

# Nutrient loss pathways from grazed grasslands and the effects of decreasing inputs: experimental results for three soil types

C. L. van Beek · C. van der Salm ·  
A. C. C. Plette · H. van de Weerd

Received: 23 November 2007 / Accepted: 15 August 2008 / Published online: 29 August 2008  
© Springer Science+Business Media B.V. 2008

**Abstract** Agriculture is a main contributor of diffuse emissions of N and P to the environment. For N the main loss pathways are  $\text{NH}_3$ -volatilization, leaching to ground and surface water and  $\text{N}_2\text{O}$  emissions. Currently, imposing restraints on farm inputs are used as policy tool to decrease N and P leaching to ground water and to surface water, and the same measure is suggested to combat emissions of  $\text{N}_2\text{O}$ . The response, however, to these measures largely depends on the soil type. In this study nutrient flows of three dairy farms in The Netherlands with comparable intensity on sand, peat and clay soils were monitored for at least 2 years. The first aim was to provide quantitative data on current nutrient loss pathways. The second aim was to explore the responses in partitioning of the nutrient loss pathways when farm inputs were altered. Mean denitrification rates ranged from  $103 \text{ kg N ha}^{-1} \text{ year}^{-1}$  for

the sandy soil to  $170 \text{ kg N ha}^{-1} \text{ year}^{-1}$  for the peat soil and leaching to surface water was about  $73 \text{ kg N ha}^{-1} \text{ year}^{-1}$  for the sandy soil,  $15 \text{ kg N ha}^{-1} \text{ year}^{-1}$  for the clay soil and  $38 \text{ kg N ha}^{-1} \text{ year}^{-1}$  for the peat soil. For P, leaching to surface water ranged from  $2 \text{ kg P ha}^{-1} \text{ year}^{-1}$  for the sandy site to  $5 \text{ kg P ha}^{-1} \text{ year}^{-1}$  for the peat site. The sandy soil was most responsive to changes in N surpluses on leaching to surface water, followed by the peat soil and least responsive was the clay soil. For P, a similar sequence was found. This article demonstrates that similar reductions of N and P inputs result in different responses in N and P loss pathways for different soil types. These differences should be taken into account when evaluating measures to improve environmental performance of (dairy) farms.

**Keywords** Soil type · Dairy farming · Denitrification · Leaching · Surplus · Nitrogen · Phosphorus

C. L. van Beek (✉) · C. van der Salm  
Alterra, Wageningen University and Research Centre,  
P.O. Box 47, 6700 AA Wageningen, The Netherlands  
e-mail: christy.vanbeek@wur.nl

A. C. C. Plette  
RIZA, P.O. Box 17, 8200 AA Lelystad, The Netherlands

*Present Address:*  
A. C. C. Plette  
Kapelsteeg 9, 3841 BJ Harderwijk, The Netherlands

H. van de Weerd  
Arcadis, P.O. Box 673, 7300 AR Apeldoorn,  
The Netherlands

## Introduction

Agriculture is a main contributor of diffuse loads of N and P to the environment (Tilman et al. 2001). In industrialized countries all agricultural practices cope with a certain nutrient surplus on farms, i.e. imported nutrients exceed export (OECD 2001; Slak et al. 1998). In The Netherlands about 75% of the national manure production is allocated to dairy farming and

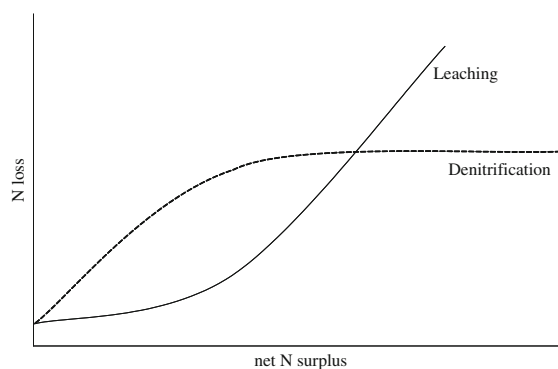
surpluses leave the agricultural system via several loss pathways. For N, the most important loss pathways are leaching to ground and surface water, emission of  $N_2$  and  $N_2O$  and  $NH_3$  volatilization. Compared to N, P is strongly bound to the soil solid phase and in soils with a low P-status the main loss pathways for P are direct losses of manure, fertilizer and/or soil to surface water by runoff, erosion and/or preferential flow. In soils with a higher P status leaching losses increase with an increasing amount of adsorbed P and may pass through the soil matrix and drain pipes (Schoumans and Groenendijk 2000).

Of all industrialized countries The Netherlands cope with the highest nutrient surpluses in agriculture (OECD 2001). Notably, in 1995 agriculture accounted for 93% of the national  $NH_3$  emissions and 42% of the national  $N_2O$  emission (Olsthoorn and Fong 1998), while agriculture accounted for 68% of the N and 49% of the P loading of surface water in 2000 (RIVM 2002). Between 1998 and 2005 the Dutch government combated the excessive use and production of nutrients by setting maximum farm surpluses for N and P. This approach, however, was abandoned in 2006 under pressure of the EU as it did not meet with the requirements of the nitrate directive and was replaced by input maxima. The same measure, i.e. reducing N inputs, is suggested to combat emissions of  $N_2O$  (Oenema et al. 2001). At this moment the maximum allowable inputs of N are set at  $345 \text{ kg N ha}^{-1} \text{ year}^{-1}$  for clay soils and  $290 \text{ kg N ha}^{-1} \text{ year}^{-1}$  for sand and peat soils, of which at most  $170 \text{ kg N ha}^{-1} \text{ year}^{-1}$  may originate from manure. For farms with more than 70% grassland (as is the case for the three experimental farms of this study) derogation to EU legislation is applied and applications via manure may add up to  $250 \text{ kg N ha}^{-1} \text{ year}^{-1}$ . For P, a total maximum application of  $48 \text{ kg P ha}^{-1} \text{ year}^{-1}$  is allowed in 2006 and is due to decrease to  $41 \text{ kg P ha}^{-1} \text{ year}^{-1}$  in 2008 (LNV 2006). A further reduction in P application, leading to equilibrium between P supply and P uptake, is foreseen in 2015.

The reduction of N and P inputs at farm level as a measure to reduce environmental losses implies a relationship between N and P inputs and N and P losses to the environment. In simple models such a relation is often assumed linear for certain soil types and environmental conditions (e.g. De Vries et al. 2001), but this is not always the case. For example, the relation between denitrification rates and soil  $NO_3$

contents (which is often related to N surplus) is generally described by a first-order process that runs asymptotically to the maximum possible denitrification rate under field conditions (Hénault and Germon 2000). When net changes in soil storage are small, leaching can be considered as the complement of denitrification and hence is also non-linearly related to the N surplus, which was confirmed experimentally by Fried et al. (1976) and Barraclough et al. (1992). Therefore, we expect that the relations between N surplus and leaching and between N surplus and denitrification are not proportional, but instead are shaped like in Fig. 1. Figure 1 shows an increasing response of leaching on changes in N surpluses when N surpluses increase, and the opposite for the relation between N surplus and denitrification. For P, a similar relation was assumed. An increase or decrease in the P surplus will lead to a certain change in the amount of P bound to the soil solid phase. The binding of P to the soil by sorption has a comparable shape as the denitrification curve presented in Fig. 1 (Van der Zee and Van Riemsdijk 1988). At relatively low P status a change in the amount of adsorbed P will lead to a small change in the soil solution concentrations and little losses to surface water, whereas at relatively high P status a similar change in adsorbed P will lead to a much larger change in P concentrations and subsequently in leaching losses.

In this article we assess the relation between N and P inputs and N and P surpluses on the partitioning over environmental loss pathways, i.e. leaching, denitrification and P sorption for three different soil types on dairy farms. The aims of this paper are to (i) experimentally quantify nutrient flows in grazed grassland and (ii) evaluate the effect of decreasing



**Fig. 1** Conceptual partitioning of N loss pathways over denitrification and leaching

N and P inputs on changes in environmental nutrient losses. To achieve these aims we collected field data and performed several experiments for three grazed grassland on different soil types.

## Material and methods

The three sites were located in den Pol (sand), Waardenburg (clay) and Vlietpolder (peat) in The Netherlands. In Table 1 a brief overview of the 3 sites is presented. To reduce the number of nutrient loss pathways the sites were selected on limited exchange with groundwater and main emphasis was given to leaching to surface water, N losses due to denitrification and P storage due to sorption. For all locations a dead-end ditch was selected and one or two neighboring fields, draining on the selected ditch, for on-site experiments. For organizational reasons measurements were performed from 1999 to 2001 for Den Pol, from 1999 to 2002 for the Vlietpolder and from 2002 to 2005 for Waardenburg. Of these years, 2003 was extremely warm and dry, while the rest of the years were somewhat warmer and wetter compared to the long term average data ([www.knmi.nl](http://www.knmi.nl)). Some results of the individual sites were previously published, but in this study the results are brought together to study the effect of soil type on the partitioning of N and P surpluses at field level.

### Site description

Den Pol is situated at the eastern border of The Netherlands (52°19'N; 6°37'E) and the soil consists of non-calcareous sands (Table 1). Since 1991 the field has been in use for grassland for yearlings and silage. Before 1991 maize was grown occasionally. The ditch is free-draining and dries up periodically in summer. About 3.5 m below the surface, a confining loam layer prevents exchange of water and solutes with deeper groundwater. The sandy topsoil was well to intermediately drain. In wet periods the field was occasionally ponded and surface run-off occurred.

Location Waardenburg is situated in the centre of The Netherlands (51°50'N; 5°16'E) on heavy clay soil. The field was drained by tile-drains and trenches. The trenches were about 50 × 50 cm (w × d) and the distance between the trenches was about 46 m. The tile drains were located in between the trenches

at about 80 cm below soil surface. Due to the low permeability of the heavy clay there was no seepage or groundwater recharge. The experimental field was used for intensive dairy farming and the grass was alternately mown and grazed.

The Vlietpolder is situated in a polder near Hoogmade in the western part of The Netherlands (52°10'N; 4°36'E). The polder is approximately 200 ha in size and about 2 m below mean sea level. In summer, when there is shortage of water, water is let in the polder via the inlet station. In winter, when there is excess of water, water is pumped out of the polder at the pumping station. The inlet station and the pumping station are on two outer ends of the polder. One field used for dairy farming was selected for intensive monitoring. In this field the man-made topsoil (0–40 cm) consisted of organic matter rich sandy clay. The subsoil consisted of woody peat. At 3 m below soil surface a dense clay layer prevented extensive groundwater recharge.

### Monitoring program

All sites were intensively monitored on nutrient fluxes to, through and from the fields. The monitoring programs were, however, adjusted to local conditions and are listed in Table 2. At the end of each ditch a weir was placed with a flow meter connected to a sampling device for flow proportional sampling (for peat soil: MagMaster, ABB Instrumentation, Switzerland; for clay soil and sand soil: ISCO 3220 flow meter and ISCO 3700 sampler). Flow rates were measured continuously and allowed for assessment of both discharge and inlet rates of water and solutes. For the clay site, discharge of both tile drains and trenches were monitored separately. The outflow from drains and trenches was led to small collectors with a V-notch opening and discharge was monitored continuously by measuring the water level in the collectors, and samples were taken at fixed discharge intervals (0.12 mm). For all sites discharge samples from the ditch were collected every 0.10–0.25 mm of through-flow and stored dark and cold (5°C) until analysis (within 1 week).

Denitrification rates were measured using the Acetylene Inhibition Technique (AIT; Mosier and Klemedtsson 1994). Briefly, 1–10% acetylene (C<sub>2</sub>H<sub>2</sub>) inhibits the reduction of N<sub>2</sub>O to N<sub>2</sub> and consequently the N<sub>2</sub>O production from soil in an atmosphere containing C<sub>2</sub>H<sub>2</sub> is a measure for the N<sub>2</sub> + N<sub>2</sub>O

**Table 1** Brief description of sampling sites

	den Pol	Waardenburg	Vlietpolder
Soil type	Sand	Clay	Peat
Sand (%)	87	3	15
Clay (%)	1	40	10
Loam (%)	12	57	75
Organic matter (%)	4.0	5.4	20
pH	5.5 <sup>a</sup>	6.7 <sup>b</sup>	5.0 <sup>b</sup>
Bulk density (kg m <sup>-3</sup> )	1357	1500	925
Drainage condition	Good to intermediate	Poor (presence of tile drains)	Poor
Ditch	Free draining	Free draining	Fixed surface water levels
Agriculture	Grazing of yearlings and silage	Grazing of lactating cows and silage	Grazing of lactating cows and silage
Precipitation (mm)	920 (2001), 847 (2001)	645 (2003), 869 (2004)	861 (2000), 1215 (2001)
Fertilization	Cattle droppings, manure	CAN, manure, cattle droppings	CAN, manure, cattle droppings

Soil characteristics refer to upper 20 cm

<sup>a</sup> pH—KCl

<sup>b</sup> pH—H<sub>2</sub>O

**Table 2** Brief descriptions of experiments performed on each location, 'x' denotes presence of experiment, '-' denotes absence of experiment

Experiment	Sand	Clay	Peat	Remark
Discharge of water and solutes of experimental ditch to higher water body	x	x	x	A dead-end ditch was sampled flow proportional using V-partitions
Discharge of water and solutes to ditch	-	x	-	Flow proportional sampling of tile-drains (no tile drains were present on sand and peat)
Surface run-off	Catchment plates and simulation modeling	Flow proportional sampling of trenches	Catchment plates	Methods were adjusted to local conditions
Mineralization of soil organic matter	Incubation of soil cores	Incubation of soil cores and balance method	Application of zero-field	Methods were adjusted to local conditions, but were only part of soil balance for peat soil (see text)
Denitrification (replicates year <sup>-1</sup> )	4	12	12	Using AIT (see text)
Composition soil solution	x	-	x	Porous suction cups
Composition groundwater	x	x	x	Groundwater wells
P sorption characteristics	x	x	x	Incubation studies on P kinetics at different P concentrations

production through denitrification. We replaced 7% of the gas phase of the incubation containers by C<sub>2</sub>H<sub>2</sub>. Concentrations of N<sub>2</sub>O in the incubation containers were measured at least twice within 8–24 h by a Bruël and Kjaer 1312 multi-gas monitor with a photo acoustic infrared gas analyzer (TGA). Potential denitrification rates, defined as the denitrification rate under anaerobic conditions with

abundant NO<sub>3</sub> at 20°C (Bijay-Singh et al. 1989), were measured at least once at each location. Volatilization of NH<sub>3</sub> was estimated at 5% of the total application of manure and cattle droppings (Misselbrook et al. 2000; Bussink 1994).

Surface run-off from field to surface water was assessed using experimental data from catchment plates and simple balance models for the peat soil

**Table 3** Potential denitrification, discharge to surface water and P saturation for each location ( $\pm$ standard deviation)

	Sand	Clay	Peat
Potential denitrification 0–20 cm (kg/ha/d) <sup>a</sup>	4–8	40 $\pm$ 10 <sup>b</sup>	27 $\pm$ 19 <sup>c</sup>
Discharge to surface water (%) <sup>d</sup>			
$Q_{\text{surface}}$	12	67	8
$Q_{\text{shallow}}$	4	32	61
$Q_{\text{deep}}$	84	–	31
DPS (%) <sup>e</sup>	37	7	15
$Al_{\text{ox}}$ (mmol kg <sup>-1</sup> )	65	59	144
$Fe_{\text{ox}}$ (mmol kg <sup>-1</sup> )	8	172	209
Langmuir adsorption constant (K)	0.37	0.11	0.18
Maximum amount of P reversible bound to Al and Fe ( $\beta$ ) (–)	0.21	0.09	0.19

Results refer to upper 40 cm, unless stated otherwise

<sup>a</sup> Samples were taken once (peat and sand) or monthly (clay)

<sup>b</sup> Mean value of monthly measurements

<sup>c</sup> Mean value of spatial replicates

<sup>d</sup>  $Q_{\text{surface}}$  = Run-off, matrix flow (0–10 cm), trenches;  $Q_{\text{shallow}}$  = Matrix flow (10–70 cm) and drains,  $Q_{\text{deep}}$  = matrix flow (>70 cm)

<sup>e</sup>  $DPS = P_{\text{ox}} / (0.5 \times (Al + Fe_{\text{ox}}))$

(Van Beek et al. 2003a), catchment plates and complex simulations for the sand soil (Torenbeek and Voskamp 2003), and direct measurements from shallow trenches for the clay soil (Van der Salm et al. 2006). Surface run-off, including both overland flow, subsurface flow (0–10 cm) and discharge by trenches, was estimated at 12, 67 and 8% of the total discharge of water to the surface water for the sand, clay and peat soil, respectively (Table 3).

To get insight in the fate of P in soil, the degree of P saturation (DPS), sorption characteristics and the amount of reversible bound P for the soil profiles at the three sites were determined. Samples were collected at the three sites up to a depth of at least 40 cm. The thickness of the soil layers ranged from 5 cm in the topsoil of the sandy site to 40 cm at the clay site. The DPS was derived from dividing the amount of sorbed P in the soil above the highest groundwater table by the P sorption capacity (PSC) of the soil according to Schoumans and Groenendijk (2000). The P sorption capacity was calculated by:

$$PSC = 0.5(Al_{\text{ox}} + Fe_{\text{ox}}) \quad (1)$$

in which  $Al_{\text{ox}}$  and  $Fe_{\text{ox}}$  are the amounts of oxalate extractable Al and Fe, respectively. Both the  $Al_{\text{ox}}$  and  $Fe_{\text{ox}}$  and the amount of sorbed P were determined by extraction with  $NH_4$ -oxalate (Schwertmann 1964). The DPS in the upper 40 cm of the soil profile equaled 37% for the sand soil, 7% for the clay soil and 15% for the peat soil (Table 3).

The reversible sorption and desorption of P can be described by a Langmuir equation according to:

$$Q = \frac{Kc\beta(Al_{\text{ox}} + Fe_{\text{ox}})}{1 + Kc} \quad (2)$$

in which  $Q$  is the amount of reversible bound P (mmol kg<sup>-1</sup>),  $c$  is P concentration in the soil solution (mmol m<sup>-3</sup>),  $\beta$  (–) is the maximum amount of P bound to Al and Fe oxides and  $K$  is the adsorption constant (m<sup>3</sup> mol<sup>-1</sup>). The parameters  $K$  and  $\beta$  were determined by measuring  $Q$  at different equilibrium concentrations. Samples were loaded with different amounts of P by shaking the samples in a solution (1:66.6 w:v) with P concentrations ranging from 0 to 150 mg l<sup>-1</sup> for a period of 7 days to obtain samples with a range in equilibrium concentrations. The solution had a background concentration of 0.005 M  $CaCl_2$  and 0.005 M KCl. At the end of this period the suspensions were filtered over 0.45  $\mu$ m and the residue on the filters was rinsed with 100 ml of a 60% ethanol solution. Samples were air dried. The amount of reversible bound P ( $Q$ ) in the dried samples was determined by extraction with FeO-strip method (Menon et al. 1989) for a period of 192 h. For each sample the equilibrium concentration was determined in a 1:4 (w:v) suspension of a mixture of  $CaCl_2$  and KCl and shaking for 7 days. After 7 days the suspensions were centrifuged (10 min at 1500g) and filtrated over 0.45  $\mu$ m. Total P concentration was measured using an Inductively Coupled Plasma-Atomic Emission Spectrometer.

Soil system balances were constructed for the three sites and were delineated by the approximate rooting depth, soil surface and surface water. Soil surface balances of N and P were defined as the N and P surpluses due to agricultural activities corrected for  $NH_3$  volatilization. Volatilization of  $NH_3$  is a very fast loss pathway (Lockyer and Whitehead 1990) and therefore has little interaction with soil.

For the peat soil mineralization was added as a balance term because of the ongoing decline of C, N and P stocks in peat soils and was determined using so-called zero-plots (Van Beek et al. 2004).

Run-off may also have limited interaction with the soil, but this is not necessarily so, as e.g. dung patches may be prone to run-off long after they were applied (Chardon et al. 2007). Therefore, the N surplus was not corrected for surface run-off. Fields were visually checked for N fixing legumes, which were generally absent and consequently biological N fixation was ignored. Details on measurements and interpretation of data are provided in Van Beek et al. (2003b, 2004), Van der Salm et al. (2006) and Torenbeek and Voskamp (2003) and some key results are presented in Table 3.

Uncertainties in balance terms arise from experimental noise (e.g. due to noise in equipment), spatial variability due to differences in management strategies and field conditions and temporal variability due to differences in meteorological conditions. For the current study the spatial variability was of most interest, because it provides information about the representativeness of the sites. However, spatial variability of balance terms demands for examination of multiple fields. The examination of multiple fields was only possible for the peat site, because of its situation in a polder. In this polder 65 fields had comparable water management, land-use and soil management (Van Beek et al. 2003b). Therefore, standard deviations are given for the peat site. Uncertainties in soil system balances due to temporal variability and experimental noise ranged from 2% for fertilizer application to 14% for the slurry application for N (Van der Salm et al. 2007). For P, comparable ranges of uncertainties in balance terms were assumed.

### Partitioning of surpluses

To partition the surpluses over different loss pathways, the relations shown in Fig. 1 were quantified using mathematical functions for each site. The non-linear relationship between N surplus ( $x'$ , kg ha<sup>-1</sup> year<sup>-1</sup>) and leaching ( $X_L$ , kg ha<sup>-1</sup> year<sup>-1</sup>) was visualized by (Van Beek et al. 2003b):

$$X_L = \frac{1}{\alpha} \ln \left( 1 + e^{\alpha(x' - \beta)} \right) \quad (3)$$

where  $\alpha$  is a curve shape parameter (year ha kg<sup>-1</sup>) and  $\beta$  is the denitrification capacity (kg ha<sup>-1</sup> year<sup>-1</sup>). The denitrification capacity was defined as the annual denitrification at prevailing temperatures and soil water contents, but with ample NO<sub>3</sub> (Koops et al. 1996) and was estimated from the potential denitrification using a reduction factor for limitations due to temperature and soil moisture ( $f_{TF_S}$ ). We used  $f_{TF_S}$  of 0.10 for the clay soil, 0.15 for the peat soil and 0.05 for the sandy soil (Heinen 2006; Van der Salm et al. 2007), because conditions for denitrification were most favorable for the peat soil (high organic matter content, shallow groundwater), followed by the clay soil (high availability of anaerobic micro-sites) and were least favorable for the sandy soil.

Considering leaching ( $X_L$ , kg ha<sup>-1</sup> year<sup>-1</sup>) being the complement of denitrification ( $X_D$ ) when changes in soil storage are limited, the following equation is valid:

$$X_L + X_D = x' \quad (4)$$

where  $x'$  is the net field N surplus.

For mineral P similar graphs were constructed based on the sorption of P to the soil solid phase. Leaching ( $X_L$ ) of P was estimated by multiplying the precipitation surplus (PS, m) with the soil solution concentration and Eq. 2 was rewritten to:

$$X_L = \text{PS} \frac{Q + \frac{\gamma x'}{\rho L} \varepsilon}{K \left( Q_m - \left( Q + \frac{\gamma x'}{\rho L} \varepsilon \right) \right)} \quad (5)$$

in which  $\gamma$  is the fraction of the P surplus which is reversible bound,  $x'$  is the net field P surplus (kg ha<sup>-1</sup> year<sup>-1</sup>),  $\rho$  is the bulk density (kg m<sup>-3</sup>),  $L$  is the thickness of the layer over which the P surplus is distributed (m),  $\varepsilon$  is a conversion factor (100/31). PS equaled the long term average precipitation surplus of 300 mm year<sup>-1</sup>. As  $Q$  is relatively large compared to the surplus, changes in  $X_L$  on an annual basis were small. Therefore, the constant  $\gamma$  could not be estimated from the annual P budgets, but was derived from the measured amount of reversible bound P and total sorbed P, assuming a constant ratio between reversible sorbed P and total soil P.  $X_L$  was calculated for a homogeneous soil layer of 30 cm and for a period of 10 years of exposure to a certain P surplus. The partitioning of P surpluses was regarded as an exploration of long-term effects of alternating P surpluses.

## Results

### Nutrient balances

The soil surface balances equaled 207, 129 and 338 kg N ha<sup>-1</sup> year<sup>-1</sup> for the sand, clay and peat site, respectively. Fertilizer and manure were major inputs, and grass harvest for silage production was the major output of N (Table 4). For P, soil surface balances equaled 20, 21 and 14 kg P ha<sup>-1</sup> year<sup>-1</sup> for the sand, clay and peat site, respectively (Table 5). Emission of N<sub>2</sub>(O) was the major loss pathway for all sites and soil system balances equaled 15, -38 and 117 kg N ha<sup>-1</sup> year<sup>-1</sup> for the sand, clay and peat site, respectively. Standard deviations of all balance entries of the peat soil were high and often exceeded

50%, which was predominantly caused by the uneven distributions of cattle droppings and manure applications over fields (Van Beek et al. 2003b) and were presumably valid for all sites.

### Nutrient loss pathways

Denitrification rates ranged between 103 and 170 kg N ha<sup>-1</sup> year<sup>-1</sup> and were highest for the peat soil and lowest for the sand soil. Leaching to surface water ranged between 15 kg N ha<sup>-1</sup> year<sup>-1</sup> at the clay site and 73 kg N ha<sup>-1</sup> year<sup>-1</sup> at the sand site (Table 4). For P this sequence was opposite with highest losses for the peat site (5 kg P ha<sup>-1</sup> year<sup>-1</sup>) and lowest losses for the sand and clay sites (2 and 3 kg P ha<sup>-1</sup> year<sup>-1</sup> respectively, Table 5). Thereby

**Table 4** N soil system balance for each soil type (kg ha<sup>-1</sup>year<sup>-1</sup>)

		Sand	Clay	Peat
IN	Fertilizer	143	153	184 ± 46
	Manure	238	306	148 ± 79
	Cattle droppings	76	26	120 ± 86
	Atmospheric deposition	44	34	31
	Net mineralization of peat soil	–	–	242 ± 19
OUT	Grazing	–93	–58	–190 ± 125
	Silage	–201	–332	–197 ± 86
Surplus		207	129	338 ± 117
	N <sub>2</sub> (O) emission (denitrification)	–103	–135	–170
	Leaching to groundwater	0	0	0
	Leaching to surface water	–73	–15	–38
	NH <sub>3</sub> volatilization	–16	–17	–13
	Sum of loss pathways	–192	–167	–221
	Surplus – Sum of loss pathways	15	–38	117

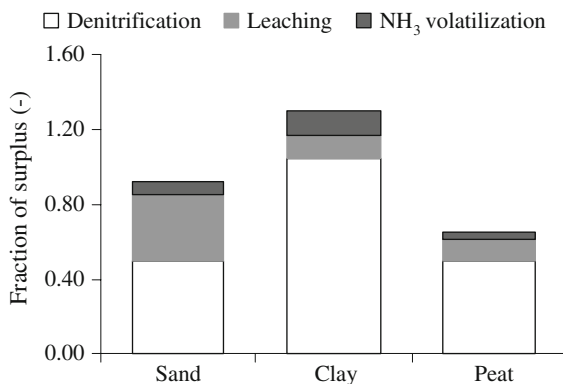
For peat soil standard deviations are provided (see text)

**Table 5** P soil system balance for each soil type (kg ha<sup>-1</sup>year<sup>-1</sup>)

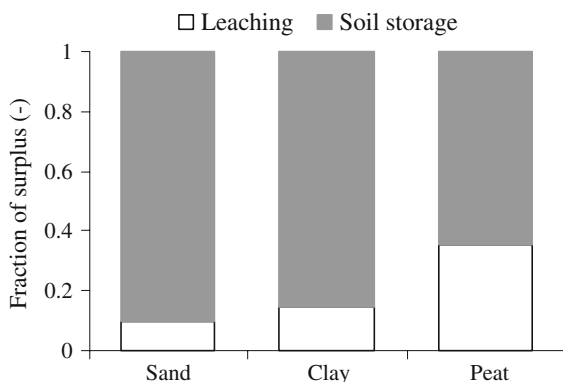
		Sand	Clay	Peat
IN	Fertilizer	0	24	10 ± 9
	Manure	32	52	20 ± 12
	Cattle droppings	8	1	11 ± 8
	Net mineralization of peat soil	–	–	10
OUT	Grazing	–6	–7	–17 ± 13
	Silage	–14	–49	–20 ± 8
Surplus		20	21	14
	Leaching to groundwater	0	0	0
	Leaching to surface water	–2	–3	–5
	Sum of loss pathways	–2	–3	–5
	Surplus – sum of loss pathways	18	18	9

For peat soil standard deviations are provided (see text)

leaching losses accounted for 3% (clay) to 16% (sand) of the agricultural N and P inputs (i.e. inputs via fertilizer, manure and cattle droppings). The relative partitioning of the N surpluses over N loss pathways is shown in Fig. 2 and demonstrates that for the sand site total losses about equaled the surplus on the soil surface balance (i.e. the surplus of the soil system balance was about zero). However, for the clay soil loss pathways exceeded the surplus of the soil surface balance, whereas for the peat soil the loss pathways accounted for about 65% of the surplus. The over- and under-partitioning of surpluses were presumably caused by the release of N from past fertilization (clay) and by overestimation of mineralization rates (peat). The relative partitioning of the P surpluses over P loss pathways is shown in Fig. 3. For P, only leaching could be quantified at field level and accounted for 10–36% of the surplus of the soil



**Fig. 2** Relative partitioning of N loss pathways for sandy site, peat site and clay site



**Fig. 3** Relative partitioning of P loss pathways for sandy site, peat site and clay site

surface balance. The remained of the surplus was supposed to be stored in soil.

### Partitioning of surpluses

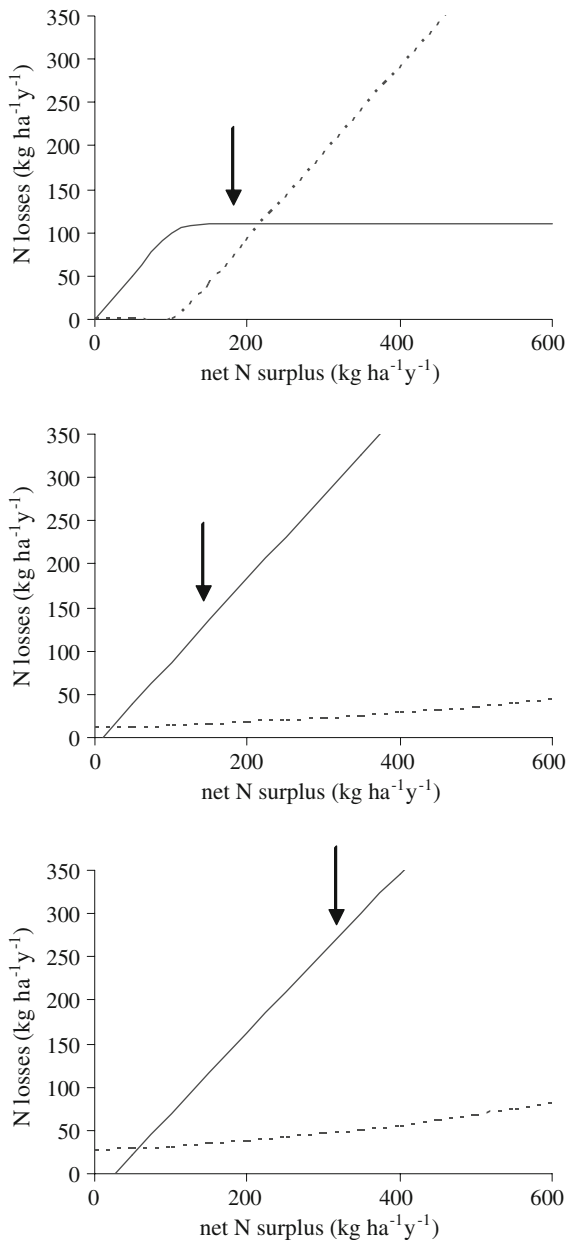
The denitrification capacities were estimated at 1478 kg N ha<sup>-1</sup> year<sup>-1</sup> for the peat soil, 1460 kg N ha<sup>-1</sup> year<sup>-1</sup> for the clay soil and 110 kg N ha<sup>-1</sup> year<sup>-1</sup> for the sandy soil. With the Eqs. 3 and 4 and the data provided in Table 4 the curves of Fig. 1 were parameterized by optimizing  $\alpha$  to a minimal sum of squares of experimental findings on leaching and denitrification at the prevailing N surpluses. Subsequently, Fig. 1 was drawn for each soil type and the present situation was pointed out in Fig. 4. The curve shape parameters  $\alpha$  were 1.21, 0.0025 and 0.0020 for the sand, clay and peat soil respectively. Figure 4 shows very different curves especially for the sand soil versus the clay and peat soil. The intersection between leaching ( $X_L$ ) and denitrification ( $X_D$ ) was reached at an N surplus of 220 kg N ha<sup>-1</sup> year<sup>-1</sup> for the sandy soil. For the clay and peat soil this point of intersection was far beyond the (realistic) range of net N surpluses (>800 kg N ha<sup>-1</sup> year<sup>-1</sup>). Apparently, for the peat soil at small N surpluses immobilization of N occurs, as at low N surpluses negative N losses occurred, while leaching set-off at about 25 kg N ha<sup>-1</sup> year<sup>-1</sup>, which is in reasonable agreement with reported background loads to surface waters in peat soils (Hendriks 1993).

Using Eq. 5 and the data in Table 5 similar figures were constructed for P leaching and P storage. The constant  $\gamma$  was found to be 0.005 for the peat and clay soils and 0.01 for the sandy soil. Differences in  $\gamma$  reflect differences in sorption characteristics and historic manure use. The amount of reversible stored P was related to the parameters  $\beta$  and K which decreased in the order sand > peat > clay. In all cases P leaching was much lower compared to the storage of P in soil. For all sites P leaching increased more rapidly per unit of change in P surplus at higher P surpluses (Fig. 5).

### Discussion

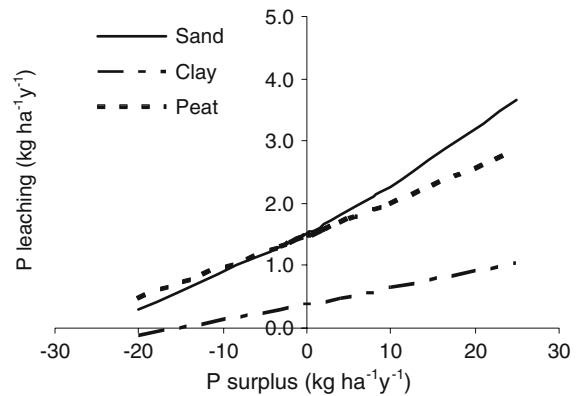
In The Netherlands agricultural production is generally very intensive, with high inputs of N and P





**Fig. 4** Partitioning of net N surplus over leaching (long dash) and denitrification (solid) for sand (upper), clay (middle) and peat (below) calculated with Eqs. 3 and 4. The net N surplus includes the N surplus at field level minus NH<sub>3</sub> volatilization. Arrows point at current net N surpluses

through fertilizers and manure, but also with high yields. In our study 10–35% of the N surplus (Fig. 2) and 3–19% of the N inputs via fertilizer and manure was lost to surface water via leaching. These figures are comparable to results obtained from Northern Ireland (Watson and Foy 2001), but are below results



**Fig. 5** Leaching of inorganic P as function of P surpluses for different soil types, calculated from Eq. 5 after 10 years exposure

obtained from Lithuania (Bucienė et al. 2003). In The Netherlands inputs of N and P to surface water have decreased considerably since the 1980s, but the improvement of water quality lags behind. As an explanation of this retardation Oenema et al. (2005) mentioned the contribution of other nutrient sources (e.g. rivers and seepage), but to our vision also the non-linear relation between inputs and losses may contribute to the uneven improvement of water quality among regions.

The N and P losses to surface water (Tables 4, 5) include both leaching through the soil matrix as well as superficial losses by run-off (including interflow through the uppermost part of the topsoil and leaching through trenches). In the clay soil superficial losses occurred through the trenches, which could be measured accurately. In the peat soil and the sandy soil superficial losses were measured using catchment plates. Accurate measurement of discharge by means of these plates is difficult due to methodological complexities and due to spatial variability as a consequence of the development of preferential flow pathways during high run-off events. Run-off losses were thus estimated using average measured concentrations and modeled fluxes leading to rather rough estimates of the N and P loads by run-off. In most cases the estimated contribution of run-off to the leaching of N and P was relatively small (<15%), except for P run-off losses for the sandy soil which equaled 88% of the total P-loss. This is a quite remarkable result as the permeability of the sandy soil was high and superficial transport of water was the lowest of the three sites (Tables 1, 3). Detailed assessment of this phenomenon demands for

additional research and is beyond the scope of this study. Equilibrium P concentrations in soil solution of the peat and sand soil were about comparable ( $0.6 \text{ mg P l}^{-1}$  in the upper 10 cm to  $0.2 \text{ mg P l}^{-1}$  at 30 cm depth, not shown). Apparently, for the sand soil the higher DPS was largely compensated by the higher adsorption constant  $K$  compared to the peat soil (Table 3). Regarding soil properties the risks for leaching were lowest for the clay soil due to a low DPS combined with an intermediate adsorption constant leading to relatively low soil solution concentrations ( $0.1 \text{ mg P l}^{-1}$ , not shown) in the topsoil.

### Quantification of nutrient fluxes

There are only few studies on experimentally quantified flows of nutrients in (and through) grazed grasslands (Jarvis 1992). Also, differences in soil type and hydrological conditions complicate the comparison of our results with those observed by others. However, for denitrification our results were high compared to others. For all sites denitrification losses exceeded  $100 \text{ kg N ha}^{-1} \text{ year}^{-1}$  (Table 4), which is high for agricultural soils according to the review of Barton et al. (1999). Presumably, the relatively high denitrification losses for all soil types were related to the relatively shallow groundwater levels that are common throughout The Netherlands and the relatively high N inputs.

### Partitioning of nutrient surpluses

When surpluses are altered, not only the extent, but also the partitioning over the various loss pathways of the surplus changes. The sensitivity of nutrient loss pathways is defined as the level of response of a loss pathway on changes in surpluses (i.e. the steepness of the curves in Figs. 1, 4, 5). Taking into account that gaseous losses of N predominantly consisted of denitrification (Table 4), the assessment of the sensitivity of nutrient loss pathways was limited to leaching to surface water and denitrification. The results of Fig. 4 demonstrate that the impacts of reducing N surpluses at farm level on reductions in N loads to surface water are limited for the clay and peat soil. For these soils, additional measures, e.g. bufferstrips or artificial wetlands should be considered. For the sandy soil there seems to be more room

for improvement, because the sensitivity for leaching was relatively high.

The relations between nutrient surpluses and nutrient losses are theoretically curvi-linear as shown in Fig. 1. In practice however they may look more linear (Figs. 4, 5), depending on the curve shape parameters. Also, there seems to be a 'breaking point' above which leaching rapidly increases (see e.g. Fig. 4 for sand). This breaking point coincides more or less with the denitrification capacity. The linear character of N surpluses or N inputs and N losses was also observed by Watson and Foy (2001) and hence, within certain limits of N surpluses, a soil type specific linear relation between N surpluses and N losses can be considered. However, the relations derived in this study remain conceptual until proven with experimental data, which are, however, hard to establish because of prolonged effects and because of (partly) unpredictable experimental conditions.

The results shown in Fig. 5 indicate that changes in P surpluses have the largest impact on P leaching on the sandy soil, followed by peat soil and finally the clay soil. This sequence is strongly influenced by the values of the constant  $\gamma$  which decreased from sand to clay due to a decline in the fraction of reversible bound P. Figure 5 provides a simplified impression of effects of changes in P surpluses on P leaching. The calculated P leaching at prevailing P surpluses was lower than the measured total P leaching (Table 5). Measured P leaching rates were 1.3 (sand), 3.4 (peat) and 8.0 (clay) times the predicted rates (Table 5, Fig. 5). Predicted leaching rates only considered inorganic P whereas measured leaching rates refer to total P. Measurements at the clay site showed that total P leaching was twice the leaching of inorganic P (Van der Salm et al. 2006). For the other sites only total P losses were measured but based on literature data for Dutch sites a factor of 2.0 to 2.5 may be expected between inorganic P and total P for the sandy soil and the peat soil (Chardon et al. 2007). Apart of these differences in absolute magnitude of the leaching fluxes the slope of the lines in Fig. 5 was affected by the time period considered and P application rate. Figure 5 was constructed assuming a constant ratio between reversible bound P and total sorbed P. On the short term changes in P surplus will mainly influence the pool of reversible bound P and accordingly the P concentrations and leaching fluxes. On the long term surpluses will lead to a change in

more strongly bound P forms as well. For example, experiments to reduce P leaching by prolonged zero P application showed that prolonged negative surpluses will lead to a gradual decline in strongly bound P forms (Koopmans et al. 2004; van der Salm et al. 2008)

The relations shown in Figs. 4 and 5 are unique for each site, and depend on the denitrification capacity of the soil for N and on the sorption characteristics of the soil for P. However, also hydrological pathways determine the potential for leaching. Losses of P from the sandy site and the clay site were comparable although the DPS was much higher for the sandy site than for the clay site. Highest P losses were found at the peat site, which had an intermediate DPS (Table 3). These differences in P losses can be explained by differences in sorption characteristics, differences in hydrological pathways and differences in the distribution of P within the soil profile between the sites. The sandy site was quite deeply drained and 84% of the water discharge was through deeper soil layers where the DPS was relatively low (Table 3). At the clay site 67% of the discharge was by means of drainage through trenches, coinciding with 75% of the P losses to surface water. Also, for the clay site high P losses were observed as a response to incidental heavy rainfall following manure application in early spring (Van der Salm et al. 2006). The peat site had an intermediate position with a somewhat higher DPS and drainage through deeper soil layers (Tables 3, 5). This intermediate position contributes to the high P losses observed for the peat soil. Another explanation for the relatively high losses from the peat soil was the release of P from eutrophic layers in the peat, contributing to about 50% of the total discharge. This contribution was not part of the mineralization mentioned in Table 5, because this methodology was related to the rooting zone (i.e. upper soil layers; Van Beek et al. 2004).

## Conclusions

Currently farm N and P inputs are reduced in order to, amongst others, reduce N and P losses to surface water. These measures are taken irrespective of soil type. The results of this paper demonstrate that with similar effort deviating results can be expected for different soil types, based on differences in soil

properties. Reducing N and P inputs as measure to reduce N and P leaching to surface water is likely to be most effective on sandy soils. For clay and peat soils hydrological conditions and high denitrification capacities hamper the effectiveness of such measures. For these soils additional measures should be considered.

## References

- Barracough D, Jarvis S, Davies G, Williams J (1992) The relation between fertilizer nitrogen applications and nitrate leaching from grazed grassland. *Soil Use Manag* 8:51–55. doi:10.1111/j.1475-2743.1992.tb00894.x
- Barton L, McLay CDA, Schipper LA, Smith CT (1999) Annual denitrification rates in agricultural and forest soils: a review. *Aust J Soil Res* 37:1073–1093. doi:10.1071/SR99009
- Bijay-Singh RJC, Ryden JC, Whitehead DC (1989) Denitrification potential and actual rates of denitrification in soils under long-term grassland and arable cropping. *Soil Biol Biochem* 21(7):897–901. doi:10.1016/0038-0717(89)90078-3
- Bučienė A, Švedas A, Antanaitis S (2003) Balances of the major nutrients N, P and K at the farm and field level and some possibilities to improve comparisons between actual and estimated crop yields. *Eur J Agron* 20:53–62. doi:10.1016/S1161-0301(03)00073-X
- Bussink DW (1994) Relationships between ammonia volatilization and nitrogen fertilizer application rate, intake and excretion of herbage nitrogen by cattle on grazed swards. *Nutr Cycl Agroecosyst* 38(2):111–121
- Chardon WJ, Aalderink GH, Van der Salm C (2007) Phosphorus leaching from cow manure patches on soil columns. *J Environ Qual* 36:17–22. doi:10.2134/jeq2006.0182
- De Vries W, Kros H, Oenema O (2001) Modeled impacts of farming practices and structural agricultural changes on nitrogen fluxes in the Netherlands. *Scientific World J* 1(Suppl 2):664–672
- Fried M, Tanji K, Van de Pol R (1976) Simplified long term concept for evaluating leaching of nitrogen from agricultural land. *J Environ Qual* 5:197–200
- Heinen M (2006) Application of a widely used denitrification model to Dutch data sets. *Geoderma* 133:464–473. doi:10.1016/j.geoderma.2005.08.011
- Hénault C, Germon JC (2000) NEMIS, a predictive model of denitrification on the field scale. *Eur J Soil Sci* 51:257–270. doi:10.1046/j.1365-2389.2000.00314.x
- Hendriks RFA (1993) Nutriëntenbelasting van het oppervlaktewater in veenweide-gebieden (Nutrient loading of surface water in peat land areas). DLO Staring report 251, 164 pp (In Dutch)
- Jarvis S (1992) Grazed grassland management and nitrogen losses: an overview. Aspects of applied biology. In: Archer JR et al (eds) Nitrate and farming systems. Association of Applied Biologists, Wellesbourne, pp 207–214
- Koopmans GF, Chardon WJ, Ehlert PAI, Dolfing J, Suurs RAA, Oenema O et al (2004) Phosphorus availability for

- plant uptake in a phosphorus-enriched noncalcareous sandy soil. *J Environ Qual* 33:965–975
- Koops JG, Oenema O, Van Beusichem ML (1996) Denitrification in the top and sub soil of grassland on peat soils. *Plant Soil* 154:1–10. doi:[10.1007/BF00029269](https://doi.org/10.1007/BF00029269)
- LNV (2006) Ministry of agriculture and food quality. Manure Policy 2006: tables for calculating manure production. [www.minlnv.nl](http://www.minlnv.nl)
- Lockyer DR, Whitehead DC (1990) Volatilization of ammonia from cattle urine applied to grassland. *Soil Biol Biochem* 22:1137–1142. doi:[10.1016/0038-0717\(90\)90040-7](https://doi.org/10.1016/0038-0717(90)90040-7)
- Menon RG, Hammon LL, Sissing HA (1989) Determination of plant-available phosphorus by iron hydroxide-impregnated filter paper (P<sub>i</sub>) soil test. *Soil Sci Soc Am J* 53:110–115
- Misselbrook TH, Van der Weerden TJ, Pain BF, Jarvis SC, Chambers BJ, Smith KA et al (2000) Ammonia emission factors for UK agriculture. *Atmos Environ* 34:871–880. doi:[10.1016/S1352-2310\(99\)00350-7](https://doi.org/10.1016/S1352-2310(99)00350-7)
- Mosier AR, Klemmedtsson L (1994) Measuring denitrification in the field. In: Weaver RW et al (eds) *Methods of soil analysis*. SSSA, Madison, pp 1047–1065
- OECD (Organisation for Economic Co-operation and Development) (2001) *Environmental indicators for agriculture. Volume 3 methods and results*. Paris, France
- Oenema O, Velthof GL, Kuikman PJ (2001) Technical and policy aspects of strategies to decrease greenhouse gas emission from agriculture. *Nutr Cycl Agroecosyst* 60:301–315. doi:[10.1023/A:1012601113751](https://doi.org/10.1023/A:1012601113751)
- Oenema O, van Liere L, Schoumans OF (2005) Effects of lowering nitrogen and phosphorus surpluses in agriculture on the quality of groundwater and surface water in The Netherlands. *J Hydrol (Amst)* 304:289–301. doi:[10.1016/j.jhydrol.2004.07.044](https://doi.org/10.1016/j.jhydrol.2004.07.044)
- Olsthoorn C, Fong N (1998) The antropogenic nitrogen cycle in The Netherlands. *Nutr Cycl Agroecosyst* 52:269–276. doi:[10.1023/A:1009724024770](https://doi.org/10.1023/A:1009724024770)
- RIVM (2002) MINAS en Milieu (MINAS and the environment) RIVM report 718201005. 205 pp
- Schoumans OF, Groenendijk P (2000) Modeling soil phosphorus levels and phosphorus leaching from agricultural land in the Netherlands. *J Environ Qual* 29:111–116
- Schwertmann U (1964) Differenzierung der Eisenoxide des Bodens durch Extraction mit Ammoniumoxalat-Lösung. *Z Pflanzenernähr Düng Bodenk* 105:194–202
- Slak M, Commagnac L, Lucas S (1998) Feasibility of national nitrogen balances. *Environ Pollut* 102(S1):235–240. doi:[10.1016/S0269-7491\(98\)80038-9](https://doi.org/10.1016/S0269-7491(98)80038-9)
- Tilman D, Fargione J, Wolff B, d'Antonio C, Dobson A, Howarth R et al (2001) Forecasting agriculturally driven global environmental change. *Science* 292:281–284. doi:[10.1126/science.1057544](https://doi.org/10.1126/science.1057544)
- Torenbek R, Voskamp T (2003) Nieuwe inzichten over uit-en afspoeling van meststoffen New insights in leaching of nutrients. *H2O* 18:19–21 (In Dutch)
- Van Beek CL, Eertwegh van den GAPH, Van Schaik FH, Van den Toorn A (2003a) Surface runoff from intensively managed grassland on peat soils; a diffuse source of nitrogen and phosphorus in surface waters. In: Walther W, Worch E, Wöhllich S (eds) *Conference proceedings of diffuse input of chemicals into soil and groundwater*. Dresden, Germany, 26–28 February, pp 9–17
- Van Beek CL, Brouwer L, Oenema O (2003b) The use of farmgate balances and soil surface balances as estimator for nitrogen leaching to surface water. *Nutr Cycl Agroecosyst* 67:233–244. doi:[10.1023/B:FRES.0000003619.50198.55](https://doi.org/10.1023/B:FRES.0000003619.50198.55)
- Van Beek CL, Van den Eertwegh GAPH, Van Schaik FH, Velthof GL, Oenema O (2004) The contribution of dairy farming on peat soil on N and P loading of surface water. *Nutr Cycl Agroecosyst* 70:85–95. doi:[10.1023/B:FRES.0000045984.93498.c3](https://doi.org/10.1023/B:FRES.0000045984.93498.c3)
- Van der Salm C, Dolging D, Van Groenigen JW, Heinen M, Koopmans G, Oenema J, Pleijter M, Van den Toorn A (2006) Diffuse belasting van het oppervlaktewater met nutriënten vanuit grasland op een zware kleigrond (Diffuse loading of surface water with nutrients from grassland on heavy clay). Alterra report 1266 (In Dutch)
- Van der Salm C, Dolging J, Heinen M, Velthof GL (2007) Estimation of nitrogen losses via denitrification from a heavy clay soil under grass. *Agric Ecosyst Environ* 119:311–319. doi:[10.1016/j.agee.2006.07.018](https://doi.org/10.1016/j.agee.2006.07.018)
- Van der Salm C, Chardon WJ, Koopmans GL, Van Middelkoop JC, Ehlert PAI (2008) Phytoextraction of phosphorus-enriched grassland soil. *J Environ Qual* (submitted)
- Van der Zee SEATM, Van Riemsdijk WH (1988) Model for long-term phosphate reaction kinetics in soil. *J Environ Qual* 17:35–41
- Watson CJ, Foy RH (2001) Environmental impacts of nitrogen and phosphorus cycling in grassland systems. *Outlook Agric* 30(2):117–127