Nutrient losses from grassland on peat soil

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Christy van Beek

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Nutrient losses from grassland on peat soil

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Abstract

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The general aim of the present thesis research was to increase the understanding of N and P routes to and from intensively managed grassland on peat soil in a polder in the western part of the Netherlands to ascertain possibilities and limitations of reducing N and P concentrations in the surface water of the polder. To achieve this general aim the following steps were made:

1. quantification of N and P surpluses at farm and field level
2. quantification of N and P pathways
3. integration of N and P surpluses with N and P loss pathways

An integrated research approach was followed, i.e. the research consisted of laboratory experiments, field experiments, field monitoring studies and mathematical simulations. Fluxes of N and P were large compared to other soils, especially for mineralization (N and P) and gaseous losses through denitrification (N). Surpluses of N and P were unevenly distributed over fields within farms, and it was hypothesized that this heterogeneity may contribute to underestimation of leaching of N and P to surface water by farm-scale modeling. The contribution of fertilizers, manure and cattle droppings to the N and P loading of surface water was estimated at 43-50% for N and at 10-48% for P. The subsoil of the eutrophic peat soil also contributed to the N and P loading of the surface water, and this contribution was estimated at 8-27% for N and at 33-83% for P. This latter contribution can be reduced by raising the surface water level, as was suggested by the results of 2 dimensional simulation modeling. The large nutrient fluxes, the uneven distribution of surpluses over fields and the limited contribution of relatively easy to control nutrient sources on nutrient loading of surface water hamper the effectiveness of reducing N and P inputs at farm level in order to decrease N and P loading of surface water.

Keywords: nitrogen, phosphorus, peat, dairy farming, denitrification, leaching, nutrient balance, nutrient surplus, eutrophication

Contents

Voorwoord

Een proefschrift is een proeve van bekwaamheid. Maar bekwaamheid in wat eigenlijk? In wetenschappelijk inzicht, in doorzettingsvermogen, in geduld? Nu, aan het eind van mijn proefschrift, weet ik dat het een combinatie is van deze drie (of misschien nog wel meer) factoren. En voor elk van deze heb ik steun ontvangen, van vrienden en familie, van collega's en van promotor en co-promotor.

Hoewel de volgorde van dankwoorden doorgaans vaststaat van (co)promotoren, via collega's naar vrienden en familie tot uiteindelijk de belangrijkste personen in iemands leven, wil ik graag beginnen met het bedanken voor de onvoorwaardelijke steun van Luc. Niet dat hij dat hele proefschrift avontuur nu zo ontzettend leuk vond, maar omdat hij me altijd weer uit de put wist te halen. Als ik weer eens begon te twijfelen of ik hier überhaupt aan had moeten beginnen, dan was Luc daar. Altijd. Hoe vaak zou je gezegd hebben dat ik het heus wel kan, maar niet in één dag? Ontelbare keren. En ik ben je ontzettend dankbaar, zonder jou was dit proefschrift nooit afgerond.

Oene, jou dank ik voor je optimisme, dat echt schier oneindig is, je enthousiasme en je capaciteit om (haast) nooit problemen te zien. Dat laatste is heerlijk, maar heeft soms een lastig staartje. Want wat nu als ik de grondboor niet de grond in krijg geduwd? Dan kon ik altijd terecht bij Gerard, die de tijd en rust had om samen met mij na te denken over de alledaagse problemen en probleempjes waar ik tegenaan gelopen ben. Oene en Gerard, jullie zijn twee heel verschillende mensen, en samen vormen jullie een prachtig team.

Bij het uitvoeren van praktisch werk heb ik veel te danken aan Eduard Hummelink en Popko Bolhuis en aan Thijs Meijer, Tamás Salánki en Bert van den Borne, die voor hun afstudeervak of stage de Vlietpolder 'ingedoken' zijn. Verder een woord van dank voor de niet wetenschappelijke, maar wel sociaal-emotioneel onmisbare steun van mijn (ex)kamergenoten, de koffiekamer, mijn loopmaatjes en vrienden die zich met raad en daad (en soms een schouder) hebben bijgestaan.

Een bijzondere plek in deze inleiding is bestemd voor Gé van den Eertwegh, de projectleider vanuit het Hoogheemraadschap van Rijnland en thans werkzaam bij het Waterschap Rivierenland. Gé weet mij altijd weer te motiveren en te inspireren en, meer dan hij wellicht zelf in de gaten heeft gehad, heeft hij me wegwijs gemaakt in het waterwereldje. Ook een woord van dank aan de medewerkers van het Hoogheemraadschap van Rijnland, in het bijzonder Frank van Schaik en Bruce Michielsen. We hebben samen heel wat afgepuzzeld en ik heb daar erg veel plezier aan beleefd. Tevens zijn de tekeningen op de voorkant en bij Hoofdstuk 2, 4 en 5 beschikbaar gesteld door het Hoogheemraadschap van Rijnland, waarvoor mijn dank.

De rol van ouders in een promotie is naar mijn mening altijd een beetje ambivalent. 'Tuurlijk hebben ze me gesteund, maar dat hadden ze ook gedaan als ik een andere keuze had gemaakt (en dat is nou het mooie van ouders, iets wat ik pas sinds de geboorte van Noortje echt begrijp). Maar toch was het iets anders, want hoe vaak komt het voor dat vader en dochter ongeveer in hetzelfde vakgebied werkzaam zijn? Kees, zonder het echt te beseffen ben ik jouw voetsporen getreden. Jij en Janneke hebben mij altijd gesteund, door mee te denken én door mee te voelen, waarvoor mijn hartelijke dank.

En nu is er toch een heel belangrijk persoon(tje) aan het einde beland: Noortje. Lieve, kleine Noortje. Met alle respect Noortje, maar eigenlijk heb jij bijzonder weinig bijgedragen aan dit proefschrift. Behalve dan dat ik door jou des te gemotiveerder was om het af te ronden en vanaf nu kunnen we 's woensdags de hele dag samen spelen. En natuurlijk met je kleine broertje of zusje die we in juli verwachten.

Ten geleide, of waarom er landbouw is in het veenweidegebied

In het boek ‘Publieke Werken’ van Thomas Rosenboom staat een prachtige uitspraak: ‘Veen is water dat te nat is om te belopen en te droog is om te bevaren’. Zo bezien is het een redelijk hachelijke onderneming om intensieve veehouderij (want dat is het) te bedrijven in het veenweidegebied. En toch is het er. Hoe is het zo gekomen?

Kort gezegd ontstond er rond het jaar 1200 AD behoefte aan landbouwgrond in de directe omgeving van de grote steden in de Randstad. Door het drassige gebied, wat nu het veenweidegebied is droog te leggen, kwam een groot areaal landbouwgrond in deze gewilde omgeving vrij. Deze drooglegging ging niet zonder slag of stoot en er zijn talloze mislukkingen geweest. Echter, na verloop van tijd, en met de verbetering van de maalcapaciteit van molens en later van gemalen, onstond er een wijds graslandschap. In eerste instantie zag dit landschap er heel anders uit dan zoals wij het thans kennen, maar hiermee was wel de basis voor het veenweidelandschap gelegd.

Door het wegpompen van water begon de bodem te dalen en een vicieuze cirkel werd in gang gezet. Immers, iedere peilverlaging heeft bodemdaling tot gevolg, en iedere bodemdaling vraagt om extra peilverlaging, etc. Bovendien is vanaf 1950 het gebruik van kunstmest, en daarmee de productiviteit van grasland, sterk toegenomen. Door de ondiepe waterpeilen komt een aanzienlijke hoeveelheid van de toegenomen hoeveelheid nutriënten in de veenweidesloten terecht en dat is schadelijk voor de waterkwaliteit.

Als ‘oplossing’ voor deze problemen wordt vaak een inkrimping van de landbouw genoemd. Dit is enigszins in de trant van de ‘vervuiler betaalt’ zoals dat gangbaar is in de industrie, en waar het naar mijn mening ook zeer terecht is. Echter, bij de landbouw vind ik het iets anders liggen. Tot aan 1967 heeft de heer Staring, naar wie tot voor kort nog een heel instituut werd vernoemd én aan wie een vooruitziende blik van ‘enkele menschenlevens’ wordt toegekend, nog actief het gebruik van kunstmest gestimuleerd. Er zijn dus wel degelijk historische redenen voor het ontstaan van het (vermestings)probleem en om dat nu allemaal op het bordje van de boer te leggen.... De wetenschap en het beleid hebben daarin ook haar verantwoordelijkheid en dit doet me denken aan een uitspraak van een veehouder tijdens één van mijn veldbezoeken. Hij zei: ‘Hoe is nou toch mogelijk dat ik gekort word op mijn mineralenbalans, terwijl jij nog moet uitzoeken hoe het werkelijk zit?’ En inderdaad, over dat laatste gaat dit proefschrift.

Chapter 1: General introduction

Grassland on peat soils

In the Western and Northern part of The Netherlands outstretched peat lands can be found (Figure 1.1). These peat lands are predominantly used as grassland for dairy farming and therefore go by the name peat pasture district (Dutch: veenweidegebied). Peat pastures are characterized by grasslands intersected by numerous, shallow ditches (Figure 1.2) that are used to regulate water supply (drainage) and to divide the fields. The peat pasture district in the western part of The Netherlands is right in the middle of one of the most densely populated parts of the world and is also called the 'Green Heart' of the Randstad metropolis. This popular name reveals the high recreational value of the area. However, because of various threats (urbanization, subsidence, flooding, eutrophication of surface water) the area is under pressure by conflicting interests of farmers, civilians and nature conservationists and lack of adequate intervention (Van der Ploeg, 2001).

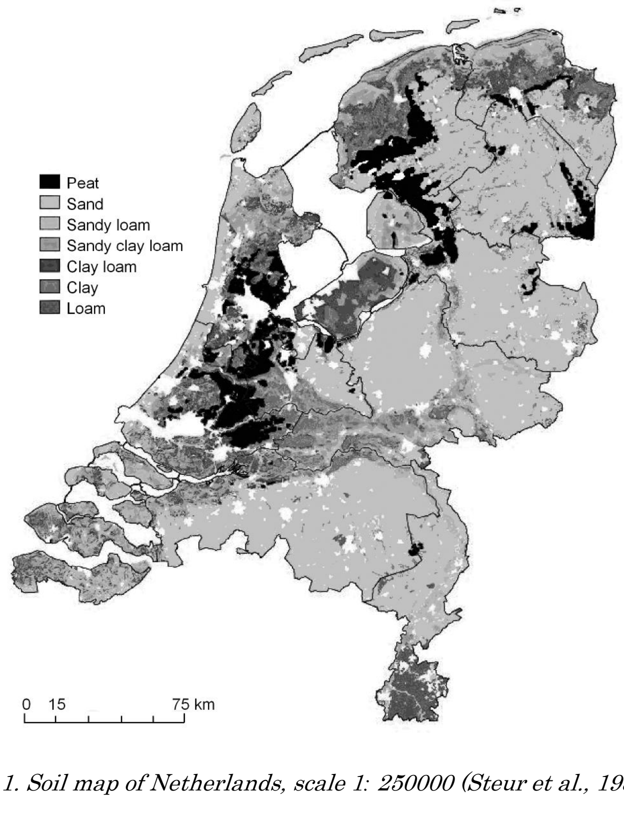

Figure 1.1. Soil map of Netherlands, scale 1: 250000 (Steur et al., 1985).

Originally, the Western peat district consisted of large Sphagnum fields, which were colonized in about 1050 AD (Van de Ven, 1993). At that time, the Sphagnum fields were still above mean sea level, but by reclaiming the area through drainage, peat digging, burning the top soil and so on, the surface level gradually lowered. As a result, it became increasingly difficult to drain the area, and finally, in the 15th century a separation was made between *polders* (low lands) and *boezem* (the water at higher elevation surrounding the polders). The water inside the polders was pumped to the boezem (against the hydraulic gradient) by windmills, which were later replaced by mechanical pumps. With this system, man had won the battle against water. At least for some time.

In the 17th century the demand for fuel by households and industry of the growing population rapidly increased and the peat in the polders was systematically exploited (Van de Ven, 1993). This resulted in an even lower elevation of the polders and some polders were even lost and became lakes (these lakes can be identified by the extension '*plas*'). However, only peat with low ash content was suitable for fuel and consequently outstretched peat areas (about 290.000 ha) can still be found in the western part of The Netherlands (Figure 1.1).

Figure 1.2. Typical sight of a Dutch peat pasture polder, also study area.

Since the introduction of windmills (circa 1400 AD), the water management in the Western peat district has been almost entirely man-controlled. The water management in this area was trusted to so-called water boards. These organisations were very powerful, as they could decide which land was drained and which was not. The water boards have a long, rich history, going back to 1200 AD when the water board of Rijnland was founded (Giebels, 2002).

Agriculture in The Netherlands

From 1950 to 1985 agricultural production (including milk production) in The Netherlands increased greatly, partly because of the increased use of fertilizers and the import of animal feed. This trend could not last very long and in the early 70's the first voices rose to restrict excessive applications of animal manure and fertilizers, because of their detrimental effects on the environment (Henkens and Koopmans, 1973). One of these detrimental effects on the environment is eutrophication of surface waters. The term eutrophication reflects a chain of effects caused by excessive nutrient loading of surface water, eventually resulting in anoxic conditions in surface water and changes in ecological equilibria (Tilman et al., 2001; Smith et al., 1999). The Dutch government responded to this problem in 1984 by prohibiting further extension of pig farming, introducing milk quota, and by setting limits to manure application. In the following years, nutrient policy became more and more stringent (e.g. Henkens and van Keulen, 2001; Neeteson, 2000; Beembroek et al., 1996; RIVM, 2002) and as a result, manure and fertilizer applications decreased. Mean fertilizer applications are now about 150 kg N $ha^{-1}y^{-1}$ and 10 kg P $ha^{-1}y^{-1}$ (Figure 1.3).

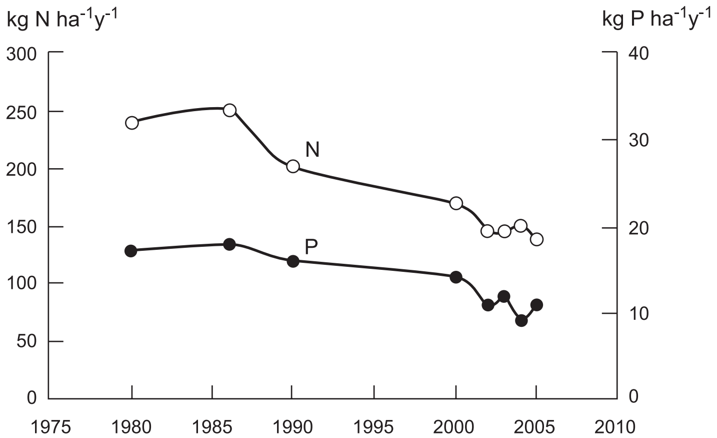

Figure 1.3. Nitrogen (N) and phosphorus (P) fertilizer application in The Netherlands (www.mnp.nl).

Dutch dairy farms are among the most productive dairy farms worldwide, i.e. achieve the highest production per unit of input and per unit of surface area. However, nutrient losses to the environment are correspondingly high (OECD, 2001). In modern, specialized dairy farms generally 4 compartments can be distinguished; the animal-compartment, the manure compartment, the soil-compartment and the plant-compartment (e.g. Kohn et al., 1997; Aarts et al., 2000). In each compartment certain losses of nitrogen (N) and phosphorus (P) may occur, resulting in nutrient use efficiencies[1] at farm-level of 0.15 to 0.30 for N and 0.30 to 0.60 for P (Van Keulen et al., 1996).

The largest nutrient inputs and outputs in intensvively managed dairy farms are imports of fertilizers and animal feed and exports of milk and animals (and sometimes also manure). Within farms there are large fluxes to and from the fields (Figure 1.4). Nutrient flows to and from the fields depend on farm management, soil type, meteorological conditions and hydrological conditions. Dairy farms on peat soils used to be less intensively managed than dairy farms on other soil types, because of the shorter working season[2]. However, in the 1990's these differences largely disappeared due to improved agricultural techniques (Reijneveld et al., 2000).

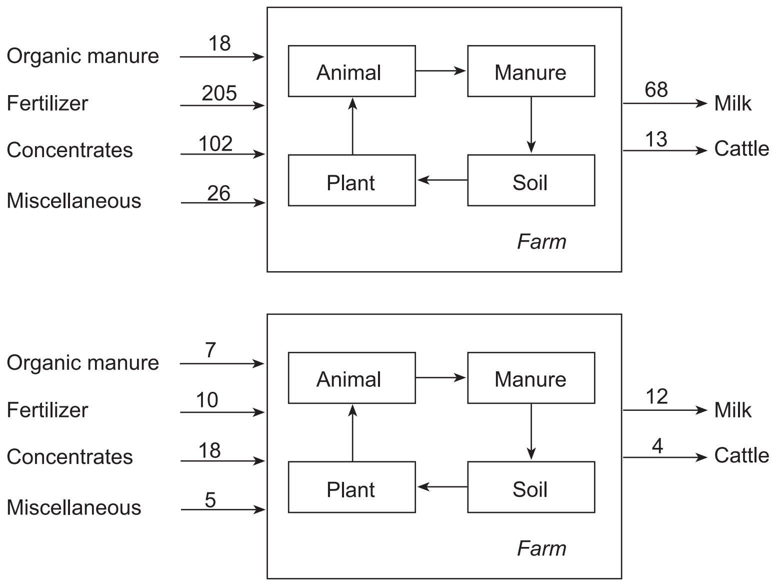

Figure 1.4. Average inputs and outputs of N (above) and P (below) of dairy farms on peat soil in The Netherlands in 1997 (Reijneveld et al., 2000).

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[1] Nutrient Use Efficiency (NUE) = Nutrients withdrawn in harvested products / Nutrients applied to the crop.

[2] Peat soils generally have a shorter working season than other soil types, because in early spring and autumn the bearing capacity of the soil prohibits the use of heavy machinery.

Nutrient losses

Ultimately, the farm nutrient surplus, i.e. the amount of nutrients applied to a farm minus the amount of nutrients exported from a farm, is lost to the environment or stored in the soil. Van Noordwijk (1999) compared the fate of this surplus with throwing a dice, i.e. the fate of the nutrient surplus being divided over the nutrient loss pathways as a matter of chance. Although the scope of his publication was different from the current thesis, the comparison with throwing dices is intriguing. Imagine the surplus, what would it do? Yet, reaction thermodynamics and environmental conditions finally determine the answer and, hence, the fate of the nutrient surplus is not a game of chance. The most important nutrient loss pathways are leaching to ground water and to surface water and volatilization of gaseous N to the atmosphere.

Leaching

Leaching is the transport of nutrients through or over the soil to groundwater and/or surface water. Leaching of nitrate (NO_3) to groundwater may violate the water quality standards set by the World Health Organisation (Joosten et al., 1998; Goodchild, 1998), while leaching of N and P to surface water leads to eutrophication of surface water. In peat areas leaching to groundwater is generally not considered as a major problem as NO_3 concentrations are generally low (Fraters et al., 2002). Leaching to surface water on the other hand is considered as a major problem in many (agriculturally used) peat land areas since quality standards for N and P in surface water are frequently violated (Portielje and Van der Molen, 1998) and severe symptoms of eutrophication can be observed. However, the relations between N and P inputs to the soil and N and P concentrations in surface water are complicated by interactions between peat biogeochemistry, hydrological conditions and agricultural activities and therefore N and P inputs to the soil can not be related directly to N and P concentrations in surface waters.

Gaseous losses

Gaseous losses of N species (P does not have volatile species) are caused by nitrification, denitrification and NH_3 volatilization. Volatilization of NH_3 (ammonia) is related to the pH-dependent equilibrium between dissolved NH_3 and gaseous NH_3. Volatilization of NH_3 from grazed grasslands has been extensively studied by, amongst others, Bussink (1996a) and Misselbrook et al. (2000). These studies showed that relatively fixed fractions of about 5-8% of the N excretion is volatilized as NH_3.

Nitrification is the microbiological oxidation of NH_4 to NO_3 with N_2O (nitrous oxide) as possible by-product (Wrage et al., 2001). Nitrification can be followed by denitrification which is the anaerobic reduction of NO_3 to N_2O and N_2 by micro-organisms. In general, gaseous losses of N_2O and N_2 via denitrification exceed gaseous losses of N_2O via nitrification and as long as the reduction of NO_3 is completed, i.e. till N_2, denitrification as such is not an environmental problem.

Consequently some measures to decrease mineral N contents in soil are based on increasing denitrification rates (e.g. Martin et al., 1999). However, the emission of N_2 can not be separated from the emission N_2O (Stevens and Laughlin, 1998). N_2O contributes to the destruction of stratospheric ozone (Crutzen, 1970) and may lead to global climate change (Wang et al., 1976). Notably, the global warming potential of N_2O equals 250 to 300 CO_2 equivalents (Houghton et al., 1996). Increasing denitrification rates is, therefore, not the way to reduce N surpluses. In peat soils, conditions for N_2O emission and for denitrification are relatively favourable. Of all soil types in The Netherlands peat soils are believed to have the highest N_2O emissions and denitrification losses (De Vries et al., 2003; Kroeze et al., 2004, Velthof and Oenema, 1995).

Motivation of the present study

Ever since the colonization around 1050 AD, the Western peat pasture district has known problems with water management, and recently in addition, with nutrient management. The Rijnland Water Board, responsible for the water quality of 105000 ha of mostly peat land, recorded nutrient concentrations in the Vlietpolder in the Western peat district since 1970 onwards (Figure 1.5).

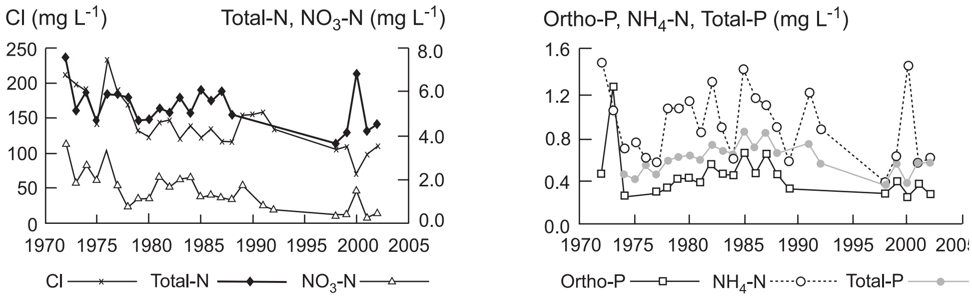


Figure 1.5. Longterm trends of Cl, total-N, ortho-P, NO_3, NH_4 and total-P concentrations in surface water of a peat pasture polder in the Western peat district.

The reductions in Cl and NO_3 concentrations (the reduction in total-N was totally accounted for by the reduction in NO_3) have been ascribed to changes in agricultural management, the reduction of N and KCl fertilizer applications, the restrictions on manure and fertilizer application from 1985 onwards and in the successful reductions of point sources nationwide (Plette et al., 2002). However, from 1998 onwards the NO_3 reductions levelled off, while total-N concentrations increased (Figure 1.5). This situation resulted in the call for action by the Rijnland Water Board in 1999 (Van den Eertwegh, 1999), which resulted in a joint project with farmers, water board, regional policy makers, national policy makers and research institutes. In this 'peat pasture project' N and P routes to surface water and relevant processes affecting these routes were studied for a small polder in the Western part of The Netherlands.

Notwithstanding extensive research on nutrient dynamics of grazed grasslands (e.g. Jarvis et al., 1996) the understanding of N and P routes through and from agricultural fields is still limited. Especially for denitrification and leaching from soil there are only few experimental studies. Denitrification is sometimes indirectly estimated as the amount 'unaccounted for' in detailed balance sheets (e.g. Watson and Atkinson, 1999), but Van der Salm et al. (2007) showed that considerable differences may occur between these kind of estimates and field measurements. Leaching, on the other hand, is often estimated by mathematical models that basically multiply the nutrient content in the soil solution with the water flux to groundwater and to surface water (e.g. SWAP, FUSSIM).

Aims and research approach

The general aim of the research described in this thesis is to increase the understanding of the N and P routes to and from intensively managed grassland on peat soil located in a polder to ascertain possibilities and limitations of reducing N and P concentrations in the surface water of the polder. To achieve this aim the following steps were made:

1. quantification of N and P surpluses at farm and field level
2. quantification of N and P pathways
3. integration of N and P surpluses with N and P loss pathways

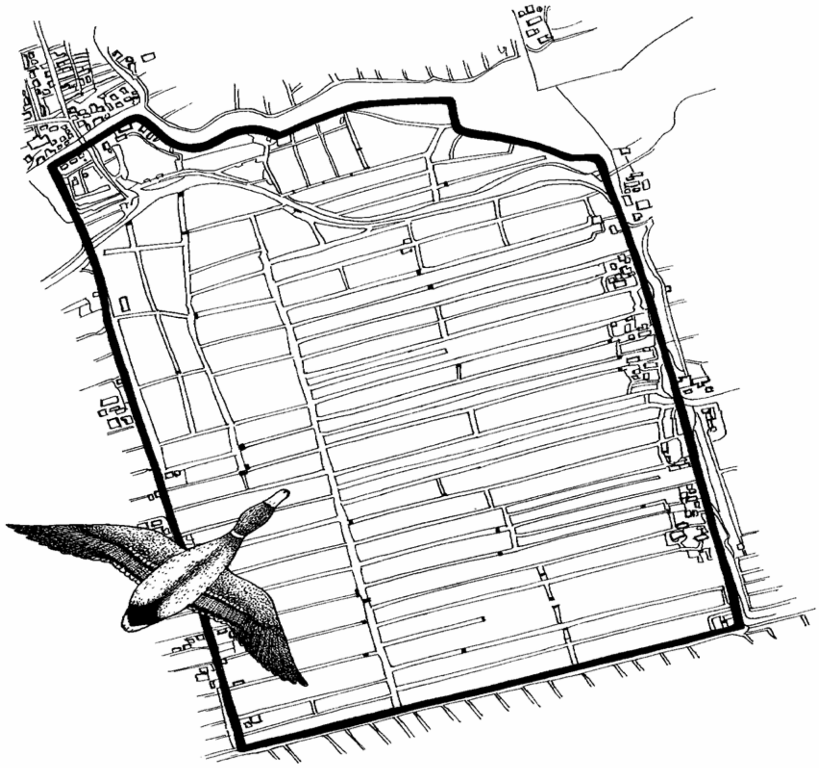
An integrated research approach was followed, i.e. the research consisted of laboratory experiments, field experiments, field monitoring studies and mathematical simulations.

The study area was a small peat pasture polder (202 ha), about 30 km from Amsterdam and about 20 km from the North Sea. The polder is almost exclusively used for dairy farming (>90% of the surface area). Main emphasis was given to the nutrients N and P, but also Cl and SO_4 are discussed. Concerning the nutrient loss pathways, most attention was given to leaching and denitrification, because for these processes few experimental data were available, whereas NH_3 volatilization was estimated by simple calculation procedures using data from literature (Misselbrook et al., 2000; Oenema et al., 2000).

Outline of this thesis

This thesis consists of 4 publications (3 published and 1 accepted for publication in international scientific journals) and a general discussion (Chapter 6). Following this introduction, first a detailed quantification of N and P surpluses at farm-level and field-level is provided (Chapter 2). Then, the major nutrient loss pathways, denitrification (Chapter 3) and leaching (Chapter 4) are presented. Leaching and the composition of the soil solution were subsequently assessed in Chapter 5. In Chapter 6 the results of the preceding chapters were integrated to come to an assessment of possibilities and limitations of reducing N and P losses to the environment for grassland on peat soil in the Western peat district of The Netherlands.

Chapter 2: The use of farmgate balances and soil surface balances as estimator for nitrogen leaching to surface water



Slightly modified from:
Van Beek C.L., Brouwer L. and Oenema O. 2003. Nutrient Cycling in Agroecosystems 67: 233-244.

Abstract

Farmgate balances (FGBs), defined as the difference between nutrient input and nutrient output at farm level, are currently used as a tool to monitor changes in nitrogen (N) and phosphorus (P) leaching to groundwater and surface water. We postulate that the estimator value of FGBs for N and P leaching to ground water and surface water depends on i) the distribution of N and P surpluses over fields within farms, and ii) the partitioning of the surplus over the various nutrient loss pathways. In this study we assessed intra farm variability of N and P surpluses and its possible consequences on N leaching to surface waters. Furthermore, we investigated the effect of policies to decrease N and P surpluses at farm level on N and P surpluses at field level. FGBs were derived for 6 dairy farms in a hydrologically isolated polder with grassland on peat soil for 3 years (1999, 2000 and 2001). Soil surface balances (SSBs), defined as the differences between nutrient input and nutrient output at field level, were derived for the accompanying 65 fields for the same years. On average, FGB surpluses decreased from 271 kg N $ha^{-1}y^{-1}$ and 22 kg P $ha^{-1}y^{-1}$ in 1999 to 213 kg N $ha^{-1}y^{-1}$ and 13 kg P $ha^{-1}y^{-1}$ in 2001. Variances in N and P surpluses between fields per farm were compared with variances between farms. For N, variances between fields per farm exceeded variances between farms for all years. A non-linear model was fitted on the measured N loading of the surface water. This model showed that N leaching to surface water was underestimated by 2 to 46% if the variability in N surpluses between fields per farm was not taken into account. We concluded that N leaching to surface water can be underestimated when the estimate of N leaching to surface water is based on data at farm level due to the large variability in N surpluses between fields per farm. The extent of this bias by a given distribution of N surpluses within farms was largely controlled by the partitioning of the N surplus over the various nutrient loss pathways, notably denitrification.

Introduction

Nutrient balances are increasingly used as environmental performance indicators or as policy support tools (e.g. EUROSTAT, 1997; OECD, 2001). The advantage of using nutrient balances is that relatively difficult to quantify balance entries (e.g. leaching) can be estimated from relatively easy to quantify balance entries. The focus is commonly not on the balance entries, but on the surplus or deficit resulting from the balance. The more balance entries are included, the more accurate the environmental performance indicator. However, complexity and uncertainty also increase with an increasing number of balance entries. Hence, the deliberation is often between the straightforward but restricted balances with a limited number of balance entries and the complicated but detailed balances with many balance entries (Watson and Atkinson, 1999; Oenema and Heinen, 1999). The link between nutrient balances used as a policy support tool and the environmental performance is still limited (Brouwer, 1998). In part this is due to a scale discrepancy between nutrient balances used as policy support tool and nutrient balances used as environmental performance indicator. For instance, in The Netherlands nutrient balances are used to decrease leaching of nutrients from agricultural fields to ground water and surface water. Best estimates of nutrient leaching are derived from balances at the field or plot scale (Watson and Atkinson, 1999). However, policy instruments to reduce nutrient leaching are normally actor-oriented, and therefore apply to the farm scale.

In The Netherlands, agriculture covers 60% of the total surface area and is a main source of N and P in surface waters (Oenema and Roest, 1998; Olsthoorn and Fong, 1998; RIVM, 2002). The Dutch agriculture is characterized by an intensive management, i.e. nutrient inputs, livestock density and crop yields are high. Amongst the common agricultural production systems, dairy farming has the second highest nutrient surplus per surface area, i.e. the difference between input of nutrients to the farm via fertiliser, feed, and manure and output via animal products (RIVM, 2002, Van Keulen et al., 1996). Moreover, dairy farms occupy more than half of the agricultural land (CBS/LEI-DLO, 2001). Surface waters cover 15% of the Dutch surface area, and intersect the agricultural land at small spatial scales via ditches, canals, lakes and rivers. As a result, there are many small-scale interactions between agricultural land and surface waters, especially in the western half of the country.

In The Netherlands farm gate balances (FGBs) in combination with levies when a certain target surplus is exceeded, used to be used as policy tool to decrease N and P leaching to ground water and surface water (Beembroek et al., 1996; Henkens and Van Keulen, 2001; Van den Brandt and Smit, 1998). FGBs are defined as the difference between nutrient input and output at farm level. We postulate that the FGB is a proper indicator for leaching when i) nutrient surpluses are equally distributed among fields within farms, and ii) all nutrient loss pathways are proportional to the nutrient surplus. In fact, propositions i) and ii) are tightly connected. When nutrient surpluses are homogeneously distributed over fields and the relationship between nutrient surplus and nutrient leaching to surface waters is linear, then the FGB is a proper relative indicator of nutrient leaching at farm level. However, when nutrient surpluses are not equally distributed and nutrient leaching to surface waters tends to be non-linear, then the FGB is not a proper indicator for nutrient leaching to surface water.

In this study we quantified inter (between farms) and intra (between fields per farm) variability of N and P surpluses over 3 years for 6 dairy farms on peat soil. During these 3 years, the levy free surpluses set by the Dutch government decreased. The farmers in this study were encouraged to be one year ahead of official levies. Consequently, a decrease in FGB surplus for both N and P was expected. We studied whether changes in FGBs affected the intra farm variability. The consequence of intra farm variability on estimates of N leaching to surface water was quantitatively evaluated. Estimations of leaching were limited to N, because the relationship between surplus and leaching to surface waters is more direct for N than for P (Van den Eertwegh, 2002).

Materials and methods

Site description

This study was carried out in the 'Vlietpolder' near Hoogmade in the western part of The Netherlands (52°10´ N; 4°36´ E). In this polder, surface water quality standards for N and P are frequently violated. The Vlietpolder is approximately 200 ha in size and lies about 2 m below mean sea level. The man-made topsoil (0-30 cm) consists of organic sandy clay that is rich in organic matter (20%). The subsoil consists of woody peat. The soil was classified as a Terric Histosol according to FAO classification (FAO-Unesco, 1988).

The Vlietpolder is surrounded by surface water and there is one inlet in the South and one outlet (pumping station) in the North (Figure 2.1). Briefly, when there is shortage of water, i.e. during dry periods in summer, water is let into the polder and when there is excess of water, i.e. during winter, water is pumped out of the polder. During its passage through the polder, water is loaded with N and P, mainly originating from agriculture, mineralization of peat and atmospheric deposition. There is a slight downward seepage to a nearby polder that is situated 5 m below sea level, but the majority of the water drains to the surface water. The Vlietpolder is typified by narrow fields (40-80 m width, 200-400 m length) surrounded by ditches with a maintained water level of 58 cm below surface level during winter and 48 cm below surface level during summer. As a result, groundwater level is shallow and during wet periods groundwater level reaches the soil surface. The shallow groundwater level and high ditch density (the surface water to land ratio of the Vlietpolder is approximately 1:10), cause shallow flow pathways and make the area vulnerable to leaching of nutrients to the surface water. Once in every 5 to 10 years, ditches are dredged and the slush is applied to the surrounding fields.

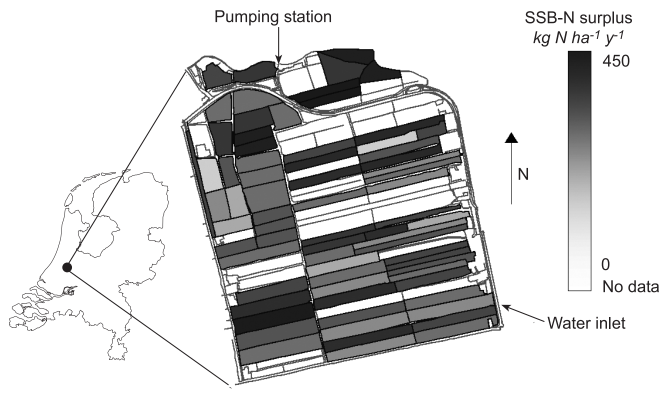


Figure 2.1. Spatial distribution of SSB-N surpluses in the Vlietpolder, The Netherlands, for the year 1999. White colour indicates not participating in project or resown field (excluded from this study). With increasing surplus, colour shifts from light to dark.

All land of the Vlietpolder is used for dairy farming. Six dairy farms, which occupy about 80% of the surface, participated in the study. Table 2.1 provides some general characteristics of the 6 dairy farms for 1999. For the years 2000 and 2001 these general characteristics only showed minor changes (data not shown).

Table 2.1. Some farm characteristics of the studied farms in the Vlietpolder, The Netherlands. n.d.= not determined.

Parameter	Dimension	Farm					
		A	B	C	D	E	F
Area	ha	25.4	20.5	32.9	28.9	34.4	62.1
Milk production	kg $cow^{-1}y^{-1}$	7923	7032	8387	7462	8108	8289
Milk production	kg $ha^{-1}y^{-1}$	18279	10399	12379	13174	15363	13093
Dairy cows	number	57.1	29.3	53.5	45.5	60.2	100.1
Concentrates	kg cow^{-1} y^{-1}	2693	n.d.	2373	2226	2450	2457
Herbage intake	kg d.m. $cow^{-1}d^{-1}$	18.5	17.9	16.8	18.1	19.1	17.0
Excretion	kg N $cow^{-1}d^{-1}$	0.50	0.48	0.45	0.49	0.51	0.46

Nutrient balances

To quantify inter and intra farm variability, FGBs and soil surface balances (SSBs), defined as the difference between nutrient input and nutrient output at field level, were derived. The system boundaries of the FGB and SSB are schematically shown in Figure 2.2. The FGB is an input-output balance at farm level, and was assessed for all 6 dairy farms for N and P for all 3 years (1999, 2000, 2001). The SSB is an input-output balance at field level and was established for 65 fields located on these farms for the same years. The SSB provides the net loading of the soil with N and P. In contrast to the FGB, ditches surrounding the fields were not included in the SSB. Consequently, ditches are inside FGB boundaries, but outside SSB boundaries and accordingly, slush application was considered as an external supply item for the SSB.

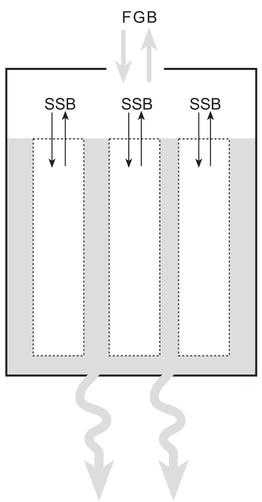


Figure 2.2. Schematic outline of the farmgate balance (FGB) and soil surface balance (SSB). The FGB is delineated by the solid line. Within the FGB several SSBs (dotted lines) are present, which interact with the FGB. Between fields ditches (grey) are located, which drain to higher water bodies

Table 2.2. Equations used to establish the FGB and the SSB. Symbols and subscripts are explained at the end of the table.

	Product	Equation used
Farm Gate Balance		
IN	Mineral fertiliser[a]	$N_{j,F} = Q_{j,F} \cdot C_F$
	Purchase of concentrates	$N_{j,Co} = Q_{j,Co} \cdot C_{Co}$
	Purchase of cattle	$N_{j,Ca} = Q_{j,Ca} \cdot C^b_{Ca}$
	Purchase of manure	$N_{j,Ma} = Q_{j,Ma} \cdot C^b_{Ma}$
	Purchase of fodder	$N_{j,Fo} = Q_{j,Fo} \cdot C^b_{Fo}$
OUT	Sale of cattle	$N_{j,Ca} = Q_{j,Ca} \cdot C^b_{Ca}$
	Sale of milk	$N_{j,M} = Q_{j,M} \cdot C_M$
	Sale of manure	$N_{j,Ma} = Q_{j,Ma} \cdot C^b_{Ma}$
Soil Surface Balance		
IN	Mineral fertiliser[a]	$N_{i,F} = Q_{i,F} \cdot C_F$
	Atmospheric deposition[c]	$N_{i,Atm} = 31$ kg N ha^{-1}y^{-1}
	Slurry application	$N_{i,S} = V_i \cdot cs^d \cdot A_i^{-1}$
	Dung and urine of grazing animals[e]	$N_{i,E} = GD_i \cdot E_j \cdot A^{-1}$
	Slush application[f]	$N_{i,Sl} =$ 211 kg N ha^{-1} and 9 kg P ha^{-1}
OUT	Mown grass	$N_{i,Mo} = Q_{i,Mo} \cdot C_G{}^g$
	Grazing	$N_{i,G} = GD_i \cdot I^h \cdot C_G{}^g \cdot A^{-1}$

a Fertiliser use included CAN (Calcium Ammonium Nitrate; 27% N), NP fertiliser (26% N; 7% P_2O_5) and TripleSuperPhosphate (43% P_2O_5).
b Standardized nutrient content (Tamminga et al., 2000).
c National Monitoring Programme Rainwater Composition (Bosschloo and Stolk, 1999)
d N and P contents of slurry were measured at each farm 4 times per year and interpolated to daily values. Average contents were 4.78 kg N m^{-3} and 0.73 kg P m^{-3}.
e Excretion by dairy cattle was calculated by mass-balance equations per farm, i.e. excretion equalled input through supplemental feeding (concentrates) and grass minus output through milk production and reproduction. For yearlings, calves and sheep no detailed information was available on feeding strategy and therefore standardized excretion rates were used (Tamminga et al., 2000).
f Unpublished data of ROC Zegveld.
g N and P contents of grass were measured at each farm 4 times per year and interpolated to daily values. Average contents were 0.035 g N g^{-1}d.m. and 0.0038 g P g^{-1}dm.
h Intake of dry matter by dairy cows was calculated from milk production level, supplemental feeding and grazing intensity (CVB, 2000). For yearlings, calves and sheep intake rates of CVB (2000) were used.

Symbol	Name	Dimension	Subscript	Name
N	Amount of nutrients (N or P)	kg ha^{-1}	i	Field
Q	Amount of product	kg ha^{-1}	j	Farm
C	Nutrient content of product	kg kg^{-1}	F	Fertiliser
V	Volume	m^3	Co	Concentrates
A	Surface	ha	Fo	Fodder
c	Concentration	kg m^{-3}	Ca	Cattle
GD	Grazing days	number of animals · d	M	Milk
I	Intake rate	kg animal^{-1}d^{-1}	Ma	Manure
E	Excretion	kg animal^{-1}d^{-1}	Atm	Atmosperic deposition
			S	Slurry
			Sl	Slush
			Mo	Mown grass
			NH_3	NH_3- volatilization
			G	Grass

Balance entries, methods of data collection and calculation procedures of the FGB and SSB are listed in Table 2.2. The balance entries of the FGB were based on the mineral accounting system used in The Netherlands at the time of study (MINAS, e.g. Breembroek et al., 1996). The balance entries not included in the FGB (i.e.

incorporated in the surplus) are atmospheric deposition, ammonia volatilization, denitrification, leaching and net mineralization. The SSB entries were chosen in correspondence with the methodology of the OECD (OECD, 2001; Parris, 1998; Van Eerdt and Fong, 1998). Fields were checked for N fixing plant species, which were absent. The SSB surplus consisted of ammonia volatilization, denitrification, leaching and net mineralization.

For the FGB, balance entries were derived from accounts and official data statistics using standardised nutrient contents for sales of organic manure and livestock (Tamminga et al., 2000). For the SSB we calculated the amount of N and P that entered or left the soil surface per field per day, which were then summed over the whole year. For that purpose, farmers daily recorded their activities per field on a so-called grassland calendar. Surpluses were equal to total input minus total output on a yearly basis.

In this study, farmers were encouraged to be one year ahead of the levy free surpluses set by the Dutch government. In 1999, 2000, 2001 and 2002 the levy free FGB surpluses for dairy farms were respectively 300, 275, 250 and 220 kg N $ha^{-1}y^{-1}$. For P the levy free surpluses were 20 kg P $ha^{-1}y^{-1}$ for 1999, 15 kg P $ha^{-1}y^{-1}$ for 2000 and 2001 and 11 kg P $ha^{-1}y^{-1}$ for 2002. The levy free surpluses for 2003 were 180 kg N $ha^{-1}y^{-1}$ and 8.7 kg P $ha^{-1}y^{-1}$, but at the time of this study still needed political confirmation.

N leaching to surface water

It was impossible to measure N leaching to the surface water for all 65 fields. Instead, the net N loading of the surface water in the polder was determined and subsequently a model was derived to calculate N leaching to surface water from each field. The net N loading of the surface water in the polder was determined by subtracting N loads of the outlet water from N loads of the inlet water. At the inlet and outlet stations, water samples were taken flow-proportional i.e. every 0.2 mm (winter) and 0.1 mm (summer) of discharge. Samples were stored dark and cold (4°C) until analysis. Concentrations of NO_3, NH_4 and organic-N were determined by Kjeldahl extraction (NH_4 and organic N) followed by standard photometric analysis at 540 mm wavelenght according to ISO 5663 (ISO, 1984).

N leaching to surface water from each field can be described as a function of SSB surplus and the contribution of other nutrient loss pathways. Besides leaching to surface water, the main fate of the SSB surplus is denitrification. Denitrification is generally described as a Michaelis-Menten function of NO_3 availability, i.e. with increasing N surpluses (and consequently NO_3 concentrations), denitrification increases less than proportionally (e.g. Hénault and Germon, 2000). This suggests that with increasing N surpluses, leaching increases more than proportionally. This rather simple derivation is supported by several field studies, but only when similar soil types were used. Generally, up to a certain threshold value, the relationship between field surplus and N load to surface water is gradual (e.g. Bechmann et al. 1998; Groeneveld et al. 1998; Watson and Foy, 2001); above the threshold value there is an increasing rate of loss (Barraclough et al., 1992; Fried et al., 1976; Watson et al., 1992). As a result, we assumed an expo-linear relationship between N surplus and leaching, according to equation 1 (modified after Goudriaan and Monteith, 1990);

$$Y = \frac{1}{\alpha}\ln(1 + e^{\alpha(x-\beta)}) \qquad \textit{eq. 2.1}$$

where Y is leaching to surface waters (kg N $ha^{-1}y^{-1}$), x is SSB surplus (kg N $ha^{-1}y^{-1}$), α (y ha kg^{-1}) and β (kg $ha^{-1}y^{-1}$) are curve shape parameters. To illustrate the possible effects of heterogeneity in SSB surpluses on the N leaching at farm level, two calculations were made with eq. 1 for each year; one with the heterogeneous distribution of SSB surpluses over fields as found in this study (per field calculation), and one with a homogeneous distribution of SSB surpluses per farm (i.e. per farm SSBs were averaged; per farm calculation).

Statistics

Inter farm variability was defined as the variation coefficient ($VC = \sigma/\mu$) of the 6 FGBs. Correspondingly, intra farm variability was defined as the pooled variance coefficient between SSBs per farm. Inter farm variability and intra farm variability were compared by a two-sided F-test. The test criterion was $F = s_1^2/s_2^2$, where s_1^2 was the estimated variance between the FGBs and s_2^2 is the pooled variance between the SSBs (Snedecor and Cochran, 1967). Statistical tests were performed with Genstat 5 release 4.2 (Genstat Committee, 2000).

Results

Farmgate balances

Table 2.3 shows the average FGBs per farm for N and P for the 3 years. Mineral fertiliser was the largest input for N, while the purchase of concentrates was the largest input for P. For both N and P milk production was the largest output term. Two farms exported animal manure to arable farms. From 1999 to 2001, FGB-N surpluses decreased significantly ($p<0.05$) by on average 58 kg N $ha^{-1}y^{-1}$ (21% of FGB-N surplus in 1999). FGB-P surpluses did not significantly decrease, although 4 out of 6 farms achieved a reduction in FGP-P surplus from 1999 to 2001. During this period purchase of N fertiliser decreased significantly ($p<0.01$), but the remainder of the balance entries, including milk production, did not change significantly. FGB-N surpluses were strongly related to fertiliser inputs, but less to milk production (Figure 2.3). Annual levy free surpluses were realised by 4 out of 6 farmers for both N and P. The levy free surpluses for the year 2003 (2 years ahead of policy) were achieved by 3 farmers in 2001. The accomplished decrease in FGB surpluses, without a decrease in production, was regarded as a major achievement by the farmers.

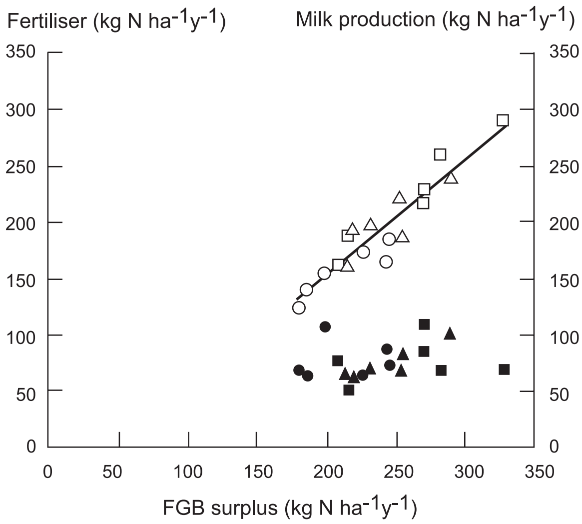


Figure 2.3. Fertiliser purchase (open symbols), milk production (solid symbols) and FGB surpluses in The Vlietpolder for the years 1999 (squares), 2000 (triangles) and 2001 (circles). Solid line shows linear regression of fertiliser purchase and FGB surplus (R^2=0.88).

Table 2.3. Average farm gate balances for the Vlietpolder, The Netherlands, and standard deviations for N and P for 1999, 2000 and 2001 (n=6).

		1999		2000		2001	
		N	*P*	*N*	*P*	*N*	*P*
		kg ha^{-1}y^{-1}					
IN	Purchase of fertiliser	222 ±42	14 ±8	190 ±38	10 ±8	157 ±22	6 ±3
	Purchase of concentrates	131 ±38	23 ±6	120 ±31	20 ±5	122 ±22	20 ±3
	Purchase of fodder	20 ±24	3 ±4	18 ±13	2 ±2	23 ±6	3 ±1
	Purchase of cattle	4 ±10	1 ±2	2 ±3	0 ±1	1 ±2	0 ±0
	Purchase of manure	0 ±0	0 ±0	2 ±4	0 ±0	2 ±3	0 ±0
OUT	Sale of cattle	12 ±2	3 ±1	10 ±3	3 ±1	12 ±4	3 ±1
	Sale of milk	81 ±20	14 ±4	77 ±15	13 ±3	78 ±7	13 ±2
	Sale of manure	13 ±25	2 ±3	2 ±3	0 ±0	2 ±2	0 ±0
	Surplus	271 ±44	22 ±10	243 ±28	16 ±9	213 ±28	13 ±4

Soil surface balances

Dry matter intake rates by dairy cows were between 16.8 and 19.1 kg dry matter cow^{-1}d^{-1} (Table 2.1). Excretion of N and P by grazing dairy cows during the summer season ranged from 0.45 to 0.51 kg N cow^{-1}d^{-1} and from 0.036 to 0.040 kg P cow^{-1}d^{-1} (for N see Table 2.1). Table 2.4 shows the average SSBs of 65 fields in the Vlietpolder for 1999, 2000 and 2001. Generally, large inputs were mineral fertiliser application, slurry application and dung and urine deposited by grazing animals. At polder level, SSB surpluses decreased by 44 kg N ha^{-1}y^{-1} and 4 kg P ha^{-1}y^{-1} from 1999 to 2001. During this period only fertiliser application decreased significantly ($p<0.001$). The spatial distribution of N surpluses for 1999 is shown in Figure 2.1.

Fields that were resown were omitted, because of the dissimilar field management on these fields. The cumulative frequency distribution of SSB-N surpluses is shown in Figure 2.4. The SSB surpluses ranged from about -50 to 500 kg N $ha^{-1}y^{-1}$. For P comparable distributions were found with a range between -30 and 30 kg P $ha^{-1}y^{-1}$ (data not shown). Strikingly, for both N and P also negative surpluses were obtained. From 1999 to 2001 the cumulative frequency distribution hardly changed, indicating equal variability between fields over the years.

Table 2.4. Average soil surface balances for the Vlietpolder, The Netherlands, and standard deviations for N and P for 1999, 2000 and 2001 (n=65).

		1999		2000		2001	
		N	*P*	*N*	*P*	*N*	*P*
		kg ha^{-1}y^{-1}					
IN	Mineral fertiliser	216 ±67	9 ±9	182 ±36	7 ± 8	153 ±35	6 ±10
IN	Atmospheric deposition	31	-	31	-	31	-
IN	Slurry application	156 ±69	25 ±13	132 ±83	21 ±12	156 ±84	21 ±11
IN	Cattle droppings	127 ±94	12 ±8	134 ±101	12 ±9	100 ±63	9 ±6
IN	Slush application	31 ±75	1 ±3	35 ±89	2 ±3	33 ±77	1 ±3
OUT	Mown grass	209 ±79	22 ±8	176 ±84	19 ±8	206 ±94	20 ±8
OUT	Grazing	198 ±139	18 ±11	214 ±147	19 ±13	157 ±90	14 ±8
	Surplus	154 ±137	7 ± 19	124 ±104	4 ±13	110 ±125	3 ±13

Farm gate balance versus soil surface balance

The FGB and SSB have different system boundaries. However, rearranging of FGB and SSB balance entries yields the following relationship:

$$FGB = SSB_j - \text{slush application} - \text{atmospheric deposition} + \varepsilon \qquad \text{eq. 2.2}$$

where FGB is the N or P surplus at farm level (kg $ha^{-1}y^{-1}$), SSB_j is the surface area weighted average SSB surplus at farm level (kg $ha^{-1}y^{-1}$), and ε an error term (kg $ha^{-1}y^{-1}$). Figure 2.5 shows the relationships between FGB and the SSB_j per farm for N and P for 3 years. For most farms FGB surpluses exceeded average SSB_j surpluses for both N and P. These results show that ε exceeded the sum of slush application and, in the case of N, atmospheric deposition. Generally, ε represents biases plus scale differences, i.e. losses and storage of N and P from stables and manure deposits. The relationship between FGB and SSB_j surpluses was not straightforward. Some farms (for N: farms A, C and F; for P: farms A, B, D and E) showed a positive relationship between FGB and SSB_j, but other farms showed no distinct relationship between FGB and SSB_j. On the farms without a relationship between FGB and SSB_j the achieved decrease in FGB surplus was not (yet) translated into adjusted field management.

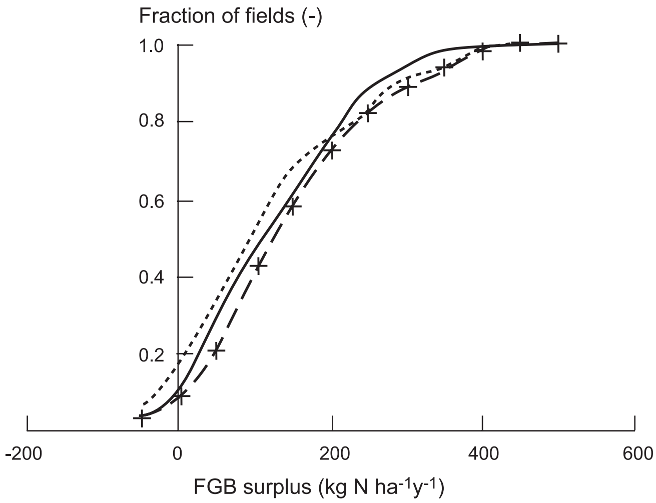


Figure 2.4. Cumulative frequency distribution of SSB-N surpluses for the years 1999 (dashed +), 2000 (solid) and 2001 (dotted).

Inter and intra farm variability

Inter and intra farm variability can be compared by means of the variation coefficient (VC). The calculation of the VC between FGBs was rather straight-forward, but for the SSB pooled variances by farm and surface weighted average SSB surpluses were used. In Figure 2.6 the VCs of the FGBs and the SSBs are shown for N for 3 years. Intra farm variability (i.e. the VC of the SSBs) always exceeded the inter farm variability. This observation was tested with an F-test for equality of variances. For N, intra farm variability significantly exceeded inter farm variability for all years (α=0.05). For P, intra farm variability exceeded inter farm variability in 2000 and 2001 (α=0.05; data not shown). From 1999 to 2001 both the average FGB and the difference between the maximum and minimum FGB decreased. However, the relative differences between farms, expressed as the VC between FBG surpluses (the inter farm variability) remained equal. Similarly, the VCs between fields per farm (the intra farm variability) did not change or slightly increased during the study period (Figure 2.6).

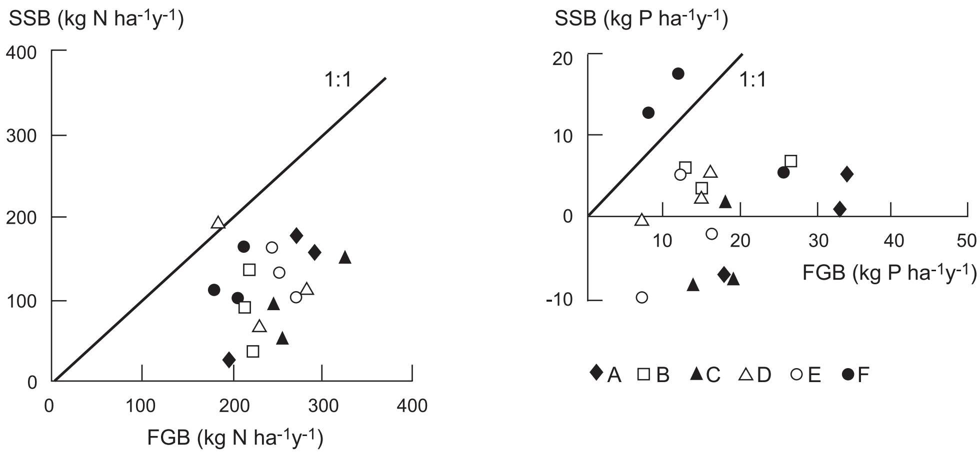


Figure 2.5. FGB surpluses and weighted by surface area average SSB surpluses per farm for N (left) and P (right). Dotted line shows 1:1 proportional line. Different farms are indicated by symbols.

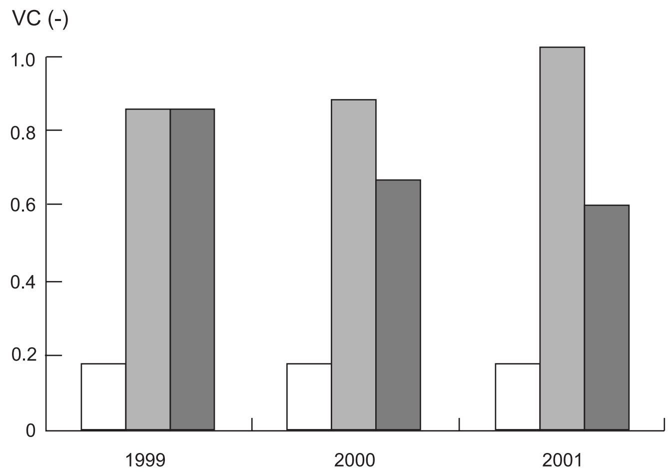


Figure 2.6. Variation coefficients (VCs) of FGB-N surpluses (white), annual SSB-N surpluses (grey) and of cumulative SSB-N surpluses (black) for the years 1999-2001.

Leaching of N to surface water

The flow proportional N analyses at the pumping station showed major outflows of N during winter (data not shown). In 1999 the net loading of the surface water in the polder equalled 43 kg N $ha^{-1}y^{-1}$ and 3 kg P $ha^{-1}y^{-1}$. To illustrate the possible effects of intra farm variability on estimates of N leaching based on farm data, we made two calculations with equation 2.1. To do so, equation 2.1 was fitted on the measured N loading of the polder with the surface weighted SSB surplus for the whole polder (141 kg N $ha^{-1}y^{-1}$), and on a background load of 15 kg N $ha^{-1}y^{-1}$ derived for grassland on peat soil by Hendriks (1993). Subsequently, in the first calculation variable x of equation 2.1 was set equal to the surface weighted average SSB per farm (farm-based estimate). In the second calculation, leaching was calculated per field and subsequently the weighted average leaching per farm was calculated (field-based estimate). We used weighted average SSB data instead of FGB data for the farm-based approach in order to account for biases between SSB and FGB data. In Figure 2.7 N leaching to surface water per farm is shown for farm-based and field-based estimates for the years 1999, 2000 and 2001. In all cases N leaching to surface waters increased when field-based estimates were used instead of farm-based estimates. Our results show that when farm-based estimates were used instead of field-based estimates N leaching to surface water per farm was underestimated with on average 27% in 1999, 24% in 2000 and 21% in 2001. The observed underestimations differed among the farms. Depending on the intra farm variability, the underestimations ranged from 5% to 38% (1999), 2% to 46% (2000) and 10% to 27% (2001).

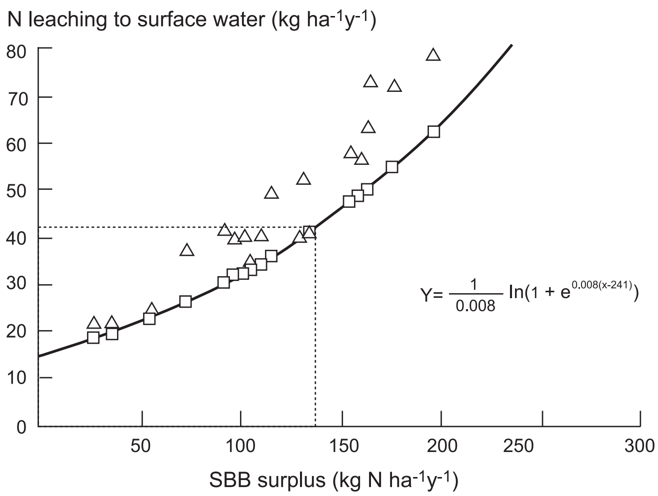


Figure 2.7. SSB-N surpluses and estimated N leaching to surface water when homogeneous distributions over fields within farms were assumed (squares) and when observed distributions were used (triangles) for the years 1999, 2000 and 2001. Solid line shows equation 2.1 (depicted in figure), dotted line shows measurement at polder level and surface area weighted average SSB-N surplus at polder level.

Discussion

Nutrient balances at farm level were used as environmental performance indicator at the time of study and as policy tool to monitor changes in N and P leaching to ground water and surface water and to monitor the effectiveness of policies and measures. This implies that nutrient balances should be discriminating at the farm level. However, we noticed a divergence between the scale on which nutrient balances are put in as policy instrument (farm) and the scale on which they ought to be effective (field or plot). We found that the intra farm variability exceeded the inter farm variability on all 6 farms in the study area. This intra farm variability may have effects on the evaluation of the effectiveness of FGB surpluses as policy instrument to reduce N leaching to surface water. Bacon et al. (1990) also found high intra farm variability for N, P and K for a study area in Pennsylvania, USA. They suggested using SSBs to identify problem fields ('hot spots') for remedial site-specific action. We focused on the consequences of this intra farm variability on estimates of N leaching based on farm data.

Estimated N leaching to surface water based on farm data in stead of field data, lead to an underestimation between 2% and 46% (Figure 2.7). This discrepancy between field-based estimates of N leaching and farm-based estimates of N leaching is basically a mathematical consideration. Notably, under non-linear circumstances, 'first averaging, then calculating' (farm-based approach) does not give the same results as 'first calculating, then averaging' (field-based approach). Consequently, the observed differences between farm-based and field-based estimates of N-leaching rely entirely on the relationship between surpluses at field level and leaching (eq. 2.1). Equation 2.1 was derived conceptually and was founded on the supposed non-linear relationship between N surplus and denitrification. We considered a relationship between SSB surplus and denitrification, based on NO_3 limitation. In our study we used relatively humid and organic matter rich soil, where denitrification is likely to be limited by NO_3 availability. In drier or more mineral soils, denitrification may be limited by organic C availability and/or aerobicity. However, the relationships between denitrification and organic C availability and between denitrification and aerobicity also tend to be non-linear (Hénault and Germon, 2000). Consequently, our results probably still hold for other soils, but the derived relationship between SSB surplus and N leaching may be modified.

Intra and inter farm variability did not change in time (Figure 2.6). However, when the intra farm variability was calculated of the cumulative SSB surpluses over 2 (1999 + 2000) and 3 (1999 + 2000 + 2001) years instead of over 1 year, intra farm variability decreased in time (Figure 2.6). This suggests that high surpluses in one year were counterbalanced by low surpluses in the next year, and vice versa. Apparently, farmers do not have a fixed management regime per field. Inter farm variability did not decrease when it was calculated from the cumulative FGB surpluses (data not shown). The observed bias made when estimates of N leaching are based on data at farm level may be partly compensated by this evening out of intra farm variability. It depends on the time dependency of the relationship between N surplus and N leaching to what extent this evening out compensates for the annual intra farm variability. Van den Eertwegh (2002) estimated that 19% to 85% of drainage water (with its constituents) is transported to the surface water within 1 year. However, for P the strong buffering capacity of soils may cause long-term effects (Yli-Halla et al., 2002). Hence, for N, the compensation due to evening out of intra farm variability in time is likely to be small.

Inter farm variability was mostly linked to the variability in purchase of fertilisers and concentrates, while intra farm variability was mostly linked to the variability in harvest, fertiliser application and manure application (e.g. Tables 2.3 and 2.4). Although the VC for grazing was high, the net effect of grazing on the surplus was small. As shown by the data, the nutrient input via dung and urine from grazing cattle was completely compensated by output via intake of grass (in part depending on supplemental feeding with concentrates). Slush application was rotated in a 5-10 year sequence and, consequently, was only applied to a few fields per farm, explaining the high standard deviation for slush application in Table 2.4. In practice, farmers do not fully take into account the nutritional value of slush for the determination of the proper amount of fertiliser and manure application. As slush application was inside FGB boundaries but outside SSB boundaries (Figure 2.2), one could argue whether slush application should be part of a test for equality of variance. However, when slush application was omitted from the F-test for equality of variances, intra farm variability was still significantly higher than inter farm variability ($\alpha=0.05$).

Estimated N leaching to surface water ranged between 20 and 80 kg N $ha^{-1}y^{-1}$ (Figure 2.7). These estimates agree reasonably with the mean estimated N leaching to surface waters in The Netherlands (RIVM, 2002). In the latter study estimated N leaching for comparable situations, i.e. grassland on shallow peat soil with an annual N surplus of 252 kg N $ha^{-1}y^{-1}$, ranged between 40 and 85 kg N $ha^{-1}y^{-1}$. Kirkham and Wilkins (1993) reported N leaching from peat soil of 67 kg N $ha^{-1}y^{-1}$ when 200 kg N $ha^{-1}y^{-1}$ was applied as ammonium nitrate. Although this fertiliser application can not be directly translated to a SSB surplus, the amount of N leaching reported by Kirkham and Wilkins is within the range of leaching found in this study. Moreover, dry matter intake rates and excretion rates agreed well with values reported in literature (Dalley et al., 1999; Smith and Frost, 2000; Van Keulen et al., 2000).

With equation 2.2 some estimates about the consistency of the data could be made. The error term ε was calculated for all farms and years. Generally, a positive ε indicates losses and/or storage of N and P in stables and manure deposits. For N, ε averaged 183 kg N $ha^{-1}y^{-1}$ with minimum and maximum values of respectively 107 and 279 kg N $ha^{-1}y^{-1}$. Those numbers largely reflect NH_3-volatilization from stables and manure deposits. During storage 0-20% of the N contained in the slurry may be lost by NH_3-volatilization (Bussink, 1996b). For P, ε varied between -8 kg P $ha^{-1}y^{-1}$ and 28 kg P $ha^{-1}y^{-1}$, showing that for most farms only small P losses occurred from stables and manure deposits, but some farms apparently had on farm storage of P. However, we can not exclude biases in our calculations. Biases may have various sources (e.g. Oenema and Heinen, 1999). First, they may originate from using different data sources. For example, FGB data were derived from accounts and data statistics with standardised nutrient contents, while SSB data were derived from data collected by farmers, supplemented with farm specific measurements and calculations. Second, the SSBs may suffer from personal biases, i.e. farmers may consistently differ in estimates of nutrient removal and nutrient supply, despite uniform instructions.

Rougoor et al. (1997) found strong relationships between FGB-N surplus and milk production and between FGB-N surplus and concentrate feeding. In our study there was no distinct relationship between concentrate feeding and FGB-N surplus (not shown) and only a slightly positive relationship between milk production and FGB-N surplus (Figure 2.3). However, the FGB surplus was significantly correlated with the fertiliser purchase for both N (r^2=0.88) and P (r^2=0.90) (for N, see Figure 2.3). Poor or no relationships between FGB surpluses and production levels suggest that nutrient inputs may be decreased without a decrease in production and nutrient outputs. This suggestion was made by Simon et al. (2000) and by Ondersteijn et al. (2003) who also found large variability between farms with similar farming systems. They concluded that high inter farm variability demonstrates that in many situations the environmental impact of a farm can be reduced without drastic changes in the productivity. Ondersteijn et al. (2002) investigated inter farm variability of 114 dairy farms and found an inter farm variability, expressed in VC, of about 0.23. This is slightly higher than the inter farm variability found in this study (about 0.17; Figure 2.6). The dairy farms in Ondersteijn's study were located all over the country, representing different regional conditions and soil types, which might explain the higher inter farm variability. Moreover, the majority of the farmers participating in the present study was member of a study group and exchanged practical information on how to cope with the decreasing levy free surpluses.

Conclusions

Estimates of N leaching to surface waters based on data derived at farm level may be biased due to the heterogeneous distribution of N surpluses over fields. The extent of the bias depended on i) the nature of the relationship between N surpluses and nutrient loss pathways, and ii) the distribution of N surpluses over fields. A strong non-linear relationship between N surpluses and nutrient loss pathways and large variability of N surpluses over fields, cause a high bias. When N surpluses at farm level decreased, the annual variability in N and P surpluses between fields remained. Consequently, accurate estimates of N leaching can not be made on basis of farm data without knowledge about the distribution of surpluses over fields and about other nutrient loss pathways.

Chapter 3: Denitrification rates in relation to groundwater level in a peat soil under grassland

Slightly modified from:
Van Beek C.L., Hummelink E.W.J., Velthof G.L. and Oenema O. 2004. Biology and Fertility of Soils 39: 329-336.

Abstract

In this study spatial and temporal relations between denitrification rates and groundwater levels were assessed for intensively managed grassland on peat soil where groundwater levels fluctuated between 0 and 1 m below soil surface. Denitrification rates were measured using the acetylene inhibition technique every 3 to 4 weeks for 2 years (2000-2002). Soil samples were taken every 10 cm until groundwater level was reached. Annual N losses through denitrification averaged 87 kg N ha^{-1} of which almost 70% originated from soil layers deeper than 20 cm below soil surface. Nitrogen losses through denitrification accounted for 16% of the N surplus at farm-level (including mineralization of peat), making it a key-process for the N-efficiency of the present dairy farm. Potential denitrification rates exceeded actual denitrification rates at all depths, indicating that organic C was not limiting actual denitrification rates in this soil. Groundwater level appeared to determine the distribution of denitrification rates with depth. Our results were explained by the ample availability of an energy source (degradable C) throughout the soil profile of the peat soil.

Introduction

Measured N losses through denitrification in peat soils appear to be of the same magnitude as N losses through denitrification in other soil types (De Klein and Logtestijn, 1994; Barton et al., 1999). This is remarkable since circumstances for denitrification seem more favourable in peat soils than in other soils, considering the often anaerobic and organic matter rich conditions. Most estimates of N losses through denitrification on different soil types, including peat soils, are based on measurements in the topsoil only (0-20 cm). The common justification for this restriction is that the main prerequisites for denitrification, i.e. the presence of NO_3^-, degradable C and anaerobic conditions only occur concurrently in the topsoil (Yeomans et al., 1992; Luo et al., 1998). Yet, in peat soils high contents of degradable C are also present in the subsoil (Jörgensen and Richter, 1992; Velthof and Oenema, 1995) and therefore, a considerable contribution of N losses through denitrification from the subsoil can be expected in peat soils when NO_3^- is present under anaerobic conditions.

In agricultural soils, NO_3^- may directly originate from fertilizer application and, after nitrification, from animal manure, atmospheric deposition, biological N_2 fixation and soil organic N. Nitrification and denitrification are oxidation-reduction processes and, in this sequence, are optimal under conditions going from high to low redox potential (Stumm and Morgan, 1996). In soils with shallow groundwater levels (0-100 cm below soil surface) groundwater fluctuations are a driving factor for changes in redox potential. In these soils, groundwater level fluctuations are expected to play a key role for creating conditions where nitrification and denitrification can occur at relative proximity. Hence, we expect that temporal changes in groundwater level are reflected in temporal changes in N losses through denitrification in soils with a shallow groundwater level.

In The Netherlands many peat soils are located in polders, i.e. at an elevation below mean sea level with a strictly regulated management of surface waters in ditches and canals. In summer, when evapotranspiration exceeds precipitation, groundwater levels drop below ditch water levels, and groundwater levels are deeper in the middle of the fields than near the ditches surrounding the fields. In winter, although lower ditch water levels are aimed for, the soil water storage capacity is frequently exceeded and then fields become waterlogged. Hence, both in summer and in winter there is a gradient in groundwater levels with increasing distance from the surface water. Velthof et al. (1996) showed that this spatial gradient in groundwater levels with increasing distance to the ditch water was reflected in the spatial pattern of nitrous oxide (N_2O) emissions from an intensively managed peat soil in summer, i.e. higher N_2O emissions were found near the ditch. Generally, N_2O emissions and denitrification rates are related (Velthof et al., 1996; Stevens et al., 1997) and therefore, under comparable conditions, we expect similar spatial patterns for actual denitrification rates in the topsoil.

So far only few estimates of annual denitrification losses on peat soils have been reported, whilst from the abovementioned studies it follows that mechanisms and magnitude of denitrification losses in peat soils may deviate considerably from those of other soil types. Therefore, the following three hypotheses were tested for an intensively managed grassland on peat soil, i) the subsoil (>20 cm) makes a considerable contribution to the total N losses through denitrification, ii) temporal changes in groundwater level are reflected in temporal changes of N losses through denitrification, and iii) spatial patterns of groundwater levels within the field are reflected in spatial patterns of denitrification rates in the topsoil (0-10 cm). In addition, N_2O production rates were calculated in order to provide an estimate of total $N_2(O)$ losses from an intensively managed grassland on peat soil.

Materials and methods

Site description

The study was carried out in the Vlietpolder near Hoogmade in the western part of The Netherlands (52°10’ N,4°36’ E). The Vlietpolder is 202 ha in size and about 2 m below mean sea level. The soil is largely composed of woody peat (Terric Histosol; FAO-Unesco, 1988) and the fields are separated by ditches, resulting in high ratios of land to surface water area (approximately 10:1). One field (approximately 40 x 400 m), surrounded by ditches on three sides with a target water level of 58 cm below the soil surface during winter and 48 cm below the soil surface during summer, was selected for intensive monitoring. Table 3.1 provides some general characteristics of the field. The grassland is intensively used for dairy farming. On average, every summer 50 cows graze the field on a 30-day rotation and about 400 kg N $ha^{-1}y^{-1}$ is harvested for silage via three to five cuts and removed by grazing.

Annual N input is about 540 kg N ha^{-1} via mineral fertilizer (200 kg N $ha^{-1}y^{-1}$ applied in three to four dressings), slurry application (150 kg N $ha^{-1}y^{-1}$ in two to four dressings), manure and urine of grazing cattle (130 kg N $ha^{-1}y^{-1}$), atmospheric deposition (31 kg N $ha^{-1}y^{-1}$) and application of slush (30 kg N $ha^{-1}y^{-1}$, Chapter 2).

Table 3.1. Selected soil properties of the experimental field. ND Not determined.

Soil layer (cm)	Clay (<2 μm)	Silt (2-50 μm)	Sand (>50 μm)	Loss on ignition	Dry bulk density	pH-H_2O
			$g\ g^{-1}$		$g\ cm^{-3}$	
0-10	0.15	0.08	0.57	0.20	0.92	5.36
10-20	0.12	0.09	0.62	0.17	0.93	4.95
20-40	0.13	0.12	0.62	0.13	0.81	4.86
40-75	0.17	0.08	0.30	0.45	0.75	4.60
75-200	ND	ND	ND	0.40	ND	ND

Denitrification measurements

We defined potential denitrification as the denitrification rate under anaerobic conditions with abundant NO_3^- at 20°C, following Bijay-Singh et al. (1989) and Aulakh et al. (1992). The potential denitrification is a measure for the organic C content available to denitrifying organisms (Jörgensen and Richter, 1992) and was determined once in May 2000. Samples were taken with a ring bore equipped with 100-cm^3 stainless steel ring samplers (5 cm diameter). Samples were taken in triplicate and were then bulked to give a composite sample. From the topsoil (0-10 cm) 16 composite samples were collected in a regular grid of 10x100 m. Fewer samples were taken from deeper soil layers because we expected more heterogeneity in the topsoil than in the subsoil. From the soil layers deeper than 10 cm below the soil surface, two composite samples were taken per 10 cm soil layer until the groundwater level was reached (70 cm below the soil surface). The composite samples were placed in PVC containers (894 ml) equipped with a screw cap with rubber sealing rings and rubber septa. Approximately 0.1 mg N g^{-1} dry soil as KNO_3 was applied to each container. To remove O_2 from the containers the containers were placed under vacuum for 1 min and flushed with N_2 for 5 min.

Measurements of actual denitrification started in September 2000 and lasted until August 2002. Soil samples were collected every 3 to 4 weeks with a ring bore yielding 24 sampling events in total. During the first year of measurements, our emphasis was on spatial relationships between groundwater level and denitrification rates in the topsoil. However, after we found that the subsoil was a major contributor to the N losses through denitrification the emphasis shifted more to the subsoil in the second year. In the first year the sampling procedure for actual denitrification measurements was the same as that for potential denitrification measurements. Note that the number of sampled layers fluctuated with the groundwater level resulting in a minimum of one sampled layer (occasionally in winter) to a maximum of nine sampled layers (in summer). In the second year the field was split into eight plots of 20 x 100 m and one ring sample was taken per soil layer of 10 cm from every plot until the groundwater level was reached. Each ring sample was incubated individually. Additionally, from February to August 2002 (eight sampling events) separate sampling rings were collected in triplicate from each 10-cm soil layer to determine N_2O production rates. On all occasions samples were taken until the groundwater level was reached, because we were not able to take undisturbed soil samples below the groundwater level. In both seasons samples for actual denitrification measurements and samples for N_2O production measurements were incubated at the prevailing air temperature.

Denitrification rates were determined by the C_2H_2 inhibition technique (Mosier and Klemedtsson, 1994; Aulakh et al., 1992). In brief, 1-10% C_2H_2 inhibits the reduction of N_2O to N_2 and consequently the N_2O production from soil in an atmosphere containing C_2H_2 is a measure for the N_2 + N_2O production through denitrification. We replaced 7% of the gas phase of the incubation containers with C_2H_2. However, C_2H_2 also impedes nitrification and therefore, when NO_3^- is rate limiting, the addition of C_2H_2 may lead to underestimation of denitrification rates (Walter et al., 1979). To minimize underestimations due to NO_3^- limitation, incubations were performed for a short as possible period. In a preliminary experiment it was found that the N_2O production rate started to become constant with time after approximately 8 h (lag phase) and remained constant for at least 24 h. Therefore, N_2O concentrations in the incubation pots were measured at least twice within 8-24 hours by a Bruël and Kjaer 1312 multi-gas monitor with a photo acoustic infrared gas analyzer (TGA). The TGA was equipped with filters for N_2O, CO_2, C_2H_2 and H_2O determination. The gas was passed through a metal cylinder filled with soda lime (KOH/NaOH) to remove CO_2, which might have interfered with the N_2O measurements. Samples for the measurement of N_2O production were treated similarly, except that no C_2H_2 was added. Denitrification rates and N_2O production rates were calculated by:

$$D = \frac{\rho_{N_2O} \cdot \Delta N_2O \cdot (V_g + \theta \cdot \alpha \cdot V_m)}{A \cdot \Delta t \cdot 2 \cdot 10^4} \qquad \textit{eq. 3.1}$$

where, D is the N_2O-N production of an incubation container (g $ha^{-1}d^{-1}$) per 10-cm soil layer, $\rho_{N_2O\text{-}N}$ is the density of N in N_2O, which was linearly dependent on temperature (range 1.15-1.25 g L^{-1}), ΔN_2O is the maximum N_2O production between two measurements corrected for the internal volume of the TGA (-), V_g is the gaseous volume of the incubation containers corrected for the sample volume (L), V_m is the soil volume in the incubation pots (L), α is the Bunsen adsorption coefficient used to correct for the amount of N_2O dissolved in water (-) (Moraghan and Buresh, 1977), θ the volumetric water content (-), A is the surface area of the samples (m^2), Δt is the time difference between two measurements (days) and $2 \cdot 10^4$ a conversion factor (ha m^{-2}) to convert from g N per m^2 per 5-cm soil layer to g N ha^{-1} per 10-cm soil layer. Note that in this paper denitrification rates imply actual denitrification rates, unless stated otherwise.

Soil and water analyses

The samples in the rings containing exactly 100 ml undisturbed soil, were weighted, dried at 105 °C for 24 hours and reweighed. Dry bulk densities were calculated per sample from the volumetric water content. In the 2001-2002 season, each time denitrification was measured separate samples were taken to determine NO_3^- and NH_4^+ contents (3 replicates per soil layer of 10 cm). Of each replicate 20 g of fresh soil was transferred to a shaking bottle and 50 ml 1 M KCl was added. The shaking bottles were shaken for 1 hour and filtered. Nitrate (NO_3^-) and NH_4^+ concentrations were determined by segmented flow analysis (Houba et al., 1995).

Groundwater tubes were installed at 0, 1, 5, 10, 20 and 22 m from the ditch, where 0 m equals the ditch water at one end of the transect and 20 m equals the middle of the field. The ditch water level at the other end of the transect (i.e. at 40 m) was equal to the ditch water level at 0 m, because the ditches surrounding the field were in open connection. Groundwater levels were measured biweekly. In addition, an automatic groundwater probe was placed at 7 m from the ditch, which recorded groundwater levels every 5 hours.

Calculations

For each sampling event, denitrification rates were integrated over depth, resulting in predicted total N losses through denitrification. Subsequently, annual N losses through denitrification were calculated by integrating N losses through denitrification over time. Because of the different sampling strategies for the 2000-2001 and the 2001-2002 seasons, separate calculation procedures were followed for each season. Briefly, for the 2000-2001 season average denitrification rates per soil layer were summed per sampling event and then linearly integrated over time. For the 2001-2002 season, annual N losses through denitrification per plot were calculated and then averaged. Standard errors were calculated from replicates and annual N losses through denitrification were calculated with 95% confidence. Consistency of spatial patterns of denitrification rates in the topsoil (0-10 cm) was tested by calculating Spearman-Rank correlation coefficients $|r_s|$ of successive measurements for 16 (2000-2001) or 8 (2001-2002) plots (Davis, 1986). Also, the depths at which highest denitrification rates were found per sampling event were compared with the prevailing groundwater level.

Results

Potential denitrification

Integrated over depth (0-70 cm) the potential denitrification totalled 41 ± 15 kg N $ha^{-1}d^{-1}$ (Figure 3.1). The upper 20 cm of the soil profile was responsible for 65% of the potential denitrification in the 0-70 cm layer. Lowest potential denitrification rates were measured at 30-40 cm below the soil surface, and increased again at greater depth.

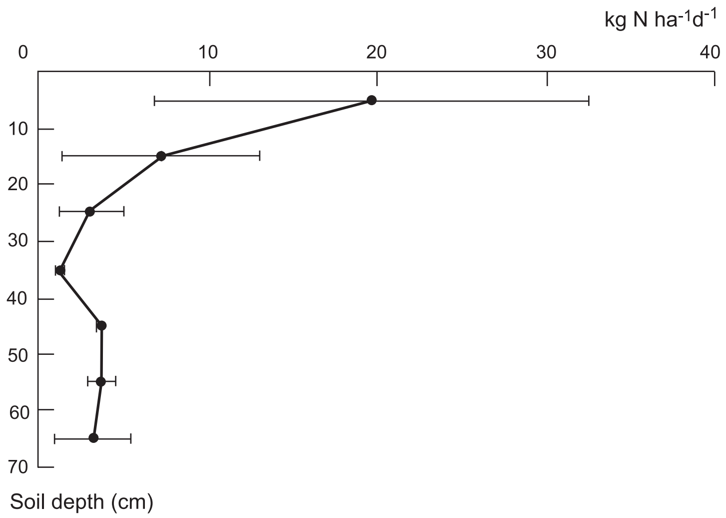


Figure 3.1. Potential denitrification rates with depth (May 2000). Error bars show SDs, n=16 for topsoil (0-10 cm) and n=2 for other soil layers.

Annual N losses through denitrification

N losses through denitrification yielded 100-120 kg N $ha^{-1}y^{-1}$ for the 2000-2001 season and 56-72 kg N $ha^{-1}y^{-1}$ for the 2001-2002 season. The differences between the years could not be related to agricultural N inputs to the field (i.e. the agricultural N surplus was 128 kg N $ha^{-1}y^{-1}$ in 2001-2002 and 124 kg N $ha^{-1}y^{-1}$ in 2000-2001). SDs between replicates of denitrification from the same soil layer were high, and in some cases the coefficient of variation exceeded 200%, but most often was around 70%. Measured N losses through N_2O production sometimes exceeded N losses through denitrification (Figure 3.2). The distribution of denitrification rates with depth varied with time (Figure 3.3).

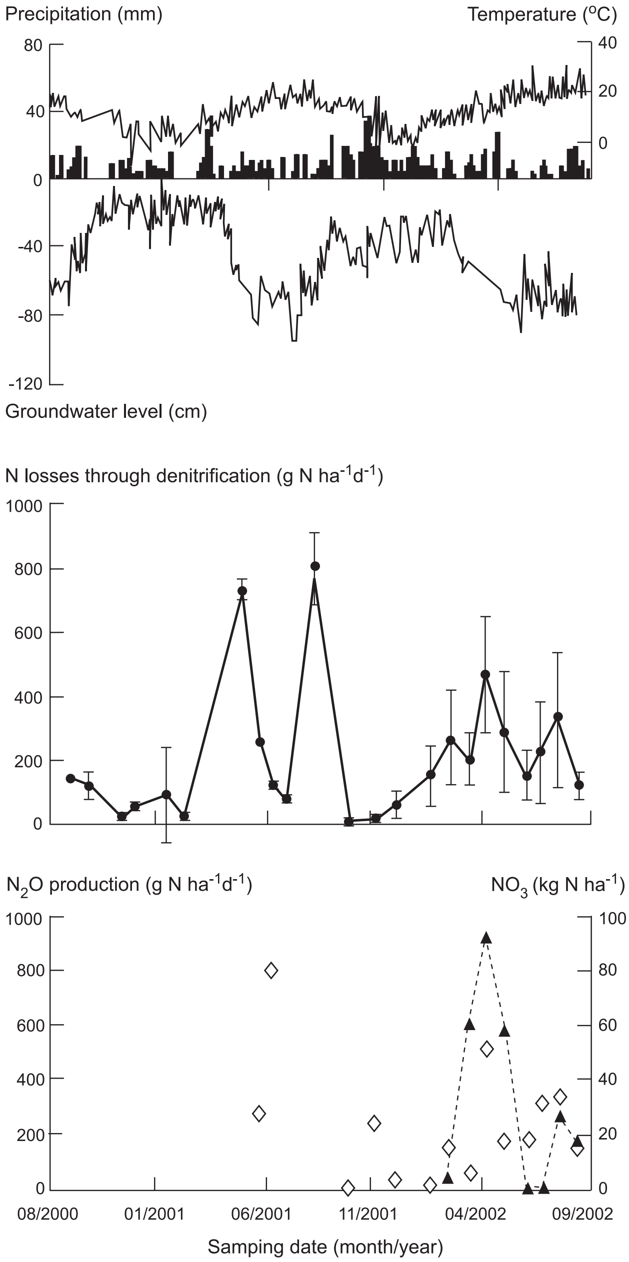


Figure 3.2. Above: groundwater level, air temperature and precipitation (bars), Centre: mean N losses through denitrification, Below: N_2O production rates (triangles) and NO_3^--N contents of the soil (diamonds). Error bars show SDs. Note that the period of denitrification measurements started before the measurement period of NO_3^- contents and N_2O production rates.

NO_3^- and NH_4^+ contents and groundwater level

NO_3^- contents of the soil fluctuated between 1 and 30 mg N kg^{-1}, with higher NO_3^- contents during the growing season. Sometimes decreasing NO_3^- contents with depth were found, but there was no consistent pattern (Figure 3.3). NH_4^+ contents of the soil occasionally showed a strong increase with depth (e.g. May, 2001), but usually were rather constant with depth and were higher in summer than in winter (Figure 3.3).

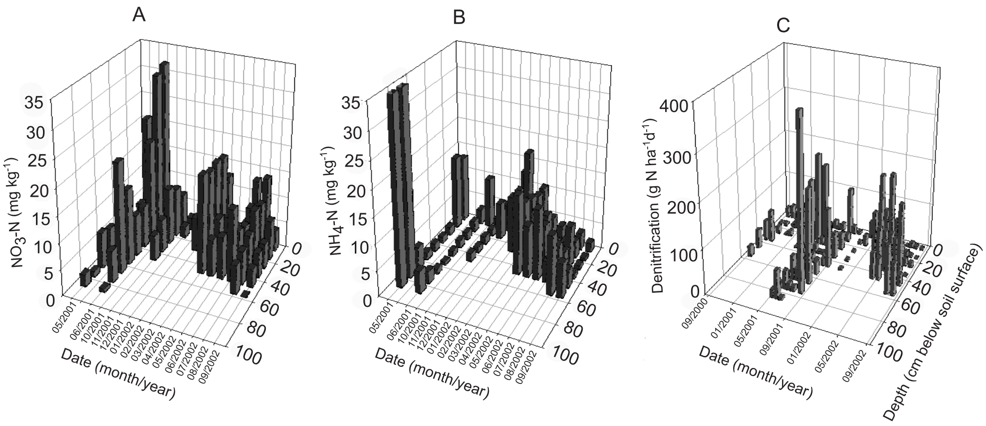


Figure 3.3. NO_3^--N (A), NH_4^+-N (B) contents of soil samples at different depths during the 2001-2002 sampling season and (C) average denitrification rates per 10-cm soil layer during the 2000-2002 sampling season. The maximum sampling depths approximately equaled prevailing groundwater levels.

Groundwater levels fluctuated between 0-100 cm below the soil surface and followed a seasonal pattern, with shallower groundwater levels in winter than in summer. In 2002 groundwater levels were generally deeper than in 2001, because more water was pumped out of the polder in 2002, while summed precipitation was higher in 2002 than in 2001 (Figure 3.2). In the summer half-year (i.e. April-September) often concave groundwater levels were found, with up to 50 cm difference between the groundwater level in the middle of the field and the surface water level in the ditches, while in the winter half-year (i.e. October-March) groundwater levels were often close to the soil surface (Figure 3.4).

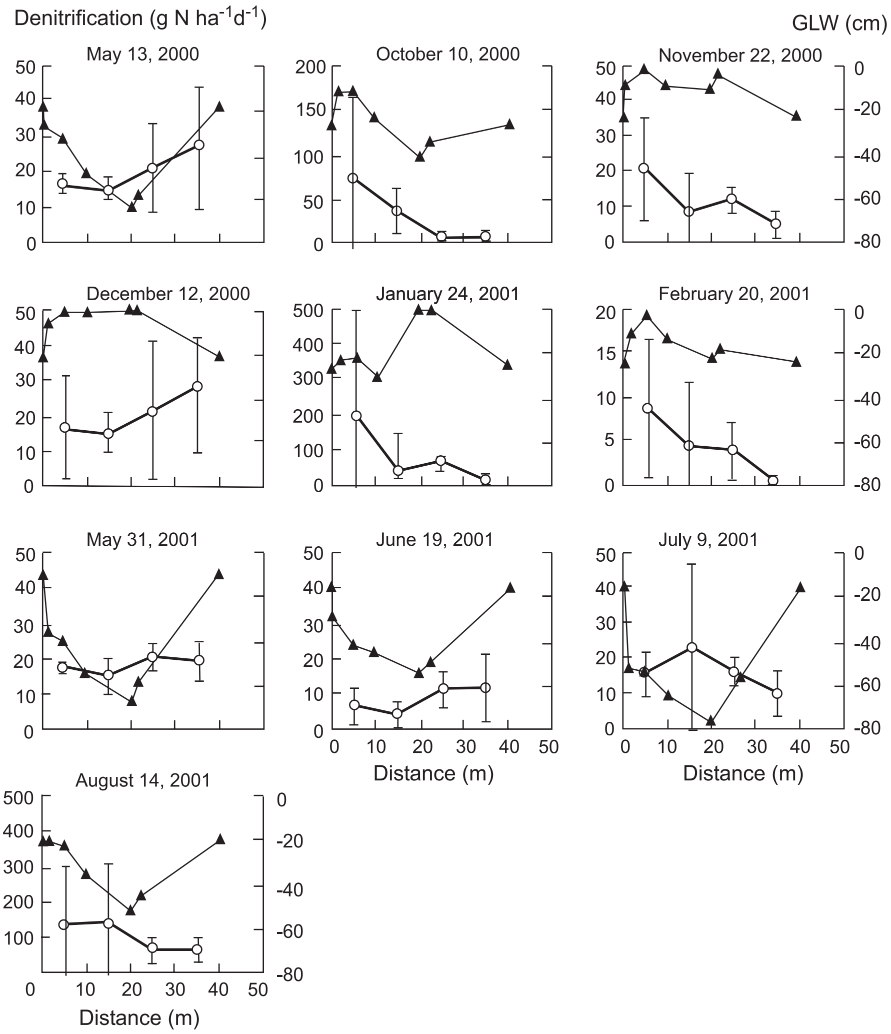


Figure 3.4. Groundwater levels (▲) and average denitrification rates (○) of the topsoil (0-10 cm). Plots represent a cross-section through the field. Ditches are located at either side of the field (0 and 40 m). Error bars show SDs of the denitrification rates (n=4). Note different scale on y-axes.

There was no clear relation between temporal changes in groundwater level and N losses through denitrification. Even when differences in temperature were taken into account, a clear relation was not found (Figure 3.5). NO_3^- contents of the soil profile were significantly related to N losses through denitrification, when one sampling event (indicated by an arrow in Figure 3.5) was omitted (r^2=0.41, p<0.001).

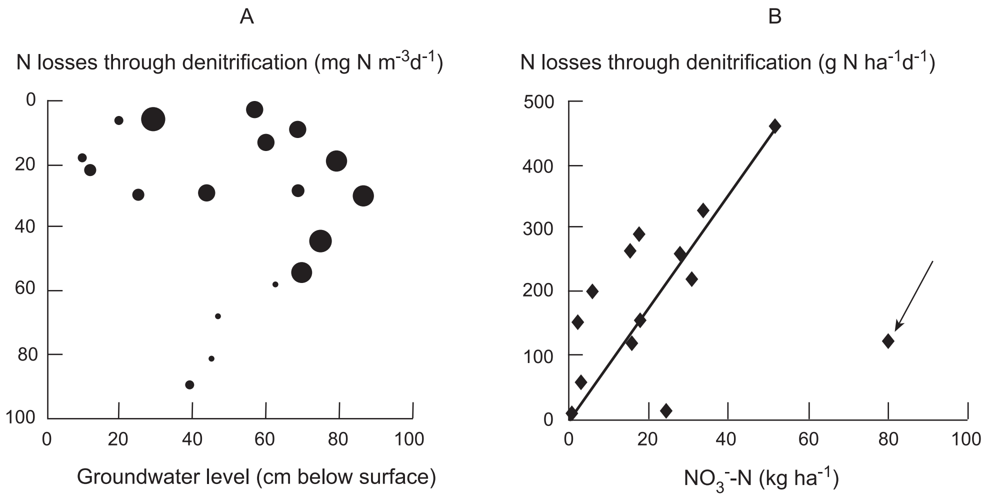


Figure 3.5. A: Total N losses through denitrification and groundwater levels at different temperatures. Symbols with increasing size represent a temperature increase from 5.5°C to 24°C. B: Total NO_3-N contents of the unsaturated soil profile and N losses through denitrification. Solid line shows linear regression (r^2=0.41), excluding outlier indicated by → (see text).

Spatial patterns of denitrification in the topsoil

For most situations the Spearman-Rank correlation coefficient $|r_s|$ between successive measurements in the topsoil was <0.4, indicating that the temporal stability of spatial patterns was poor (data not shown). Furthermore, the hypothesis that spatial variations in groundwater levels are reflected in spatial patterns of denitrification rates in the topsoil was tested. In the 2000-2001 season average denitrification rates were calculated for each sampling event per grid distance perpendicular to the ditch (n=4) and compared with measurements of groundwater level at seven distances from the ditch. The spatial dependency of denitrification rates in the topsoil with distance from the ditch water showed contrasting results. For some measurements (e.g. 10 October 2000 and 20 February 2001) denitrification and groundwater level followed a comparable spatial pattern, but for other events such relations were absent (Figure 3.4).

Groundwater level and depth of maximum denitrification

Both increasing and decreasing denitrification rates with depth were found (Figure 3.3). The depth where highest denitrification rates were found per sampling event appeared to be deeper when groundwater levels were deeper. In Figure 3.6 average denitrification rates per depth were calculated per sampling event. Then, per sampling event denitrification rates were ranked and the depth with the highest rank (i.e. highest denitrification rate) was plotted against the groundwater level (from the automatic groundwater probe) during that event. Note that measurements were made as deep as groundwater level and that the depths at which highest denitrification rates occurred were delimited by the prevailing groundwater level at the time of sampling (this limitation is indicated by the 1:1 line in Figure 3.6). The positive relation between groundwater level and depth of highest denitrification rate suggests that highest denitrification rates were predominantly found in soil layers 0–40 cm above groundwater level (Figure 3.6).

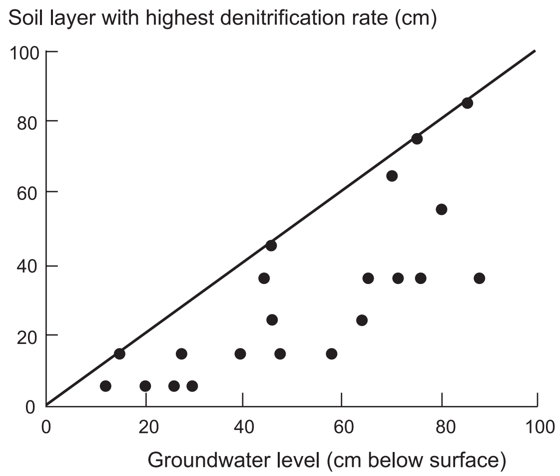


Figure 3.6. Soil layer with the highest denitrification rate per sampling event and groundwater level. No samples were taken below groundwater level, and therefore the number of sampled soil layers increase with depth of groundwater level (indicated by line).

Discussion

Mean annual N losses through denitrification equaled 87 kg N ha^{-1}. In the 2000-2001 season, the N losses through denitrification of 2 sampling events accounted for 78% of the total annual N loss through denitrification (Figure 3.2). There were no extraordinary circumstances during these events; probably a combination of favorable meteorological, agricultural and hydrological conditions occurred. In comparison to other soil types, the measured N losses through denitrification found in this study are high (Barton et al., 1999), and are considerable in terms of N inputs and N surpluses at farm level. N surpluses at farm level, consisting of inputs via fertilizers and feed and outputs via sale of milk, manure and cattle, averaged 280 kg N $ha^{-1}y^{-1}$ of which 220 kg N $ha^{-1}y^{-1}$ was applied as mineral fertilizer (Chapter 2). Moreover, annually 265 kg N ha^{-1} was released by mineralization of peat (Chapter 4, average of 2000-2002), thus N losses through denitrification accounted for 16% of the total N surplus at farm-level and for 40% of the mineral fertilizer application.

N_2O production rates, which included N_2O produced by nitrification and N_2O produced by denitrification, sometimes exceeded denitrification rates (Figure 3.2) and yielded in total 126 kg N $ha^{-1}y^{-1}$. N_2O produced by denitrification and N_2O

produced by nitrification can not be separated easily (Wrage, 2003; Aulakh et al., 1992), but the high N_2O production in comparison with denitrification indicates that N_2O produced during nitrification was an important source of N_2O emission. The N emission measured with and without C_2H_2 partly overlapped (viz. the N_2O produced during denitrification) and consequently we could not calculate total N_2+N_2O emission by nitrification and denitrification. Instead, minimum and maximum ranges of total N_2 + N_2O emission were calculated by assuming all N_2O production originated from denitrification and by assuming all N_2O production originated from nitrification, respectively. This estimate yielded 126 - 213 kg N $ha^{-1}y^{-1}$.

In literature, only few estimates of annual N losses through denitrification in managed grasslands on peat soils have been reported. De Klein and Logtestijn (1994) and Berendse et al. (1994) reported N losses through denitrification for grassland on peat soils of 4-16 kg N $ha^{-1}y^{-1}$ and 17 kg N $ha^{-1}y^{-1}$, respectively and in both studies measurements were limited to the topsoil (<20 cm below the surface). In the present study annually about 12 kg N ha^{-1} originated from the top 10 cm, which is consistent with the results of De Klein and Logtestijn (1994) and Berendse et al. (1994). However, De Klein and Van Logtestijn (1994) and Berendse et al. (1994) may have considerably underestimated total denitrification rates, because we observed here that 69% of the annual N loss through denitrification originated from soil layers deeper than 20 cm (e.g. Figure 3.3). To our knowledge, the only other denitrification study on managed peat soil at greater depth (0-40 cm) was performed by Koops et al. (1996), who reported N losses through denitrification from intensively managed grasslands of 70 kg N $ha^{-1}y^{-1}$, which agrees reasonably well with our results (in the present study annually 58 kg N $ha^{-1}y^{-1}$ originated from the upper 40 cm). Thus, considerable N losses through denitrification may originate from the subsoil (>20 cm) of grassland on peat soil.

In the present study, samples were taken up to groundwater level, because we were not able to take undisturbed soil samples from saturated peat. However, there are indications that denitrification rates may be significant also below the groundwater level (Well et al., 2001). We assumed that denitrification rates below groundwater level were small at our site, because of the decreasing NO_3^- contents with depth (Figure 3.3). To verify this assumption, NO_3^- removal from the soil was calculated by summing reductions in NO_3^- contents between 2 successive measurements. Annual NO_3^- removal from the unsaturated zone yielded 135 kg NO_3^--N ha^{-1}, which included crop uptake, leaching and denitrification. Because N_2+N_2O losses could completely account for the NO_3^--N removal from the unsaturated zone, we believe that we did not miss large peaks in denitrification.

Nitrogen losses through denitrification appeared to be largely controlled by NO_3^--N contents of the soil profile (Figure 3.5). During one event (June 19, 2001) NO_3^--N contents were exceptionally high in the topsoil (Figure 3.3) as a result of fertilization, and did not coincide with high N losses through denitrification (Figure 3.5, indicated by an arrow). At the same time, the groundwater level was relatively deep (61 cm below surface, e.g. Figure 3.2). Apparently, the depth where favorable conditions occurred for denitrification (i.e. close to the groundwater level) did not correspond with the depth with the highest NO_3^- contents. Hence, relatively large (vertical) distances between the groundwater level and NO_3^- caused deviating results of denitrification in relation to NO_3^- contents of the soil. Vice versa, relatively high N losses through denitrification are expected when relatively high NO_3^- contents (e.g. after fertilization) coincide with shallow groundwater levels.

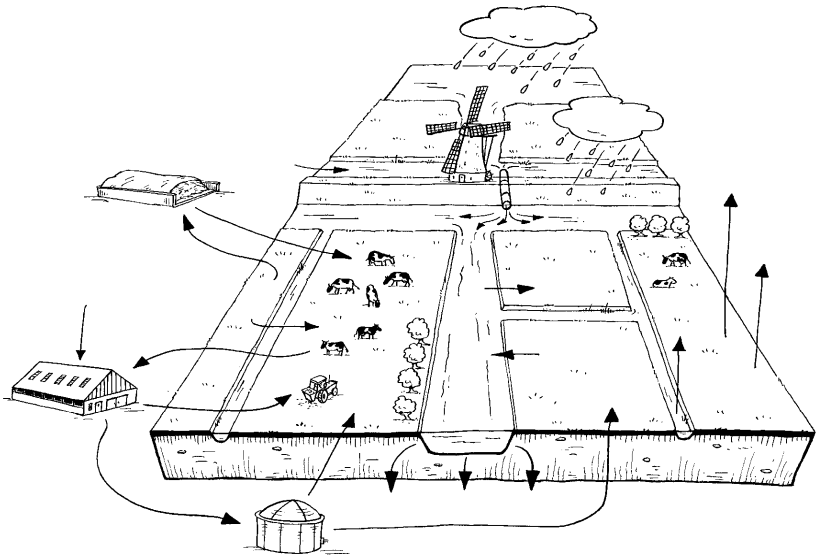
In low-land areas, groundwater levels tend to determine the magnitude of N losses via denitrification. In these areas, rising groundwater levels often coincide with increasing denitrification rates (Steenvoorden, 1983; Kliewer and Gilliam, 1995; Abdirashid et al., 2002). However, Koops et al. (1996) found no significant differences in N losses through denitrification for two intensively managed grasslands on peat soil with different mean groundwater levels. In the present study no clear relation between temporal changes in groundwater level and N losses through denitrification was found (Figure 3.5). Highest denitrification rates were predominantly found 0-40 cm above the groundwater level (Figure 3.6). Just above the groundwater level, aerobic and anaerobic niches may occur in relative proximity (Grundmann et al., 1995; Abbasi and Adams, 2000) resulting in optimal conditions for simultaneous nitrification and denitrification. Consequently, in shallow soils (like the present peat soil) temporal changes in groundwater level are expressed in changes in the depth where denitrification occurs, and to a lesser extend in changes in the magnitude of N losses through denitrification, providing that degradable C is not limiting denitrification. In the present study, potential denitrification rates (as a measure of degradable C availability) were always more than twice the actual denitrification rate at all depths, indicating that actual denitrification rates were not limited by C throughout the soil profile (Figures 3.1 and 3.3). Hence, as the depth at which the denitrification rates were highest approximately followed the groundwater level, the magnitude of N losses through denitrification was largely controlled by the NO_3^- contents of the soil (Figure 3.5).

During some events, spatial patterns of denitrification rates in the topsoil were related to spatial patterns of groundwater level, but these relationships were not consistent in time (Figure 3.4). In literature, contrasting effects of distance to surface water on denitrification rates in the topsoil are reported. Schnabel et al. (1996) and Clément et al. (2002) reported decreasing denitrification rates with increasing distance from surface water, while Pinay et al. (1993), Schipper et al. (1993) and Lowrance (1992) found increasing denitrification rates with increasing distance from surface water. Clément et al. (2002) showed that spatial patterns of denitrification rates with distance to surface water were often weak (i.e. not significant) and were not consistent for different vegetation covers and time periods. The absence of a consistent relation between denitrification rates in the topsoil and distance to surface water was ascribed to the high intersite variability. The present study and the study of Clément et al. (2002) indicate that spatial patterns of denitrification are not consistent in time. Therefore, conclusions about spatial relations between denitrification rates and groundwater level should be temporally validated, before regarded as general patterns.

Conclusions

Annual N losses through denitrification from an intensively managed grassland on peat soil yielded 87 kg N ha^{-1}, which equaled about 40% of the fertilizer N input and about 16% of the total N surplus at farm level, including mineralization of peat. Almost 70% of the total N loss through denitrification originated from soil layers deeper than 20 cm below the soil surface, showing that the subsoil should be taken into account when measuring N losses through denitrification from grassland on peat soils. Because degradable organic C was not limiting actual denitrification at any depth in our soil, favorable conditions for denitrification could develop throughout the soil profile. As a consequence, NO_3^- contents of the soil largely governed the magnitude of N losses through denitrification, while groundwater levels acted as a mechanism determining the depths at which denitrification occurred.

Chapter 4: The contribution of dairy farming on peat soil on N and P loading of surface water



Slightly modified from:
Van Beek C.L., Van den Eertwegh G.A.P.H., Van Schaik F.H., Velthof G.L. and Oenema O. 2004. Nutrient Cycling in Agroecosystems 70: 85-95.

Abstract

In agriculturally used peat land areas, surface water quality standards for nitrogen (N) and phosphorus (P) are frequently exceeded, but it is unclear to what extent agriculture is responsible for nutrient loading of the surface water. We quantified the contribution of different sources on the N and P loading of a ditch draining a grassland on a peat soil (Terric Histosol) used for dairy farming in the Netherlands. Measurements were performed on N and P discharge at the end of the ditch, supply of N and P via inlet water, mineralization of soil organic matter, slush application, composition of the soil solution, and on N losses through denitrification in the ditch for 2 years (September 2000 - September 2002). Discharge rates at the end of the ditch were 32 kg N $ha^{-1}y^{-1}$ and 4.7 kg P $ha^{-1}y^{-1}$. For N, 43-50% of the discharge was accounted for by applications of fertilizers, manure and cattle droppings, 17 to 31% by mineralization of soil organic matter, 8 to 27% by nutrient rich deeper peat layers, 8 to 9% by atmospheric deposition and 3 to 4% by inlet water. For P, these numbers were 10 to 48% for applications of fertilizers, manure and cattle droppings, 2 to 14% mineralization of soil organic matter, 33 to 82% nutrient rich peat layers and 5 to 6% inlet water. The results of this paper demonstrate that nutrient loading of surface water in peat areas involves several sources of nutrients, and therefore, reducing one source to reduce nutrient inputs to surface water is likely to result in modest effectiveness.

Introduction

In agriculturally used peat land areas (PLA), surface water quality standards are frequently exceeded, resulting in a general increase of water impairment in these areas (Klapwijk, 1988; Best et al., 1993). In addition to agriculture, other sources of nutrients, e.g. seepage and mineralization of peat, may also contribute nutrients to surface waters in PLA. The contributions of different sources of nutrients to the total nutrient loading of the surface water in PLA are often unknown, which leads to variable results of similar measures in different PLA to improve the surface water quality.

In the western part of The Netherlands, large parts of the grasslands on peat soils are located in polders, which are commonly used for dairy farming (grazing and foraging). Typically, the fields are long and narrow (e.g. 55 x 1250 m^2) and are separated by ditches to improve drainage. Most peat land polders have been cultivated since around 1000 AD and, due to artificial drainage, flow pathways to the surface water are shallow (0-1 m below soil surface, Schothorst, 1982). When precipitation exceeds evapotranspiration (most often during winter), water is pumped out of the polder. When evapotranspiration exceeds precipitation (most often during summer), water is supplied to the polder from the surrounding lakes and channels. Van Huet (1991) reported that this inlet introduced about 0.5 kg P $ha^{-1}y^{-1}$ into a peat land polder in the northern part of The Netherlands.

Net soil mineralization rates in PLA generally range between 100 and 400 kg N $ha^{-1}y^{-1}$, depending on drainage conditions and peat quality. Deeper groundwater levels and nutrient-rich peat result in higher mineralization rates (Schothorst, 1982; Berendse et al., 1994). Nitrogen and P inputs via applications of fertilizers, manure and cattle droppings in intensively managed peat lands equal about 450 kg N $ha^{-1}y^{-1}$ and 40 kg P $ha^{-1}y^{-1}$ (Chapter 2). Both N and P originating from applications of fertilizers, manure and cattle droppings, and N and P originating from mineralization of soil organic matter may be prone to leaching, as are N and P originating from atmospheric deposition and seepage. Unraveling the different sources and pathways that contribute to N and P loading of surface waters is essential for proper evaluation of regulations that aim at reducing their inputs.

In the Western part of the Netherlands many PLA are eutrophic and, due to long term shallow drainage, nutrients are leached from the topsoil. As a consequence, rainwater 'lenses' can be observed with lower solute concentrations in a bell shape between the ditches than below the lens and below the ditches (Wassen and Joosten, 1996). To compensate for a gradual surface decline of 0.5-1.0 cm y^{-1} in most PLA, water boards stepwise decrease the surface water level to secure sufficient drainage. Hence, occasionally drainage pathways intersect deeper, nutrient rich, soil layers and carry-on nutrients from these nutrient rich soil layers to the surface water. This process is referred to as supply from nutrient rich peat layers. Apart from this, seepage may contribute to the N and P loading of the ditch as most PLA are below mean sea level. With seepage, nutrients from deeper soil layers are transported to the ditch. In contrast to seepage, which is triggered by a vertical water flux, supply from nutrient rich peat layers is triggered by dominant horizontal water fluxes. Also, supply from nutrient rich peat layers can be regarded as a local phenomenon, while seepage is mainly the result of the regional hydrology. To estimate the supply from nutrient rich peat layers, concentrations profiles of the soil solution were measured.

We distinguished six possible sources of N and P loading of surface water in agriculturally used PLA, (1) applications of mineral fertilizer, manure, and dung and urine from grazing cattle, (2) mineralization of soil organic matter, (3) inlet water, (4) atmospheric deposition, (5) seepage and (6) supply from nutrient rich peat layers (Figure 4.1). To quantify these sources, experimental data were collected on discharge at the end of the ditch, inlet water, denitrification in surface water and ditch sediments, slush application (i.e. removal of ditch sediments), mineralization of soil organic matter, and fertilizer applications.

The aim of the present paper was to experimentally quantify the contribution of each of the above mentioned sources to the N and P loading of a ditch draining a grassland on peat soil in a polder. Therefore, N and P balances of the ditch were set-up, including reuse of N and P via slush application and N losses through denitrification (Figure 4.1).

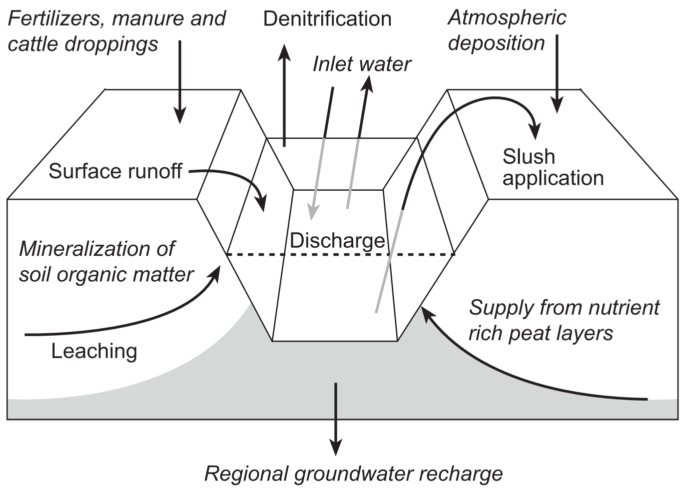


Figure 4.1. Schematic presentation of the ditch. Arrows show transport routes of N and P. Italic text refers to N and P sources that contribute to the N and P loading of the ditch. The grey area shows nutrient rich peat layers kept from year-long drainage (i.e. below the rain water lenses, see text).

Materials and methods

Site description

The site was located in the 'Vlietpolder' in the western part of The Netherlands (52°10' N; 4°36' E). The majority of the soils in the Vlietpolder consist of woody peat (Terric Histosol; FAO-Unesco, 1988), with a peat layer of about 3 m overlying marine clay deposits, which act as a physical barrier, preventing extensive infiltration and/or seepage (Meinardi, 2005). The area of the Vlietpolder is 202 ha and the elevation is about 2 m below mean sea level. The Vlietpolder is separated from a neighboring polder at an elevation of about 5 m below mean sea level by a channel at the east-side. Almost all land is used for intensive dairy farming (>90% of the surface area). In the period 1999 to 2001, an annual rate of approximately 452 kg N ha^{-1} and 41 kg P ha^{-1} was applied via mineral fertilizer, cattle slurry application, and cattle droppings (Chapter 2). The inlet station is located in the south and the pumping station in the north of the Vlietpolder (Figure 4.2). Target levels of the surface water are 58 cm below mean soil surface during winter and 48 cm below mean soil surface during summer. One dead-end ditch of approximately 800 x 3 x 1 m^3 (length x width x depth) was selected. Two fields drained on this ditch, yielding a drainage area of the ditch of 3.2 ha (Figure 4.2).

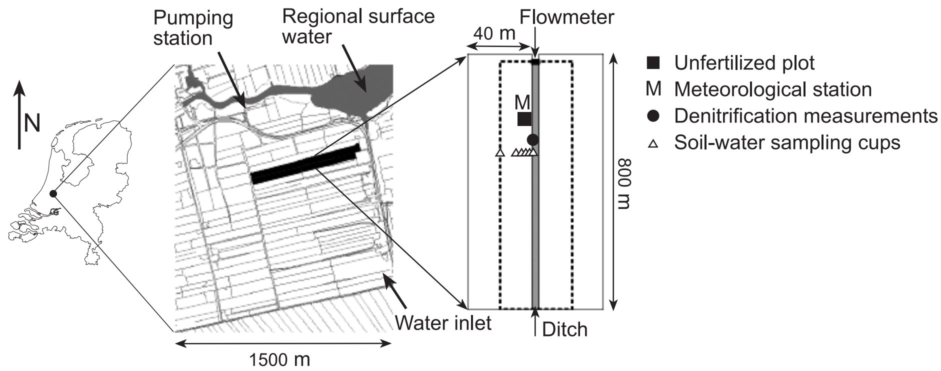


Figure 4.2. The location (left) and the design of the Vlietpolder (middle), the Netherlands. The catchment area of the ditch is shown by dashed lines (right). Water samples were collected at the flowmeter, the inlet station and at the pumping station. Not on scale.

The following sections describe several experiments. All samples were analyzed using identical procedures, which are therefore provided in the last section.

Hydrology and composition of the soil solution

Precipitation, irradiation and temperature were continuously measured at the meteorological station and were used to calculate the water balance of the polder, which was reported by Van den Eertwegh et al. (2003) and by Meinardi (2005), and is presented in Table 4.1. Recharge to the regional groundwater equaled about 25 mm y^{-1}, which was caused by the neighboring polder at lower elevation than the Vlietpolder (Meinardi, 2005). Consequently, net seepage did not occur, and the majority of the precipitation surplus (i.e. precipitation minus evapotranspiration) drained to the ditches.

Table 4.1. Average water balance of the Vlietpolder for the years 2000 and 2001 (mm y^{-1}). Data taken from Van den Eertwegh et al. (2003) and Meinardi (2005).

Precipitation	1114
Inlet water	98
Evapotranspiration	-579
Discharge at pumping station	-572
Regional groundwater recharge	-25
Sum (changes in storage)	36

The Vlietpolder was a net source of Cl^-, which could only be explained when also Cl^- discharge from Cl^- rich soil layers below the predominantly drained upper 1 m of soil was taken into account (unpublished data Rijnland Water Board). Similar processes are likely to occur for N and P, which are, however, more difficult to quantify. Meinardi (2005) estimated, on basis of on-site measurements of geophysical and hydrochemical profiles of the soil using Multi Layer Sampling, that about 90% of the precipitation surplus drained towards the ditch through the upper 1 m of the soil in the Vlietpolder. As 1 m below soil surface approximately equals the depth of the ditch, about 10% of the precipitation surplus drained through soil layers lower than the ditch floor. This amount was assumed to potentially interact with the nutrient rich subsoil. A simple estimate of potential minimum and maximum N and P loads from these nutrient rich peat layers to the ditch was derived by multiplying 10% of the annual water discharge with observed mean highest and mean lowest N and P concentrations in the soil solution.

The soil solution was sampled using ceramic suction cups, which were placed in a transect perpendicular to the ditch at 15, 25, 35, 50, 70 and 120 cm below surface and at 1, 2, 4, 7, 10, 20 m from the ditch, where 20 m equals the middle of the field (Figure 4.2). Also, 2 cups were inserted below the ditch at 78 and 106 cm below soil surface. Every 2 weeks from October 2001 till March 2003 vacuum bottles (approximately -90 kPa) were connected to the suction cups to collect water samples. With ceramic suction cups, the mobile fraction of the soil solution is sampled (Corwin, 2002). Predictions between ceramic suction cups were made by inverse squared distance interpolations (Davis, 1986).

N and P discharge and recharge of the ditch

In 2001, an electromagnetic flowmeter (MagMaster, ABB Instrumentation, Switzerland) was placed at the end of the ditch (Figure 4.2). Flow rates were measured continuously and allowed for assessment of both discharge and inlet of water and solutes. Every 0.1 mm (summer) or 0.2 mm (winter) of through-flow water samples were taken and stored dark and cold (4°C) until analysis (within 1 week).

Mineralization of soil organic matter and atmospheric deposition

Net N mineralization of soil organic matter was estimated by determining the N uptake of grass in zero-N plots. From 1999 onwards, 4 plots (10 x 20 m^2) in the Vlietpolder were fenced. Phosphorus and K fertilizers were applied at normal rates. Five times during the growing season all grass was harvested from the plots and 3 composite soil samples (approximately 500 g consisting of 15 sub-samples) were taken from each soil layer (0-20 cm, 20-40 cm and 40-60 cm). The soil samples were analyzed for mineral N. As grass is very effective in taking up mineral N from the upper 40-60 cm of the soil (Prins, 1983), we used the N uptake in the grass corrected for denitrification and atmospheric deposition as an estimator of the total amount of N mineralized in the rooting zone. Uptake of N in roots and stubbles was disregarded, as we assumed that this amount equaled the mineralization of N from old roots and stubbles. Denitrification losses from the soil were calculated from soil NO_3-N contents, using a site-specific regression equation based on field measurements (Chapter 3).

$$\textit{Denitrification} = 8.9\ NO_3\text{-}N \textit{ content of soil, } r^2=0.41 \qquad \textit{eq.4.1}$$

where denitrification is in g N $ha^{-1}d^{-1}$ and NO_3-N content in kg ha^{-1}. Atmospheric deposition was taken from a nearby (approximately 20 km) weather station and yielded on average 31 kg N $ha^{-1}y^{-1}$ (Boschloo and Stolk, 1999). N mineralization of soil in the rooting zone was calculated by:

$$\begin{aligned} &\textit{Net N mineralization} = \textit{N uptake} + \textit{denitrification from soil} - \\ &- \textit{atmospheric deposition} - \textit{changes in storage of soil mineral N} \end{aligned} \qquad \textit{eq.4.2}$$

where all units are in kg N $ha^{-1}y^{-1}$. In October 2001, 6 soil samples were taken at 50 cm below soil surface and total N:total P ratios of the peat soil were determined in order to estimate P mineralization from N mineralization.

Denitrification in sediment and ditch water

To set up the N balance of the ditch, also output of N via denitrification was assessed. To measure denitrification rates from sediment and ditch water, the acetylene inhibition methodology of Chan and Knowles (1979) was modified. PVC tubes (1 m length, 0.20 m internal diameter) were carefully inserted into the sediment of the ditch until they stood firmly. Approximately 0.20 m of the tube remained elevated above the surface water. The lids of the tubes had 3 holes of which 2 were closed with airtight septa and the third was closed with an airtight stopper. When placing the tube, the stopper was removed. Acetylene (C_2H_2) was produced by slowly adding water to CaC_2 and was led to the surface water in the tube via an aeration stone. Acetylene was bubbled from the aeration stone through the water column for approximately 8 min to inhibit the reduction of N_2O to N_2. Then, the tube was closed by the stopper and N_2O and C_2H_2 concentrations in the headspace were measured immediately and after 20 and 24 hours by a Bruël and Kjaer 1312 multi-gas monitor with a photo acoustic infra red analyzer (TGA; De Klein et al., 1996). The TGA was equipped with filters for N_2O, CO_2, C_2H_2 and H_2O determination. In each experiment 10 tubes were installed in a row, of which one did not contain C_2H_2 (control measurement).

Denitrification rates were estimated from the increase in N_2O concentrations in the headspace with time and were corrected for the internal volume of the TGA and for the amount of N_2O dissolved in water using the Bunsen adsorption coefficient (Moraghan and Buresh, 1977). At each sampling event, water samples were taken and analyzed for NO_3 and NH_4 concentrations.

Slush application

Every 5 to 10 years farmers mechanically (sprayer technique) apply slush from the ditches to the fields to irrigate and fertilize the fields and to prevent overgrowing of the ditches. To quantify the re-use of N and P by slush application, a slush experiment was performed on a field close to the experimental field (approximately 1000 m) in July 2001. The field was 57 x 380 m^2 and 60 open basins of 0.60 x 0.40 x 0.15 m^3 (length x width x height) were placed in 12 transects with about 10 m inner space perpendicular to the ditch at 5, 11, 18, 25 and 31 m from the ditch to collect the slush and estimate the amounts applied. Before and after slush application the basins were weighted. Slush samples were taken from the basins and analyzed for total-N and total-P contents.

Analytical procedures and calculations

Water samples were analyzed for total-N, NH_4, NO_3, total-P, P-ortho, Cl- and pH by continuous flow analysis (CFA). Soil, slush and grass samples were dried at 105˚C (soil and slush) or 40˚C (grass) for 24 h, sieved (2 mm) and digested with H_2SO_4 for determination of total-N and total-P by CFA (Mulvaney, 1996), according to international standards (ISO 11905). Dry matter contents were determined by weighing the samples before and after drying. Mineral N contents of the soil samples were determined by segmented flow analysis after extraction with 1 M KCl (Mulvaney, 1996).

To quantify the contributions of different nutrient sources on the N and P loading of the ditch, N and P balances of the ditch were set-up, with the ditch delineated at the flowmeter and at the ditch floor, including the upper slush layer. Nitrogen and P loads to the ditch included nutrient applications to the field that were subsequently transported to the ditch via leaching and surface run-off (e.g. nutrients from fertilization) and processes that affected the N and P concentrations in the ditch directly (e.g. denitrification in the ditch). Therefore, N and P fluxes were presented in loads towards the ditch, i.e. normalized to the surface area of the ditch. We used averaged N and P balances of the ditch over the years 2000-2002, to account for differences in response times. The contribution of each nutrient source to N and P loading of the ditch was calculated as shown in Table 4.2. If applicable, calculations were made with minimum and maximum values to account for uncertainties, errors and variabilities in measurements and determinations, under the restriction that the N and P balances of the ditch remained balanced. A proportional distribution of N and P mineralized, deposited and applied by fertilizers, manure and cattle droppings in the leaching water was assumed. Minimum and maximum net leaching rates from the field to the ditch were calculated following

Net leaching = discharge at flowmeter + denitrification in ditch + slush application – inlet water – atmospheric deposition on surface water – supply from nutrient rich peat layers *eq. 4.3*

(all units in kg y^{-1}), using minimum and maximum estimates for denitrification in the ditch and supply from nutrient rich peat layers. For discharge at the flowmeter, slush application, inlet water and atmospheric deposition no minimum and maximum values were determined.

Table 4.2. Equations used to calculate relative contributions of different sources on the N and P loading of surface water and magnitudes (average, minimum and maximum) of different nutrient sources in the catchment area. F = fertilizer application including mineral fertilizers, manure and cattle droppings, M = mineralization of soil, A = atmospheric deposition, NRPL = contribution from nutrient rich peat layers, NL = net leaching (eq. 4.3). Total inputs equal total N and P loading of the ditch as calculated in Table 4.4. Atmospheric deposition refers to atmospheric deposition on land, unless stated otherwise.

Source	Average[1] (kg y^{-1})		Minimum-maximum[1] (kg y^{-1})		Procedure to calculate relative contribution of source to N and P loading of surface water	Relative contribution of source to N and P loading of surface water (%)	
	N	P	N	P		N	P
Fertilizer, manure and cattle droppings	1446	131	n.a.	n.a.	$\frac{\frac{F}{F+M+A}\cdot NL}{total\ input}\cdot 100\%$	43-50	10-48
Mineralization of soil	781	35	579-896	32-38	$\frac{\frac{M}{F+M+A}\cdot NL}{total\ input}\cdot 100\%$	17-31	2-14
Nutrient rich peat layers	26	11	15-37	7-15	$\frac{NRPL}{Total\ input}\cdot 100\%$	8-27	33-82
Seepage	0	0	n.a.	n.a.	n.a.	0	0
Atmospheric deposition	99	<1	n.a.	n.a.	$\frac{A_{surface\ water}}{Total\ input}+\frac{\frac{A}{F+M+A}\cdot NL}{Total\ input}\cdot 100\%$	8-9	0
Inlet water	19	3	n.a.	n.a.	$\frac{Inlet}{Total\ input}\cdot 100\%$	3-4	5-6

[1] Average, minimum and maximum nutrient fluxes are expressed in kg y^{-1}, i.e. linked to the catchment area of the ditch (3.2 ha), and calculated over a minimum of 2 years (September 2000 - September 2002).

Results

Composition of the soil solution

The composition of the soil solution showed a clear spatial pattern with relatively high N and P concentrations directly below the ditch and with little variation in time (Figure 4.3). Using minimum concentrations of 8 mg N L^{-1} and 4 mg P L^{-1} and maximum concentrations of 20 mg N L^{-1} and 8 mg P L^{-1} (Figure 4.3) and an annual water flow of 1827 m^3 (10% of the annual discharge), resulted in an estimated contribution of nutrient rich peat layers to the N and P load of the ditch of 15 to 37 kg N y^{-1} and 7 to 15 kg P y^{-1}.

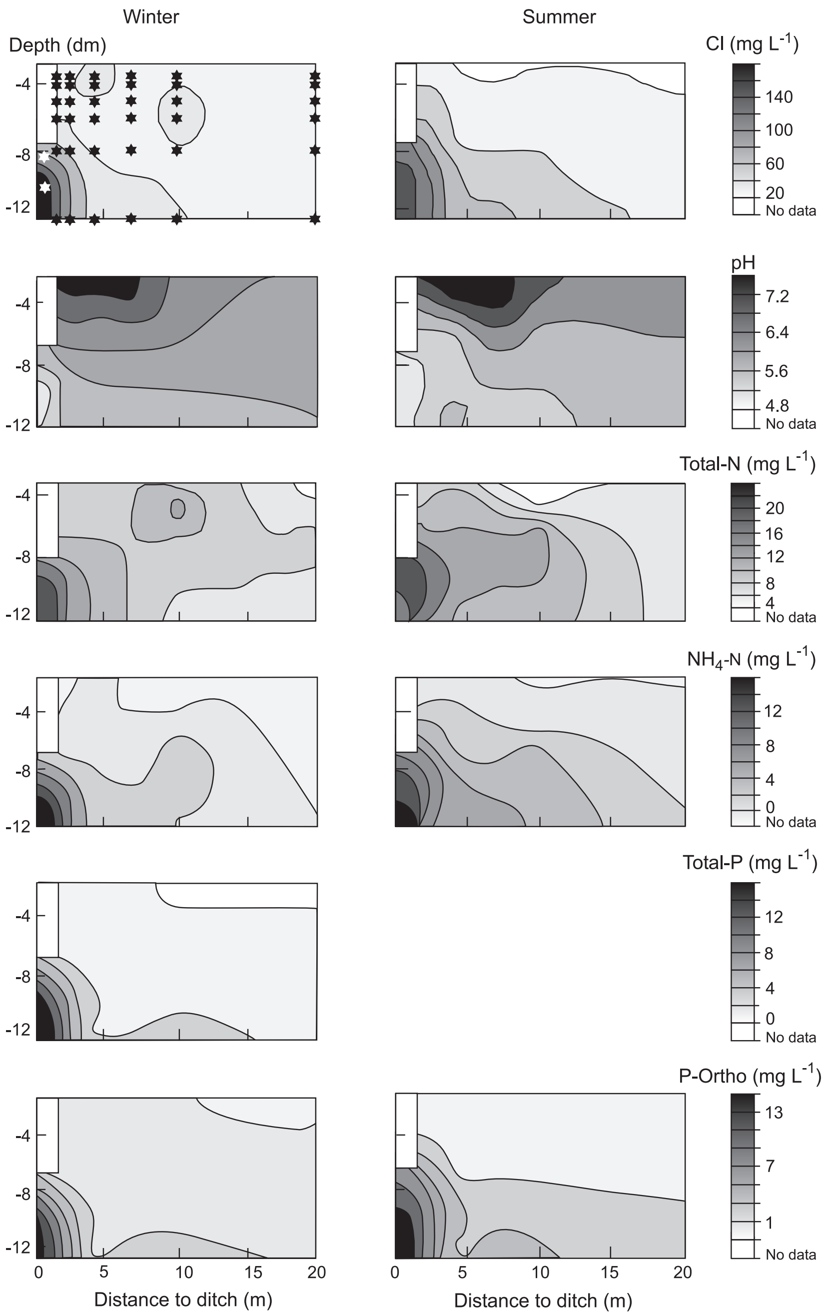


Figure 4.3. Typical results of water samples of ceramic suction cups in winter (left) and in summer (right) for Cl, pH, total-N, NH_4-N, total-P and ortho-P. In upper left graph locations of the ceramic suction cups are shown (black and white asterisks). The white surface in above left corner represents the experimental ditch. Total-P concentrations were only measured occasionally, omitting measurements during summer.

N and P mineralization of soil organic matter

Average N uptake by the grass in the zero-N plots ranged between 179 and 291 kg N $ha^{-1}y^{-1}$. Mean amounts of soil mineral N ranged between 0.5 and 16.0 mg N kg^{-1}, but showed no temporal trend. After corrections were made for atmospheric deposition and denitrification losses, N mineralization from soil was estimated between 181 and 280 kg N $ha^{-1}y^{-1}$ (Table 4.3). Measured total-N: total-P ratios of the peat soil were 23 ± 9 (n=6) and consequently P mineralization was estimated between 8 and 12 kg P $ha^{-1}y^{-1}$.

Table 4.3. N uptake of unfertilized plots and calculation of N and P mineralization from soil (± standard deviation, n=4) in kg $ha^{-1}y^{-1}$.

		2000	2001	2002	2003	Data source
A	N uptake by grass	291 ±18	228 ±28	286 ±18	179 ±16	Measurements
B	Atmospheric deposition	31	31	31	31	National monitoring program
C	Denitrification	20 ±5	35 ±12	23 ±8	40 ±10	Equation 4.1
D	Changes in soil storage (0-60 cm)	0	0	0	0	Measurements
E	N mineralization from soil	280 ±15	232 ±38	278 ±18	181 ±5	A-B+C+D

Nutrient discharge and nutrient inlet

About 20% of the total N and P discharges at the flowmeter were achieved in 10 days with the highest discharge rates. Cumulative N discharge rates yielded 33.6 and 31.5 kg $ha^{-1}y^{-1}$, for 2001 and 2002, respectively. For P, these numbers were 4.9 and 4.5 kg $ha^{-1}y^{-1}$. Cumulative inlet rates equaled 1.9 kg N $ha^{-1}y^{-1}$ for 2001 and 3.0 kg N $ha^{-1}y^{-1}$ for 2002. For P these numbers were 0.3 kg P $ha^{-1}y^{-1}$ for 2001 and 0.5 kg P $ha^{-1}y^{-1}$ for 2002.

Denitrification in ditch sediment and surface water

Nitrous oxide concentrations in the tubes with C_2H_2 always largely exceeded N_2O concentrations in the control tubes (without C_2H_2, not shown). This result indicates that denitrification was the dominant N_2O source in the ditch. Using linear interpolation between data points, annual N losses through denitrification in the ditch sediment and ditch water equaled 2.3 kg N y^{-1}. Denitrification rates were not clearly related to temperatures and/or NO_3 concentrations in the ditch water, which were often below 0.01 mg L^{-1}, but in winter sometimes reached 0.70 mg L^{-1} (Figure 4.4). A drawback of using C_2H_2 inhibition technique is that denitrification rates may be underestimated, because the addition of C_2H_2 also inhibits nitrification and thereby the supply of NO_3 for denitrifiers (Walter et al., 1979; Rudolph et al., 1991). In a simultaneous experiment at the same site, denitrification rates in the surface water were estimated by the ^{15}N pairing technique. This technique yielded denitrification rates of 18.6 kg N y^{-1} (De Klein and Gillisen, 2003), which is considerably higher than our results. However, this technique may result in overestimations of denitrification rates because of the very low NO_3 concentrations in the surface water, and the additions of $^{15}NO_3$ may have increased denitrification rates. To value both approaches, minimum (C_2H_2 inhibition methodology) and maximum (^{15}N pairing technique) denitrification rates were used to calculate net leaching (equation 4.3) and the N and P balance of the ditch (Table 4.4).

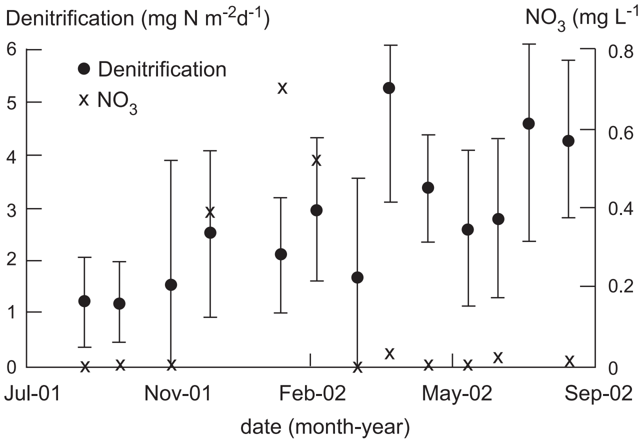


Figure 4.4. Denitrification rates in the ditch and NO_3 concentrations in the ditch water. Error bars show standard deviations of denitrification measurements (n=9). The surface area of the ditch is 2400 m^2.

Table 4.4. Average N and P balances of the ditch (kg y^{-1}) for the years 2001 and 2002. Values show minimum and maximum values, if applicable.

		N	P
IN	Inlet water	6	1
	Atmospheric deposition	7	<1
	Seepage	0	0
	Nutrient rich peat layers	15-37	7-15
	Net leaching from fields[1]	87-157	2-13
	Total input to ditch	115-207	10-29
OUT	Denitrification in ditch	2-19	0
	Groundwater recharge[2]	<1	<1
	Slush application	32-64	3-6
	Discharge at flowmeter	103	15
	Total output from ditch	137-186	18-21

1) Calculated according to eq. 4.3.
2) Considering the small groundwater recharge term on the water balance (Table 4.1), N and P recharge to groundwater was estimated to be less than 1 kg y^{-1}.

Slush application

About 40 mm of slush with an average dry matter content of 6.8% was applied to the field. Average N and P contents of the slush were 20.7 g N kg dry matter^{-1} and 1.93 g P kg dry matter^{-1}. Per slush application 102 kg N ha^{-1} and 9.6 kg P ha^{-1} were applied to the field. Because slush is generally applied every 5 to 10 years, the long-term average N and P relocations via slush application were estimated at 10 to 20 kg N ha^{-1}y^{-1} and 1 to 2 kg P ha^{-1}y^{-1}. The catchment area of the ditch was 3.2 ha and consequently minimum and maximum N and P removals from the ditch via slush applications yielded 32 to 64 kg N y^{-1} and 3 to 6 kg P y^{-1} (Table 4.4).

Sources of N and P in ditch water

Total N and P inputs to the ditch ranged between 115 and 207 kg N y^{-1} and between 10 and 29 kg P y^{-1} (Table 4.4) and net leaching yielded 87 to 157 kg N y^{-1} and 2 to 13 kg P y^{-1}. Apparently, variability in input processes exceeded variability in output processes, resulting in larger ranges in input terms than in output terms (Table 4.4). Following the equations in Table 4.2, applications of fertilizers, manure and cattle droppings accounted for 43 to 50%, mineralization of soil organic matter for 17 to 31%, nutrient rich peat layers for 8 to 27%, atmospheric deposition for 8 to 9% and inlet water for 3 to 4% of the annual N loading of the ditch. For P, these numbers were 10 to 48% for applications of fertilizers, manure and cattle droppings, 2 to 14% for mineralization of soil organic matter, 33 to 82% for nutrient rich peat layers and 5 to 6% for inlet water (Table 4.2).

Discussion

The present study shows that besides nutrients originating from fertilizers and manure applications also others sources of nutrients contribute to the N and P loading of surface water in PLA. Notably, applications of fertilizers, manure and cattle droppings accounted for 43 to 50% of the total N loading of surface water and for 10 to 48% of the total P loading of the surface water in the Vlietpolder. The remainder originated from soil mineralization, inlet water, atmospheric deposition and from nutrient rich peat layers (Table 4.2). Van Liere et al. (2002) reported even lower contributions of fertilizers and manure to the diffuse (i.e. via leaching and run-off) N and P loading of surface water in polder Bergambacht, a grassland on peat soil polder located at approximately 50 km from the Vlietpolder. In polder Bergambacht fertilizer applications and manure accounted for 24% of the total nutrient loading of the surface water, while seepage accounted for 31% (N) and 45% (P) of the total nutrient loading of the surface water. In the Vlietpolder seepage was absent (Table 4.1), while in polder Bergambacht the soil was more clayey and dairy farming was less intensive than in the Vlietpolder. Yet, for both polders the contribution of fertilizer applications and manure to the N and P loading of the surface water was limited to less than 50%.

For the Vlietpolder, mineralization of soil organic matter and nutrient rich peat layers were large contributors to the N and P loading of the ditch (Table 4.2). The occurrence of a strong concentration gradient with depth of nutrient concentrations in the soil solution (Figure 4.3) was interpreted as the result of long-term drainage through shallow soil layers, replenishing the soil solution between the ditches, but hardly reaching the soil solution below the ditch. Plausibly, deeper flow pathways carried-on nutrient rich peat water to the ditch via diffusion, dispersion and/or local convection. However, we were unable to precisely quantify any of these terms and therefore the contribution of nutrient rich peat layers should be considered as a compilation of interactions between the ditch water and the nutrient rich peat layer below the predominantly drained soil layers. The quantification of the N and P loads from nutrient rich peat layers to the surface water warrants additional research. Nevertheless, we estimated the contribution from nutrient rich peat layers to be 15 to 37 kg N y^{-1} and 7 to 15 kg P y^{-1} (Table 4.4). To evaluate this approach, leaching of N and P through the upper 1 m of soil was calculated similarly, i.e. using 90% of the total drainage and average concentrations of 7 mg N L^{-1} and 1 mg P L^{-1} (Figure 4.3). For N, this approach deviated less than 5% from the net leaching as calculated using equation 4.3. However, for P this approach resulted in overestimations of the net leaching, which was presumably caused by sorption of P to soil. In general, we believe that the analysis of the contributions of different nutrient sources to the nutrient loading of surface water is better applicable to N than to P, because of this sorption of P to soil. Nevertheless, annual net leaching from the field to the ditch (27 to 49 kg N $ha^{-1}y^{-1}$ and 0.6 to 4.1 kg P $ha^{-1}y^{-1}$; Table 4.4, the catchment area of the ditch is 3.2 ha), compared well with previously reported values of 10 to 80 kg N $ha^{-1}y^{-1}$ and 0 to 6 kg P $ha^{-1}y^{-1}$ leaching rates from grassland on peat soil (Van Huet, 1991; Kirkham and Wilkins, 1993; Van Liere et al., 2002). Strikingly, approximately 80% of the discharged N and P at the flowmeter consisted of organic N and P (not shown), but it is not clear whether the majority of N and P leached as organic compounds from the field, or leached as mineral compounds from the field and were transformed to organic compounds in the ditch.

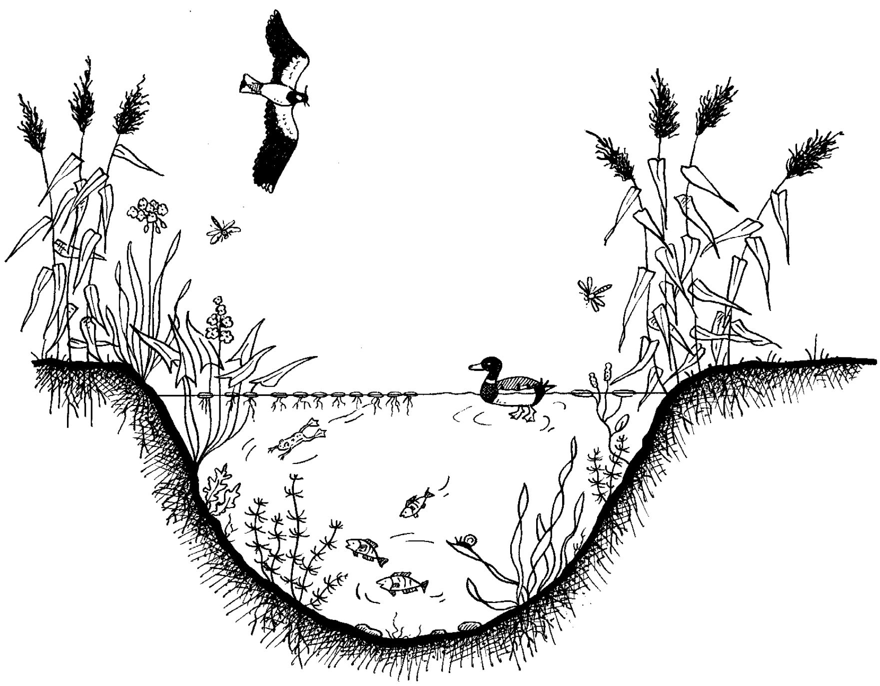
In the Vlietpolder, mineralization rates from soil were higher than most previously reported mineralization rates from peat soils (Schothorst, 1982; Berendse et al., 1994; Best and Jacobs, 2001). However, the results of the zero-N plots refer to all N released in soil, i.e. including mineralization from roots, stubbles and clayey and sandy soil layers. Also, the relatively high mineralization rates in the Vlietpolder may be due to warm weather conditions in the summers of 2000 and 2002 (KNMI, 2004), but we cannot exclude some contribution of past fertilization as the zero-N plots were fenced only just in 2000.

Mineralization of soil organic matter and the contribution from nutrient rich peat layers can both be regarded as the mining of the indigenous soil nutrient stock and accounted for a considerable part of the N and P loading of the ditch (Table 4.2). Also, both processes are largely driven by hydrological conditions. Considering the large stock of nutrients in the soil solution below the ditch (Figure 4.3), changes in drainage conditions will strongly affect the N and P balances of the ditch, e.g., higher groundwater levels will reduce the N and P mineralization and presumably also the contribution of nutrient rich peat layers, but may increase total leaching rates and reduce the workability of the fields, because of prolonged waterlogged conditions (Armstrong et al., 1988).

Conclusions

The results of this paper show that nutrient loading of surface water in peat land areas involves several sources of nutrients, of which fertilizer application, manure and cattle droppings, mineralization of soil organic matter and contributions from nutrient-rich peat layers are the most important. Due to the number of nutrient sources contributing to the N and P loading of surface water in peat land areas, reducing one source under unchanged hydrological conditions is likely to result in modest improvement of the surface water quality.

Chapter 5: Leaching of solutes from an intensively managed peat soil to surface water



Van Beek C.L., Droogers P., Van Hardeveld H.A., Van den Eertwegh G.A.P.H., Velthof G.L. and Oenema O. Water, Air and Soil Pollution. *Accepted for publication.*

Abstract

In many peat land areas in The Netherlands target concentrations for nitrogen (N) and phosphorus (P) in surface water are exceeded. A considerable, but poorly quantified fraction of the N and P loading of surface water in these areas originate from the subsoil. Waterboards, responsible for the watermanagement, are currently exploring options to improve surface water quality, whilst sustaining agricultural production. Therefore, insight into dynamics of nutrient pools in peat soils is required. The aim of this study was to measure concentration profiles (0-12 m) of the soil solution in an intensively managed grassland on peat soil and to explore the effects of a rise in surface water level on N and P loading of surface water, using budgeting approaches and two dimensional simulation modeling. The concentration profiles of N, P and Cl reflect the presence of nutrient-rich anaerobic peat and a nearly impermeable marine clay in the subsoil. Concentrations of N, P and Cl tended to increase with depth till about 6 m below soil surface and then decreased. In the top soil, inputs of N and P via fertilizers and animal manure were only partly retrieved in the soil solution, suggesting that biogeochemical processes, uptake and lateral transport processes had a dominant influence on dissolved N and P. Exploring scenario simulations showed that major drainage fluxes passed through the peat layer that transported nutrients to the adjacent surface water. Raising surface water levels with 20 cm suppresses this kind of nutrient loading of surface water by more than 30%, but nutrient rich peat layers will remain persistent as a potential source of nutrients in surface water in many peat polders in the western part of The Netherlands.

Introduction

In the western part of The Netherlands are many polders with grassland on peat soils that are used for intensive dairy farming. The grassland fields in these polders are intersected by shallow ditches for drainage of excess rainfall and land:water ratios range between 8:1 and 15:1 (De Boer, 1982). Nitrogen (N) and phosphorus (P) loads to surface water are typically in the order of 40-85 kg N $ha^{-1}y^{-1}$ and 1-6 kg P $ha^{-1}y^{-1}$ (RIVM, 2002) and contribute to the deterioration of surface water quality. A considerable part of the N and P loads is related to intensive dairy farming, but the peat soil itself may also contribute to the discharge of nutrients (Chapter 4).

Extensively managed peat soils can act as a source as well as a sink of nutrients, depending on peat type and drainage conditions (Peverly, 1982; Wieder and Lang, 1984; Heathwaite, 1991). Intensively managed peat soils are generally considered as a source of nutrients, because the indigenous peat can contain considerable amounts of nutrients that are available for leaching (Heathwaite, 1990). For intensively managed grasslands on peat soils various sources of nutrients may contribute to surface water loading, viz. fertilizers, manure, cattle droppings, seepage, atmospheric deposition, mineralization of soil organic matter and dissolved nutrients in the subsoil. The quantification of the contribution of each individual source of nutrients is complex, because different sources are hard to discriminate and because the relations between sources and transport processes to surface water are still poorly understood.

In general, shallow groundwater levels coincide with relatively high N and P loads to surface water in agricultural areas (RIVM, 2002), because of the small water storage capacity of the soil and because drainage water passes through shallow soil layers. Shallow soil layers are commonly enriched with nutrients from agricultural inputs, and it is usually assumed that drainage of these shallow, enriched soil layers has a large impact on N and P loading of surface water. For peat soils however, this generality may not hold as i) a major part of the N surpluses at the soil surface is quickly denitrified (Koops et al., 1996, De Klein and Logtestijn, 1994) and ii) indigenous peat layers may contain large amounts of nutrients that can contribute significantly to the N and P loading of surface water (Heathwaite, 1991).

Drainage of peat soils is a prerequisite for intensive agricultural management, but also enhances subsidence of peat soils (Schothorst, 1977). Subsidence of the surface necessitates the lowering of surface water levels and these intimately linked processes may continue until all peat has been oxidized. Subsidence contributes to a sequence of side-effects, including damage of infrastructure and increased risks for salt water intrusion and flooding. Water boards are currently exploring options to raise surface water levels to slow down subsidence, whilst sustaining agricultural production. Therefore, insights into mechanisms of nutrient leaching from peat soils under present and possible future drainage conditions are required.

The objectives of the study presented in this paper were to analyze current distributions of nutrients in shallow soil layers of an intensively managed peat soil and to assess possible impacts of surface water rising on nutrient leaching to surface water. Monitoring results of the composition and distribution of solutes in the soil solution of an intensively managed peat land are presented, along with results of simulation modeling of soil water movements in the upper 3.25 m and flow patterns in soil, to assess the implications of surface water rising on transport of solutes from shallow soil layers to surface water.

Materials and methods

Site description

The study was carried out in the 'Vlietpolder' in the Western part of The Netherlands (52°10' N; 4°36' E). The Vlietpolder is 202 ha in size and at present has a surface level of about 2 m below mean sea level. Almost all fields in the Vlietpolder are intensively used for dairy farming (grassland) and are intersected by shallow ditches and the land-water ratio is about 10:1. The soil consists of a man-made topsoil of about 30 cm, oxidized eutrophic peat (0.3-1 m), reduced eutrophic peat (1-3 m), calcareous marine deposits containing light and heavy clay (3–9 m) and Pleistocene sands (> 9 m, Figure 5.1). The hydrology of the Vlietpolder is described by Meinardi (2005), and can be characterized by decreasing hydraulic heads with depth with relative small gradients in the Holocene peat and clay and a relative large gradient in the Pleistocene sands. Groundwater levels in the middle of the fields fluctuate between 0 and 100 cm below soil surface. Groundwater recharge was estimated at 25 mm y^{-1}.

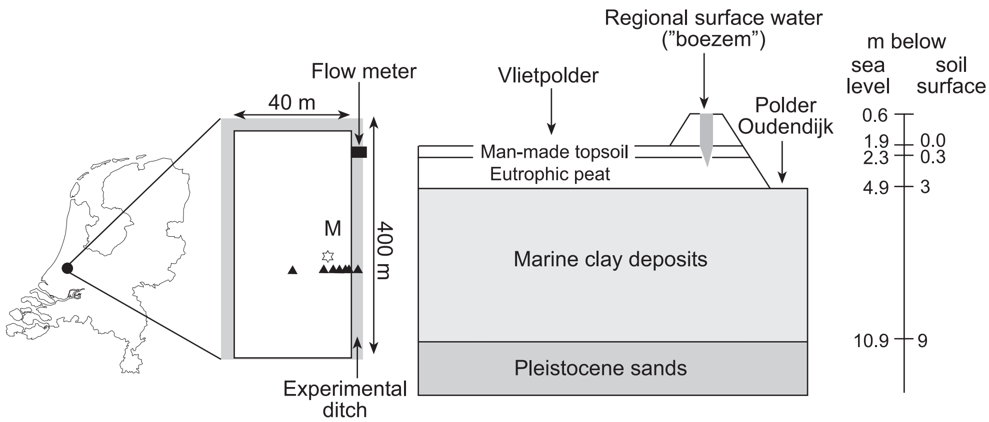


Figure 5.1. Location of the Vlietpolder (left), the experimental field (centre) with ceramic suction cups (triangles), groundwater probe (asterisk) and meteorological station (M), and cross-section of the soil profile of the Vlietpolder and Polder Oudendijk (right).

One field (approximately 40 x 400 m) was selected for intensive monitoring and is referred to as experimental field. The experimental field was surrounded by ditches at 3 sides. The field was made concave to increase surface drainage by dragging soil from the sides of the field towards the middle and nowadays the field slightly declines from the centre towards the ditch (~2 cm m^{-1}). Surface water levels were kept at about 58 cm below soil surface during winter and 48 cm below soil surface during summer. In one dead-end ditch a flow meter was placed (Figure 5.1) and ditch discharge was determined, as described in Chapter 4. During the experimental period total N and P inputs in the Vlietpolder were about 515 kg N $ha^{-1}y^{-1}$ and 40 kg P $ha^{-1}y^{-1}$ via mineral fertilizer, cattle slurry, manure and urine of grazing cattle, atmospheric deposition and application of slush from ditch cleaning, of which most was applied in spring and early summer (Chapter 2).

Soil balances

To relate above ground inputs to below ground processes we analyzed two kinds of element balances; the soil surface balance (SSB) and the soil system balance (SSysB). The SSB accounted for all inputs and outputs at the soil surface viz. inputs via atmospheric deposition, applications of fertilizers, slurry and cattle droppings, and outputs via harvests of mown and grazed grass. The SSysB is a more elaborated soil balance and also accounts for NH_3 volatilization, denitrification, leaching and net mineralization. The surplus of the SSB gives the net balance at the soil surface and is a descriptor of the nutrient management at field level. The net balance of the SSysB is an indicator of the net depletion or accrual of the soil system, and is also an indicator of our process understanding. As the SSysB accounts for all inputs and outputs of the soil system, the net balance is a measure for the correctness of balance entries, provided that changes in soil storage are limited (Oenema and Heinen, 1999). The SSB and the SSysB of the experimental field were constructed for N, P and Cl, using results from previous studies on the same site (Chapter 2, Chapter 3, Chapter 4) complemented with results from the present study. We used the methodology described in Chapter 2 which, briefly, included that so-called grassland calendars on which farmers recorded day-to-day field activities were analyzed. Inputs and outputs of N, P and Cl were calculated based on experimentally quantified contents of N, P and Cl in manure and grass and on known relations between grazing intensity, excretion and dry matter uptake of cows. Volatilization of NH_3 from the experimental field was estimated at 7 kg N $ha^{-1}y^{-1}$, based on an 8% emission from excreta of grazing cattle (Bussink, 1992).

Soil solution

In previous research on diagenetic properties of the Vlietpolder using natural abundance ratios of ^2H and ^{18}O it was concluded that the groundwater system below about 3 m below soil surface has had little interaction with the topsoil for the past 1000 years (Meinardi, 2005). Therefore, more emphasis was put on the upper soil layers than on the deeper soil layers. The soil solution was sampled using ceramic suction cups which were inserted in triplicate in a transect perpendicular to the ditch at 0.15, 0.25, 0.35, 0.50, 0.70 and 1.20 m below surface and at 1, 2, 4, 7, 10, 20 m from the ditch, where 20 m equals the middle of the field (Figure 5.1). Below the ditch ceramic suction cups were inserted at 78 and 106 cm below soil surface. With ceramic suction cups the mobile fraction of the soil solution is sampled (Corwin, 2002). From October 2001 till September 2002, every 2 weeks vacuum bottles (approximately -90 kPa) were connected to the suction cups to collect mixed water samples (one per location). Additionally, groundwater samples were taken using groundwater tubes with 1 m filters at 1, 3, 6, 9 and 12 m below soil surface. The groundwater tubes were placed in 4 fields in the Vlietpolder and samples were taken 4 times a year from December 1999 to December 2002. Water samples were analyzed for NH_4, Cl, ortho-P, total-P, NO_3+NO_2, NO_2, N-Kjeldahl, SO_4, and pH by continuous flow analysis according to ISO 5663. Total-N was calculated as the sum of N-Kjeldahl and NO_3+NO_2. Spatial interpolations between ceramic suction cups were made by inverse squared distance (Davis, 1986).

Mathematical simulations

A representative cross-section of the experimental field (20 m length, 3.25 m depth) was simulated in HYDRUS2D to analyze the behavior of water and tracer through the shallow soil solution. The HYDRUS2D program is a finite element model for simulating the movement of water, heat, and multiple solutes in variably saturated media (Simunek et al., 1999). The experimental field was schematized by 989 triangle elements taking into account the slight decline of the field towards the ditch. Because temporal dynamics of the infiltration rates were largely unknown, and because the clayey layer at 3-9 m had a very low permeability, the lower boundary was assumed impermeable (no flow). The left boundary equaled the middle of the field and was regarded as the water divide.

The aim of the mathematical simulations was to explore the behavior of water and solutes for different hydrological scenarios rather than to give an exact representation of the field measurements. Hydraulic properties were measured in duplicate samples at a depth of 30-70 cm below soil surface and were assumed representative for the whole profile. The dimensionless anisotropy factor K^A was set at 2 to account for the horizontal orientation of the deposits and to allow surface run-off. Two dimensionless conservative tracers (initial concentration = 100) were introduced to the model that were constantly released to simulate a constant source. The tracers were spatially located in such a way that water movements affecting solutes in the peat layer could be discriminated from water movements affecting solutes originating from agricultural practices.

Simulations of transport of water and solutes were run for 30 years using actual weather data from a nearby (50 km) weather station from 1974 onwards. The numerical stability of water and tracer was good (< 2.1%). We present the results of three runs. The first run, run 1, resembled the current conditions and was regarded as the reference situation. In the second run, run 2, surface water levels were raised by 20 cm, and in the third run, run 3, an additional downward seepage of 1 mm d^{-1} was added to the second run, because downward seepage is likely to increase when surface water levels are raised. For each run cumulative leaching rates in units of a conservative tracer are presented.

Results

Inputs of N, P and Cl at the soil surface

For N and Cl, SSB surpluses were 105 and 38 kg $ha^{-1}y^{-1}$, respectively. Major balance entries were mineral fertilizers and slurry application (Table 5.1). For P no surplus was calculated. The SSysB showed surpluses of 95 kg N $ha^{-1}y^{-1}$ and 8 kg P $ha^{-1}y^{-1}$ and a deficit of 12 kg $ha^{-1}y^{-1}$ for Cl. The SSysB surpluses were less than 10% of the total turnover for N, P and Cl. These relatively small surpluses on the SSysB indicate that we likely did not overlook major processes affecting transport of solutes in the shallow soil solution of the experimental field.

Concentrations of solutes in the soil solution

Solute concentrations in the shallow soil layers showed large spatial variations and relatively little variation in time. In general, we found high concentrations of solutes below the ditch that leveled-off in a bell shaped curve towards the middle of the field. For Cl this 'high concentration bell' was most apparent with concentrations of > 100 mg L^{-1} below the ditch that decreased to about 20 mg L^{-1} in the middle of the field (Figure 5.2). For total-N, NH_4, total-P and ortho-P comparable spatial distributions were found (not shown).

Table 5.1. Soil surface balance and soil system balances of N, P and Cl for the experimental field in 2002 (kg ha^{-1}y^{-1}).

				N	P	Cl
Soil System Balance	Soil Surface Balance	IN	Mineral ferilizer[1]	159	0	11
			Atmospheric deposition[2]	31		71
			Slurry application[3]	159	22	81
			Dung and urine of grazing cattle[3]	87	9	33
		OUT	Mown grass[4]	-231	-24	-79
			Grazing[4]	-100	-10	-79
		Surplus SSB		105	-3	38
		IN	Supply from peat layer[5]	8	3	120
			Mineralization of soil organic matter	244	10	<1
		OUT	Leaching to surface water	-38	-2	-170
			Groundwater recharge	-4	<1	<1
			Denitrification[6]	-213	0	0
			NH_3 volatilization	-7	0	0
		Surplus SSysB		95	8	-12

1 CAN (Calcium Ammonium Nitrate, 27% N, 2% Cl)
2 National Rainwater Monitoring Programme
3 Average N and P contents of manure were 4.78 kg N m^{-3} d.m. and 0.73 kg P m^{-3} d.m.. Cl contents were taken from Pratt (1978) and equaled 2% and were comparable to unpublished Dutch results.
4 Average N, P and Cl contents of grass samples were 0.035 g N g^{-1} d.m., 0.0038 g P g^{-1} d.m. and 0.016 g Cl g^{-1} d.m..
5 N and P taken from Chapter 4. In the reduced peat layer N:Cl ratios were about 1:15 (Figure 4.3) and this ratio was used to estimate the Cl supply from the N supply.
6 Total N losses through denitrification were estimated at 126-213 kg N ha^{-1}y^{-1} (Chapter 3). To account for underestimations due to methodological difficulties (Bollman and Conrad, 1997) we used the upper range.

Figure 5.3 shows depth profiles of solute concentrations in the soil solution, using the ceramic suction cups for Cl, total-P, NH_4 and total-N. Chloride, total-P and NH_4 showed a strong increase of concentration with depth. For dissolved total-N high concentrations were found occasionally in the upper two ceramic suction cups that resulted in large standard deviations. NO_3-N concentrations in the soil solution were most often below the detection limit of 0.05 mg N L^{-1} and were therefore omitted from further analysis. The groundwater probes showed a continuation of the concentration increase with depth until about 6 m below soil surface for all solutes. At greater depth (6-12 m below soil surface) solute concentrations decreased (Figure 5.4). With the groundwater tubes different water was sampled than with the ceramic suction cups, i.e. with the groundwater tubes mobile water in the saturated zone was sampled, while with the ceramic suction cups water was extracted from fast-flowing pores in the vadose zone. Consequently, we could not directly relate the results of the ceramic suction cups with those from the groundwater tubes, although there was a reasonable correspondence (compare Figures 5.3 and 5.4).

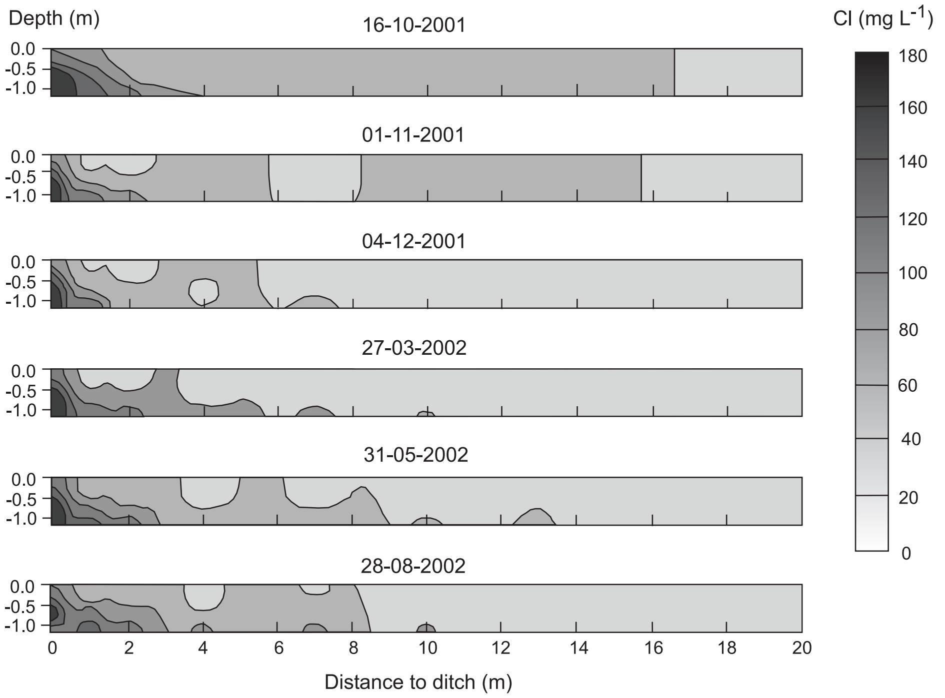


Figure 5.2. Contour plots of Cl concentrations in the soil solution. Concentrations increase from light to dark and white asterisks show locations of the ceramic suction cups. On the x-axes the distance from the ditch is shown and on the y-axes the depth below soil surface (m). The ditch is located in the upper left corner.

Model performance

The model simulations were validated on measured discharge rates of the ditch and on Cl concentrations in the soil solution. The simulated drainage and surface run-off agreed within 3% of the measured ditch discharge rates on a longer time-span (2 years). On a daily basis, however, response times of drainage and infiltration were overestimated by the model, i.e. the measurements showed less fluctuations in drainage and infiltration rates than the simulation model (Droogers et al., 2005). There were three possible explanations for these deviations: 1) the flow meter was placed at the end of the ditch (400 m), while the model simulated a cross-section of one side of the ditch, 2) in the model simulation the ditch water level was assumed constant, while in practice some fluctuations were allowed that could buffer fluctuations in drainage and infiltration rates, and 3) effects of hysteresis were not taken into account in the simulations, which could have caused overestimation of infiltration rates. Nonetheless, simulated tracer concentrations in the soil solution agreed reasonable with measured Cl concentrations when averaged over distance (R^2 = 0.74, Figure 5.5) and consequently it was concluded that the performance of HYDRUS2D to simulate movement of water and conservative solutes in the reference situation of the Vlietpolder was adequate to explore the possible effects of various scenarios.

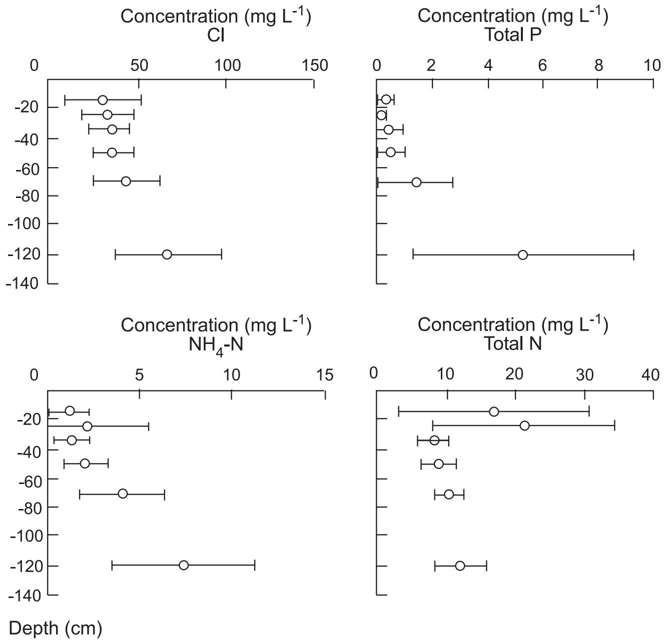


Figure 5.3. Concentration profiles with depth measured with the ceramic suction cups. Error bars show standard deviations (n=6-148).

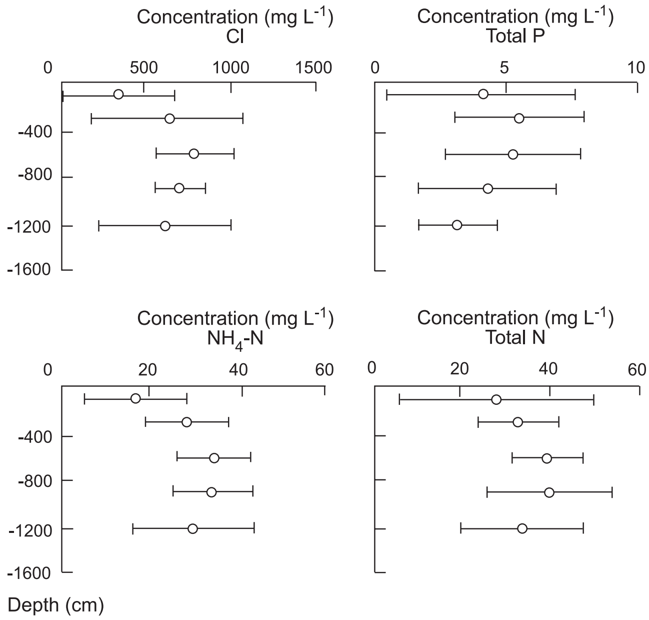


Figure 5.4. Concentration profiles with depth measured with groundwater tubes. Error bars show standard deviations (n=4-16).

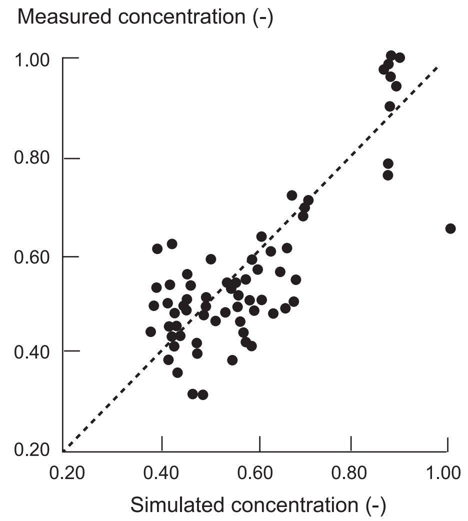


Figure 5.5. Measured and simulated Cl concentrations at several depths (0.25, 0.50 and 1.20 m below soil surface) averaged over 5 distances from the ditch (1, 2, 4, 7 and 10 m). Results are converted to dimensionless units and dashed line is 1:1 line.

Scenario analysis

The reference run (run 1) showed a strong horizontal orientation of flow pathways during discharge periods. In dry periods (e.g. August) some infiltration of ditch water and little water movement in the middle of the field were simulated (Figure 5.6). The horizontal orientation of discharge pathways was maintained when ditch water levels were increased in run 2 (not shown) and when an additional groundwater recharge of 1 mm d^{-1} (i.e. much higher than the estimated 25 mm y^{-1}) was introduced in run 3 (not shown). Also, the spatial distribution of the tracer hardly changed in 30 years of simulation. In the reference situation (run 1) leaching of the peat layer to the surface water was about 70 units tracer y^{-1}. The majority (about 90%) of the tracer was leached during winter, which is visible in Figure 5.7 by the stair wise increase of cumulative leaching. An increase of the ditch water level with 20 cm (run 2) resulted in an average decrease of 30% in leaching of solutes originating from the peat layer over 30 years. When an additional increase of downward seepage was taken into account in run 3 the decrease in surface water loading equaled 73% (Figure 5.7).

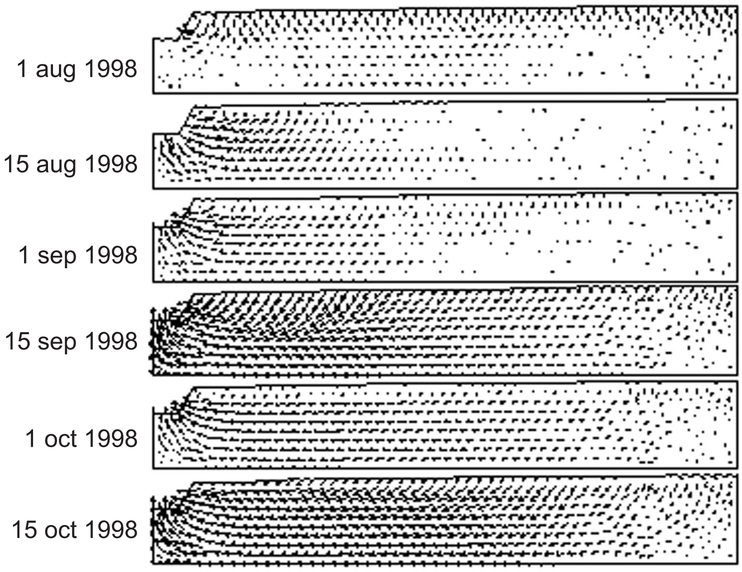


Figure 5.6. Simulated flow pathways of run 1 (reference) at some typical moments for a representative cross-section in the experimental field (20 m length, 3.25 m depth).

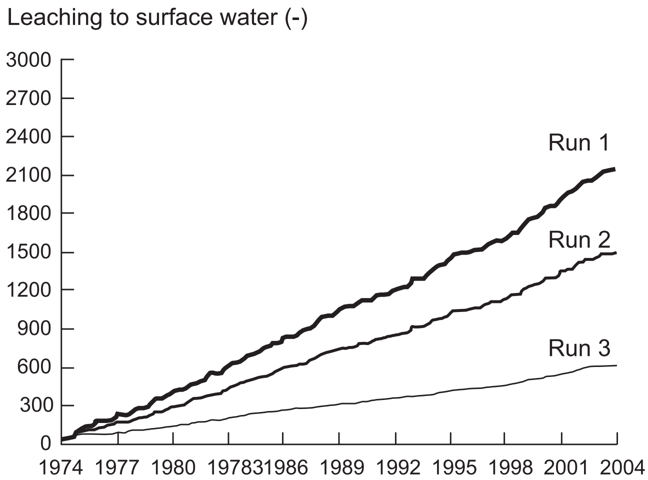


Figure 5.7. Cumulative leaching to surface water of an inert tracer representing the solutes in the peat layer for different scenarios. Run 1 = reference situation, run 2 = run 1 + 20 cm surface water level, run 3 = run 2 + downward seepage of 1 mm d^{-1}

Discussion

The Dutch peat soils are rather unique in the sense that there are only few other places in the world where peat lands are used for intensive agriculture. In literature, most studies on peat soils refer to extensively managed wetlands, and therefore are difficult to compare with our results. Nonetheless, several authors reported significant leaching losses from peat soils when drained irrespective of fertilization (Heathwaite, 1990; Nilsson and Lundin, 1996; Wieder and Lang, 1984). Heathwaite (1990, 1991) showed that a substantial nutrient source is stored within the peat and undrained peat samples in general contained more nutrients per unit of dry weight than drained peat samples.

Grasslands management on peat soils is almost as intensive as on mineral soils in The Netherlands (Reijneveld et al., 2000). In fertilizer recommendations, allowance is made for supply of nutrients by the soil, but this is not always done in practice. Farmers consider the nutrient release from the soil unpredictable and point at the poor nutrient status of the man-made topsoil consisting of sand, clay and wastes. As a consequence, the inputs of N and P via fertilizers and animal manure are relatively high, but decreasing because of the manure policy (RIVM, 2002), and have contributed to the nutrient enrichment of the top soil. This enrichment of the topsoil combined with the shallow and lateral flow of drainage water contributes to the nutrient enrichment of the surface waters in the Vlietpolder. However, our results show that also the nutrient rich subsoil has a strong impact on the composition of the soil solution (Figures 5.2, 5.3 and 5.4) and on the nutrient leaching to surface water.

For the experimental field, surpluses were observed for N and Cl on the SSB and for N and P on the SSysB (Table 5.1). These surpluses were only partly retrieved in the shallow soil solution of the topsoil. In general, solute concentrations in the shallow soil solution increased in the direction of the ditch (Figure 5.2) and with depth (Figure 5.3) until about 6 m below soil surface (Figure 5.4). For total-N relatively high concentrations were found at 15 and 25 cm below soil surface with high standard deviations (Figure 5.3). These high concentrations were probably caused by fertilizer application, but it was not possible to relate events of fertilizer application directly to moments of increased N concentrations in the soil solution. Apparently, effects of agricultural inputs on solute concentrations in the soil solution were masked by biochemical processes (mineralization, nitrification and denitrification), meteorological conditions and lateral transport processes.

Denitrification and mineralization likely have a strong effect on dissolved N concentrations as their rates were relatively large (Table 5.1). The spatial pattern of N concentrations in the soil solution may also relate to spatial variation in denitrification. On a nearby grassland, Velthof et al. (1996) observed a strong spatial relation between nitrous oxide (N_2O) emissions and distance to the ditch. As denitrification is a major source of N_2O in peat soil (Koops et al., 1997) it may suggest that rates of denitrification are also dependent on the distance to the ditch.

However, no spatial dependence of denitrification rates were observed at the study site (Chapter 3). Therefore, we presume that the observed spatial distribution of solutes including N was mainly caused by a combination of drainage and upwards diffusion of solutes as described by Fraser et al. (2001) and Giller and Wheeler (1986) for extensively used peat lands. These transport processes led to the development of a rainwater lens in the middle of the field as was found also by Wassen and Joosten (1996) and Poot and Schot (2000) for shallow soils. In a rainwater lens the indigenous soil solution is gradually replaced by rainwater with lower concentrations and eventually the rainwater lens floats on top of the indigenous soil solution.

In the Vlietpolder denitrification rates were governed by NO_3 contents in soil (Chapter 3). Total N mineralization from soil organic matter in the rootable zone in the Vlietpolder was estimated at 243 kg $ha^{-1}y^{-1}$ (Chapter 4). Apparently, a large fraction of the mineralized NH_4 was quickly taken up by grass and/or nitrified to NO_3, and subsequently nearly completely denitrified, considering the fact that NO_3 concentrations in the soil solution were generally below the detection limit. This intimate linkage of N processes was supported by the presence of ample anaerobic micro sites in the upper oxidized zone of the experimental field, as measured with an O_2 micro sensor (unpublished results Dr. K. Rappoldt). Hence, little nitrate was leached to the surface water.

Our results suggest that a major part of the soil N that leached to surface water was NH_4 and dissolved organic N that originated from the deeper, reduced peat layers, and from surface run-off. This suggestion was supported by the Cl concentrations of the ditch water. Chloride concentrations in ditch water were on average 118 ± 58 mg L^{-1} (n=126). Using a simple mixing model with Cl concentrations of 150 mg L^{-1} just below the ditch, 80 mg L^{-1} just besides the ditch and 40 mg L^{-1} at the soil surface (Figure 5.2) we calculated that more than 50% of the Cl in surface water originates from the high concentrations just below the ditch.

This study exemplifies that the presence of eutrophic peat and marine clay in the soil profile and hydrological conditions strongly affect the distribution of solutes in the soil solution, and thereby the response of nutrient leaching on measures to reduce nutrient leaching. The simulation results demonstrated that increasing surface water levels resulted in decreasing loads of a conservative tracer to surface water (Figure 5.7). This effect can be explained by the fact that increasing the surface water level will eventually result in shallower drainage pathways and consequently less nutrient rich soil layers are drained. For N, it is unlikely that changes in denitrification activity will affect this mechanism, as the magnitude of denitrification was hardly affected by groundwater level. Notably, vast amounts of degradable organic matter were found throughout the soil profile and consequently groundwater levels determined the depth of major denitrification activity, but had little effect on the total magnitude of N losses through denitrification (Chapter 3). For P however, the topsoil is enriched by yearlong application of fertilizers and manure and increasing groundwater levels will result in increased desorption of P

(Schoumans and Groenendijk, 2000) and probably in increased P leaching losses for some time.

Raised water tables reduce mineralization of soil organic matter of peat soils (Best and Jacobs, 1997; Schothorst, 1977) and consequently decrease subsidence. However, the bearing capacity and temperature of the soil will also decrease, causing less productive agriculture. In general, farmers are reluctant to raising surface water levels, but recently some promising results were achieved in the Northern peat soils of The Netherlands where, however, surface water levels are generally lower than in the Western peat soils of The Netherlands (Hoekstra et al., 2005). Moreover, farmers might adapt their fertilizer applications under a regime of raised surface water levels. Hence, the net effect of raised water tables on nutrient loading to surface water is the result of interactions between adjustments in fertilizer applications, biochemical responses and hydrological pathways. The responses to changes in water level may differ for the various nutrients. Drexler and Bedford (2002) concluded that for N and Ca groundwater was the main pathway of nutrient loading in a peat land area, whereas P and K were predominantly transported by overland flow. For the Vlietpolder surface run-off was measured for N and P using catchment plates and yielded 1.30 ± 0.76 kg N $ha^{-1}y^{-1}$ and 0.11 ± 0.03 kg P $ha^{-1}y^{-1}$ (Van Beek et al., 2003a), which is equivalent to respectively 3.4% and 5.5% of the annual N and P leaching losses. These results support the conclusions of Drexler and Bedford (2002) that for P overland transport is more important than for N.

The groundwater samples in the Vlietpolder were classified as suboxic or Fe-anoxic pH neutral to slightly acid and Ca-saturated using the methodology of Graf Pannatier et al. (2000). This groundwater type is common for areas with a Holocene toplayer with clay or peat (pers. comm. van der Grift, 2004). About 35% of the peat soils in the western part of the Netherlands were created under eutrophic conditions (Pons, 1992; De Vries and Brouwer, 2005) and consequently shallow nutrient rich peat layers as found in the experimental field of the Vlietpolder are expected in more locations. This suggests that conclusions from this study are valid for many peat areas in the western part of the Netherlands.

Conclusions

In many polders in the western part of The Netherlands eutrophic peat layers and marine clays are present at shallow depth. The results of this study suggest that these layers largely determined the composition of the soil solution and subsequently contributed to the N and P loading of surface water. Rising surface water levels can suppress this contribution with more than 30% for Cl and presumably also for N, but for P the consideration with increased desorption is unclear.

Chapter 6: General discussion

Introduction

Water boards and regional policy makers in the Western peat district of The Netherlands increasingly face the challenge to reduce emissions of nitrogen (N) and phosphorus (P) from agricultural lands whilst sustaining profitable agriculture (mainly dairy farming). In this study N and P flows to, through and from an intensively managed grassland on peat soil in the Vlietpolder in the Western peat district of The Netherlands were assessed. The aim of this study was to increase the understanding of nutrient dynamics in managed grasslands on peat soils and thereby to asses the effects of possible measures to reduce detrimental emissions of N and P. Field experiments, laboratory experiments, monitoring studies and mathematical simulations were performed and combined. With this approach a unique dataset was obtained which enabled the drawing of the following conclusions:

- Nutrient surpluses greatly differed between fields within farms and this heterogeneity may affect the estimation of nutrient loadings of surface waters at farm level (Chapter 2);
- Losses of N through denitrification were high and were predominantly governed by NO_3 contents of the soil. Therefore, denitrification losses will strongly respond to changes in soil NO_3 contents. Groundwater fluctuations did not have a distinct effect on the magnitude of the N losses through denitrification, but determined the soil depth at which denitrification predominantly occurred (Chapter 3);
- The N and P in the surface water originated from several sources. Agricultural sources (manure, fertilizers, cattle droppings) accounted for about 50% of the N loading and for about 30% of the P loading. Other main sources of N and P in surface water were mineralization of soil organic matter in the rooting zone and nutrient rich peat layers in the subsoil. Seepage, inlet water and atmospheric deposition were minor sources on an annual basis (Chapter 4);
- Biogeochemical processes and transport processes predominantly governed the composition of the soil solution in the upper meter. At greater depths (> 1 m) the composition of the soil solution reflected the presence of a nutrient rich peat layer. This nutrient rich peat layer was a considerable contributor of N and P to surface water. Mathematical simulations indicated that raising the mean surface water level with 20 cm will decrease N and P inputs from this nutrient rich soil layer to surface water by more than 30% (Chapter 5).

In this chapter the results of the previous chapters are brought together to come to an integrated assessment of nutrient losses from grassland on peat soil. Following Jarvis et al. (1996), single nutrient flows can be influenced by management, but the intervention in one part of the nutrient cycle may affect flows elsewhere in the cycle. Therefore, a whole farm assessment of reducing nutrient losses is required and applied.

In Chapter 1 three steps were distinguished to achieve the final aim to 'increase the understanding of the N and P routes to and from an intensively managed grassland on peat soil located in a polder to ascertain possibilities and limitations of reducing N and P concentrations in the surface water of the polder'. These steps were:

- Quantification of N and P surpluses at farm and field level;
- Quantification of N and P pathways;
- Integration of N and P surpluses with N and P loss pathways.

In this chapter each of these steps is discussed, supplemented with a section on the effectiveness of measures to reduce N and P loading of surface water and some final remarks.

Quantification of N and P surpluses at farm and field level

In Chapter 2 N and P surpluses at farm level and field level were quantified and discussed. These surpluses reflect the extent of N and P losses to the environment. In the current section N and P flows at field level and farm level, as presented in Chapter 2, are summarized in a farm system balance together with measured N and P losses, as presented in Chapters 3, 4 and 5. The outline of such a farm system balance was presented in Chapter 1 for average dairy farms on peat soil in The Netherlands. Here the same outline as in Chapter 1 was used, supplemented with two environmental compartments, viz. atmosphere and surface water. Note that the data provided in this thesis do not allow for a distinction between manure produced in farm buildings from manure applied to fields. Therefore, the compartment 'manure' as shown in Figure 1.4 of Chapter 1 was replaced by the compartment 'farm buildings' which covers all incoming and outgoing flows to the farm building, including manure produced in stables. To provide a clear overview of the large number of data the results are presented in so-called 'from-to' (or n-squared) tables. These 'from-to' tables indicate all of the possible movements of elements between different compartments, as shown in Table 6.1.

Table 6.1. 'From-to' table of net mass flows of the experimental field of the Vlietpolder. An 'x' denotes absence of flow.

		TO						
		Market	Farm buildings	Grass	Cattle	Soil	Atmosphere	Surface water
FROM	Market	x	fertilizers, concentrates and cattle	x	x	x	x	x
	Farm buildings	milk and cattle	x	x	concentrates, silage	fertilizers, manure	NH_3 volatilization	x
	Grass	x	silage	x	grazed grass	x	x	x
	Cattle	x	manure, milk	x	x	dung and urine	x	x
	Soil	x	x	uptake of nutrients	x	mineralization in rooting zone, subsoil supply	NH_3 volatilization, $N_2(O)$ emission	leaching
	Atmosphere	x	x	N_2 fixation	x	atmospheric deposition	x	atmospheric deposition
	Surface water	x	x	x	x	x	$N_2(O)$ emission	x

Table 6.1 shows that some compartments have multiple interactions, e.g. the soil compartment and the atmosphere compartment, while others are restricted to one interaction, e.g. the market compartment. Also, in Table 6.1 some simplifications were made, e.g. all gaseous N losses from farm buildings were considered NH_3 volatilization, while in practice also N losses through denitrification may occur in farm buildings. Moreover, NH_3 volatilization in the fields was allocated to the soil compartment, but this allocation is not very strict, as most NH_3 from grazed grasslands originates from fresh urine patches (Bussink, 1996a) and therefore may just as well have been allocated to the cattle-compartment. Nevertheless, a 'from-to' table is a valuable tool to i) provide insight into mass flows within complex (i.e. with multiple compartments) systems, ii) locate sinks and sources within farming systems, and iii) check the consistency of the data as eventually all 'to' flows should equal all 'from' flows.

In Tables 6.2a-c the 'from-to' tables for N, P and Cl are presented. To check the consistency of data, the total sums of N, P and Cl should remain equal throughout the system. However, N and P are reactive elements (i.e. react with soil) and in general reactive processes are highly variable in time and in space and reactive processes are more complex to quantify than non-reactive mass flows. Therefore, the inert (i.e. non-reactive) tracer Cl was included as an indicator for the consistency of data. It was collected in a comparable way as N and P. For N and P, average numbers for the polder were used as presented in Chapter 2 for 1999 to 2001 supplemented with field data presented in Chapter 4 for 2001 and 2002. For Cl average numbers for 2002 were used as presented in Chapter 5.

Table 6.2a. 'From-to' matrix for mean N flows of the experimental field of the Vlietpolder in the period 1999 to 2002 (kg $ha^{-1}y^{-1}$).

		TO							Total from
		Market	Farm buildings	Grass	Gattle	Soil	Atmosphere	Surface water	
FROM	Market	x	314	x	x	x		x	314
	Farm buildings	96	x	x	321	332	5[1]	x	754
	Grass	x	197	x	190	x	x	x	387
	Cattle	x	227	x	x	120	x	x	347
	Soil	x	x	386	x	269	177	38	870
	Atmosphere	x	x	0	x	31	x	2	33
	Surface water		x	x	x	x	1	x	1
Total To		96	738	386	511	752	183	40	

[1)] NH_3-N volatilization from farm building was estimated at 10.5 kg N per cow (Oenema et al., 2000).

Table 6.2b. 'From-to' matrix for mean P flows of the experimental field of the Vlietpolder in the period 1999 to 2002 (kg $ha^{-1}y^{-1}$).

		TO							Total from
		Market	Farm buildings	Grass	Gattle	Soil	Atmosphere	Surface water	
FROM	Market	x	34	x	x	x	x	x	34
	Farm buildings	17	x	x	41	31	x	x	89
	Gras	x	20	x	17	x	x	x	37
	Cattle	x	36	x	x	11	x	x	47
	Soil	x	x	25	x	22	x	2	49
	Atmosphere	x	x	x	x	x	x	x	0
	Surface water		x	x	x	x	x	x	0
Total To		17	90	25	58	64	0	2	

Table 6.2c. 'From-to' matrix for mean Cl flows of the experimental field of the Vlietpolder in 2002 (kg $ha^{-1}y^{-1}$).

		TO							Total From
		Market	Farm buildings	Grass	Gattle	Soil	Atmosphere	Surface water	
FROM	Market	x	32[1]	x	x	x	x	x	32
	Farm buildings	8[2]	x	x	41	92	x	x	141
	Grass	x	79	x	79	x	x	x	158
	Cattle	x	8	x	x	33	x	x	41
	Soil	x	x	158	x	120	x	170	448
	Atmosphere	x	x	x	x	x	x	x	0
	Surface water		x	x	x	x	x	x	0
Total To		8	119	158	120	245	0	170	

[1] The Cl content of concentrates is about 0.3% (Chan et al., 2005).
[2] The Cl content of milk is about 0.1% (Vink and Wolbers, 1997).

In Tables 6.2a-6.2c data from different experiments were used that differed in spatial scales and in time. Yet data consistency of the total balance was evaluated as good, with less than 1% difference between total 'to's' and total 'from's' for all considered elements. However, not all data were entirely independent. For instance, the production of manure in the farm buildings was calculated from the application of manure to the soil (Chapter 2) and consequently the production of manure equals the application of manure to the soil by definition. For the internal flows related to the cattle compartment large discrepancies were found between total 'to's' and total 'from's'. These discrepancies were at least partly caused by systematic underestimations of manure applications to the fields. When comparing officially registered purchase of fertilizers with farmrecords on application of fertilizers it appeared that farmers systematically underestimated the application of fertilizer to their fields (Van Beek and Oenema, 2002). Apparently this also occurred with the registration of manure application as was corresponded with the farmers in a discussion session on this topic. The underestimation of manure application to the fields may have consequences for the soil surface balances as provided in Chapter 2 and possibly explains some of the differences in farm gate balances and soil surface balances (Chapter 2, Figure 2.5). The impact on other N and P pathways as determined on this thesis are expected to be small, as they were independent of manure application.

For N and Cl highest fluxes originated from the soil compartment (source), while for P the highest flux was found to the farm buildings compartment (sink), which was due to the relatively high input of P fertilizers from the market compartment to the farm buildings compartment. Several compartments could act as a sink as well as a source at the same time. Despite the large inputs of N, P and Cl to the soil via manure and fertilizers, the soil compartment was net depleted with N and Cl (i.e. was a nutrient source). This was caused by mineralization of organic matter (N, P) and by supply from the subsoil (N, P, Cl). For P the soil was a net sink and apparently sorption of P to the soil exceeded the release of P from mineralization and supply from the subsoil. This finding was supported by the ample availability of sorption sites for P in the soil as analyzed by chemical extractions for the upper 75 cm (Van Beek et al., 2003b). The difference between total inputs and total outputs for the external compartments 'surface water' and 'atmosphere' reflect the total net emissions of N, P and Cl from the soil to these compartments. The difference between total inputs and total outputs for the farm buildings compartment reflects changes in the storage of animal feed and animal manure.

Mean total flows of N, P and Cl to surface water equaled 40, 2 and 170 kg ha^{-1}, respectively (Tables 6.2a-c). Hence N:Cl and P:Cl ratios of total inputs to the surface water were about 1:4 and 1:85, respectively. As N:Cl and P:Cl ratios can be used to assess the presence of nutrient consuming or producing processes in ground and surface water (Mengis et al., 1999; Altman and Parizek, 1995), these ratios were compared with measured N:Cl and P:Cl ratios in the ditch water. Averaged measured total N, P and Cl concentrations of the surface water in the Vlietpolder equaled 5.2 mg N L^{-1}, 0.9 mg P L^{-1} and 118 mg Cl L^{-1} (Chapter 5) and consequently

N:Cl ratios and P:Cl ratios in the surface water were about 1:23 and 1:130. Hence, N and P concentrations in the surface water were relatively low compared to the input ratios of Tables 6.2a-c and apparently considerable amounts of N and P were removed from the surface water. There are various processes that can explain this finding, e.g. denitrification of N in the ditch, sedimentation and uptake by aquatic plants. But also inlet of Cl rich water or underestimation of direct supply of Cl to the ditch water from the nutrient rich soil layer below the ditch (Chapter 5) may explain the relatively low N:Cl and P:Cl ratios in surface water.

Tables 6.2a-c show that total inputs to the soil compartments were 752 kg N $ha^{-1}y^{-1}$, 64 kg P $ha^{-1}y^{-1}$ and 245 kg Cl $ha^{-1}y^{-1}$, while total outputs from the soil compartment were 870 kg N $ha^{-1}y^{-1}$, 49 kg P $ha^{-1}y^{-1}$ and 448 kg Cl $ha^{-1}y^{-1}$. These numbers are high in comparison with a study performed by Aarts et al. (2000) on experimental dairy farm 'De Marke' on sandy soil in the Eastern part of The Netherlands. They found a total input of 389 kg N $ha^{-1}y^{-1}$ to the soil compartment and a total output of 353 kg N $ha^{-1}y^{-1}$ from the soil compartment. The large differences between De Marke and the Vlietpolder in N flows to and from the soil system is mainly caused by differences in soil properties, i.e. net mineralization of soil organic matter in the Vlietpolder, while Aarts et al. (2000) found net accumulation in the soil compartment. The Vlietpolder can be characterized by its intensive cycling of N, P and Cl, not only due to managed flows like inputs through fertilizers and concentrates but also through 'unmanaged' flows like mineralization in the rooting zone and gaseous emissions. Keeping in mind the comparison of the fate of nutrients as a matter of chance (Chapter 1, Van Noordwijk, 1999) the risk for losing nutrients to the environment in absolute numbers is much higher for the Vlietpolder than for the De Marke. However, as already pointed out in Chapter 1, the fate of nutrients is not merely a matter of coincidence, as preferences and likelihoods may occur as is shown in the subsequent sections.

Quantification of N and P pathways

In this thesis emphasis was given to the experimental quantification of N and P loss pathways from the soil to surface water and atmosphere. Experimental data on these pathways are rather scarce and fragmented, in part because of methodological complexities and spatial and temporal variabilities (Kroeze et al., 2004). In this section some of the possible experimental pitfalls in the determinations of N and P loss pathways are discussed to illustrate the complexity and the achievements made in this study, but also to illustrate the uniqueness of these experiments and to set challenges for future research.

Actual denitrification rates in soil

Actual denitrification rates in soil were quantified using the Acetylene Inhibition Technique (AIT, Chapter 3). This technique is easy to apply and has the advantage of allowing for many replicates in space and time, but has the tendency to underestimate denitrification (Walter et al., 1979; Bolmann and Conrad, 1997; Knowles, 1990; Ryden and Dawson, 1982) due to:

1) Inhibition of nitrification by acetylene and subsequent NO_3 limitation of denitrification.
2) Incomplete inhibition of the reduction of N_2O to N_2 due to uneven diffusion of acetylene through the soil.
3) Adaptation of denitrifiers to the presence of acetylene.

Restrictions 1 and 3 demand for prompt detection of N_2O accumulation in the headspace after incubation of the soil, but restriction 2 demands for postponed detection. Consequently the timing of detection of N_2O accumulation in the headspace is a compromise between contrasting demands. For best estimation of denitrification by AIT, measurements should be performed in the linear phase, i.e. when N_2O accumulation in the headspace is linearly related to time after addition of acetylene. In this study, the time window of the linear phase was determined several times (but not each time), and was found to range between 8 and 24 hours after addition of acetylene (Chapter 3). However, the timing of this linear phase depended on moisture conditions of the soil sample and prevailing air temperature. Considering the high variability of moisture conditions and temperatures of the samples, it is plausible that sometimes the linear phase was (partly) missed causing underestimation of actual denitrification rates in soil. Also, occasionally the N_2O production of the control treatments (those without acetylene) exceeded the N_2O production of the treatments with acetylene. Although N_2O produced during nitrification might explain this finding, it is unlikely to totally cover the sometimes high discrepancies between the treatments with and without acetylene. Therefore, in my view, denitrification rates using AIT were presumably underestimated and I prefer to use the estimate based on the sum of the N_2O production in the control (-acetylene) and experimental (+ acetylene) treatments instead of the estimate based on the experimental treatments corrected for the control treatments. This estimate yielded 126 to 213 kg N $ha^{-1}y^{-1}$ (Chapter 3), which is fairly high compared to other estimates of denitrification in agricultural soils (Barton et al., 1999). These relatively high denitrification rates can be explained by the favourable conditions for denitrification in the soil of the present study (Chapter 3).

Processes in ditches

Leaching of nutrients from soil to surface water was assessed indirectly using the flow meter (weir) at the end of the ditch and measured nutrient concentrations in the surface water. This is a common method as, apart from tile drained soils, it is nearly impossible to measure leaching from soil to surface water directly. The discharge measured at the location of the flowmeter was subsequently corrected for in-ditch processes to obtain an estimate of leaching of N and P from soil to surface water. This approach resulted in relatively large ranges of 87 to 157 kg N y^{-1} ($\approx$ 27 to 49 kg N $ha^{-1}y^{-1}$) and 2 to 13 kg P y^{-1} ($\approx$ 0.6 to 4.1 kg P $ha^{-1}y^{-1}$) due to high variability in balance entries (Chapter 4, Table 4.4). One of the most variable balance entries was denitrification in the ditch water which was experimentally quantified in Chapter 4. Accurate quantification of denitrification in sediment and surface water is very complex (Seitzinger, 1990) and there are only few studies where AIT was used to determine actual denitrification rates in ditches (e.g. Christensen and Sørensen, 1988). Another method to estimate denitrification from soil and surface water is based on tracing ^{15}N labeled N. Comparison with this ^{15}N pairing technique showed considerable differences. However, there are also possible artifacts associated with this ^{15}N technique, as for example, the NO_3 concentrations in the ditch water were very low and the addition of labeled NO_3 may lead to overestimating actual denitrification rates in the ditches. Hence, this possible artifact could have caused the deviation between the results of the AIT and the ^{15}N pairing technique (Chapter 4). Consequently, accurate validation of the AIT methodology was not possible. Therefore, additional research on methodologies to quantify N losses from ditches in general, and through denitrification in particular, is warranted considering that denitrification is generally assumed to cause significant reductions of N concentrations in surface water (e.g. Yan et al., 1998).

Mineralization in the rooting zone

The release of N and P due to mineralization of organic matter in the rooting zone was determined using so-called zero-N plots (plots omitted from agricultural N applications) corrected for denitrification, atmospheric deposition and changes in soil mineral N contents (Chapter 4). Although this method is straightforward and is widely used, it may have caused overestimation of the N mineralization of organic matter in the rooting zone due to contributions from past fertilization. The data showed that N uptake decreased during the experimental period (Chapter 4, Table 4.3). Visual extrapolation of the mineralization rates in time eventually runs to about 160 kg N $ha^{-1}y^{-1}$. This is well in line with previously reported estimates of 100 to 200 kg N $ha^{-1}y^{-1}$ for peat soils (Berendse et al., 1994; Best and Jacobs, 2001; Schothorst, 1982). Also, the uncorrected N uptake rates (i.e. the N supply capacity of the soil) corresponded with those reported by Van der Meer et al. (2004). However, the time span of the experiment in the present study was too small to unambiguously relate the observed decline in mineralization rates to the contribution of past fertilization. Therefore, the estimation of mineralization from soil organic matter as presented in this study refers to mineralization of N in the rooting zone, with possible residual effects of past fertilizations.

Mineralization in the subsoil

The subsoil appeared to be a considerable contributor of N and P loading of surface water as discussed in Chapters 4 and 5. The presence of such a 'subsoil' source was previously observed by mathematical simulation of movements of water and solutes through peat soil by Hendriks (1993) and Hendriks et al. (2002), but was part of a wider 'soil' balance entry and is therefore only partly comparable to the definition used in this thesis. The quantification of this nutrient supply by the subsoil was very difficult and is still far from satisfying. The local hydrology predominantly governed the extent of this contribution as discussed in Chapter 5, but the understanding of the mechanisms and controlling factors is still poor. Therefore, future research should aim at:

- Understanding of the dynamics, mechanisms and controlling factors of the N and P supply by the subsoil in order to evaluate the impact of e.g. increased groundwater fluctuations beforehand.
- Improving the quantification of the contribution of subsoil supply to N and P loading of surface water in peat land areas and factors affecting it.
- Exploring the spatial distribution of this subsoil supply of N and P to surface water in peat land areas.

Discrimination of nutrient sources

An important result of this thesis was the quantification of the contributions of different sources to the total N and P loading of surface water. In Chapter 4 this was done by quantifying all inputs to the soil solution and to the surface water and assuming a proportional contribution of each input, except for those inputs that directly emitted to the surface water.

There are various methods to estimate the contribution of different sources on the N and P loading of surface water at field level:

- Estimation of the distribution of flow pathways to the surface water and associated (combinations of) sources to each flow pathway (e.g. Michielsen and Van Schaik, 2004; Grieve and Gilvear, 1994).
- Determining isotopic ratios of N and/or O of different sources and of the receiving water body (e.g. McLaughlin et al., 2006; Vander Zanden et al., 2005)
- Using known concentration ratios of different sources and the distribution of travel times of solutes through soil (Meinardi, 2005).
- Mathematical simulation modeling and excluding one source at the time to quantify its contribution (Hendriks et al., 2007; Wolf et al., 2003; Kronvang et al., 1995).

Each of the abovementioned methods has been applied to the Vlietpolder (reported in the first reference provided by each method above) and each has its own advantages and disadvantages. Comparison between the different methods was complicated by differences in source definitions, but roughly speaking results were comparable in the sense that agricultural inputs accounted for about 40% of the N and P loading of the ditch water in the Vlietpolder (Van den Eertwegh and Van Beek, 2004).

The natural abundance of ^{15}N of main N inputs to soil and surface water as a discriminator between N sources in surface water was applied in the Vlietpolder in spring 2003. The $\delta^{15}N$ values of various sources (N fertilizer, soil organic matter, soil mineral N, manure, fresh cattle droppings, etc) and of the surface water were determined. However, the range of $\delta^{15}N$ values found for N in surface water exceeded the range of $\delta^{15}N$ values of the possible N sources. Therefore, it was not possible to quantify the contribution of these sources to the N loading of surface water on basis of differences in natural abundance of ^{15}N. However, this method can be improved by also analyzing the $^{16/18}O$ ratio of NO_3 of the different sources (pers. comm. N. Wrage, 2006) and would then be a possible tool for experimental discrimination of the contribution of different sources to the N loading of surface water.

Still other methods for discriminating nutrient sources in surface water are applied at higher spatial scales, e.g. GIS mapping in combination with (statistically derived) nutrient export coefficients (Smith et al., 2005), or a combination of the abovementioned methods (e.g. Pieterse et al., 2003). However, with differing spatial scales also the discrimination of sources differs. For instance, a GIS based approach would likely result in the statement that agriculture is the only source of nutrients in the surface water of the Vlietpolder, because the land use of the Vlietpolder is classified as agriculture. Small-scale studies like this study have the advantage to demonstrate that various sources may contribute to the broadly defined 'agricultural' source, sometimes reported in large-scale studies. Identifying the various sub-sources has the advantage also to define specific measures for decreasing the emissions of nutrients to surface waters. This study has clearly shown that not all nutrient inputs to the surface water were directly related to 'agriculture' and that these sub-sources respond different to e.g. hydrological measures. Therefore, all different sources should be specified and understood to be able to define effective and efficient abatement strategies.

Other sources

In this thesis nutrient loading of surface waters was related to transport of nutrients from the soil solution to the surface water and to direct emission of nutrients on surface water. However, nutrients may be also released from the surface water compartment itself via biogeochemical processes, e.g. due to interactions between inlet water and ditch sediments. This inlet water eventually originates from the river Rhine and is, compared to peat waters, alkaline and rich in SO_4^{2-} (Smolders and Roelofs, 1995). This inlet water may contribute to enhanced mineralization rates. Also, SO_4^{2-} may be reduced to FeS and FeS_2, thereby reducing the availability of sorption sites for P and releasing P to surface waters (Lamers et al., 1998; Roelofs, 1991).

Inlet water is not the only source of SO_4^{2-} in the Vlietpolder as SO_4^{2-} discharge to surface water exceeded SO_4^{2-} inlet with a factor 14. This suggests that the Vlietpolder was a net source of SO_4^{2-}. Moreover, in a strip of about 5 m alongside the ditch SO_4^{2-} concentrations increased from about 500 mg L^{-1} in summer to about 1000 mg L^{-1} in winter in the upper 30 cm (Van Schaik, 2004) which equals about 250 kg SO_4^{2-} $ha^{-1}y^{-1}$. This amount can not be achieved by atmospheric deposition (about 16 kg SO_4^{2-} $ha^{-1}y^{-1}$) and/or animal manure and fertilizer applications (about 50 kg SO_4^{2-} $ha^{-1}y^{-1}$). Hence, there was a source of SO_4^{2-} in the ditch and/or in the fields draining on the ditch.

Most likely the origin of this SO_4^{2-} was in the ditch sediments. In reclaimed peat soils of marine or estuarine origins often reduced S species are present

(Christensen and Olesen, 1986; Shotyk, 1988). These sulfides can be found especially in ditch sediments (Holmer and Storkholm, 2001; Murray, 1995). Farmers in Dutch peat land polders are obliged to dredge their ditches annually, thereby placing ditch sediments on the ditch banks (Higler, 1989). Also, ditch dredging is recommended to farmers in order to re-use leached nutrients (Van Strien et al., 1991). Possibly, with this ditch dredging sulfides are oxidized and SO_4^{2-} is supplemented to the soil. Eventually SO_4^{2-} may be reduced again in the soil solution resulting in the release of P from soils and in increased P concentrations in soil solution and subsequently in surface waters of peat land polders. However, this is still a hypothesis that needs further validation. If it turns out that this hypothesis is correct, it will put the effectiveness of nutrient re-use through improved dredging techniques in part under pressure.

Integration of N and P surpluses with N and P loss pathways

Nutrients from agriculture are lost to the wider environment via different pathways. The contribution of each of these pathways depends on the nutrient inputs and outputs, on soil characteristics and environmental conditions. The relative contribution (f_i) of each pathway (x_i) to the total nutrient surplus follows from:

$$f_i = x_i / (\text{denitrification} + NH_3 \text{ volatilization} + \text{leaching to groundwater} + \text{leaching to surface water} + \text{changes in soil storage}) \quad \text{eq. 6.1}$$

which equals according to the system definition used in Chapter 2:

$$f_i = x_i / (\text{fertilizer application} + \text{manure application} + \text{atmospheric deposition} + \text{cattle droppings} + \text{slush application} + \text{net mineralization of soil} + \text{subsoil contribution} - \text{mown grass} - \text{grazing}) \quad \text{eq. 6.2}$$

where all units are in kg $ha^{-1}y^{-1}$. This definition deviates somewhat from more common definitions in which mineralization of soil organic matter and subsoil supply are not taken into account (e.g. Schröder et al., 2003; Oenema and Heinen, 1999), because these entries especially apply to the Vlietpolder. With data provided in the previous chapters and equation 6.2 the average partitioning of the nutrient loss pathways was estimated for N and P for the experimental field of the Vlietpolder for the years 2000-2002.

So far, leaching of N and P to deeper groundwater was neglected in this thesis because of the presence of a dense clay layer at shallow depth. However, still an annual recharge of 25 mm y^{-1} was found for the Vlietpolder (Chapter 4, Table 4.1), which corresponds to an estimated groundwater recharge of about 7.5 kg N $ha^{-1}y^{-1}$ and 1.0 kg P $ha^{-1}y^{-1}$, assuming average N and P concentrations of 30 mg N L^{-1} and 4 mg P L^{-1} in the soil solution (Chapter 5, Figure 5.4). This estimate confirms the modest leaching to groundwater in comparison with leaching to surface water for

N, but for P leaching to groundwater still accounted for 9% of the P losses (Table 6.3).

Table 6.3. Average partitioning of N and P over different nutrient loss pathways for the experimental field of the Vlietpolder (%). Data calculated from Tables 6.2a and 6.2b and Eq. 6.2.

	Loss pathway	*N*	*P*
Gaseous losses	N_2 and N_2O (denitrification)	70	
	NH_3 volatilization	8	
Leaching	Surface water	16	18
	Groundwater	3	9
Unaccounted for		3	73

Table 6.3 shows that gaseous losses and leaching losses accounted for 97% of the N surplus and for 27% of the P surplus. The remainder ('unaccounted for') is attributed to storage in the soil, but may also be attributed to errors in data sources and data administration (Chapter 2). For P, the considerable amount unaccounted for was interpreted as changes in soil storage, which was confirmed by the net soil P 'to' flux of Table 6.2b.

Relation between nutrient inputs and nutrient loss pathways

The Vlietpolder is rather unique in the sense that there are only few places in the world where intensive dairy farming is performed on peat soil with such a strict water management. Therefore, a divergent partitioning of nutrient inputs over nutrient loss pathways was expected for the Vlietpolder compared to other soils. In Figure 6.1 the fractions of N inputs to the soil that were lost via denitrification and leaching in the Vlietpolder were compared with published results obtained on other (mineral) soils.

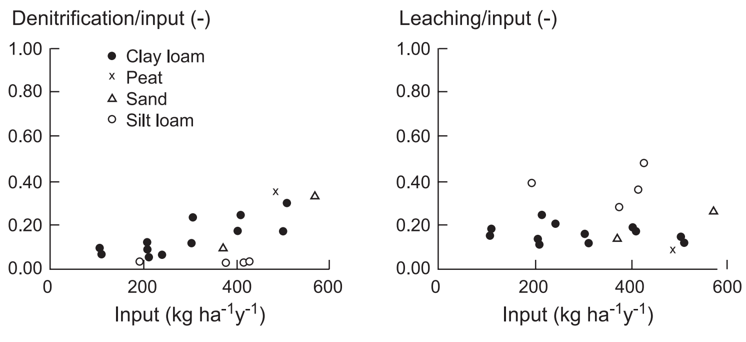


Figure 6.1. Fractions of N inputs at field level for grassland sites on sand, clay loam, silt loam and peat soils that were lost via denitrification (left) and leaching (right). Data taken from Ledgard et al. (1999), Watson et al. (1992) and Aarts et al. (2000). Inputs were recalculated from published data and included fertilizer applications, atmospheric deposition, slurry, dung and urine of grazing animals and symbiotic N fixation. Leaching included leaching to groundwater and to surface water. Data for the peat soil refer to the results of the Vlietpolder presented in this thesis.

For reasons of comparison, N loss pathways were related to N inputs (and not to N surpluses), which explains that for the peat soil of the Vlietpolder loss fractions in Figure 6.1 are smaller than the values in Table 6.3. Figure 6.1 shows that the peat soil of the Vlietpolder had relatively high denitrification losses and relatively low leaching losses compared to other soil types. The assumed positive relation between N input and N loss pathway was supported to some extent for denitrification, but for leaching this relation was absent. The effect of soil type on the partitioning of the N inputs was somewhat scattered. The silt loam soil had relatively low denitrification losses and relatively high leaching losses.

It is sometimes assumed, e.g. in simple models (De Vries et al., 2001) that N loss pathways are proportional to the N surplus for a certain soil type and environmental condition. However, this is not always the case. For example, the relation between denitrification and NO_3 content of the soil (which is often related to N surplus) is generally described by a Michaelis-Menten function that runs asymptotically to the maximum possible denitrification rate under field conditions (e.g. Hénault and Germon, 2000). In Chapter 3 a linear relation was found between actual denitrification rates and NO_3 contents of the soil and hence denitrification rates versus field N surpluses for this site were considered curvi-linear, with a linear phase at relatively low N surpluses and approaching a maximum rate at relatively high N surpluses. 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 as discussed in Chapter 2 (Figure 2.7). For the experimental field in the Vlietpolder, the partitioning of the N surplus over leaching and denitrification as a function of N surplus can then be visualized as in Figure 6.2.

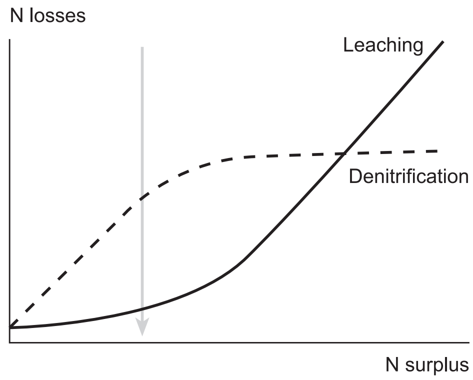


Figure 6.2. Conceptual partitioning of the N surplus (corrected for NH_3 volatilization) over denitrification and leaching for the experimental field of the Vlietpolder. Arrow indicates the average situation of the experimental field in the Vlietpolder based on the partitioning between denitrification and leaching (Table 6.3).

Figure 6.2 indicates that at relatively low N surpluses (corrected for NH_3 volatilization) denitrification is the major N loss pathway and is most strongly related to N surplus. At increasing N surpluses the relation between N surplus and leaching becomes stronger and eventually denitrification reaches its maximum and leaching becomes proportional to the additional N surplus.

For the experimental field the ratio between N losses through denitrification and leaching was about 4:1 (Table 6.3) and consequently the situation of the experimental field was located in the left part of Figure 6.2 (indicated by arrow). This situation is characterized by the limited response of leaching to changes in N surpluses and by the strong, nearly linear, response of denitrification to changes in N surpluses. The partitioning of N surplus over denitrification and leaching is soil type specific. For other soils the axes rescale, but the outline (i.e. the processes behind the partitioning) remain equal. Figure 6.2 demonstrates that for the Vlietpolder achieving considerable reductions in N leaching to surface water demands for large efforts, compared to situations with a lower denitrification capacity.

Although Figure 6.2 is conceptual and does not take into account response times and fast nutrient loss processes like run-off, it may provide insight in expected responses to changes in N surpluses, once the shape of the curves is known. For instance, the silt loam soil presented in Figure 6.1 was characterized by relatively high leaching losses, suggesting that for this soil the steepness of the curves is much lower for denitrification and much higher for leaching than presented in Figure 6.2. Therefore, for the silt-loam soil a higher effectiveness and efficiency in improving surface water quality might be achieved under similar reductions of N inputs, compared to the Vlietpolder.

The concept presented in Figure 6.2 may be used also to illustrate the partitioning of the P surplus over leaching and sorption to the soil. Sorption of P to soil has, like denitrification, a maximum that is related to the availability of sorption sites (Del Campillo et al., 1999). In the Vlietpolder there were still ample sorption sites for P (Van Beek et al., 2003b) and it is reasonable to assume that reduction of P inputs will predominantly result in less P sorption under the current biogeochemical conditions. However, it should be emphasized that, in contrast to denitrified N that is lost from the system forever, sorbed P may be released again when biogeochemical conditions change or when the soil is subjected to mining soil P, as was demonstrated for sandy soils by Koopmans (2004).

Effectiveness of measures to reduce nutrient leaching to surface water

One of the main measures taken at this moment to reduce N and P loading of surface water in agricultural areas is reducing inputs of N and P at farm-level. In 2006, application rates of 'effective' N and P to grassland on peat soil were limited at 290 and 24 kg $ha^{-1}y^{-1}$, respectively. The maximum application of animal manure N, including N in the droppings of grazing animals, was 250 kg N $ha^{-1}y^{-1}$ (LNV, 2006). These limits aim at restraining N and P losses to surface waters, groundwater and atmosphere. However, the effectiveness of decreasing nutrient loads of surface water by reducing the nutrient inputs depends on several factors:

1. Nutrient inputs are evenly distributed among fields within farms.
An uneven distribution of nutrient inputs over fields within a farm may obscure established mean relationships between nutrient input and nutrient leaching at farm level. This follows from the non-linear relation between nutrient input and nutrient leaching as suggested in Figure 6.2. Hence, when the nutrient input at farm level is decreased, but the remaining input is concentrated on a few fields of the farm, the nutrient loading to surface water may exceed that of a farm with a higher nutrient input, but with a more even distribution over fields (Chapter 2).

2. Nutrient inputs are manageable.
Inputs of N and P via fertilizers, manure and cattle droppings are relatively well manageable and can be decreased by putting restrictions on inputs and/or grazing intensity. Inputs of N and P via nutrients supplied by the subsoil are hardly controllable, while nutrient inputs via soil mineralization have an intermediate position, as it can be managed to some extent, but only when hydrological measures are taken.

3. Relatively large fractions of the N and P inputs leach to the surface water.
The eventual effect of a reduction in N and P inputs on reducing N and P loads to surface water follows from the decrease in input (reduction fraction), leaching fraction and a non-linearity parameter. The reduction fraction is the fraction by which N and P inputs are reduced, the leaching fraction is the fraction of the N and P inputs that leach to the surface water and the non-linearity parameter indicates the non-linear relation between inputs to the soil and leaching to surface water (Chapter 2). Hence, at a given reduction of N and P inputs, higher leaching fractions coincide with a higher effectiveness of the measure in terms of loads to surface water.

4. Leaching is a main contributor of N and P in surface water.
When non-soil borne sources (e.g. inlet water, atmospheric deposition on surface water and sewage plants) govern the nutrient concentrations of surface water, reduction of N and P inputs to the soil will be little effective.

In this thesis abovementioned conditions were tested. There are, however, no quantitative criteria available and a qualitative evaluation was used.

1) Condition 1 was hardly met considering that intra-farm variability exceeded inter-farm variability always for N and most of the time for P. This means that

a farm with a low mean farm N surplus, may still have one or more fields with a higher N surplus than a farm with a high mean N surplus, but with a more homogeneous distribution (Chapter 2).

2) Condition 2 refers to the manageability of nutrient flows, i.e. reducing nutrient inputs or nutrient surpluses at field level in order to reduce nutrient leaching to surface water, implies that nutrient inputs or nutrient surpluses are manageable. Manageable nutrient inputs include fertilizer and manure applications, while soil mineralization, atmospheric deposition, and supply from the subsoil are hard to manage or unmanageable. Table 6.4 shows average total nutrient inputs for the experimental field in the Vlietpolder, subdivided in controllable and less-controllable inputs.

Table 6.4. Average N and P inputs to the experimental field of the Vlietpolder (1999 – 2002, data derived from Chapters 2, 3 and 4, kg ha^{-1}y^{-1}).

		N	*P*
Controllable	Fertilizers	184	7
	Manure (slurry)	148	24
	Cattle droppings	120	11
Less controllable	Atmospheric deposition	31	0
	Mineralization of organic matter in rooting zone	243	11
	Supply from subsoil	26	11
Total		752	64

Table 6.4 shows that about 40% of the total nutrient inputs was related to hard to manage or less controllable nutrient inputs. This finding demonstrates that there are only limited options for managing nutrient inputs to the soil at the present hydrological conditions.

3) The third condition refers to leaching as a main loss pathway of the N and P surplus at field level. In the Vlietpolder only 16% of the field N surplus and 18% of the P surplus were lost via leaching to surface water (Table 6.3). Hence, even in the linear part of Figure 6.2 a reduction of 10% in N and P inputs will only result in less than 2% reduction in leaching of N and P to surface water.

4) Condition 4 refers to nutrient inputs directly on the surface water, i.e. without interaction with the soil. In the Vlietpolder there were neither sewage treatment plants nor domestic sewage disposals, but inlet water accounted for about 4% of the N loading and for about 5% of the P loading of surface water. Atmospheric deposition on surface water accounted for about 8% of the N loading of surface water. Although these numbers are small, the contribution of inlet water will be higher in summer and was estimated to increase to 10-20% of the N loading of surface water and to 30-35% of the P loading of surface water (Van den Eertwegh and Van Beek, 2004).

The analysis of the 4 conditions for effective reduction of nutrient loading by decreasing N and P inputs at field and farm level show that apparently none of the conditions was entirely met. Notwithstanding the qualitative nature of this analysis, it may be concluded that the scope of reducing N and P loadings of surface water by decreasing N and P inputs at farm level is limited for the Vlietpolder.

At a national level also limited effects of reducing N and P inputs at farm level on decreasing N and P concentrations in surface water were suggested in a modeling study by Oenema et al. (2005). They attributed this limited response to i) spatial differences between sites with a relatively large input reduction and sites which greatly contribute to surface water loading, ii) contribution of nutrient rich seepage and iii) continued release of P from P-rich soils. The explanations given by Oenema et al. (2005) are related to conditions 2 and 4 mentioned above, but defined at a different spatial scale. Hence, at different scales, i.e. the field scale (this study) and at national scale (Oenema et al., 2005) there are mechanisms preventing high effectiveness of decreasing N and P inputs as a measure to reduce N and P loading of surface water.

Final remarks

In the near future, developments in Dutch peat pasture polders will concentrate on decreasing agricultural nutrient inputs and on raising groundwater water levels (LNV, 2003). Raising groundwater levels as measure to decrease mineralization rates and thereby the rate of subsidence, appears to be an attractive measure also for decreasing N and P loading of surface water via subsoil supply (Chapter 5) and may benefit ecological values as well (Best et al., 1993). However, it may also increase total leaching rates including surface run-off. Most peat land polders (including the Vlietpolder) have been loaded with P for decades, resulting in a P stock of more than 3000 kg ha^{-1} in the upper 75 cm. For these soils, raising the groundwater level may result in increased P leaching (Turner and Haygarth, 2001; Kalbitz et al., 1999), at least for some time. Also, increasing groundwater levels will affect biochemical equilibria as was described by Van Dijk (2006), but it is unclear how this will affect N and P losses to surface water.

Also, the discussion about target N and P concentrations in surface water in different water types, including those of managed peat soils is still ongoing (CIW, 2002). In Chapter 1 the course of N and Cl concentrations in surface water over time was shown since 1970. It was indicated that major improvements had been achieved, but that improvements had leveled off and even worsened since 1998. However, comparing current data with those of 1941, hence before the strong increase in fertilizer application, shows that during the last 60 years Cl concentrations decreased while ortho-P concentrations increased. For NO_3-N and NH_4-N the results were somewhat scattered, but these data demonstrate that nutrient loading of surface water in peat land areas is not restricted to our present day (Table 6.5).

Table 6.5. Solute concentrations (mg L^{-1}) in surface water in 1941 (De Gruyter and Molt, 1952) and at present. Data of De Gruyter and Molt (1952) originated from Polder Oudendijk (a reclaimed, deep polder adjacent to the Vlietpolder) and from a shallow polder (like the Vlietpolder). Samples were taken in March and for reasons of seasonal fluctuations were compared with samples in the Vlietpolder taken in the same month

	1941	*1941*	*2001-2003*
	Polder Oudendijk	*Polder Boskoop*	*Vlietpolder ($n \geq 6$)*
pH	7.8	7.8	8.0 ± 0.50
Cl	123	368	64.5 ± 23.2
NO_3-N	3.8	< detection limit	0.32 ± 0.21
NH_4-N	2.6	0.23	1.13 ± 1.38
PO_4-P	0.16	0.20	0.31 ± 0.23

Although the data of Table 6.5 may seem to suggest that current N and P concentrations in surface water about equal N and P concentrations in surface water in pre-fertilizer times, the data are not entirely comparable because of differences in analyzing procedures, hydrological conditions and the presence of domestic sewage discharges.

The Vlietpolder has the typical characteristics of managed coastal peat polders in the Western part of the Netherlands (Van der Grift, 2003). Therefore, the results of the present thesis give -to my view- a valuable contribution to the understanding of the N and P dynamics in managed peat soils in the western part of the Netherlands. Apart from the conclusions mentioned at the beginning of this chapter, the integration of all chapters allows for drawing the following additional conclusions:

- The effectiveness of measures to improve surface water quality by reducing farm inputs is limited, because the conditions underlying these measures were poorly met. In Van den Eertwegh and Van Beek (2004) the maximum possible reduction of N and P loads to surface water was estimated at 15%, but with high uncertainties and efforts.
- Reduction of nutrient inputs presumably mainly results in decreased denitrification rates (N) and decreased accumulation of P in the topsoil.
- Nutrient flows through the experimental field were intense, but the different flows were generally consistent throughout this thesis.
- High nutrient concentrations in surface water are not restricted to our present time, but occurred already in pre-fertilizer times in the study area.

There are major scientific challenges remaining concerning the management of the western peat land area in The Netherlands. Some of these challenges were already mentioned while discussing the quantification of N and P pathways. Others involve the management of nutrient flows in this area, of which I consider the following items still open-ending:

- Figure 6.2 of this chapter suggests that different loss pathways respond differently to changes in nutrient inputs. However, the exact shapes of the relations require (experimental) verification. Knowing the sensitivity of a loss pathway to changes in N and P inputs allows for more accurate estimation of the effectiveness of measures beforehand.

- The suggested large contribution of nutrient rich subsoil layers to the N and P loading of surface water needs further experimental verification as well. Furthermore, it is still unknown how this source of nutrients can be managed in the long term.
- So far, the ditch dredging and the application of slush to the fields is considered to contribute to decreasing the nutrient concentration in surface waters, because of the re-use of nutrients. However, the data collected in this thesis suggest that application of slush to agricultural land may also contribute to increasing sulfate concentrations in the soil solution and to subsequent changes in soil pH and P chemistry. To my knowledge, these aspects have not been taken into consideration in the evaluation of the consequences of slush application.
- The results of the present study and other field studies like those of Van der Salm et al. (2007) need to be upscaled to the regional level to be able to assess its full impact. Detailed simulation modelling generally does not allow for upscaling to regional levels and often lacks the transparency needed for policy making. Simple decision support systems (DSS) like that of Janssen et al. (2005) seem to be preferred. With a DSS the effects of different scenarios can be explored on surface water quality, but also on the prospects for dairy farming, recreation and infra-structure. These scenarios should focus on spatial planning, water management and opportunities for agriculture. To my view the opportunities for agriculture should be put in a more fundamental discussion: Do we really want intensive dairy farming in the low-lying peat polders? Or should we give the low-lying peat polder multifunctional purposes, including nature management, flood water storage and extensive agriculture? Or would we rather have a landscape controlled by (subsidized) farmers? In this respect Van der Ploeg (2001) mentions the possibilities of 'wider' farm developments e.g. on-farm cheese making, on-farm tourism, and farm excursions. A process-based DSS can demonstrate possible consequences of certain decisions, showing all pro's and con's and raising the awareness that no measure in the Western peat land district goes without detrimental consequences for one or more aspects.

References

Aarts H.F.M., Habekotté B. and Van Keulen H. 2000. Nitrogen (N) management in the "De Marke" dairy farming system. Nutr Cycl Agroecosyst 56:231-240.

Abbasi M.K. and Adams W.A. 2000. Gaseous N emission during simultaneous nitrification-denitrification associated with mineral N ferilization to a grassland soil under field condition. Soil Biol Biochem 32:1251-1259.

Abdirashid A.E., Madramootoo C., Egeh M., Liu A. and Hamel C. 2002. Environmental and agronomic implications of water table and nitrogen fertilization management. J Env Qual 31:1858-1867.

Altman S.J. and Parizek R.R. 1995. Dilution of nonpoint-source nitrate in ground water. J Env Qual 24:707-718.

Armstrong A.C., Rands J.G. and Castle D.A. 1988. Drainage benefits: watertable control, workability and crop yields. Agric Water Man 14:43-52.

Aulakh M., Doran J. and Mosier A. 1992. Soil denitrification - significance, measurement, and effects of management. Adv Soil Sci 18:1-57.

Aulakh M.S., Rennie D.A. and Paul E.A. 1981. Gasous nitrogen losses from cropped and summer-fallowed soils. Can J Soil Sci 62:187-196.

Bacon S., Lanyon L. and Schlauder R.J. 1990. Plant nutrient flow in the managed pathways of an intensive dairy farm. Agron J 82:755-761.

Barraclough D., Jarvis S.C., Davies G.P. and Williams J. 1992. The relation between fertilizer nitrogen applications and nitrate leaching from grazed grassland. Soil Use Man 8(2):51-56.

Barton L., McLay C.D.A., Schipper L.A. and Smith C.T. 1999. Annual denitrification rates in agricultural and forest soils: a review. Austr J Soil Res 37:1073-1093.

Bechmann M., Eggestad H. and Vagstad N. 1998. Nitrogen balances and leaching in four agricultural catchments in southeastern Norway. Environ Poll 102:493-499.

Beembroek J.A., Koole B., Poppe K.J. and Wossink G.A.A. 1996. Environmental farm accounting: the case of the Dutch nutrients accounting system. Agric Syst 51:29-40.

Berendse F., Oomes M.J.M., Altena H.J. and De Visser W. 1994. A comparative study of nitrogen flows in two similar meadows affected by different groundwater levels. J Appl Ecol 31:40-48.

Best E.P.H. and Jacobs F.H.H. 2001. Production, nutrient availability, and elemental balances of two meadows affected by different fertilization and water table regimes in The Netherlands. Plant Ecol 155:61-73.

Best E.P.H., Verhoeven J.T.A. and Wolff W.J. 1993. The ecology of The Netherlands wetlands - characteristics, threats, prospects and perspectives for ecological research. Hydrobiologia 265:305-320.

Best E.P.H. and Jacobs F.H.H. 1997. The influence of raised water table levels on carbon dioxide and methane production in ditch-dissected peat grasslands in The Netherlands. Ecol Engin 8:129-144.

Bijay-Singh R.J.C., Ryden J.C. and Whitehead D.C. 1989. Denitrification potential and actiual rates of denitrification in soils under long-term grassland and arable cropping. Soil Biol Biochem 21:897-901.

Bollmann A. and Conrad R. 1997. Acetylene blockage technique leads to understimation of denitrification rates in oxic soils due to scavenging of intermediate nitric oxide. Soil Biol Biochem 29(7):1067-1077.

Boschloo D.J. and Stolk A.P. 1999. National Monitoring Program Rainwater Composition. RIVM report 733101054. Bilthoven, The Netherlands.

Breembroek J.A., Koole B., Poppe K.J. and Wossink G.A.A. 1996. Environmental Farm Accounting: the case of the Dutch Nutrients Accounting System. Agric Syst 51: 29-40.

Brouwer F. 1998. Nitrogen balances at farm level as a tool to monitor effects of agri-environmental policy. Nutr Cycl Agroecosys 52:303-308.

Bussink D.W. 1992. Ammonia volatilization from grassland receiving nitrogen fertilizer and rotationally grazed by dairy cattle. Fert Res 33:257-265.

Bussink D.W. 1996a. Ammonia volatilization from intensively managed dairy pastures. PhD thesis Wageningen Agricultural University.

Bussink D.W. 1996b. Relationships between ammonia volatilization and nitrogen fertilizer application rate, intake and excretion of herbage nitrogen on grazed swards. Fert Res 38:11-121.

CBS/LEI-DLO 2001. Land- en Tuinbouwcijfers 2001. Agricultural and Horticultural Data 2001.

Chan P.S., West J.W., Bernard J.K. and Fernandez J.M. 2005. Effects of dietary cation-anion difference on intake, milk yield, and blood components of the early lactating cow. J Dairy Sci. 88:4384-4392.

Chan Y. and Knowles R. 1979. Measurement of denitrification in two freshwater sediments by an in situ acetylene inhibition method. Appl Environ Microbiol 37:1067-1072.

Christensen L.B. and Olesen S.E. 1986. Leaching of ferrous iron after drainage of pyrite-rich soils and means of preventing pollution of streams. Paper presented at the Agricultural water management: proceedings of a Symposium on Agricultural Management. pp. 279-289. Balkema.

Christensen P. and Sorensen J. 1988. Denitrification in sediment of lowland streams: regional and seasonal variation in Gelbaek and Rabis Baek, Denmark. FEMS Microbiol Ecol 53:335-344.

CIW (Commissie Integraal Waterbeheer). 2002. Gedifferentieerde normstelling voor nutriënten in oppervlaktewateren. www.rws.nl.

Clément J.C., Pinay G. and Marmonier P. 2002. Seasonal dynamics of denitrification along topohydrosequences in three different riparian wetlands. J Env Qual 31:1025-1037.

Corwin D.L. 2002. Suction cups. In: J.H. Dane and G.C. Topp (Eds), Methods of Soil Analysis. Part 4: Physical Analysis. SSSA, Madison, pp. 1261-1266.

Crutzen P.J. 1970. The influence of nitrogen oxides on the atmospheric ozone content. Quat J Royal Meteorol Soc 96:320-325.

CVB 2000. Tabellenboek veevoeding 2000.

Dalley D., Roche J., Grainger C. and Moate P. 1999. Dry matter intake, nutrient selection and milk production of dairy cows grazing rainfed perennial pastures at different herbage allowances in spring. Aust J Exp Agric 39:923-931.

Davis J.C. 1986. Statistics and Data Analysis in Geology. John Wiley & Sons, Toronto, 646 pp.

De Boer, T.A. 1982. The use of peat soils for grassland. Paper presented at the symposium on peat lands below sea level. pp. 214-221. ILRI.

De Gruyter P. and Molt E.L. 1952. Rijnlands Boezem. Een technisch-hygienische studie over de beteekenis en het beheer van den boezem, de waterhuishouding van het boezemgebied, het gebruik en de hoedanigheid van het boezemwater. Hoogheemraadschap van Rijnland.

De Klein C.A.M. and Logtestijn R.S.P. 1994. Denitrification in the top soil of managed grasslands in The Netherlands in relation to soil type and fertiliser level. Plant Soil 163:33-44.

De Klein C.A.M., Harrison R. and Lord E.I. 1996. Comparison of N_2O flux measurements using gas chromatography and photo-acoustic infra-red spectroscopy. Progress in Nitrogen Cycling Studies, 533-536.

De Klein J. and Gillissen F. 2003. Measurement of denitrification in a peat ditch in the Vlietpolder. Wageningen University report M327. Wageningen, The Netherlands.

Del Campillo M.C., Van der Zee S.E.A.T.M. and Torrent J. 1999. Modelling long-term phosphorus leaching and changes in phosphorus fertility in excessively fertilized acid sandy soils. Europ J Soil Sci 50:391-399.

De Vries W., Kros H. and Oenema O. 2001. Modeled impacts of farming practices and structural agricultrual changes on nitrogen fluxes in the Netherlands. The Scientific World 1(s2):664-672.

De Vries W., Kros H., Oenema O. and de Klein J. 2003. Uncertainties in the fate of nitrogen II: A quantitative assessment of the uncertainties in major fluces in the Netherlands. Nutri Cycl Agroecosyst 66:71-102.

De Vries F. and Brouwer F. 2005. Veengronden in Nederland. In Rienks W. and Gerritsen A. (eds). Veenweide 25x belicht. Alterra speciale uitgaven 2005/11. Wageningen, The Netherlands.

Drexler J.Z. and Bedford B.L. 2002. Pathways of nutrient loading and impacts on plant diversity in a New York peatland. Wetlands 22:263-281.

Droogers P., van Dam J.C. and Loeve R. 2005. Decreasing leaching of peat water; a 2D analysis. FutureWater, Wageningen, The Netherlands.

EUROSTAT (Statistical office of the European Union) 1997. Soil surface nitrogen balances in EU countries. Document Agrienv/97/4.

FAO-Unesco 1988. Soil Map of the World. Revised Legend. World Soil Resources report 60. Rome.

Fraser C.J.D., Roulet N.T., and Lafleur M. 2001. Groundwater flow patterns in a large peatland. J of Hydr 246:142-154.

Fraters B., Boumans L., Van Leeuwen T. and De Hoop W. 2002. Monitoring nitrogen and phosphorus in shallow groundwater and ditch water on farms in peat regions of The Netherlands, 6th International Conference on Diffuse Pollution, pp. 575-576.

Fried M, Tanji K.K. and Van der Pol R.M. 1976. Simplified long term concept for evaluating leaching of nitrogen from agricultural land. J Env Qual 5(2):197-200.

Genstat Committee 2000. GenStat release 4.2 reference manual. Oxford University Press, Oxford.

Giebels L. 2002. Hollands Water: Het Hoogheemraadschap van Rijnland na 1857. ISBN 9053451994. Matrijs, The Netherlands.

Giller K. and Wheeler B. 1986. Peat and peat water chemistry of a flood-plain fen in Broadland, Norfolk, U.K. Freshwater Biology 16:99-114.

Goodchild R. 1998. EU policies for the reduction of nitrogen in water: the example of the Nitrates Directive. Environ Poll 102:737-740.

Goudriaan J. and Monteith J.L. 1990. A mathematical function for crop growth based on light interception and leaf area expansion. Ann Bot 66:695-701.

Graf Pannatier E., Broers H.P., Venema P. and Van Beusekom G. 2000. A new process-based hydro-geochemical classification of groundwater, application to the Netherlands national monitoring system. TNO-report NITG 00-143-B. Utrecht, The Netherlands.

Grieve I.C. and Gilvear D.J. 1994. Quantifying anthropogenic nutrient sources and loadings within a small catchment with conservation interests, Eastern Scotland. Aquatic Conservation-Marine and Freshwater Ecosyst 4:273-287.

Groeneveld R., Bouwman A.F., Kruitwagen S. and Van Ierland E.C. 1998. Nitrate leaching in dairy farming: economic effect of environmental restrictions. Environ Poll 102:755-761.

Grundmann G.L., Renault P., Rosso L. and Bardin R. 1995. Differential effects of soil water content and temperature on nitrification and aeration. Soil Sci Soc Am J 59:1342-1349.

Heathwaite A.L. 1990. The effect of drainage on nutrient release from fen peat and its implication for water quality - a laboratory simulation. Water, Air Soil Poll 49:159-173.

Heathwaite A.L. 1991. Solute transfer from drained fen peat. Water, Air Soil Poll 55:379-395.

Hénault C. and Germon J.C. 2000. NEMIS, a predictive model of denitrification on the field scale. Europ J Soil Sci 51:257-270.

Hendriks R.F.A. 1993. Nutriëntenbelasting van het oppervlaktewater in veenweidegebieden. 164 p. DLO-Staring Centrum report 251. Wageningen, The Netherlands.

Hendriks R.F.A., Walvoort D.J.J. and Jeuken M.H.J.L. 2007. Evaluation of the applicability of the SWAP-ANIMO model for simulating nutrient loading of surface water in a peat land area Calibration, validation, and system and scenario analysis for a study area in the Vlietpolder. Alterra report 619. Wageningen, The Netherlands.

Hendriks R.F.A., Kruijne R., Roelsma J., Oostindie K., Oosterom H.P and Schoumans O.F. 2002. Berekening van de nutriëntenbelasting van het oppervlaktewater vanuit landbouwgronden in vier poldergebieden. Analyse van de bronnen. Alterra report 408. Wageningen, The Netherlands.

Henkens C.H. and Koopmans J.J. 1973. Tracing and treating mineral disorders in dairy cattle: prep. by the Committee on mineral nutrition. PUDOC. Wageningen, The Netherlands.

Henkens P.L.C.M. and van Keulen H. 2001. Mineral policy in The Netherlands and nitrate policy within the European Community. Neth J Agric Sci 49:117-134.

Higler L.W.G. 1989. Hydrobiological research in peat polder ditches. Hydrobiol Bull 23(2):105-109.

Hoekstra J.R., Bos L. and De Ruiter H.R.G. 2005. Fean-Wetter-Buorkje – Experiments with higher summer surface water levels in the Frisian peat land area. www.clm.nl.

Holmer M. and Storkholm P. 2001. Sulphate reduction and sulphur cycling in lake sediments: a review. Freshwater Biol 46(4):431-451.

Houba V.J.G., Van der Lee J. and Novozamski I. 1995. Soil Analysis Procedures. Wageningen University and Research Center report 06175208. Wageningen, The Netherlands.

Houghton J.T., Meiro Filho L.G., Callander B.A., Harris A., Kattenbert A. and Maskell K. 1996. Climate Change 1995. Cambridge University press.

ISO (International Organization for Standardization) 1984. Water quality. Determination of Kjeldahl nitrogen; method after mineralization with selenium. ISO 5663.

Janssen, R., Goosen H., Verhoeven M.L., Verhoeven J.T.A., Omtzigt A.Q.A. and Maltby E., 2005. Decision support for integrated wetland management. Env Mod Softw 20:215-229.

Jarvis S.C., Wilkins R.J. and Pain B.F. 1996. Opportunities for reducing the environmental impact of dairy farming management: a system approach. Grass Forage Sci. 51:21-31.

Joosten L.T.A., Buijze S.T. and Jansen D.M. 1998. Nitrate in sources of drinking water? Dutch drinking water companies aim at prevention. Environ Poll 102:487-492.

Jörgensen R.G. and Richter G.M. 1992. Composition of carbon fractions and potential denitrification in drained peat soils. J Soil Sci 43:347-358.

Kalbitz K., Rupp H., Meissner R. and Braumann F. 1999. Effects of fen restoration on nitrogen, phosphorus, and carbon in soil solution and groundwater. Zeitschrift für Kulturtechnik und Landentwicklung. 40:22-28.

Kirkham F. and Wilkins R. 1993. Seasonal fluctuations in the mineral nitrogen content of an undrained wetland peat soil following differing rates of fertiliser nitrogen application. Agric Ecosyst Environ 43:11-29.

Klapwijk S.P. 1988. Biological assessment of the water-quality in South-Holland (The Netherlands). Internationale revue der gesamten hydrobiologie 73:481-509.

Kliewer B.A. and Gilliam J.W. 1995. Water table management effects on denitrification and nitrous oxide evolution. Soil Sci Soc Am J 59:1694-1701.

KNMI (Royal Dutch Meteorological Insitute) 2004. www.knmi.nl.

Knowles R. 1990. Acetylene inhibition technique: development, advantages, and potential problems. Denitrification in soil and sediment. Plenum press. New York.

Kohn R.A., Dou Z., Ferguson J.D. and Boston R.C. 1997. A sensitivity analysis of nitrogen losses from dairy farms. J Environ Manag 50:417-428.

Koopmans G.F. 2004. Characterization, desorption, and mining of phosphorus in noncalcareous sandy soils. PhD thesis Wageningen University.

Koops J.G., Oenema O. and van Beusichem M.L. 1996. Denitrification in the top- and subsoil of grassland on peat soils. Plant Soil 1154:1-10.

Koops J.G., van Beusichem M.L. and Oenema O. 1997. Nitrogen loss from grassland on peat soils through nitrous oxide production. Plant Soil 188:119-130.

Kroeze C., Aerts R., Van Breemen N., Van Dam D., Hofschreuder P., Hoosbeek M., De Klein J., Van der Hoek K., Kros H., Van Oene H., Oenema O., Tietema A., Van der Veeren R., and De Vries W. 2004. Uncertainties in the fate of nitrogen I: An overview of sources of uncertainty illustrated with a Dutch case study. Nutr Cycl Agroecosyst 66:43-69.

Kronvang B., Grant R., Larsen S.E, Svendsen L.M. and Kristensen P. 1995. Non-point-source nutrient losses to the aquatic environment in Denmark: impact of agriculture. Marine Freshw Res 46(1):167-177.

Lamers L.P.M., Tomassen H.B.M. and Roelofs J.G.M. 1998. Sulfate-induced eutrophication and phytotoxicity in freshwater wetlands. Environ Sci Techn 32:199-205.

Ledgard S.F., Penno J.W. and Sprosen M.S. 1999. Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertilizer application. J Agric Sci 132:215-225.

LNV 2003. Structuurschema groene ruimte 2. http://www.minlnv.nl/thema/groen/ruimte/.

LNV 2006. Mestbeleid 2006: tabellen. Dienst regelingen.

Lowrance R.R. 1992. Groundwater nitrate and denitrification in a coastal plain riparian forest. J Env Qual 21:401-405.

Luo J., Tillman R.W., White R.E. and Ball P.R. 1998. Variation in denitrification activity with soil depth under pasture. Soil Biol Biochem 30:897-903.

Martin T.L., Kaushik N.K., Trevors J.T. and Whiteley H.R. 1999. Review: denitrification in temperate climate riparian zones. Water Air Soil Poll 111:171-186.

McLaughlin K., Cade-Menum B.J, and Paytan A. 2006. The oxygen isotopic composition of phosphate in Elkhorn Slough, California: A trace for phosphate sources. Estuarine Coastal Shelf Sci 70:499-506.

Meinardi C.R., 2005. Transport of water and solutes from soil to surface water in the Vlietpolder. RIVM report 500003004/2005. Bilthoven, The Netherlands.

Mengis M., Schiff S.L., Harris M., English M.C., Aravena R., Elgood R.J. and MacLean A. 1999. Multiple geochemical and isotopic approaches for assessing ground water NO_3 elimination in a riparian zone. Ground Water 37:448-457.

Michielsen B.F. and van Schaik F.H. 2004. Stofstromen van en naar het oppervlaktewater in de Vlietpolder. Deelrapport Veenweide Project fase I. HHRS Rijnland.

Misselbrook T.H., van der Weerden T.J., Pain B.F., Jarvis S.C., Chambers B.J., Smith K.A., Phillips V.R. and Demmers T.G.M. 2000. Ammonia emission factors for UK agriculture. Atmos Environ 34:871-880.

Moraghan J.Y. and Buresh R. 1977. Correction for dissolved nitrous oxide in nitrogen studies. Soil Sci Soc Am J 41:1201-1202.

Mosier A.R. and Klemedtsson L. 1994. Measuring denitrification in the field. In: Weaver R.W., Angle J.S. and Bottomley P.S. (eds) Methods of Soil Analysis, vol 2 - Microbiological and biological properties. SSSA, Madison, pp 1047-1065.

Mulvaney R.L. 1996. Nitrogen – inorganic forms. Methods of Soil Analysis, part 3 – Chemical Methods. SSSA, Madison, pp 1123-1184.

Murray T.E. 1995. The correlation between iron sulfide precipitation and hypolimnetic phosphorus accumulation during one summer in a softwater lake. Canad J Fish Aquat Sci 52(6):1190-1194.

Neeteson J. 2000. Nitrogen and phosphorus management on Dutch dairy farms: legislation and strategies to meet the regulations. Biol Fertil Soils 30:566-572.

Nilsson T. and Lundin L. 1996. Effects of drainage and wood ash fertilization on water chemistry at a cutover peatland. Hydrobiologia 335:3-18.

OECD (Organisation for Economic Co-operation and Development) 2001. Environmental indicators for agriculture. Volume 3 Methods and Results. Paris, France

Oenema O. and Heinen M. 1999. Uncertainties in nutrient budgets due to biases and errors. In: Smaling E.M.A., Oenema O. and Fresco L.O. (eds). Nutrient Disequilibria in Agroecosystems. CABI Publishing, Wallingford, United Kingdom.

Oenema O. and Roest C.J.W. 1998. Nitrogen and phosphorus losses from agriculture into surface waters; the effects of policies and measures in The Netherlands. Wat Sci Technol 37:19-30.

Oenema O., Van Liere L. and Schoumans O.F. 2005. Effects of lowering nitrogen and phosphorus surpluses in agriculture on the quality of groundwater and surface water in the Netherlands. J Hydr 304:289-302.

Oenema O., Velthof G.L., Verdoes N., Groot Koerkamp P.W.G, Bannink A., Monteny G.J., Van der Meer H.G. and Van de Hoek K. 2000. Forfaitaire waarden voor gasvormige stikstofverliezen uit stallen en mestopslagen. Alterra report 107. Wageningen, The Netherlands.

Olsthoorn C.S.M. and Fong N.P.K. 1998. The anthropogenic nitrogen cycle in The Netherlands. Nutri Cycl Agroecosyst 52:269-276.

Ondersteijn C.J.M., Beldman A.C.G., Daatselaar C.H.G., Giesen G.W.J. and Huirne R.B.M. 2003. Farm structure or farm management: effective ways to reduce nutrient surpluses on dairy farms and their financial impacts. Livest Prod Sci 84(2):171-181.

Ondersteijn C.J.M., Beldman A.C.G., Daatselaar C.H.G., Giesen G.W.J. and Huirne R.B.M. 2002. The Dutch mineral accounting system and the European Nitrate directive: implications for N and P management and farm performance. Agric Ecosyst Environ 92(2-3):283-296.

Parris K. 1998. Agricultural nutrient balances as agri-environmental indicators: an OECD perspective. Environ Poll 102(S1):219-225.

Peverly J.H. 1982. Stream transport of nutrients through a wetland. J Env Qual 11(1):38-42.

Pieterse N.M, Bleuten W. and Jørgensen S.E. 2003. Contribution of point sources and diffuse sources to nitrogen and phosphorus loads in lowland river tributaries. J Hydr 271:213-225.

Pinay G., Rogues L. and Fabre A. 1993. Spatial and temporal patterns of denitrification in a riparian forest. J Appl Ecol 30:581-591.

Plette A.C.C., Verstappen G.G.C. and Boers P.C.M. 2002. Mest en oppervlaktewater; Een terugblik 1985-2000. RIZA report 2002.019. Lelystad, The Netherlands.

Pons L.J. 1992. Holocene peat formation in the lower parts of the Netherlands. In: J.T.A. Verhoeven (ed). Fens and Bogs in the Netherlands: Vegetation, History, Nutrient Dynamics and Conservation, 7-79. Kluwer Academic Publishers, The Netherlands.

Poot A. and Schot P. 2000. Rainwater lenses; shape and dynamics. Stromingen 6(4):13-26.

Portielje R. and van der Molen D. 1998. Trend-analysis of eutrophication variables in lakes in the Netherlands. Wat Sci Techn 37:235-240.

Pratt P.F. 1978. Leaching of cations and chloride from manure applied to an irrigated soil. J Env Qual 7:513-516.

Prins W.A. 1983. Effect of a wide-range of nitrogen applications on herbage nitrate content in long-term fertilizer trials on all-grass swards. Fert Res 4:101-113.

Reijneveld J.A., Habekotté B., Aarts H.F.M. and Oenema J. 2000. 'Typical Dutch'; View on the diversity of the Dutch dairy farming. Plant Research International report 8. Wageningen, The Netherlands.

RIVM 2002. MINAS en Milieu (MINAS and the environment) RIVM report 718201005. 205 pp. Bilthoven, The Netherlands.

Roelofs J.G.M. 1991. Inlet of alkaline river water into peaty lowlands - effects on water-quality and Stratiotes-Aloides L stands. Aquatic Botany 39(3-4):267-293.

Rougoor C.W., Dijkhuizen A.A., Huirne R.B.M., Mandersloot F. and Schukken Y.H. 1997. Relationships between technical, economic and environmental results on dairy farms: an explanatory study. Livest Prod Sci 47:235-344.

Rudolph J., Frenzel P. and Pfennig N. 1991. Acetylene inhibition technique underestimates in situ denitrification rates in intact cores of freshwater sediments. FEMS Microbiol Ecol 85:101-106.

Ryden J. and Dawson K. 1982. Evaluation of the acetylene-inhibition technique for the measurement of denitrification in grassland soils. J Sci Food Agric 33:1197-1206.

Schipper L.A., Cooper A.B., Harfoot C.G. and Dyck W.J. 1993. Regulators of denitrification in an organic riparian soil. Soil Biol Biochem 25:925-933.

Schnabel R.R., Cornish L.F., Stout W.L. and Shaffer J.A. 1996. Denitrification in a grassed and a wooded, valley and ridge, riparian ecotone. J Env Qual 25:1230-1235.

Schothorst C.J. 1977. Subsidence of low moor peat soils in western Netherlands. Geoderma 17:265-291.

Schothorst C.J. 1982. Drainage and behaviour of peat soils. ICW report 3. Wageningen, The Netherlands.
Schoumans O.F. and Groenendijk P. 2000. Modelling soil phosphorus levels and phosphorus leaching from agricultural land in The Netherlands. J Env Qual 29:111-116.
Schröder J.J., Aarts H.F.M., Ten Berge H.F.M., Van Keulen H. and Neeteson J. 2003. An evaluation of whole-farm nitrogen balances and related indices for efficient nitrogen use. Europ J Agron 20:33-44.
Seitzinger S.P. 1990. Denitrification in aquatic sediments. In: Revsbech N.P. and Sørensen J. (eds). Denitrification in soil and sediment. Plenum Press, New York.
Shotyk W. 1988. Review of the inorganic geochemistry of peats and peatland waters. Earth-Science Reviews 25:95-176.
Simon J.C., Girgnani C., Jacquet A., Le Corre L. and Pagès J. 2000 Typologie des bilans d'azote de divers type d'exploitation agricole: recherche d'indicateurs de fonctionnement. Agronomie 20:175-195
Simunek J., Sejna M. and Van Genuchten M. Th. 1999. The HYDRUS-2D software package for simulating two-dimensional movement of water, heat, and multiple solutes in variably saturated media. Version 2.0, IGWMC - TPS - 53, International Ground Water Modeling Center, Colorado School of Mines, Golden, Colorado, 251 pp.
Smith K.A. and Frost J.P. 2000. Nitrogen excretion by farm livestock with respect to land spreading requirements and controlling nitrogen losses to ground and surface waters. Part 1. Bioresour Technol 71:173-181.
Smith R.V., Jordan C. and Annett J.A. 2005. A phosphorus budget for Northern Ireland: inputs to inland and coastal waters. J Hydr 304:193-202.
Smith V., Tilman G. and Nekola J. 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestral ecosystems. Environ Poll 100:179-196.
Smolders A.J.P. and Roelofs J.G.M. 1995. Internal eutrophication, iron limitation and sulfide accumulation due to the inlet of river Rhine water in peaty shallow waters in The Netherlands. Archiv fur Hydrobiologie 133(3):349-365.
Snedecor G.W. and Cochran W.G. 1967. Statistical Methods, The Iowa State University Press, Ames, Iowa.
Steenvoorden J.H.A.M. 1983. Nitrate load of groundwater in sandy soils: denitrification in the subsoil. ICW report 1435. Wageningen, The Netherlands.
Steur G.G.L, De Vries F and Van Wallenburg C. 1985. De Bodemkaart van Nederland 1 : 250 000. Stichting voor Bodemkartering. Wageningen, The Netherlands.
Stevens R.J. and Laughlin R.J. 1998. Measurement of nitrous oxide and di-nitrogen emissions from agricultural soils. Nutr Cycl Agroecosyst 52:131-139.
Stevens R.J., Laughlin R.J., Burns L.C., Arah R.M., Hood R.C. 1997. Measuring the contributions of nitrification and denitrification to the flux of nitrous oxide from soil. Soil Biol Biochem 29:139-151.
Stumm W. and Morgan J.J. 1996. Aquatic Chemistry, 3 edn. John Wiley & Sons.

Tamminga S., Jongbloed A., Van Eerdt M.M., Aarts H.F.M., Mandersloot F., Hoogervorst N. and Westhoek H. 2000. The fixed rate excretion of nitrogen by livestock. report ID-Lelystad, The Netherlands.

Tilman D., Fargione J., Wolff B., d'Antonio C., Dobson A., Howarth R., Schindler D., Schlesinger W., Simberloff D. and Swackhamer D. 2001. Forecasting agriculturally driven global environmental change. Science 292: 281-284.

Turner B.L. and Haygarth P.M. 2001. Phosphorus solubilization in rewetted soils. Nature 411:258.

Van Beek C.L. and Oenema O. 2002. Nutriëntenbalansen in de Vlietpolder in het jaar 1999. Alterra report 482. Wageningen, The Netherlands.

Van Beek C.L., Van den Eertwegh G.A.P.H., Van Schaik F.H. and Van den Toorn A. 2003a. Surface runoff from intensively managed grassland on peat soils; a diffuse source of nitrogen and phosphorus in surface waters. Paper presented at the conference on diffuse input of chemicals into soil and groundwater, Dresden, Germany.

Van Beek C.L., Schoumans O.F. and Schuurmans W. 2003b. Fosfaatresorptie- en desorptiekarakteristieken van bodemmonsters. Deelrapport Veenweideproject fase I.

Van de Ven G.P. 1993. Man-made lowlands. Stichting Matrijs. Utrecht.

Van den Brandt H.M.P. and Smit H.P. 1998. Mineral accounting: the way to combat eutrophication and to achieve the drinking water objective. Environ Pollut 102(S1): 705-709.

Van den Eertwegh G.A.P.H. 1999. Projectplan veenweideproject. HHRS Rijnland.

Van den Eertwegh G.A.P.H. 2002. Water and nutrient budgets at field and regional scale. Travel times of drainage water and nutrient loads to surface water. PhD thesis Wageningen University.

Van den Eertwegh G.A.P.H. and Van Beek C.L., 2004. Veen, Water en Vee. Water- en nutriëntenhuishouding van een veenweidepolder. Eindrapport Veenweideproject fase I.

Van den Eertwegh G.A.P.H., Kroes J.G., Smit A. and Van Schaik F.H. 2003. Management of surface water levels using FIW Multi-SWAP. H_2O 20:28-31.

Van der Grift B. 2003. Samenstelling grondwater VlietpolderDeelrapport veenweideproject fase I.

Van der Meer H., van Middelkoop J. and De Visser W. 2004. Het stikstofleverend vermogen (NLV) van veengronden. In: Van Kekem (ed). Veengronden en stikstofleverend vermogen. Alterra report 965. Wageningen, The Netherlands.

Van der Ploeg B. 2001. Het Weigevoel in het Groene Hart van de Randstad. PhD thesis Wageningen University.

Van der Salm C., Dolfing J., Heinen M. and Velthof G.L. 2007. Estimation of nitrogen losses via denitrification from a heavy clay soil under grass. Agric Ecosys Environ 119:311-319.

Vander Zanden M.J, Vadeboncoeur Y., Diebel M.W. and Jeppesen E. 2005. Primary consumer stable nitrogen isotones as indicators of nutrient source. Environ Sci Techn 39:7509-7515.

Van Dijk J. 2006. The effects of increased groundwater levels on biogeochemical and ecological characteristics of degraded peat meadows. PhD thesis Vrije Universiteit Amsterdam.

Van Eerdt M.M. and Fong N.P.K. 1998. The monitoring of nitrogen surpluses from agriculture. Environ Poll 102 (S1):227-233.

Van Huet H.J.W.J. 1991. Phosphorus loads from peaty polders in the SW Frisian lake district, The Netherlands. Water Air Soil Poll 55:321-335.

Van Keulen H., Aarts H.F.M., Habekotté B., Van der Meer H.G. and Spiertz J.H.J. 2000. Soil-plant-animal relations in nutrient cycling: the case of dairy farming system 'De Marke'. Europ J Agron 13:245-261.

Van Keulen H., Van der Meer H.G. and De Boer I.J.M. 1996. Nutrient balances of livestock production systems in The Netherlands In: Groen A. and Van Bruchem J. (eds) Utilization of local feed resources by dairy cattle: perspectives of environmentally balanced production systems. Symposium proceedings Wageningen Institute of Animal Sciences (WIAS), Wageningen, The Netherlands.

Van Liere L., Janse J.H., Jeuken M., Van Puijenbroek P.J.T.M., Schoumans O.F., Hendriks R.F.A., Roelsma J. and Jonkers D.A. 2002. Effect of nutrient loading on surface water in polder Bergambacht, The Netherlands. In: Steenvoorden J.H.A.M., Claessen F. and Willems J. (eds), Agricultural effects on ground and surface waters: research at the edge of science and society. IAHS, Wageningen, pp. 213-218.

Van Noordwijk M. 1999. Nutrient cycling in ecosystems versus nutrient budgets of agricultural systems. In: Smaling E.M.A., Oenema O. and Fresco L.O. (eds). Nutrient Disequilibria in Agroecosystems. CABI Publishing, Wallingford, United Kingdom.

Van Schaik F.H. 2004. Kwaliteit vna het ondiepe grondwater, greppelwater en afspoeling. Deelrapport veenweideproject fase I.

Van Strien A.J., Van der Burg T., Rip W.J. and Strucker R.C.W. 1991. Effects of mechanical ditch management on the vegetation of ditch banks in Dutch peat areas. J Appl Ecol 28:501-513.

Velthof G.L. and Oenema O. 1995. Nitrous oxide fluxes from grassland in the Netherlands: II. Effects of soil type, nitrogen fertilizer application and grazing. Eur J Soil Sci 46:541-549.

Velthof G.L., Koops J.G., Duyzer J.H. and Oenema O. 1996. Prediction of nitrous oxide fluxes from managed grassland on peat soil using a simple empirical model. Neth J Agric Sc 44:339-356.

Vink I. and Wolbers H. 1997. Handboek Melkveehouderij. Praktijkonderzoek Rundvee, schapen en paarden.

Walter H.M., Keeney D.R. and Fillery I.R. 1979. Inhibition of nitrification by acetylene. Soil Sci Soc Am J 43:195-196.

Wang W., Lacis Y., Mo T. and Hansen J. 1976. Greenhouse effects due to man made perturbations of trace gases. Science 194:685-690.

Wassen M.J., and Joosten J.H.J. 1996. In search of a hydrological explanation for vegetation changes along a fen gradient in the Biebrza Upper Basin (Poland). Vegetatio 124(2):191-209.

Watson C.A. and Atkinson D. 1999. Using nitrogen budgets to indicate nitrogen use efficiency and losses from whole farm systems: a comparison of three methodological approaches. Nutr Cycl Agroecosyst 53:259-267.

Watson C.A. and Foy R.H. 2001. Environmental impacts of nitrogen and phosphorus cycling in grassland systems. Outlook Agric 30:117-127.

Watson C.A., Jordan C., Taggart P.J., Laidlaw A.S., Garrett K.M. and Steen R.W.J. 1992. The leaky N-cycle on grazed grassland. Aspects Appl Biol 30:215-222.

Well R., Augustin J. and Meyer K. 2001. In situ measurement of denitrification and N_2O production in the saturated zone of three Eutric Histosols and of a Mollic Gleysol. In: Broll G, Merbach W, Pfeiffer EM (eds). Wetlands in Central Europe Soil organisms, Soil Ecological Processes and Trace Gas Emissions. Springer, pp 165-176.

Wieder R.K. and Lang G.E. 1984. Influence of wetlands and coal mining on stream water chemistry. Water Air Soil Poll 23:381.

Wolf J., Beusen A.H.W., Groenendijk P., Kroon T. Rötter R. and Van Zeijts H. 2003. The integrated modeling of STONE for calculating nutrient emissions from agriculture in The Netherlands. Environ Mod Softw 18:597-617.

Wrage N. 2003. Pitfalls in measuring nitrous oxide production by nitrifiers. PhD thesis Wageningen University.

Wrage N., Velthof G.L., Van Beusichem M. and Oenema O. 2001. Role of nitrifier denitrification in the production of nitrous oxide. Soil Biology and Biochemistry 33:1723-1732.

Yan W. Yin C. and Tang H. 1998. Nutrient retention by multipond systems: Mechanisms for the control of nonpoint source pollution. J Env Qual 27(5):1009-1017.

Yeomans J.C., Bremner J.M. and McCarty G.W. 1992. Denitrification capacity and denitrification potential of subsurface soils. Commun Soil Sci Plant Anal 23:919-927.

Yli-Halla M., Hartikainen H. and Väätäinen P. 2002. Depletion of soil phosphorus as assessed by several indices of phosphorus supplying power. Europ J Soil Sci 53:431-438.

Summary

Ever since the colonization around 1050 AD, the peat district in the Western part of The Netherlands has known problems with water management, which were solved by building dikes, pumping stations and a strict management of inlet and discharge rates. Nowadays, the Rijnland Water Board, responsible for the water quantity and quality of 105000 ha of mostly peat land, faces another challenge: the water quality.

Dairy farming is the dominant land use and is a main contributor of diffuse pollution of surface waters. Since 1970 concentrations of chloride (Cl) and nitrate (NO_3) decreased due to reductions of point sources, but concentrations of nitrogen (N) and phosphorus (P) are still above target levels and disturb sound ecological functioning of the surface water.

In this thesis N and P dynamics of an intensively managed grassland on peat soil were studied in order to assess possibilities and limitations of reducing N and P concentrations in the surface water of the polder. The study area (Vlietpolder) was a small peat pasture polder (202 ha), about 30 km from Amsterdam and about 20 km from the North Sea. The polder is almost exclusively used for dairy farming (> 90% of the surface area).

To provide detailed information about N and P fluxes at field level and at farm level, farmgate balances (FGBs), defined as the difference between nutrient inputs and nutrient outputs at farm level, and soil surface balances (SSBs), defined as the differences between nutrient inputs and nutrient outputs at field level were set-up for all farms in The Vlietpolder. It then appeared that inter-farm (between-farm) variabilities exceeded intra-farm (within-farm) variabilities. This has consequences for the estimated N leaching to surface water based on farm-level data as was shown for N. Due to the variability of N surpluses at field-level, N leaching to surface water was underestimated by 2-46% (Chapter 2).

The main N loss pathway was gaseous losses via microbial denitrification (the reduction of soluble NO_3 in gaseous N_2 and N_2O). Annual N losses through denitrification were estimated at 126-213 kg N ha^{-1} of which almost 70% originated from soil layers deeper than 20 cm below soil surface. Groundwater levels governed the distribution of denitrification rates with depth which was explained by the ample availability of an energy source (degradable C) throughout the soil profile of the peat soil (Chapter 3).

A key element of this thesis was the estimation of the contribution of dairy farming to the N and P loading of surface water. This could not be done without the information provided in the previous chapters, supplemented with data on N and P discharge at the end of the ditch (103 kg N y^{-1} and 15 kg P y^{-1}), supply of N and P via inlet water (6 kg N y^{-1} and 1 kg P y^{-1}), mineralization of soil organic matter

(181-280 kg N $ha^{-1}y^{-1}$ and 8-12 kg P $ha^{-1}y^{-1}$), slush application (102 kg N ha^{-1} $application^{-1}$ and 9.6 kg P ha^{-1} $application^{-1}$), and on N losses through denitrification in the ditch (2-19 kg N y^{-1}). For N, 43-50% of the discharge was accounted for by applications of fertilizers, manure and cattle droppings, 17-31% by mineralization of soil organic matter, 8-27% by nutrient rich deeper peat layers, 8-9% by atmospheric deposition and 3-4% by inlet water. For P, these numbers were 10-48% for applications of fertilizers, manure and cattle droppings, 2-14% mineralization of soil organic matter, 33-82% nutrient rich peat layers and 5-6% inlet water (Chapter 4).

In Chapter 4 it was observed that also the subsoil had an important impact on the N and P loading of the surface water. This 'source' of N and P was explored in more detail in Chapter 5 by analyzing concentration profiles of the soil solution and 2D simulation modelling. Concentrations of N, P and Cl tended to increase with depth till about 6 m and then decreased. Exploring scenario simulations showed that major drainage fluxes passed through the peat layer that transported nutrients to adjacent surface water. Raising surface water levels with 20 cm suppresses this kind of nutrient loading of surface water by more than 30%, but nutrient rich peat layers will remain persistent as a potential source of nutrients in surface water in many peat polders in the western part of The Netherlands (Chapter 5).

Due to the high intensity of N and P fluxes in the Vlietpolder, the limited contribution of relatively easy to control nutrient sources on N and P leaching to surface water and the uneven distribution of N and P surpluses over fields, the scope for measures taken at farm level to reduce N and P leaching to surface water is limited. Notably, it is plausible that decreased N and P inputs to the fields predominantly result in less denitrification for N and in less sorption for P. In comparison to other (mineral) soil types, presumably, for peat soils more efforts have to be made to achieve similar reductions in N and P leaching to surface water. The relatively high N and P concentrations in surface water are not unique for our present time, as already in 1941 comparable concentrations were reported. However, the surface water in the Vlietpolder still suffers from the detrimental effects of eutrophication and therefore new measures are demanded, taking into account all sources and processes that affect the nutrient loading of surface water. This thesis may facilitate the development of such measures.

Samenvatting

In het westelijk veenweidegebied, onderdeel van het groene hart van de Randstad, worden kwaliteitsnormen voor voedingsstoffen in het water dikwijls overschreden. Dit heeft groene sloten ('groene soep') en verlies van ecologische functies van het oppervlaktewater tot gevolg. Het grootste deel van de veenweidepolders wordt gebruikt voor de melkveehouderij en het lijkt dan ook aannemelijk dat de melkveehouderij een belangrijk aandeel heeft aan de hoge concentraties van voedingsstoffen in het oppervlaktewater. Om te beoordelen of dit ook daadwerkelijk zo is, zijn in dit proefschrift de stromen van de voedingsstoffen stikstof en fosfaat van en naar de bodem en van de bodem naar de sloot in beeld gebracht en is de bijdrage van de landbouw aan de belasting van het oppervlaktewater gekwantificeerd.

De verdeling van overschotten van stikstof en fosfaat over percelen van bedrijven is geanalyseerd in Hoofdstuk 2. Het overschot geeft de netto belasting van de bodem aan en is daarmee een maat voor de hoeveelheid stikstof en fosfaat die mogelijk uit kan spoelen naar het oppervlaktewater. Uit deze analyse bleek dat de overschotten erg ongelijk werden verdeeld. Zodanig zelfs dat bedrijven met een gemiddeld lager overschot, toch percelen konden hebben met een hoger overschot dan bedrijven met een gemiddeld hoger overschot. In eerste instantie hoeft dit niet heel erg te zijn en kan een boer goede redenen hebben om het ene perceel meer te bemesten dan het andere, maar het heeft wel gevolgen voor de berekening van uitspoeling van stikstof en fosfaat naar het oppervlaktewater. Dit wordt immers vaak op bedrijfsniveau gedaan omdat die gegevens doorgaans goed voorhanden zijn. Echter, de relatie tussen overschot en uitspoeling is niet lineair en door geen rekening te houden met de ongelijke verdeling van overschotten binnen bedrijven worden aanzienlijke onderschattingen gemaakt van de uitspoeling van stikstof.

Het grootste deel van het stikstofoverschot ging verloren door denitrificatie, de gasvormige emissie van stikstof. Door deze denitrificatie verdween jaarlijks 126-213 kg stikstof per hectare de lucht in. Er wordt vaak aangenomen dat de omvang van de denitrificatie gestuurd wordt door de grondwaterstand. Voor de veengrond in deze studie was dat echter niet zo. De grondwaterstand bepaalde wel de diepte waarop de denitrificatie plaatsvond, die bewoog ongeveer met de grondwaterstand mee. De afwezigheid van de relatie tussen fluctuaties in grondwaterstand en omvang van denitrificatie werd verklaard door het hoge gehalte aan organische stof in de bodem (Hoofdstuk 3).

Met de informatie uit hoofdstukken 2 en 3, en nog een aantal aanvullende bepalingen kon de bijdrage van de landbouw aan de stikstof- en fosfaatbelasting van het oppervlaktewater worden vastgesteld. De directe bijdrage van de landbouw (d.w.z. datgene wat actief door de boer wordt aangevoerd) bleek voor stikstof 43-50% te zijn en voor P 10-48%. Andere belangrijke bronnen waren oxidatie van veen (17-31% voor stikstof en 2-14% voor fosfaat), bijdrage vanuit de ondergrond (8-27%

voor stikstof en 33-82% voor fosfaat), atmosferische depositie (8-9% voor stikstof) en inlaatwater (3-4% voor stikstof en 5-6% voor fosfaat) (Hoofdstuk 4).

In hoofdstuk 4 viel de aanzienlijke bijdrage van de veenbodem op. Op zo'n 1-3 m diepte bevond zich een veenlaag die rijk was aan voedingsstoffen. Het watertransport was overwegend horizontaal gericht naar de sloot, maar af en toe bereikte de stroombanen ook deze veenlaag. Hierdoor werden voedingsstoffen 'opgepikt' en meegenomen naar de sloot. Met behulp van een 2 dimensionaal model is gekeken hoe de bijdrage van deze 'bron' verandert als het peil wordt opgezet (verhoogd). Het bleek dat in dat geval de bijdrage van de veenlaag terugliep met meer dan 30%. Echter, de veenlaag verdwijnt niet en blijft een potentiële bron van stikstof en fosfaat in het oppervlaktewater (Hoofdstuk 5).

De mogelijkheden om stikstof- en fosfaatbelasting van het oppervlaktewater in de Vlietpolder (eenvoudig) te reduceren zijn beperkt. Dit komt vooral door i) de ongelijke verdeling van overschotten over percelen, ii) de beperkte bijdrage van goed-te-hanteren bronnen aan de stikstof- en fosfaatbelasting van het oppervlaktewater en iii) de beperkte uitspoelingsfractie van het stikstof- en fosfaatoverschot. Terugdringing van het stikstof- en fosfaatoverschot of van de stikstof- en fosfaataanvoer op bedrijfsniveau zal vermoedelijk vooral resulteren in minder verliezen door denitrificatie van stikstof en in minder vastlegging in de bodem van fosfaat. In vergelijking met andere, meer minerale, bodems is de verwachting dat voor de Vlietpolder, een organische bodem, veel meer inspanning (d.w.z. reductie van het overschot) moet worden verricht om eenzelfde verbetering van de waterkwaliteit te realiseren (Hoofdstuk 6).

Curriculum Vitae

Ik (Christina Laetitia van Beek) ben geboren op 18 mei 1974 te Renkum. Na het bepalen van mijn VWO diploma aan het Oosterlicht College te Nieuwegein ben ik naar Wageningen gegaan. In eerste instantie om Bosbouw te studeren, maar na ruim een jaar heb ik de overstap naar Bodemkunde gemaakt. Het buitenland heeft me altijd getrokken en in 1997 heb ik 6 maanden stage gelopen bij het IFDC in Togo (West-Afrika) om vervolgens in Nederland een afstudeervak te doen naar fosfaatdesorptie in zandgronden. Na mijn afstuderen in 1998 heb ik 9 maanden gewerkt als trainee bij het LNEC in Lissabon, Portugal. In die periode heb ik onderzoek gedaan naar het transport van zware metalen in de bodem. Weer terug in Nederland kreeg ik een aanstelling bij het AB-DLO, wat later (deels) werd ondergebracht bij Alterra. Eén van de Alterra projecten was het DOVE-veen project welke de basis vormt voor dit proefschrift. Ik werk nog steeds met veel plezier bij Alterra en na het afronden van het DOVE-veen project in 2003 heb ik mij steeds breder ingezet, van procesonderzoek met stabiele isotopen tot analyse van nutriëntenmanagement in de Tropen. En het is deze breedte in onderwerpen die mijn werk afwisselend en uitdagend maken en houden.