most frequently in commercial arable farms in the northeast (ArC-ne) of the Netherlands.

Table 1 Type, location and some soil characteristics of the farms involved

Farm type and region	Code	Soil type	Peat layers[a]	OM%[b]	Groundwater[c]	Number of farms
Arable farming						
Commercial, north east	ArC-ne	Sand	36	9.6	145	3
Commercial, south east	ArC-se	Sand	2	3.2	126	4
Commercial, south west	ArC-sw	Clay	0	2.3	133	5
Experimental, south east	ArE-se	Sand	0	3.7	96	1
Horticulture						
Commercial, central Brabant	HoC-cb	Sand	19	4.3	152	4
Commercial, south east	HoC-se	Sand	0	3.1	196	4
Experimental, south east	HoE-se	Sand	0	2.8	345	1
Bulb cultivation						
Commercial, north west	BuC-nw	Sand	3	1.5	72	6
Experimental, north west	BuE-nw	Sand	15	1.1	117	1
Nursery stock production						
Commercial, south east	NuC-se	Sand	0	3.4	215	4
Experimental, south east	NuE-se	Sand	0	2.5	337	1

[a] Expressed as the percentage of the total number of sampling points in the corresponding group where a peat layer occurred within sampling depth

[b] OM% = percentage organic matter in top 30 cm

[c] Mean of the average farm groundwater table at the time of sampling of groundwater (cm below ground level)

Table 2 Groundwater table (Gt) classes

Gt class	MHW[a] (cm)	MLW[a] (cm)
Well drained	> 80	> 120
Reasonably drained	40–80	> 120
Slightly drained	40–120	80–120
Poorly drained	0–40	0–120

[a] MHW = mean highest groundwater table, MLW = mean lowest groundwater table

Groundwater table classes

Groundwater classes in The Netherlands are defined by a combination of the mean highest and the mean lowest groundwater table. For this study, some classes were grouped and four levels were distinguished (Table 2).

Groundwater table classification was carried out in the field for each individual sampling point at the horticultural and arable farms on sandy soil (Finke 2000). Nursery stock production was not included in this classification because of the relative small area. At the arable farms on clay soil and the bulb farms on coarse dune sand groundwater table classes were not determined as nitrate concentrations appeared to be low on these soils.

Nitrogen balances

Total N surpluses ($N_{\text{tot-surplus}}$) per field and crop were calculated as the differences of major inputs (atmospheric deposition, seeds/planting material, biological N fixation, mineral and organic fertilizers including straw from outside) and commercial outputs (harvested crops) between January 1st and December 31st in 2000, 2001, 2002 and 2003. The N input via fertilizers was calculated by multiplication of the amount used and its N content. In the case of manure, the N content of a sample was analyzed per separate batch. The N content of composts was either determined in a sample or retrieved from the supplier. Regional averages on atmospheric deposition were taken from data of RIVM, the National Institute for Public Health and the Environment. Inputs through seed, planting material and straw were calculated by multiplying the amount used by a standard N content per product (Beukeboom 1996). The amount of N fixed by legumes was calculated from crop yields (as recorded) and their 'standard' N content (as tabulated by Beukeboom 1996) and by multiplying the aboveground N-uptake with 4/3 (according to Van Leeuwen-Haagsma and Schröder, 2003). The commercial N output was calculated by multiplying the crop yield with the 'standard' N content per type of crop product (Beukeboom 1996).

In addition to the $N_{tot\text{-}surplus}$, another surplus based on mineral N fluxes ($N_{min\text{-}surplus}$) was calculated (Table 3). Input in $N_{min\text{-}surplus}$ was the amount of N (becoming) available as mineral N from mineral and organic inputs (including cover crops, if applicable) between January 1st and December 31st. Output of $N_{min\text{-}surplus}$ is the N in harvested products, N lost as ammonia from manures, and the N taken up by subsequent cover crops, if applicable. Mineralization of organic N from manure and compost was calculated using a model (Janssen 1984). Mineralization of N from and uptake of N into cover crops was based on standard values used in Dutch recommendations (Van Dijk 2003).

Soil mineral nitrogen

To determine soil mineral nitrogen after harvest (SMN_{ph}), the upper 90 cm was sampled (60 cm on bulb growing farms) in October/November by taking 40 cores per field. Mineral nitrogen was extracted in 1 M KCl, and ammonium and nitrate contents were determined by segmented flow analysis. SMN (kg/ha) was calculated using the bulk density calculated from organic matter content (Whitmore et al. 1992).

Dilution and duration of percolation

The observed nitrate concentration in the upper groundwater in any year ('year *n*') can be linked to measures and resulting explanatory variables (inputs, surpluses, SMN_{ph}'s) in the preceding year ('year $n - 1$'). However, downward movement of nitrate and dilution in the precipitation surplus determines to what extent the nitrate concentration in the upper groundwater is determined by the management of just 'year $n - 1$'. ONZAT (Van Drecht 1983; OECD 1989) was used to calculate an index for dilution and duration of percolation for each measured nitrate concentration (Boumans et al. 2005). Year- and location-specific weather data from KNMI (the Royal Netherlands Meteorological Institute) were used as model input. The index allowed us to attribute the nitrate concentration to the weighted 'effective' values of explanatory variables (indicated by the suffix_eff) of more than the one preceding year.

Explanatory variables and statistical analysis

Various indicators were tested as explanatory variables to account for the observed nitrate concentrations (Table 4). Nitrate concentrations appeared to be low on clay soils and on coarse dune sand (bulb production). Nitrate concentrations were also low when peat layers were present in the profile of sandy soils. In a subsequent analysis, we focused on data from arable farming and horticulture on sandy soils, excluding sampling points with a peat layer. Nursery stock production was excluded because of the relative small area. Relationships between indicators and nitrate concentrations were studied at four levels

Table 3 Inputs and outputs of a balance based on mineral N fluxes ($N_{min\text{-}surplus}$)

N inputs as:	N outputs as
Mineral fertilizer	Harvested products
Part of the N in manure and compost (applied in the year under study), i.e.:	Following cover crops
• The mineral fraction of the N-content	Gaseous losses from applied manures
• The amount of organic N that supposedly mineralized between the day of application and December 31st	
The amount of organic N that supposedly mineralized between January 1st and December 31st from manure and compost applied in the preceding year	
Atmospheric deposition	
Mineralized from cover crops grown in the preceding autumn	
Fixed by legumes[a]	
Seeds and in planting material	

[a] N fixation is no mineral N but this input compensates the N output of leguminous crops

Table 4 Indicators used as explanatory variable for the nitrate concentration in the upper groundwater. When variables are averaged over years, the suffix 'A' is added

Variable	Explanation
SMN_{ph}	Post harvest soil mineral nitrogen in the autumn of the year prior to nitrate sampling (N in 0–90 cm, kg/ha)
$N_{tot\text{-}surplus}$	Total N-surplus (kg/ha) of the balance of all inputs and outputs in the year prior to nitrate sampling (see text for definitions)
$N_{min\text{-}surplus}$	Mineral N-surplus (kg/ha) of the balance of inputs and outputs in the year prior to nitrate sampling (see text for definitions)
SMN_{ph_eff}	Soil mineral nitrogen in autumn corrected for dilution and duration of percolation (year effect)
$N_{tot\text{-}surplus\_eff}$	Total N-surplus (kg/ha) of the balance of all inputs and outputs, corrected for dilution and duration of percolation (year effect)
$N_{min\text{-}surplus\_eff}$	Mineral N-surplus (kg/ha) of balance of inputs and outputs, corrected for dilution and duration of percolation (year effect)
$N_{tot\text{-}input}$	Input of total N (kg/ha) in manure and fertilizers in the year prior to nitrate sampling
$N_{min\text{-}input}$	Input of mineral N (kg/ha) in manure and fertilizers in the year prior to nitrate sampling, (including N that mineralizes during the first year)
Gt-class/Gt-%well drained	'well drained', 'reasonably drained', 'slightly drained' and 'poorly drained' (see Table 2). For analysis of individual sampling points the four classes were used as factor. For analysis of farm averages, the percentage of sampling points in the class 'well drained' was used as variable.

Table 5 Aggregation levels and applied models

Level of aggregation	Fixed effects[a]	Random effects
1: Individual sampling points	Gt-class + variable	(Farm*year)/field/sampling point
2: Individual sampling points, averaged over years	Gt-class + variable-A	Farm/field/sampling point
3: Yearly farm average	Gt-%well drained + variable	Year
4: Yearly farm average, averaged over years	Gt-%well drained + variable-A	–

[a] Variables as in Table 4; Variable-A = averaged over years

of aggregation (Table 5). Averages over years for nitrate were based on the years 2002 through 2004. Averages over years for $N_{tot\text{-}surplus}$, $N_{min\text{-}surplus}$ and SMN_{ph} were based on the years 2000 through 2003. The effective values of $N_{tot\text{-}surplus\_eff}$, $N_{min\text{-}surplus\_eff}$ and SMN_{ph_eff} were calculated with data from the years 2000 through 2003. Farm averages were calculated as the average of the values from the individual sampling points of the farm.

The statistical analysis was carried out with mixed linear models using REML (residual maximum likelihood; Genstat 8 Committee 2005; Table 5) and with regression models. The REML algorithm estimates treatment effects and variance components in a linear model with both fixed and random variables. Like regression, REML can be used to analyse unbalanced data sets; but unlike regression, it can also account for more than one source of variation in the data, providing an estimate of the variance components associated wit the random variables in the model.

The relationships between nitrate in groundwater and different variables are compared using the 'effect' of a variable and the Wald (REML) and *t*-statistic (regression). In the following the Wald statistic is also indicated by the *t*-statistic. The effect of a variable was calculated, using the models, as the difference between the nitrate concentrations corresponding with the 25%-quantile and 75%-quantile of that variable.

Results

Commercial and experimental farms

Figure 2 shows $N_{tot\text{-}surplus}$, SMN_{ph} and nitrate concentrations in groundwater for the different groups of commercial farms and for the experimental farms. For all three variables, the

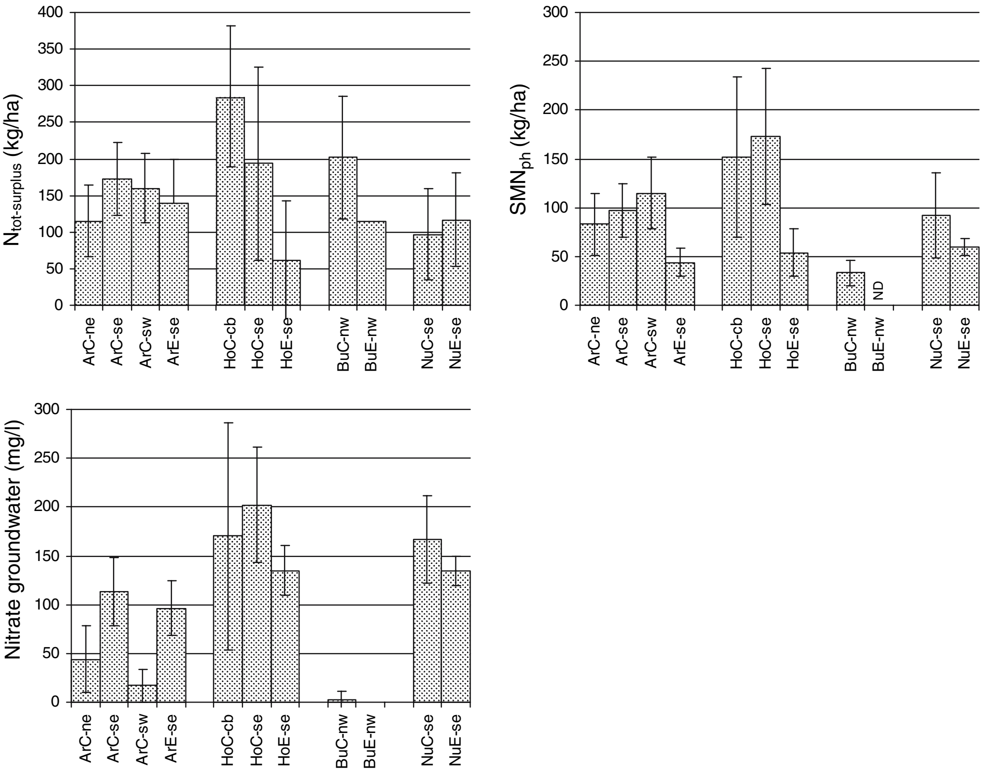

Fig. 2 Mean values per group of commercial farms or experimental farm for average farm levels of $N_{tot-surplus}$ and SMN_{ph} for the years 2000 through 2003 (in kg/ha; 0–60 cm on bulb growing farms, 0–90 cm on all other farms), and mean values of nitrate in groundwater (in mg/l) for the years 2002 through 2004. Error bars indicate one standard deviation. ND = not determined. See Table 1 for explanation of abbreviations

Table 6 Mean values of nitrate in groundwater (NO_3), $N_{tot-surplus}$, $N_{min-surplus}$ and SMN_{ph} for four groundwater table classes and the presence of peat layers

	Well drained			Reasonably drained			Slightly drained			Poorly drained		
	n	Mean	SD	*n*	Mean	SD	*n*	Mean	SD	*n*	Mean	SD
Peat layers absent												
NO_3 (mg/l)	1201	158	92	568	124	97	120	117	96	256	77	89
$N_{tot-surplus}$ (kg/ha)	1120	176	187	496	209	154	110	209	147	213	159	132
$N_{min-surplus}$ (kg/ha)	1120	116	130	496	172	131	110	163	128	213	139	117
SMN_{ph} (0–90 cm, kg/ha)	1098	105	86	498	135	100	100	111	60	214	124	78
Peat layers present												
NO_3 (mg/l)	30	73	86	72	38	53	57	14	21	33	30	62
$N_{tot-surplus}$ (kg/ha)	30	239	89	72	205	105	54	108	68	32	157	108
$N_{min-surplus}$ (kg/ha)	30	167	62	72	131	60	54	82	70	32	93	51
SMN_{ph} (0–90 cm, kg/ha)	30	89	57	66	80	48	54	103	66	30	95	68

Means based on data of arable farming and horticulture on sandy soil. *n* = number of observations (sampling points), SD = standard deviation

horticulture group had a higher standard deviation (variation between farms) than the others.

As for commercial farms, the group with nursery stock production had, on average, the lowest $N_{tot-surplus}$ of almost 100 kg/ha. The group

Table 7 Relationship between indicators and eventual nitrate concentration in upper groundwater for sandy soils without peat layers

(A) Analysis with individual sampling points
Model: NO_3 = $Intercept_{Gt}$ + slope*variable; random = (farm*year)/field/sample

Variable	*n*	Slope	$Intercept_{Gt}$				Effect	*t*-value
			Well drained	Reasonably drained	Slightly drained	Poorly drained		
SMN_{ph}	1910	0.13 (0.04)	126 (11)	101 (12)	108 (14)	47 (13)	13	3.2
$N_{tot\text{-}surplus}$	1939	0.03 (0.02)	132 (12)	110 (12)	118 (14)	58 (14)	3	1.8
$N_{min\text{-}surplus}$	1939	0.08 (0.02)	129 (11)	107 (12)	115 (14)	55 (13)	14	3.2
$SMN_{ph\text{-}eff}$	1136	0.64 (0.13)	108 (16)	90 (16)	110 (19)	49 (18)	27	4.9
$N_{tot\text{-}surplus\text{-}eff}$	1418	0.25 (0.06)	118 (14)	101 (14)	120 (17)	62 (16)	18	4.4
$N_{min\text{-}surplus\text{-}eff}$	1418	0.26 (0.07)	122 (14)	105 (15)	124 (17)	66 (16)	17	3.8
$N_{tot\text{-}input}$	1939	0.01 (0.02)	135 (12)	113 (13)	121 (15)	60 (14)	2	0.5
$N_{min\text{-}input}$	1939	0.06 (0.03)	127 (12)	105 (13)	113 (15)	53 (14)	7	2.0

(B) Analysis with individual sampling points, averaged over years
Model: NO3 = $Intercept_{Gt}$ + slope*variable; random = farm/field/sample

Variable	*n*	Slope	$Intercept_{Gt}$				Effect	*t*-value
			Well drained	Reasonably drained	Slightly drained	Poorly drained		
SMN_{ph}-A	654	0.10 (0.06)	128 (12)	97 (13)	94 (16)	46 (15)	9	3.7
$N_{tot\text{-}surplus}$-A	654	0.05 (0.04)	130 (13)	100 (14)	97 (16)	49 (15)	7	1.9
$N_{min\text{-}surplus}$-A	654	0.10 (0.05)	127 (12)	98 (13)	95 (16)	47 (14)	13	3.1
$SMN_{ph\text{-}eff}$-A	417	0.84 (0.21)	105 (16)	81 (18)	94 (21)	34 (19)	25	6.2
$N_{tot\text{-}surplus\text{-}eff}$-A	477	0.33 (0.11)	113 (15)	89 (16)	103 (19)	47 (17)	18	4.3
$N_{min\text{-}surplus\text{-}eff}$-A	477	0.25 (0.12)	123 (15)	99 (16)	114 (19)	57 (18)	11	3.8
$N_{tot\text{-}input}$-A	649	0.00 (0.04)	137 (14)	108 (14)	105 (17)	56 (16)	0	0.8
$N_{min\text{-}input}$-A	649	0.04 (0.05)	131 (14)	102 (15)	99 (17)	50 (16)	4	1.9

(C) Analysis with farm averages
Model: NO_3 = constant + slope*variable + slope_Gt* (%well drained); random = year

Variable	*n*	Slope	Constant	Slope_Gt	Effect	*t*-value
SMN_{ph}	70	0.69 (0.08)	7 (16)	0.75 (0.13)	43	8.7
$N_{tot\text{-}surplus}$	71	0.27 (0.05)	43 (16)	0.62 (0.16)	33	5.1
$N_{min\text{-}surplus}$	71	0.37 (0.07)	37 (17)	0.72 (0.16)	41	5.2
$SMN_{ph\text{-}eff}$	47	1.25 (0.24)	18 (19)	0.95 (0.19)	42	5.2
$N_{tot\text{-}surplus\text{-}eff}$	65	0.68 (0.16)	36 (19)	0.60 (0.17)	38	4.4
$N_{min\text{-}surplus\text{-}eff}$	65	0.78 (0.20)	43 (19)	0.62 (0.18)	37	4.0
$N_{tot\text{-}input}$	71	0.27 (0.07)	32 (21)	0.58 (0.17)	22	3.8
$N_{min\text{-}input}$	71	0.41 (0.09)	13 (23)	0.64 (0.17)	29	4.3

(D) Analysis with farm averages, averaged over years
Model: NO_3 = constant + slope*variable + slope_Gt*(%well drained); no random effects

Variable	*n*	Slope	Constant	Slope_Gt	Effect	*t*-value
SMN_{ph}-A	23	0.81 (0.09)	− 15 (17)	0.84 (0.15)	48	8.5
$N_{tot\text{-}surplus}$-A	23	0.44 (0.12)	14 (29)	0.67 (0.24)	46	3.8
$N_{min\text{-}surplus}$-A	23	0.49 (0.15)	20 (30)	0.73 (0.26)	47	3.3
$SMN_{ph\text{-}eff}$-A	19	1.76 (0.33)	− 11 (25)	0.92 (0.23)	45	5.4
$N_{tot\text{-}surplus\text{-}eff}$-A	21	0.98 (0.27)	7 (31)	0.69 (0.26)	54	3.6
$N_{min\text{-}surplus\text{-}eff}$-A	21	1.05 (0.34)	20 (32)	0.71 (0.28)	46	3.1
$N_{tot\text{-}input}$-A	23	0.44 (0.14)	− 8 (38)	0.72 (0.27)	40	3.2
$N_{min\text{-}input}$-A	23	0.54 (0.18)	− 15 (42)	0.75 (0.28)	33	3.0

Data of arable farming and horticulture on sandy soil. Between brackets the standard deviation (SD). The effect is calculated from the regression results and is the difference between nitrate concentration corresponding with the 25%-quantile and the 75%-quantile. *n* = number of observations

with horticulture farms in Central Brabant had the highest $N_{tot\text{-}surplus}$ of 285 kg/ha whereas the horticultural experimental farm had the lowest N surplus of 62 kg/ha.

SMN_{ph} was, on average, higher than 150 kg/ha on commercial horticultural farms. SMN_{ph} for the groups with arable and nursery stock production was around 100 kg/ha. SMN_{ph} associated with bulb production was low. The experimental farms all showed a lower $\mathrm{SMN_{ph}}$ than the commercial farms.

Nitrate concentrations in groundwater under bulb production were low and often below the detection limit. Nitrate concentrations of the arable farms were lower than those of horticulture farms or nursery stock production. Among the commercial arable farms, nitrate concentration was lowest on clay soil (ArC-se).

Groundwater table and peat layers on sandy soil

Groundwater table had a clear effect on the nitrate concentration. Nitrate concentrations were highest on well drained soils and lowest on poorly drained soils (Table 6). In the presence of peat layers, nitrate concentrations were on average less than 50% than where peat layers were absent. Peat layers occurred occasionally on well drained soils and more frequently on soils with other groundwater table classes. In general, peat layers were present in less than ten% of the sampling points.

Note that, as opposed to nitrate concentration in groundwater, the mean values of the variables $\mathrm{N_{tot\text{-}surplus}}$, $\mathrm{N_{min\text{-}surplus}}$ and $\mathrm{SMN_{ph}}$ showed relatively small differences between the classes distinguished in Table 6, especially in the absence of peat layers.

Indicators of nitrate in groundwater

As nitrate concentrations often were low on clay soils and on sandy soils where peat layers were present, we restricted the analysis of the predictive value of the various indicators to sandy soils without peat layers. The response of the nitrate concentration to changes of the indicator values did not differ significantly between the four groundwater table classes. Therefore, one slope was fitted for each variable (Table 7A–D). At the aggregation level of individual sampling points, the slope for $\mathrm{SMN_{ph}}$ is 0.13, which corresponds to an increase in nitrate concentration of 13 mg/l per 100 kg/ha $\mathrm{SMN_{ph}}$ (Table 7A). The variables $\mathrm{N_{tot\text{-}surplus}}$ and $\mathrm{N_{min\text{-}surplus}}$ have a weaker slope than $\mathrm{SMN_{ph}}$. These slopes, however, cannot simply be compared because the absolute values of the variables differ in magnitude. For comparison the effect is calculated using the regression equations. The effect is the difference between the nitrate concentrations calculated with the 25%-quantile and the 75%-quantile of a variable. This shows how nitrate is affected by the available range of a variable. The effects of $\mathrm{SMN_{ph}}$ and $\mathrm{N_{min\text{-}surplus}}$ are similar; the effect of $\mathrm{N_{tot\text{-}surplus}}$ is small.

The effect and the *t*-value increased when the effects of dilution and duration of percolation (the suffix '-eff') were included. The effect of $\mathrm{SMN_{ph}}$_eff was larger than the effects of $\mathrm{N_{tot\text{-}surplus\text{-}eff}}$ and $\mathrm{N_{min\text{-}surplus\text{-}eff}}$. The total input of N by fertilizers and manure ($\mathrm{N_{tot\text{-}input}}$) or the availability of mineral N from fertilizers ($\mathrm{N_{min\text{-}input}}$) had a poor correlation with nitrate in groundwater with low effects and *t*-values.

The next aggregation level (averaging of values per sampling point over years) yielded similar relationships as those found for individual years (Table 7B).

The third aggregation level (averaging of data of individual sampling points to the farm level) reduced the number of observations but increased effects and *t*-values of almost all variables (Table 7C). Additional averaging over years at farm level (fourth aggregation level) further increased the effects but reduced *t*-values (Table 7D). At farm level and in individual years, $\mathrm{SMN_{ph}}$ was the variable with the highest effect and the highest *t*-value. At farm level and when averaged over years, effects were equal for $\mathrm{SMN_{ph}}$-A, $\mathrm{N_{tot\text{-}surplus}}$-A and $\mathrm{N_{min\text{-}surplus}}$-A but $\mathrm{SMN_{ph}}$-A had the highest *t*-values. $\mathrm{N_{tot\text{-}input}}$ and $\mathrm{N_{min\text{-}input}}$ had lower effects and lower t-values than $\mathrm{SMN_{ph}}$, $\mathrm{N_{tot\text{-}surplus}}$ and $\mathrm{N_{min\text{-}surplus}}$, both at individual years and averages over years. At farm level, correction for effects of dilution and duration of percolation (indicated by suffix '-eff') increased the effect with $\mathrm{N_{tot\text{-}surplus}}$ and slightly reduced the effects with $\mathrm{SMN_{ph}}$ and $\mathrm{N_{min\text{-}surplus}}$ whereas *t*-values were reduced.

Discussion

Nitrate concentrations in groundwater exceeded the target value of 50 mg/l at 20 of the 34 farms.

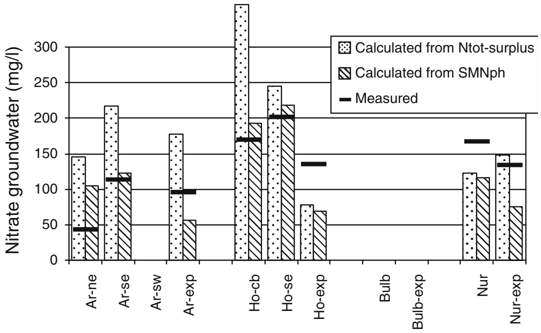

Fig. 3 Calculated nitrate concentrations from $N_{tot\text{-}surplus}$ and SMN_{ph} (calculated by dividing the data from Fig. 2 by an assumed average precipitation surplus of 350 mm) The horizontal lines indicate the measured nitrate concentrations as shown in the bottom left graph of Fig. 2

Both local circumstances and fertilization affected the nitrate concentrations.

Local circumstances

In our study, nitrate concentrations in groundwater were also affected by soil type, and by other local circumstances such as the occurrence of peat layers and the depth of the groundwater table. The low nitrate concentrations on clay soil compared to sandy soil agree with Simmelsgaard (1998) who found a negative relationship between clay content and N leaching. When the average clay content of 25% of our study is used in the regression formula of Simmelsgaard (1998), nitrate concentrations lower than 50 mg/l are calculated.

On sandy soil, the occurrence of peat layers was associated with nitrate concentrations that were—on average—about 80 mg/l less than where peat layers were absent. The reduced nitrate concentration in presence of peat layers can be explained by increased denitrification, facilitated by the breakdown of organic matter under anaerobic circumstances (McCarty and Bremner 1992). The stimulation of denitrification by breakdown of peat layers (providing a C-source) in the subsoil is not ever lasting. Comparison of data of a recent survey in the Netherlands with data from 20–40 years ago showed that peat layers were indeed smaller or had disappeared (Velthof et al. 2004).

Sandy soils with high groundwater tables (poorly drained) had nitrate concentrations that were less than half of those at sandy soils with low groundwater tables (well drained). This corresponds with results of Boumans et al. (2005). As with the effect of peat layers, the effect of groundwater tables can be explained by differences in denitrification. At a shallow groundwater table, leached dissolved organic matter (DOM) may reach the groundwater and stimulate denitrification. At a deep groundwater table, however, DOM is decomposed aerobically before it reaches the groundwater (Starr and Gillham 1993).

Indicators of nitrate in groundwater

Denitrification in clay soils and in soils where peat layers are present, results in reduced nitrate concentrations in groundwater and makes it easier to achieve nitrate concentrations below 50 mg/l. Therefore, we focused on sandy soils without peat layers to study the relationship between N management and nitrate concentrations in groundwater.

$N_{tot\text{-}surplus}$ poorly correlated with nitrate concentrations in groundwater when individual sampling points were studied. An explanation can be that changes in the soil N pool were not taken into account. Especially at the aggregation level of individual sampling points and individual years, changes in the soil N pool occur because consecutive crops add different amounts of organic matter to the soil, and crops receive different amounts of organic manure. In the course of years, increases in the soil N pool will be alternated by decreases. Averaging over years would therefore reduce variability and strengthen the relationship between $N_{tot\text{-}surplus}$ and nitrate in groundwater (Oenema et al. 2003; Van Beek et al. 2003). However, this was not found at the level of individual sampling points. The correlation between $N_{tot\text{-}surplus}$ and nitrate in groundwater clearly increased when data were averaged at the farm level. Averaging at the farm level also reduced the variability within a single year. As crops are grown in a rotation, overestimation at one field-crop combination is counterbalanced with an underestimation at another field-crop combination. A possible explanation for the better relationship between $N_{tot\text{-}surplus}$ and nitrate in groundwater at the farm level compared to the level of individual sampling points, is that at farm level the complete crop rotation is taken into account, whereas the three or four year averages of individual sampling points do not represent the complete crop rotation.

We expected that $N_{min\text{-}surplus}$ would correspond better with nitrate in groundwater than $N_{tot\text{-}surplus}$, especially within individual years. However, when both variables were corrected for dilution and duration of percolation, the effects of $N_{min\text{-}surplus}$ and $N_{tot\text{-}surplus}$ were similar. Apparently there were too many omissions in the calculation of $N_{min\text{-}surplus}$. For example, mineralization from the soil N pool was not accounted for, whereas its contribution to the mineral N pool may have exceeded the annual N immobilization. Especially some farms are in a transition phase towards lower input levels. This can be derived from data of Fig. 2 and an assumed average precipitation surplus of 350 mm (Fig. 3). A net mineralization from the soil N pool will have occurred at the horticultural experimental farm and the nursery stock production group, as the calculated nitrate concentrations from $N_{tot\text{-}surplus}$ or SMN_{ph} and the precipitation surplus were lower than the measured nitrate concentrations.

Processes of mineralization from and immobilization into the pool of organic N in the soil are included in the value of SMN_{ph}. SMN_{ph} is a measurement of mineral N that is susceptible to leaching and is therefore closer related to nitrate in groundwater than $N_{tot\text{-}surplus}$ and $N_{min\text{-}surplus}$. Of the studied variables, SMN_{ph} had the highest *t*-values and often also the highest effects. Although SMN_{ph} is affected by temperature and precipitation in autumn (Schweigert et al. 2004), the effects of management including fertilizer use are evident.

Policy implications

The Dutch national action plan aims at reducing nutrient emissions to achieve 'good water quality for all purposes'. Recently introduced legislation is based on soil type and crop type-specific N standards (i.e. fixed N application rates) per crop. N standards are less related to nitrate concentrations in groundwater than N surpluses (Schröder et al. 2003). Whether or not an indicator can be used to achieve a desired nitrate concentration does not only depend on the type of indicator, but also on the maximum values of that indicator allowed. At present, the crop-specific N standards are based on the current N recommendations. The high nitrate concentrations on sandy soils, especially those with low groundwater tables, indicate that the N standards will have to be cut to levels below the N recommendations. However, reduced N inputs do not automatically result in proportionally lower N surpluses or nitrate concentrations. Reductions of inputs could lead to reduced N outputs to some extent, so that the eventual effect on the N surplus or nitrate concentration is lower than initially was expected. According to the regression data from Table 7D, nitrate concentrations in groundwater of 50 mg/l are achieved with a surplus of only 80 kg/ha for poorly drained soils. For well drained soils, already the intercept (at zero surplus) surpasses the level of 50 mg nitrate per liter groundwater. Theoretically it is impossible to have a high nitrate concentration at zero values of $N_{tot\text{-}surplus}$. It indicates that a steady state of mineralization and immobilization has not yet been reached.

The need to reduce the N standards to attain good water quality at the regional level will also depend on the contribution of agriculture in that region, the contribution of other types of land use and the soil type or groundwater tables. As agriculture in the Netherlands is the major type of land use, drastic measures will be inevitable on well-drained sandy soils. Such measures could consist of changes in land use (adjusted crop rotations or even set-asides, buffer strips, creation of wetlands, etc.), reduced N inputs (bluntly or facilitated by an improved fertilizer use efficiency), or combinations of these measures. However, many measures to reduce nitrate leaching may have extra costs or may increase the risk of reduced yields or reduced crop quality. These aspects slowed down the implementation of measures in Denmark (Grant and Blicher-Mathiesen 2004).

The Water Framework Directive (EC 2000) may require additional reduction of threshold values of indicators. Sooner or later groundwater may become surface water, and desired N concentrations for surface water are lower than for groundwater. Moreover, water quality is not only determined by the nitrate concentration. The bulb cultivation on coarse dune sand had sufficiently low concentrations of nitrate in groundwater, but the groundwater also contained N as ammonia and as dissolved organic N. Moreover, P concentrations in groundwater of bulb cultivation on coarse dune sand were high.

References

Beukeboom JA (1996) Kiezen uit gehalten 3. Forfaitaire gehalten voor de mineralenboekhouding. IKC-landbouw, Ede. (in Dutch)

Boumans LJM, Fraters B, Van Drecht G (2001) Nitrate in the upper groundwater of 'De Marke' and other farms. Neth J Agric Sci 49:163–177

Boumans LJM, Fraters D, Van Drecht G (2005) Nitrate leaching in agriculture to upper groundwater in the sandy regions of the Netherlands during the 1992–1995 period. Environ Monit Assess 102:225–241

CBS (2006) http://statline.cbs.nl

D'Haene K, Moreels E, De Neve S, Daguilar BC, Boeckx P, Hofman G, Van Cleemput O (2003) Soil properties influencing the denitrification potential of Flemish agricultural soils. Biol Fertil Soils 38:358–366

EC (1991) Council directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources. Official Journal L 375, 31.12.1991, pp 1–8. European Communities, Brussels

EC (2000) Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Official Journal L 327, 22/12/2000 pp 01–73. European Communities, Brussels

Elmi AA, Madramootoo C, Egeh M, Liu AG, Hamel C (2002) Environmental and agronomic implications of water table and nitrogen fertilization management. J Environ Qual 31:1858–1867

Finke PA (2000) Updating the (1:50,000) Dutch groundwater table class map by statistical methods: an analysis of quality versus cost. Geoderma 97:329–350

Fraters D, Boumans LJM, van Drecht G, de Haan T, de Hoop WD (1998) Nitrogen monitoring in groundwater in the sandy regions of the Netherlands. Environ Pollut 102:479–485

Fraters D, Hotsma PH, Langenberg VT, Van Leeuwen TC, Mol APA, Olsthoorn CSM, Schotten CGJ, Willems WJ (2004) Agricultural practice and water quality in the Netherlands in the 1992–2002 period. Background information for the third EU Nitrates Directive Member States report. RIVM-report 500003002, Bilthoven, the Netherlands

Genstat 8 Committee (2005) Genstat Release 8 Reference Manual, Part 2: directives. VSN International, Wilkinson House, Jordan Hill Road, Oxford, UK

Grant R, Blicher Mathiesen G (2004) Danish policy measures to reduce diffuse nitrogen emissions from agriculture to the aquatic environment. Water Sci Technol 49:91–99

Janssen BH (1984) A simple method for calculating decomposition and accumulation of 'young' soil organic matter. Plant Soil 76:297–304

Langeveld JWA, van Keulen H, de Haan JJ, Kroonen Backbier BMA, Oenema J (2005) The nucleus and pilot farm research approach: experiences from The Netherlands. Agric Syst 84:227–252

LEI (2006) Agricultural and horticultural figures 2006 (Land-en tuinbouwcijfers 2006) LEI-Wageningen UR, Centraal Bureau voor de Statistiek (CBS), 's Gravenhage (In Dutch)

Matson PA, Parton WJ, Power AG, Swift MJ (1997) Agricultural intensification and ecosystem properties. Science 277:504–509

McCarty GW, Bremner JM (1992) Availability of organic carbon for denitrification of nitrate in subsoils. Biol Fertil Soils 14:219–222

OECD (1989) Compendium of environmental exposure assessment methods for chemicals. OECD Environ Monogr 22:181–188

Oenema O, Boers PCM, van Eerdt MM, Fraters B, van der Meer HG, Roest CWJ, Schröder JJ, Willems WJ (1998) Leaching of nitrate from agriculture to groundwater: the effect of policies and measures in the Netherlands. Environ Pollut 102:471–478

Oenema O, Kros H, de Vries W (2003) Approaches and uncertainties in nutrient budgets: implications for nutrient management and environmental policies. Euro J Agron 20:3–16

Oenema O, van Liere L, Plette S, Prins T, van Zeijts H, Schoumans O (2004) Environmental effects of manure policy options in The Netherlands. Water Sci Technol 49:101–108

Schröder JJ, Aarts HFM, ten Berge HFM, van Keulen H, Neeteson JJ (2003) An evaluation of whole farm nitrogen balances and related indices for efficient nitrogen use. Eur J Agron 20:33–44

Schröder JJ, Scholefield D, Cabral F, Hofmans G (2004) The effects of nutrient losses from agriculture on ground and surface water quality: the position of science in developing indicators for regulation. Environ Sci Policy 7:15–23

Schweigert P, Pinter N, van der Ploeg RR (2004) Regression analyses of weather effects on the annual concentrations of nitrate in soil and groundwater. J Plant Nutr Soil Sci 167:309–318

Simmelsgaard SE (1998) The effect of crop, N-level, soil type and drainage on nitrate leaching from Danish soil. Soil Use Manage 14:30–36

Smith VH (2003) Eutrophication of freshwater and coastal marine ecosystems – a global problem. Environ Sci Pollut Res 10:126–139

Starr CR, Gillham RW (1993) Denitrification and organic carbon availability in two aquifers. Ground Water 31:934–947

Van Beek CL, Brouwer L, Oenema O (2003) The use of farmgate balances and soil surface balances as estimator for nitrogen leaching to surface water. Nutr Cycling Agroecosyst 67:233–244

Van Dijk W (2003) Adviesbasis voor de bemesting van akkerbouw- en vollegrondsgroentegewassen. Praktijkonderzoek Plant & Omgeving, Lelystad (in Dutch)

Van Drecht G (1983) Simulation of vertical non-stationary transport of water and solute. Mededeling No 83–11, RID, Leidschendam (In Dutch)

Van Leeuwen-Haagsma WK, Schröder JJ (2003) Groenbemesters en rustgewassen. In: Op weg naar goede biologische praktijk; resultaten en ervaringen uit BIOM. Wijnands FG, Holwerda J (eds) Praktijkonderzoek Plant en Omgeving. Lelystad, pp 105–122

Velthof GL, Van Beek CL, Brouwer F, Burgers SLGE, Fraters B, Groenendijk P, Hack-tenBroeke MJD, Van Kekem AJ, Oosterom HP, Schoumans OF, De Vries F, Willems WJ, Zwart KB (2004) Denitrificatie in de zone tussen bouwvoor en het bovenste grondwater in zandgronden. Alterra rapport 730.1, Wageningen

Watson CA, Atkinson D (1999) Using nitrogen budgets to indicate nitrogen use efficiency and losses from whole farm systems: a comparison of three methodological approaches. Nutr Cycling Agroecosyst 53:259–267

Whitmore AP, Bradbury NJ, Johnson PA (1992) Potential contribution of plowed grassland to nitrate leaching. Agric Ecosyst Environ 39:221–233