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ORIGINAL ARTICLE

Use of long-term monitoring data to derive a relationship between nitrogen surplus and nitrate leaching for grassland and arable land on well-drained sandy soils in the Netherlands

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The decrease in nitrogen (N) use in agriculture led to improvement of upper groundwater quality in the Sand region of the Netherlands in the 1991–2009 period. However, still half of the farms exceeded the European nitrate standard for groundwater of 50 mg/l in the 2008–2011 period. To assure that farms will comply with the quality standard, an empirical model is used to derive environmentally sound N use standards for sandy soils for different crops and soil drainage conditions. Key parameters in this model are the nitrate-N leaching fractions (NLFs) for arable land and grassland on deep, well-drained sandy soils. NLFs quantify the fraction of the N surplus on the soil balance that leaches from the root zone to groundwater and this fraction represents N available for leaching and denitrification. The aim of this study was to develop a method for calculating these NLFs by using data from a random sample of commercial arable farms and dairy farms that were monitored in the 1991–2009 period. Only mean data per farm were available, which blocked a direct derivation of NLFs for unique combinations of crop type, soil type and natural soil drainage conditions. Results showed that N surplus leached almost completely from the root zone of arable land on the most vulnerable soils, that is, deep, well-drained sandy soils (95% confidence interval of NLF 0.80–0.99), while for grassland only half of the N surplus leached from the root zone of grassland (0.39–0.49). The NLF for grassland decreased with 0.015 units/year, which is postulated to be due to a decreased grazing and increased year-round housing of dairy cows. NLFs are positively correlated with precipitation surplus (0.05 units/100 mm for dairy farms and 0.10 units/100 mm for arable farms). Therefore, an increase in precipitation due to climate change may lead to an increase in leaching of nitrate.

Keywords: nitrate; nitrogen leaching fraction; regional scale approach; root zone leaching; sandy soils; upper groundwater

Introduction

Nitrogen (N) is one of the most important plant nutrients and is essential for good crop production. However, the excessive use of N fertilisers in agricultural systems during the twentieth century has led to the widespread pollution of groundwater and surface water with nitrate (Strebel et al. 1989; Stoate et al. 2009). Since the mid-1980s, the agricultural use of N fertilisers has decreased in many countries of the European Union (EU) due to a combination of agricultural policies, such as the establishment of the milk quota in 1984 (Alliance Environment 2008), and environmental legislation, including the implementation of the Nitrates Directive in 1991 (Van Grinsven et al. 2012; Velthof et al. 2014). A decrease in N fertiliser use is also observed in the agricultural sector in the Netherlands (Baumann et al. 2012). The result has been a decrease in soil N surplus and, consequently, a decrease in nitrate concentrations in the upper groundwater of farms in the Sand region of the Netherlands, from an average of about 140 mg/l in 1992 to about 60 mg/l in 2010 (Baumann et al. 2012). Nevertheless, the EU nitrate standard for

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groundwater of 50 mg/l was exceeded at about 50% of the farms in this region in the 2008–2011 period (Baumann et al. 2012). The environmental goals set by the Nitrates Directive (European Commission 1991) necessitate a further restriction of N use in agriculture in the Sand region of the Netherlands. N use is regulated by setting standards for each combination of crop, soil type and soil drainage condition (Schröder & Neeteson 2008). To assure that farms in the Sand region will comply with the nitrate standard for groundwater in the future, an empirical model is used to derive environmental sound N use standards for sandy soils for different crops and soil drainage conditions (Schröder et al. 2007). Key parameters in this model are the nitrate-N leaching fractions (NLFs) for arable land and grassland.

NLFs in this model quantify the fraction of the N surplus on the soil balance that leaches as nitrate-N from the root zone to groundwater. The N surplus represents only N available for leaching and N available for denitrification, as net mineralisation and ammonia volatilisation are included in the soil balance. Nitrate-N accounted for about 85% of the total N concentration in the upper groundwater of farms in the Sand region in the Netherlands (Baumann et al. 2012). In the Sand region, the upper metre of groundwater is defined as water leaching from the root zone (Baumann et al. 2012). NLFs are used in more models (Velthof et al. 2009; Owens et al. 2012), but NLFs used in models in the USA and the UK related N use instead of N surplus to nitrate leaching from the root zone (Owens et al. 2012). NLFs are normally derived using data from lysimeter plot and field experiments (Simmelsgaard 1998; Wachendorf et al. 2004, 2006; Barton et al. 2009; Perego et al. 2012) or model calculations (Dalgaard et al. 2006; Owens et al. 2012).

The aim of this study was to develop a method for calculating NLFs by using data from a random sample of commercial arable farms and dairy farms that were monitored in the framework of a national monitoring programme in the 1991–2009 period. NLFs were calculated for arable land and grassland on deep, well-drained sandy soils – the most vulnerable soils for nitrate leaching in the Netherlands. The empirical model calculates NLFs for sandy soils with a different drainage condition by multiplying the NLFs for deep, well-drained sandy soils by a factor that relates drainage condition to nitrate leaching. These factors are derived from an in-depth study at 10 dairy farms in the Sand region (Boumans et al. 1989). The advantage of this approach is that these key model parameters are based on real-life situations and this may increase acceptance by farmers of the environmental sound N use standards derived with the empirical model.

There are limitations associated with the use of data from commercial farms, as there are with the use of data from field experiments. For field experiments, upscaling is a challenge. For whole-farm data, for example, as registered and collected within the framework of the national monitoring programme, the challenge is to derive parameters for models, as data do not provide concrete, direct information on N use and/or nitrate concentration in groundwater for unique combinations of crop, soil type and drainage conditions. On dairy farms in the Sand region, for example, forage maize is grown in fields adjacent to grassland; farm average nitrate concentration is, therefore, due to the combined contributions of both crops. A second limitation is that soil types and natural soil drainage conditions may differ significantly from one farm to another as well as between fields within any one farm. This is exemplified in the northern part of the Sand region where organic matter-rich peat and reclaimed peat soils frequently occur in combination with organic matter-poor sandy soils. Peats in this region are commonly raised bogs on sandy soils. Most bogs were originally excavated to a large extent for fuel production; the remaining lower peat layer was then mixed with the poor, Pleistocene sand to improve the soil structure (Van der Veer 2006).

This study discusses the method developed to calculate NLFs and addresses the following questions with the aim of underpinning the NLFs used to derive environmentally sound N use standards in the Netherlands:

(1) Does the soil N surplus affect nitrate-N leaching from the root zone of sandy soils?
(2) Which fraction of the soil N surplus leaches from the root zone and how are nitrate-N leaching losses affected by land use (arable land versus grassland)?
(3) Do NLFs changes in time and which factors may cause such a change?

Materials and methods

Study area

The Sand region, one of the four major soil type regions (Figure 1), consists of several districts that occur in the eastern and southern part of the Netherlands with the exception of the dunes in the west. The landscape is chiefly flat to slightly undulating. Most soils used for agriculture are formed in aeolian, periglacial, Pleistocene sand deposits (east and south) and in Holocene peats developed on sandy deposits underlain by glacial bolder clay deposits (north east). The climate is moderate marine west coast with mean annual precipitation

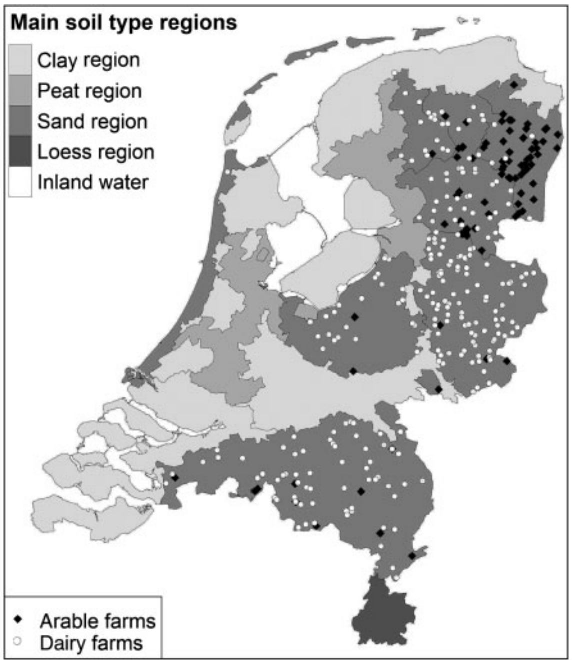

Figure 1. Locations of arable farms and dairy farms monitored in the Sand region in the Netherlands in one or more years in the 1991–2009 period.

per weather districts ranging from 770 mm/year in the southeast up to 890 mm/year in the central part of the country (KNMI 2014a) and a mean winter temperature of 3°C and summer temperature of 17 °C (KNMI 2014b). The Sand region covers about 46% of the Dutch agricultural area that totals almost 2 million ha. Dairy farms (48%), arable farms (16%) and other grazing farms (15%) are the largest land users in the Sand region (De Goffau et al. 2012).

Data on farm practices and water quality

Data were collected on commercial farms located in the Sand region which participated in the national monitoring programme to show the effectiveness of the Nitrate Directive action programmes and the national minerals policy, the Minerals Policy Monitoring Programme (LMM), during the 1991–2009 period (Figure 1; Tables 1 and 2). No groundwater sampling was carried out related to farm practices on arable farms in 1993 due to the use of budget for intensification of research on dairy farms. No groundwater sampling was carried out at all related to farm practice in 1995 due to an evaluation and redesigning of the programme. The methodology was reported elsewhere (Fraters et al. 1998, 2005; De Goffau et al. 2012), this includes selection of farms, the collection of data, the sampling of the upper metre of groundwater and the chemical analyses. The farms were a stratified random sample of arable farms and dairy farms in the Sand region. The upper metre of groundwater in this region, sampled in the period April–October represents water leached from the root zone reflecting mainly the agricultural practices of the previous agricultural year (Boumans et al. 2001). However, Verloop et al. (2006) showed that the nitrate concentration in the upper groundwater was also influenced by

Table 1. Number of arable farms, main characteristics per growing season, precipitation characteristics and nitrate concentration in the upper metre of groundwater the following year.

Year	Number of farms	N surplus (kg/ha)	Area under grass (%)	$f_{ors[y]}$[a] (–)	$Pn50_{[y]}$[b] (mm/year)	$Pr_{[y]}$[c] (–)	NO_3[d] (mg/l)
1991	16	164	0	0.71	340	1.48	132
1992	19	179	0	0.70	342	1.57	140
1994	16	170	0	0.67	341	0.49	63
1996	9	141	0	0.73	350	1.28	56
1997	11	138	2	0.67	343	1.67	109
1998	8	176	2	0.65	346	0.75	40
1999	8	122	1	0.40	338	0.93	75
2002	14	120	3	0.47	340	0.79	54
2003	13	132	3	0.38	331	1.25	79
2004	10	120	0	0.56	346	1.09	79
2005	12	118	2	0.50	342	1.18	76
2006	28	152	1	0.53	342	1.34	101
2007	30	139	1	0.48	340	0.88	84
2008	29	138	1	0.48	339	1.02	66
2009	27	133	1	0.50	338	1.36	80

[a]Fraction of organic-rich soils.
[b]Year specific long-term median precipitation surplus based on fractions of crop types, soil types and GRCs (overlays with maps) and crop type, soil type and GRC specific long-term median precipitation surplus (Van Bakel et al. 2008).
[c]Year specific relative precipitation surplus, high figure indicates sampling of more concentrated leachate (Boumans et al. 2001).
[d]Nitrate concentration measured in summer period of the next year (Fraters et al., 1998).

Table 2. Number of dairy farms, main characteristics per growing season, precipitation characteristics and nitrate concentration in the upper metre of groundwater the following year.

Year	Number of farms	N surplus (kg/ha)	Area under grass (%)	$f_{ors[y]}$[a] (–)	$Pn50_{[y]}$[b] (mm/year)	$Pr_{[y]}$[c] (–)	NO_3[d] (mg/l)
1991	55	315	80	0.13	301	1.66	196
1992	54	339	78	0.13	301	1.57	184
1993	22	329	81	0.23	299	0.54	76
1994	46	343	77	0.16	302	0.60	85
1996	13	301	70	0.12	304	1.33	160
1997	16	260	64	0.19	310	1.83	149
1998	16	268	64	0.14	309	0.77	77
1999	22	253	73	0.13	303	0.91	83
2001	24	191	63	0.16	312	0.86	50
2002	31	187	73	0.16	305	0.81	46
2003	53	174	75	0.18	305	1.31	64
2004	59	179	69	0.19	309	1.21	56
2005	101	208	71	0.18	307	1.20	57
2006	112	190	76	0.17	303	1.32	63
2007	100	178	77	0.18	303	0.96	50
2008	110	166	75	0.17	304	1.05	46
2009	105	191	76	0.18	304	1.50	53

[a]Fraction of organic-rich soils.
[b]Year specific long-term median precipitation surplus based on fractions of crop types, soil types and GRCs (overlays with maps) and crop type, soil type and GRC specific long-term median precipitation surplus (Van Bakel et al. 2008).
[c]Year specific relative precipitation surplus, high figure indicates sampling of more concentrated leachate (Boumans et al. 2001).
[d]Nitrate concentration measured in summer period of the next year (Fraters et al. 1998).

agricultural practices of earlier years for a forage maize–grass rotation at an experimental farm with groundwater level between 1 and 3 m below soil surface level (m-SSL). The average groundwater level at sampling was 1.4 m-SSL and annual mean values varied between 0.6 and 1.7 m-SSL.

Grassland covered on average 1.2% of the total agricultural area on the LMM arable farms in the Sand region (Table 1), while 73% of the area of dairy farms was covered in grassland during the study period (Table 2). Arable land on dairy farms was mainly used for forage maize (*Zea mays* L.) production. Grassland was intensively managed and, in general, fields were used for grazing as well as cutting. Grazing was usually restricted to daytime only. The nitrate concentration in the upper metre of groundwater was on average 80 mg/l for arable farms and 88 mg/l for dairy farms, and the concentrations decreased for both farm types during the monitoring period (Tables 1 and 2).

Additional data

National default values were used when farm-specific data were lacking for specific parameters, such as those on the N content of crops. Precipitation and evapotranspiration were available as 10-day average values per district (Royal Dutch Meteorological Institute).

For each farm, the distribution of different groundwater depth regime classes (GRCs) – an indicator for the soil drainage condition – and soil types were determined by means of overlays of the farm area with the national GRC map and the soil map (Boumans et al. 2005). In total, 11 GRCs were distinguished on the map (Table 3). GRCs were

Table 3. Fraction of groundwater depth regime classes at arable farms and dairy farms monitored in the Sand region; mean value and standard deviation for 1991–2009 period.

Farm type		Groundwater depth regime class										
		1	2	3	4	5	6	7	8	9	10	11
Arable	Mean	0.00	0.02	0.00	0.05	0.20	0.14	0.07	0.12	0.28	0.10	0.03
	SD	0.00	0.03	0.00	0.03	0.08	0.05	0.05	0.07	0.07	0.04	0.03
Dairy	Mean	0.00	0.06	0.01	0.19	0.12	0.04	0.13	0.09	0.24	0.09	0.03
	SD	0.00	0.03	0.01	0.02	0.03	0.02	0.02	0.03	0.03	0.03	0.01

Table 4. Ratio of soil types at arable farms and dairy farms monitored in the Sand region; mean value and standard deviation for 1991–2009 period.

		Soil type			
Farm type		Sand	Clay	Reclaimed	Peat
Arable	Mean	0.43	0.02	0.28	0.27
	SD	0.11	0.02	0.08	0.07
Dairy	Mean	0.80	0.03	0.10	0.07
	SD	0.03	0.02	0.03	0.02

classified based on the combination of long-term average lowest (ALG) and highest groundwater level (AHG) in a hydrological year (April–April) (Locher & De Bakker 1993). GRCs 1–6 are commonly natural poorly drained soils with shallow groundwater levels (ALG <1.20 m), and GRCs 10 and 11 are well-drained soils (AHG >0.80 m). GRCs 7–9 are moderately drained soils with intermediate groundwater levels (ALG >1.20 m and AHG <0.80 m).

Four main soil types were distinguished based on clay and organic matter content: (1) sand and loess soils, (2) clay soils, (3) reclaimed peat soils and (4) peat soils (Table 4). The fraction of soils rich in organic matter was defined as the fraction of reclaimed peat soils and peat soils relative to the total farm area, and this fraction differed between years (Tables 1 and 2). On average, arable farms had more organic matter-rich soils than dairy farms (Table 4).

Calculations

Nitrate-N leaching fraction

NLFs were calculated per farm type per year as the ratio of nitrate-N leaching (kg N/ha/year) and the N surplus on the soil balance (kg N/ha/year), that is, NLF = N leaching/N surplus. Only farms with all necessary data available were used in calculations. Both N leaching and N surplus were averaged over all farms per farm type and per year. If there were too few farms of a certain type per year then no average surplus and leaching were calculated in order to suppress the effect of outliers. The cut-off point was arbitrarily set to fewer than seven farms as this is the number often used in LMM reports and farm accountancy data in the Netherlands.

NLFs per crop type (arable crops and grass) per year were derived from NLFs per farm type, under the assumption that the NLF for arable crops was equal to the NLF for arable farms, and that the NLF for grassland was equal to the NLF for dairy farms, accounting for the percentage of arable land (100 – %grassland; Table 2, 4th column), while assuming that N load and N leaching for the arable land on dairy farms in a specific year were equal those on arable farms in that year.

N surplus

The N surplus on the soil balance of each farm was calculated as described by Baumann et al. (2012). In short, the N surplus on the soil balance was calculated by adding the following items to the N surplus on the farm gate balance: net N mineralisation for organic matter-rich soils (long-term annual average), atmospheric N deposition (average data per province per year) and biological N fixation by legumes; NH_3-N losses were then subtracted from the result (Fraters et al. 2007). The NH_3-N losses concern the collective loss of ammonia-N from inorganic fertiliser and from manure from the housing, from the storage, during grazing and during mechanical application. The surplus on the farm gate balance was calculated as the difference between input (mainly N in imported fertiliser, concentrates, fodder, organic manures and crop products) and output (mainly N in exported animal products, animals, organic manures, crops and other crop products). The N input and output of specific products were calculated by multiplying the registered amount of the product with a fixed N content. Differences in stock supplies between years were accounted for in the calculations.

Nitrate-N leaching

Nitrate-N leaching (N_{leach}, kg N/ha/year) was calculated for the reference soil, that is, a deep, well-drained sandy soil (GRC = 11), using average data per farm type per year. N_{leach} is equal to the nitrate concentration in upper groundwater of this reference soil (NO_{3ref}, mg NO_3/l) multiplied by the year specific precipitation surplus or recharge ($R_{[y]}$, mm/year) and a factor accounting for differences in dimensions, that is:

$$N_{leach[y]} = NO_{3ref[y]} \times R_{[y]} \times 2.26 \times 10^{-3} \qquad (1)$$

NO_{3ref} was the sum of (1) the mean measured concentration ($NO_{3measured}$, mg/l; Tables 1 and 2, last column) divided by an average relative concentration factor ($\hat{C}_{GRC}$) accounting for the influence of the GRC on the nitrate concentration and (2) a factor accounting for the influence of the presence of organic-rich soils (C_{ORS}) multiplied by the fraction of these soils (f_{ors}; Tables 1 and 2, 5th column), that is:

$$NO_{3ref[y]} = NO_{3measured[y]}/\hat{C}_{GRC} + C_{ORS} \times f_{ors[y]} \quad (2)$$

$\hat{C}_{GRC}$ was the sum of the multiplications of the experimentally derived relative concentration factors for individual GRCs (Table 5; Boumans et al. 1989) and the fraction of those GRCs obtained via overlays of the farm area and the GRC map. C_{ORS} (0.0081 ± 0.0045 kg/m^3) was determined by regression analysis using the restricted maximum likelihood (REML) procedure of GENSTAT (Payne 2007) and individual farm data for nitrate concentration, fractions of soil drainage classes (three groups of GRCs; see 'Additional data'), fraction of organic matter-rich soils and relative precipitation surplus.

Recharge

Annual recharge ($R_{[y]}$) was calculated using both model calculations and field data. The long-term median precipitation surplus was calculated with a hydrological model for 3 crop types (grass, forage maize and arable crops), 5 soil types (sand, loess, clay, peat and reclaimed peat) and 11 GRCs using weather data for the 1970–2000 period (Van Bakel et al. 2008). The year specific long-term median precipitation surplus ($Pn50_{[y]}$) takes into account the differences between year in ratios of crop types, soils types and GRCs on farms in the sample. $R_{[y]}$ was calculated by multiplying the year specific long-term median precipitation surplus ($Pn50_{[y]}$, mm/year; Tables 1 and 2, 6th column) by the year specific relative precipitation surplus ($Pr_{[y]}$; Tables 1 and 2, 7th column) and by a factor (C_{it}). This factor is to assure that the overall average value of year specific long-term median precipitation surplus ($Pn50_{[y]}$) for the study period (1991–2009) was equal to the overall average value of the calculated annual recharge ($R_{[y]}$) for this period. C_{it} was 1.007167 for arable farms and 1.021264 for dairy farms. In summary, the year specific precipitation surplus per farm type was calculated as:

$$R_{[y]} = Pn50_{[y]} \times Pr_{[y]} \times C_{it} \quad (3)$$

$Pn50_{[y]}$ was the sum of the multiplications of a crop type, soil type and GRC specific long-term median precipitation surplus and the fractions of crop type, soil type and GRCs present on the farms in a specific year.

$Pr_{[y]}$ was calculated by averaging per farm type per year the farm averages of the monitoring point specific relative precipitation surpluses. These were calculated with a hydrological model, national available weather data and monitoring data on groundwater levels at farms during sampling as described by Boumans et al. (2001) and they reflect the influence of the precipitation and evapotranspiration on the measured nitrate concentration.

Results and discussion

The NLF for a deep, well-drained sandy soils (GRC = 11) at arable farms, that is, for arable land, with mean N surpluses between 115 and 180 kg/ha is 0.90 (95% confidence interval of NLF is 0.80–0.99) (Figure 2). The NLF for dairy farms with mean N surpluses between 165 and 343 kg/ha is 0.52 (0.48–0.56) (Figure 2). The NLF for grassland on deep, well-drained sandy soils is 0.44 (0.39–0.49). This means that on average 90% of the N surplus leaches as nitrate to groundwater at arable fields on soils most vulnerable to nitrate leaching, and that the leaching percentage of the N surplus at grassland fields on those types of soils is half of the N leaching at arable fields. NLFs for other drainage conditions (GRC 1–10) are calculated by multiplying the NLF for GRC 11 with the relative nitrate concentration factors given in Table 5.

The difference in leaching between arable and grassland reported here is in agreement with results reported in other studies. Higher leaching for arable land was reported by Simmelsgaard (1998) based on data collected on 22 fields situated on sandy soils to sandy clay loams in Denmark during a 6- to 21-year study period. For a standardised situation (precipitation surplus of 403 mm/year, soils with 12% clay and N level of 168 kg/ha/year), Simmelsgaard (1998) calculated a leaching of 10% of the N application for

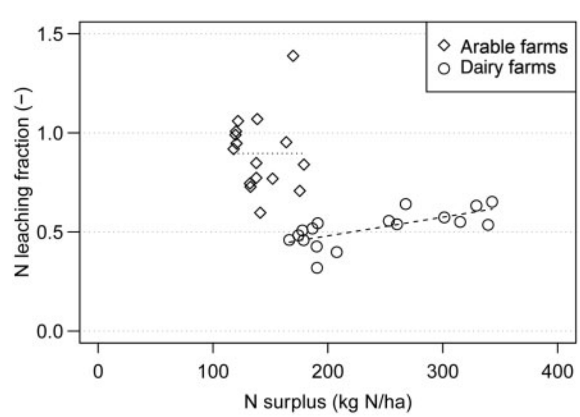

Figure 2. Effect of nitrogen surplus on the nitrate-nitrogen leaching fraction for deep, well-drained sandy soils (GRC = 11) at arable farms and dairy farms.
Note: Each dot in the plot represents the annual mean value of all arable or all dairy farms monitored a specific year.

Table 5. Relative nitrate concentration factors (C_{GRC}) for sandy soils per groundwater depth regime class (GRC) and their standard deviation.

	Groundwater depth regime class								
	1–3	4	5	6	7	8	9	10	11
Average	0.05	0.08	0.31	0.43	0.50	0.48	0.65	0.83	1.00
SD	0.09	0.07	0.06	0.06	0.06	0.06	0.04	0.07	0.09

Note: C_{GRC} for GRC 11 is 1.00 by definition (Boumans et al. 1989).

grassland and 21–46% for arable land. Based on a review of the literature, Jürgens-Gschwind (1989) reported much lower N leaching percentages for grassland (<20% of N application) than for arable land (>45%) for N applications of between 100 and 400 kg N/ha/year. The lower NLFs for grassland are attributed to higher denitrification rates in soils under grassland (Hofstra & Bouwman 2005).

Wachendorf et al. (2004) reported NLFs of between 0.30 and 0.40 for grassland on a peaty, coarse-textured sandy podzol, with relatively deep groundwater tables (a depth of 1.0–1.5 m in winter and 1.5–2.0 m in summer) in a field experiment in Germany. Dalgaard et al. (2006) provided data from the Danish Farm Accountancy Data Network and model calculations on nitrate leaching for farms on sandy soils with an N surplus ranging from 78 to 209 kg/ha. Using these data, we calculated NLFs ranging from 0.64 to 0.94 for farm types with less than 15% grassland and from 0.29 to 0.63 for farm types with 34–54% grassland. These results clearly demonstrate that farm types with the highest percentage of grassland had the lowest NLFs. Our results and those of Dalgaard et al. (2006) are in agreement with the findings of Simmelsgaard (1998), who showed that N leaching was negatively related to the time during which the soil was covered by a growing crop.

Data of Perego et al. (2012) for six sites in the Po valley in Italy with maize on different soil types, with N surpluses between 62 and 337 kg/ha, lead to an average NLF of 0.70. However, Wachendorf et al. (2006) reported a relationship between N leaching and N surplus for maize grown on a coarse-textured sandy Gley podzol with relatively deep groundwater tables, which results in NLFs of between 0.27 and 0.33 for N surpluses between 50 and 120 kg/ha.

Relationship between NLF and N surplus/time

The apparent relationship for dairy farms between NLF and N surplus (Figure 2) is probably due to a decrease in NLF for dairy farms in time (Figure 3). First, this relationship between NLF and N surplus does not exist for arable farms. This might have been caused by the limited trajectory of N surpluses at arable farms in combination with a relative high variability in NLFs (Figure 2) perhaps as a consequence of the relative small number of available arable farms per year (Table 1). Second, the highest N surpluses on dairy farms occurred in the early 1990s and the lowest in the 2001–2009 period (Figure 6). The relationship between NLF and N surplus is unknown (Schröder et al. 2007). Literature provides data that show an increase of NLF with increase of N surplus (Van Beek et al. 2009) as well as data that show the opposite (Schröder & Van Keulen 1997). The most plausible explanation for a decrease of the NLF for dairy farms in time, and thereby for the apparent relationship between NLF and N surplus, is a decrease in grazing in the period 1991–2009. Evidence for a decrease in grazing are the increased mowing percentage by a factor 1.7 (LMM data not shown) and the decreased ammonia emission during grazing, from 13.1 million kg in 1990 to 0.8 million kg in 2012 (CBS 2014). Lower nitrate concentrations in grass cutting systems than in grass grazing systems were reported by others (Nevens & Rehuel 2003; Verloop et al. 2006). Hansen et al. (2012) showed that urinations of ruminants on grazed pastures increased the risk of nitrate leaching. However, these findings might have been caused by a decreased N surplus as N in animal manure is used more efficiently in cutting systems

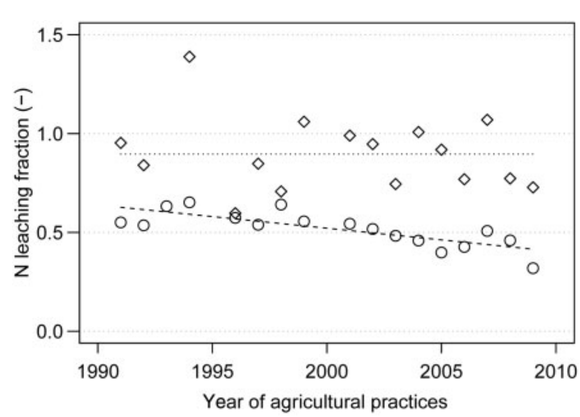

Figure 3. Trends in nitrate-nitrogen leaching fraction for deep, well-drained sandy soils (GRC = 11) at arable farms and dairy farms.
Note: Each dot in the plot represents the annual mean value of all arable or all dairy farms monitored a specific year.

(Wachendorf et al. 2004). Oenema et al. (2010) showed that increased grazing led to a significant higher nitrate concentration also when the effect on N surplus was accounted for. Other possible causes for a decrease in NLF for dairy farms were eliminated. There were no changes in ratio between grassland and arable land on dairy farms that could explain a reduced NLF; the percentage of grassland was even slightly lower in later years (Table 2). As the NLF for grassland is lower than the NLF for arable land, an increase in the percentage of grassland at dairy farms would lead to a decrease in NLF. Also, there was only a slight increase in the ratio N manure/N total (10% increase; not shown). An increase in this ratio could lead to increase in denitrification (more organic matter), but the observed relative increase in use of manure N was probably too small to cause the calculated decrease in NLF.

Relationship between NLF and recharge

NLFs showed a clear increase with increasing recharge (Figure 4); for arable farms 0.10 units per 100 mm/year ($R^2_{adj} = 0.41$, $p = 0.004$) and for dairy farms 0.05 units per 100 mm/year ($R^2_{adj} = 0.33$, $p = 0.009$). That higher recharge led to lower nitrate concentration due to dilution is reported earlier (Fraters et al. 1998; Simmelsgaard 1998). Wick et al. (2012) showed that an increase in precipitation led to lower nitrate concentrations and higher temperatures resulted in higher concentrations. The former, they attributed to increased plant uptake as well as dilution, the later, they explained by an increased evapotranspiration. Current results seem to indicate that more nitrate-N leached from the root zone when the recharge increased, even though nitrate concentration were lower. This finding was in accordance with results of, for example, Liao et al. (2012), who also included data of previous model studies carried out in the USA, found an increase in the NLF (N leach/N use) of 0.05 units per 100 mm of recharge. Owens et al. (2012) measured subsurface flow in four small micro-watersheds (about 1 ha each) under pasture in the USA and reported that nitrate-N transport followed a pattern similar to subsurface flow. This higher nitrate concentration might be due to shorter travel times through the unsaturated zone at higher recharge levels. As a consequence the water leaching from the root zone is younger and nitrate is less affected by denitrification.

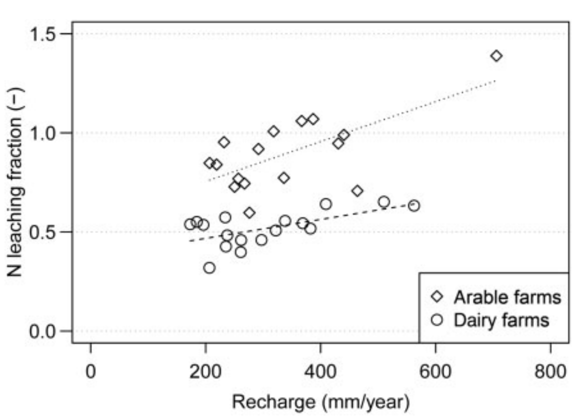

Figure 4. Effect of recharge on the nitrate-nitrogen leaching fraction for deep, well-drained sandy soils (GRC = 11) at arable farms and dairy farms.
Note: Each dot in the plot represents the annual mean value of all arable or all dairy farms monitored a specific year.

Effects of lag time on calculated NLFs

NLFs in this study were calculated by coupling N surplus in year x with the calculated N leaching based on the measured nitrate concentration in the upper metre of groundwater in the summer of year $x + 1$. Verloop et al. (2006) showed that effects of forage maize–grass rotation on nitrate concentrations in upper groundwater on a sandy soil lasted for three to four years. Peu et al. (2007), who performed a macro-lysimeter study with a silt loam on which ryegrass was grown in Western Brittany in France, reported that nearly 92% of the nitrate leaching occurred within the first three years after cessation of pig slurry application.

Based on the observed decrease in the N surplus for arable farms (Figure 5) and dairy farms (Figure 6) in the 1991–2004 period and our observation that N leaching in this study was related to the N surplus in the preceding year only, the calculated

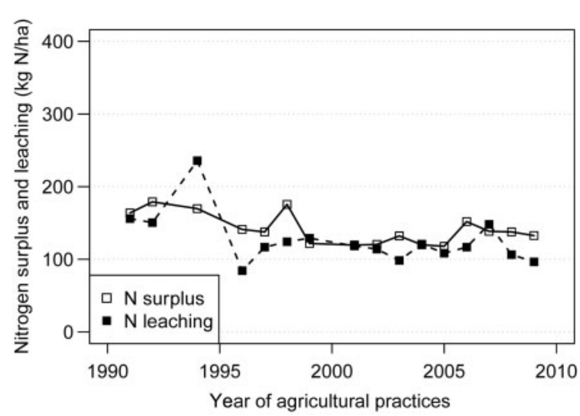

Figure 5. Trends in nitrogen surplus and nitrate-nitrogen leaching from deep, well-drained sandy soils (GRC = 11) at arable farms. Nitrogen surplus and resulting nitrogen leaching in the summer of the next year are both plotted in the year of agricultural practice.
Note: Each dot in the plot represents the annual mean value of all arable farms monitored a specific year.

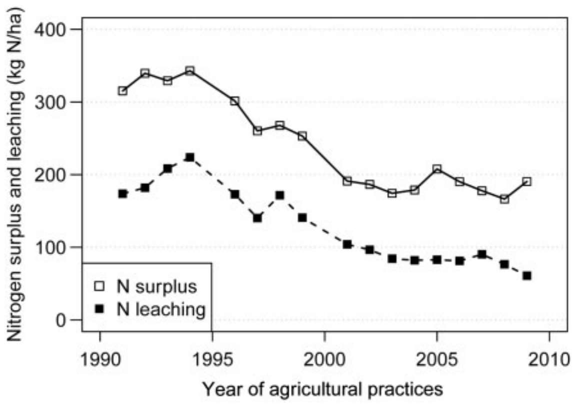

Figure 6. Trends in nitrogen surplus and nitrate-nitrogen leaching from deep, well-drained sandy soils (GRC = 11) at dairy farms. Nitrogen surplus and resulting nitrogen leaching in the summer of the next year are both plotted in the year of agricultural practice.
Note: Each dot in the plot represents the annual mean value of all dairy farms monitored a specific year.

NLFs presented here may be too high if nitrate leaching was truly influenced by the N surplus in preceding years as well. Two scenarios were considered in the calculation of the effect of lag time. The first scenario assumed that 50% of the N leaching was determined by the N surplus in the preceding year, 30% by the N surplus of two years ago and 20% by the surplus of three years ago. The second scenario assumed that the relative contribution of the N surplus in the four subsequent years to N leaching amounted to 40%, 30%, 20% and 10%, respectively. The NLFs calculated for arable farms were on average 0.9% and 1.3% lower for the first and second scenario, respectively, and for dairy farms 2.7% and 3.9% lower than NLFs calculated without accounting for lag time. The stronger effect of lag time for dairy farms is due to a stronger decrease in N surplus at dairy farms in the 1991–2009 period.

Use of NLFs for deriving environmentally sound N use standards

NLFs were used to calculate N use standards (Schröder et al. 2007). The points discussed earlier may have distinct consequences for farmers, for example, the 2006–2009 average NLF for grassland is 0.34, while the average value for the entire study period is 0.44. Use of the former value may seem logical given the decreasing trend in NLFs for grassland, and use of a NLF of 0.34 in model calculations will result in higher N use standards than the NLF of 0.44. Further study is required to determine whether this is environmentally sound. Second, in the Netherlands precipitation has increased with 25% in the last 100 years to around 820 mm/year (Buishand et al. 2013), notably winter precipitation has increased (35%). If this trend continues, NLFs may increase with about 2% the next 10 years which results in a NLF of 0.92 for arable farms and 0.53 for dairy farms. These changes are within the confidence intervals of the NLFs. Lag time has a relatively small, but reversed effect on the calculated NLFs, the estimation is −1% for arable farms and −4% for dairy farms. This means that a lower value for the NLF could perhaps be used in calculations of N use standards.

There are no other comparable studies we know off that use monitoring data of national programmes to calculate NLFs. The Dutch approach of monitoring the effectiveness of programmes of measures on a random sample of commercial farms is quite unique (Fraters et al. 2011). However, the method developed can be used elsewhere, for example, to link stream water N loads at the outlet of catchments to agricultural practices in those catchments.

Conclusions

The simple, straightforward method to link N leaching to N surplus on the soil balance, using data from long-term monitoring at a random sample of commercial farms has been developed successfully.

Based on the results reported here, which are derived of monitoring farm practices and water quality at farms in the Sand region of the Netherlands during a period of almost 20 years, the following conclusions can be drawn:

(1) Nitrate-N leaching from the root zone of sandy soils increased when the N surplus on the soil balance increased.
(2) The N surplus appears to leach almost completely from the root zone of arable land (80–99%) on deep, well-drained sandy soils, while about half of the N surplus appears to leach from the root zone of grassland (48–56%).
(3) The fraction of the N surplus that leaches from the root zone may change in time due to developments in agricultural practices that may influence the N stock in soil or denitrification rate, such as grazing regime, and due to climate change.

The NLFs have been used to derive ranges for environmentally sound N use standards for arable crops and grassland. The present Dutch action programme for the Nitrates Directive (2014–2017) sets stricter N use standards (−20% for arable crops susceptible to N leaching in the Southern Sand district and Loess region), a higher N working coefficient for pig slurry (from 70% to 80% for all sand and loess soils) and restrict derogation to farms with at least 80% grassland instead of 70% with a limited derogation standard of manure N use of 230 kg/ha in the Central and Southern Sand districts and the Loess region instead of the regular standard of 250 kg/ha.

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