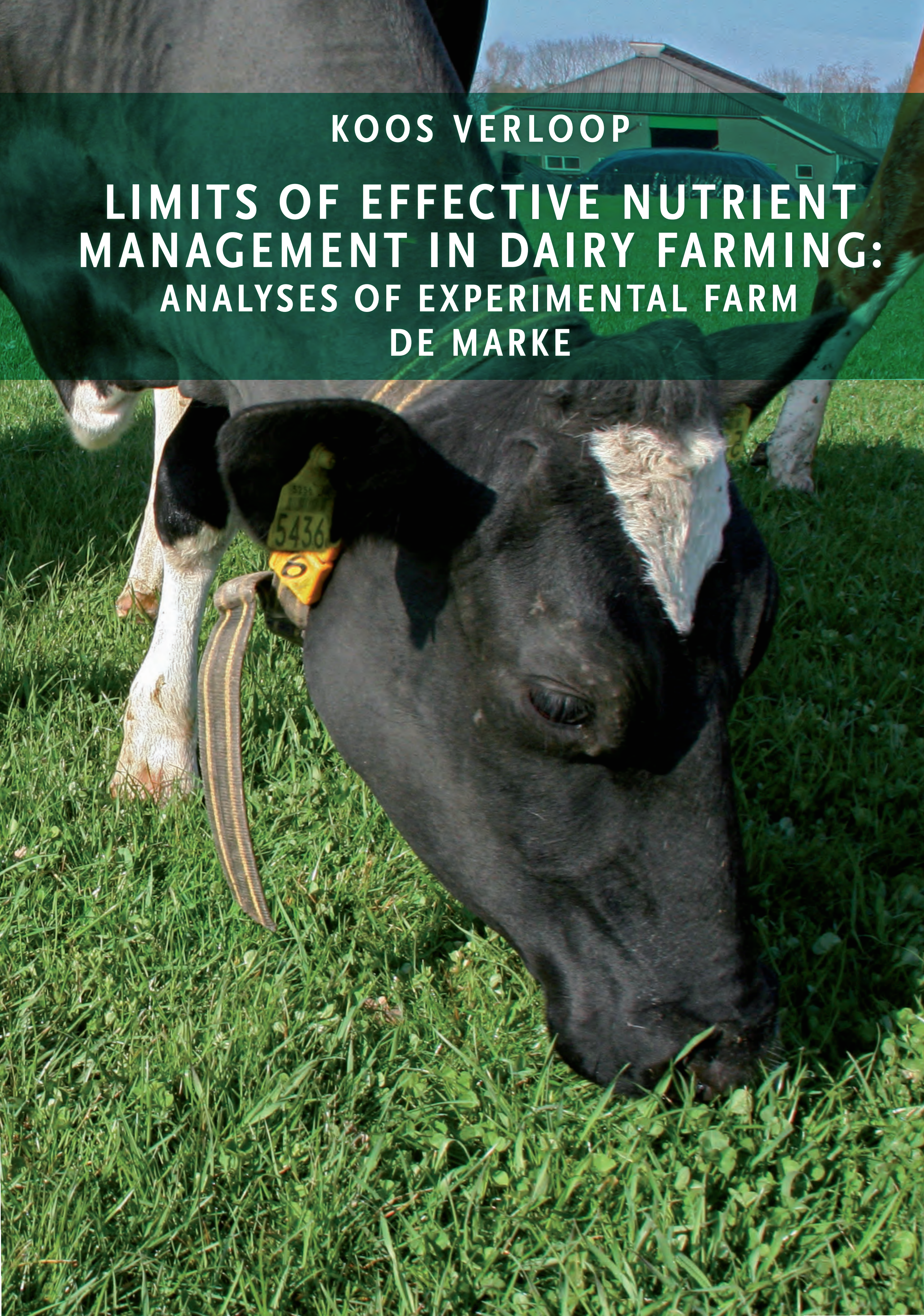
KOOS VERLOOP

LIMITS OF EFFECTIVE NUTRIENT MANAGEMENT IN DAIRY FARMING: ANALYSES OF EXPERIMENTAL FARM DE MARKE



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ABSTRACT

Intensive dairy production in the Netherlands is associated with high farm nutrient (N and P) inputs and high losses to the environment. The Dutch government and the dairy sector stimulate farmers to reduce losses through more efficient use of N and P inputs on their farms. This study explores for a dairy farm on dry sandy soil with average Dutch production intensity (12,000 kg milk per ha) the possibilities to meet strict environmental standards related to N and P by maximizing N and P use efficiency at the level of the farm and of the soil. Moreover, the study addresses the effects of efficient nutrient management on soil fertility.

The research was conducted on experimental dairy farm De Marke, that is designed to meet strict environmental standards, implemented in practice in 1989 and modified continuously to meet its targets by prototyping, i.e. a cyclic procedure of designing, implementing, testing and evaluating measures. The thesis evaluates system development since 2000, while results from 1993-1999 were used to analyse long-term developments.

After implementation of the farming system in 1989, the nitrate concentration in groundwater 'stabilized' at a level exceeding the environmental standard: 55 mg l^{-1}. Causes of excessive nitrate leaching were examined by relating measured nitrate concentrations to management. Grazing was associated with higher leaching in spite of careful management with rotational grazing. Leaching under permanent grassland was similar to the overall leaching in crop rotations in which grass was alternated with maize and grains. Spatial and temporal patterns of soil-N mineralization were explored to improve the synchronization of N application and crop-N requirements. This study indicated that fertilizing a 1st year maize crop, following grassland, is not necessary.

Measures implemented since 2000 to improve nutrient efficiency, included reduced grazing, adoption of anaerobic digestion, application of manure in the rows of maize, growing spring barley as the last crop in the arable phase, and, since 2004, the abolishment of fertilizer N. These measures contributed to an increase in the manure-N utilization and to an increase in the farm-N use efficiency up to 2008 to values exceeding the value of 33% that was realized in the period 1993-1999. Farm-N use efficiency was 35% in 2000-2003, 43% in 2004-2008 and 37% in 2009-2010. Farm-P use efficiency also increased as compared to the 87% that was realized in 1993-1999, i.e., it was 103% in 2000-2003 and 91% in 2004-2008. In 2009-2010, however, the farm-P use efficiency decreased to 69%, lower than the value realized in 1993-1999. The lower N and P use efficiency in 2009-2010 can be attributed to the lower N and P yields in grassland as a delayed effect of N limitation resulting from the abolishment of fertilizer N in grassland since 2004. Hence, despite the increase in manure-N utilization, mineral-N use is not yet completely redundant.

P-equilibrium fertilization seems to be compatible with highly efficient crop production, in the short and in the long term. Soil organic matter (SOM) percentage in the upper topsoil decreased by 0.03 yr^{-1} (average across all land uses) at a constant rate over the last 20 years. The possibilities to stop this decline by higher organic matter inputs to the soil seem conflicting with efficient nutrient use. Hence, the long term dynamics of SOM may become critical for future farm performance. It was concluded that N and P use efficiency can be enhanced substantially by on-farm nutrient management, but that efficient nutrient management may conflict with maintenance of SOM.

Key words: nutrient management, dairy, prototyping, organic matter, soil fertility, nitrogen, phosphor.

VOORWOORD

Eén van de mooie dingen van onderzoek doen op proefbedrijf voor duurzame melkveehouderij ‘De Marke’ is samenwerken in een team. Bedankt, Zwier, Gerjan, Léon, André, Henry, Stef, Jenny, Alwies, Gerjo en Paul, huidige en voormalige leden van ‘team De Marke’, voor jullie steeds weer geweldige inzet. Bedankt Frans! Je bent als één van de grondleggers nog steeds beschikbaar als vraagbaak. Je eerlijke, enthousiaste, soms kritische reacties zijn steeds heel waardevol geweest. Jouke, je bent als ‘promotiemaatje’ op verschillende manieren van belang geweest. Je enthousiasme en doorzettingsvermogen werken aanstekelijk. En, eerlijk is eerlijk, jij wint, met een voorsprong van enkele uren.

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Het project ‘Koeien & Kansen’ wordt gedragen door een samenwerkingsverband van de overheid (vertegenwoordigd door het Ministerie van Economische Zaken en het Ministerie van Infrastructuur en Milieu, de melkveesector (vertegenwoordigd door LTO en het Productschap Zuivel) en het onderzoek. In deze samenwerking zijn ook adviseurs, onderwijs, loonwerkers en leveranciers belangrijk. Het samenspel tussen al deze betrokkenen is van onschatbare waarde. Het ondersteunt de kennisoverdracht in alle richtingen. ‘Lang leve dit samenspel. Samenwerkers, bedankt!’

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Koos

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CHAPTER 1

GENERAL INTRODUCTION

1 BACKGROUND

Nitrogen (N) and phosphorus (P) are indispensable external inputs in agricultural production systems to replenish the nutrients exported in produced products and those removed from the agricultural systems through unavoidable losses. From the second half of the 20th century onwards, production levels in various animal production regions, but particularly in Dutch dairy farming, increased continuously in response to increasing inputs. Inputs of N and P from external sources increased disproportionally in comparison to their outputs (Aarts, 2000). As a consequence, surpluses, defined as nutrient inputs minus nutrient outputs in products, increased dramatically. On dairy farms, in the mid-1980s only 14 and 32% of the N and P, respectively imported to the farm was exported in products. The average annual surpluses of N and P amounted to 470 and 32 kg ha^{-1}, respectively (Aarts, 2000). Since the mid-1980s awareness has been growing of the adverse effects of these surpluses on surface water quality, groundwater quality, air quality and nature. In Western Europe, enrichment of ecosystems with nutrients (eutrophication) became more and more severe (Hosper, 1997; Hooda, 2000), and nitrate and phosphorus were considered the most problematic substances in agricultural catchments (Reijnders et al., 1998; Hooda, 2000). Water quality problems appeared in many animal production regions in the European Union (De Walle and Sevenster, 1998) and in the United States (Stout et al., 2000). It was recognized that farm management had to be adjusted to reduce nutrient surpluses by using nutrients more efficiently within the systems. Farm nutrient use efficiency is defined as the proportion of nutrients imported into the farm that is transferred into exported products (milk and meat in the case of dairy farms). An increase in nutrient use efficiency implies that the same production can be realized with lower inputs and losses of nutrients. Development of improved nutrient management practices was stimulated by national legislation (Schröder and Neeteson, 2008), implemented to realize water quality standards, defined in the EU Nitrates directive (EC, 1991) and the Water framework directive (EC, 2000), and to restrict ammonia emissions in agreement with the EU Directive on National Emission Ceilings for Atmospheric Pollutants (NEC; EC, 2001).

As a result of the efforts of policy, research and the dairy sector and endorsed by policy regulations, efficient nutrient management has increasingly been adopted since the early 1990s. This has resulted in an increase in nutrient use efficiency in Dutch dairy farming and associated reductions in N and P surpluses. Estimated N and P use efficiency for the average dairy farm in 2006 amounted to 27 and 57%, respectively, and estimated N and P surpluses amounted to 218 and 11 kg ha^{-1}, respectively (Aarts et al., 2008). This improvement in nutrient use efficiency was mainly the result of a reduction in the use of mineral fertilizer-N and of reductions in N and P inputs in feeds (Aarts et al., 2008; Van den Ham et al., 2011). From 1990 to 2010 emissions of nitrate and ammonia to the environment gradually decreased (Vellinga et al., 2011; Baumann et al., 2012; Willems and Van Schijndel, 2012). Over that period, various tools were developed to support efficient nutrient management and a number of research networks were started to enhance adoption of these tools (Vellinga et al., 2011), e.g. 'Cows and Opportunities' (Oenema et al., 2001). Also in the EU, environmental policies have resulted in lower N losses from agriculture Oenema et al. (2009). However, further improvements in nutrient management are required to meet environmental standards. Reports on EU-water quality indicate that the environmental status of groundwater bodies, surface waters and natural areas in the EU countries is still below the target values (EC, 2007). Moreover, on-going changes in dairy farming urge to further reduce nutrient losses through a more efficient use of inputs, as will be explained next.

In the course of these developments, average production intensity, i.e. milk production per unit surface area, has gradually increased. In the Netherlands, from 2000 to 2011, the number of dairy farms decreased from 29,466 to 19,247 and the remaining farms have expanded in size (PZ, 2012). As land is very expensive, farms expand by increasing milk production rather than by enlarging the area of farmland. This intrinsic trend of expansion and intensification takes place in other EU animal production regions (Anonymous, 2000; De Vries et al., 2013). Currently, in the EU milk production is regulated by quota (Kempen et al., 2011), a system that will be abolished in 2015. In the vision of the Dutch dairy sector, the abolishment of the quota after 2015 will lead to a milk production of Dutch dairy farms that will be determined by market developments and a gradual increase in production is expected. However, it is realized that such an increase in production is only possible within the boundaries of environmental standards and societal demands (LTO, 2011). Thus maximizing the efficiency of nutrient use becomes essential for developing environmentally sustainable dairy farming systems. The dairy sector also faces demands with respect to animal health, landscape quality and greenhouse gas emissions (ELI, 2011; UDV, 2012). Nevertheless, nutrient management is one of the central issues (EZ, 2013a; De Vries et al., 2013) in the development of sustainable dairy farming systems.

Generally, high manure-N and -P inputs to farmland are associated with high N and P losses to the environment and, in particular to aquatic systems (Tunney et al., 1997; Van der Zee and Van Riemsdijk, 1988; Kronvang et al., 2007). Consequently, upper limits have been established by the EU member states with respect to the application of N and P on farmland (Van Grinsven et al., 2012). In intensive dairy farms the P input into farmland through manure and inorganic fertilizer must be balanced with P output in crop products, e.g. by applying P-equilibrium fertilization. With respect to N, an application threshold has been established in the Nitrates Directive of the EU (EC, 1991) of 170 kg manure-N ha^{-1}. In the Netherlands, dairy farmers having at least 70% of their farmland in grassland can apply for a derogation that implies a threshold of 250 kg manure-N ha^{-1} (LNV, 2009). On many intensive dairy farms, the production of manure-N and -P exceeds the quantity allowed for application on farmland. These farms have to export the excess manure out of the farm. This, however, is not taken as the preferred solution (EZ, 2013b), because export of manure is expensive and massive manure export from intensive animal production regions has adverse off-farm effects: (*i*) export by truck is energy-demanding and, consequently, negatively affects the energy efficiency of production; (*ii*) mineral fertilizer-N and, occasionally, fertilizer-P is imported to the farm to meet crop requirements, while manure-N and -P is exported to protect the environment, which is again energy-and resource-consuming; (*iii*) it is questionable whether exported manure is utilized efficiently, keeping in mind that exported manure may be treated rather as a waste product than as a valuable resource (Schröder et al., 2011). Hence, there is increasing consensus that more attention should be given to on-farm management of manure. Therefore, in the Netherlands, the national government and the dairy sector (in addition to other stakeholders) agreed to endorse modifications of manure management so that: (*i*) the relation between manure-N and -P rates and environmental pressure at farm level is nullified, and (*ii*) the need for mineral fertilizers is drastically reduced (Anonymous, 2008). This requires optimisation of existing techniques or implementation of innovations. For dairy production this is referred to as efficient dairy farming.

2 EFFICIENT DAIRY FARMING - PRESENT AND FUTURE CHALLENGES

Despite reductions in nitrate emissions from dairy farms to ground- en surface water since 1990 in The Netherlands (Baumann et al., 2012), nitrate leaching still is a critical issue, in particular on light deep-draining sandy soils that are susceptible to nitrate leaching (Boumans et al., 2001; Van der Grift et al., 2002). To further reduce nitrate leaching, management strategies for dairy farming systems have to be developed that lead to increased N utilisation: the transformation of N applied to farmland into crops should be as 'complete' as possible. Two coherent types of measures can contribute to improvement of this transformation: (*i*) closing 'leaks', i.e., those parts in the system with enhanced N transformation from farming system to the environment, especially from soil to ground- and surface water (Aarts, 2000; Shepherd and Chambers, 2007), and (*ii*) synchronizing the flow of nutrients through the soil by adjusting input rates to the N uptake by crops (Schröder, 1998; Schröder, 2005; Webb et al., 2013).

Considering the disadvantages of manure export, efficient nutrient management on dairy farms should focus on the use and management of the manure produced on the farm. It is important to gain insight in the level of nutrient efficiency that can ultimately be realized on farms where crop production fully relies on the use of farm manure (Jarvis and Aarts, 2000). However, efficient nutrient management should not jeopardize soil fertility in the long term. P equilibrium fertilization is commonly considered a sustainable strategy for intensive production systems, because a built up of P in soil, already rich in P, is not accepted, to prevent future environmental pressure by losses of P to water. However, there is concern that application of P equilibrium fertilization, in the long term, may lead to P limitation in crop production, due to a reduction in plant-available P (Neeteson et al., 2006; Syers et al., 2008; Masterplan Mineralenmanagement, 2011). Moreover, there is general concern about decreasing soil organic matter (SOM) contents in agricultural soils (Eglin et al., 2010; Lal, 2011; EC, 2012), though Reijneveld et al. (2009) conclude that SOM in agricultural soils in the Netherlands tends to increase slightly. Soils, low in SOM, are more susceptible to both drought stress in crops (Bell and Van Keulen, 1995) and leaching of nitrate to ground- and surface water (Boumans et al., 2001). The extent to which measures applied to enhance nutrient management may put SOM contents under pressure, i.e. lead to reduced SOM contents, is not well-known.

3 CONTRIBUTION OF THIS STUDY

This study was set up in response to the challenges on development of a sustainable dairy production system, described in terms of efficient use of nutrients (particularly N and P). The general objective was to explore for a dairy production system on dry sandy soil,

1. the possibilities to meet strict environmental standards related to N and P. This objective was addressed with emphasis on reduction of nitrate leaching to groundwater.
2. the possibilities to maximise N and P use efficiency at the level of the farm and of the soil. This objective was addressed with emphasis on the transformation of nutrients from manure (after application to soil) into crops.
3. the potential consequences of efficient nutrient management on soil fertility dynamics.

The research questions emerging from these objectives are:

1. How to restrict nitrate leaching to values below 50 mg l^{-1} in groundwater to meet pre-defined environmental standards related to N?
2. To what extent can nutrient use efficiency at the level of soil and the whole farm, be improved by management with the techniques that are available at present?
3. What are the effects of improved N and P management on long term soil fertility?

The first question was addressed by an analysis of the causes of nitrate leaching on the farm to explore the options for reduced nitrate leaching by management adjustments (Chapter 2). This exploration was extended to an analysis of N mineralization observed in *in situ* measurements on the farm (Chapter 3). Elements of this study also contributed to knowledge about the second issue. The second question was addressed by exploring the effects of adaptation of manure management on nitrogen utilisation by crops (Chapter 4). To answer the third question, an analysis was performed of long-term effects of P-equilibrium fertilization on short- and long-term P availability and on the response of crops to P-fertilization in soils with low P status (Chapter 5). Furthermore the dynamics of SOM were explored (Chapter 6). This was relevant because there may be an interaction between efficient N and P use on dairy farms and SOM dynamics.

The objectives and the research questions of this study are directly related to farm management. Therefore, the research is conducted within the framework of a whole farm, as will be motivated in Section 1.4. The research approach and methods are described in detail and more specifically in Section 1.5.

4 FARMING SYSTEM RESEARCH AND PROTOTYPING

To support the required adaptation of dairy farming systems (see Sections 1.1 and 1.2), knowledge is required on the effects and consequences of nutrient management on farm performance. Farmers must know the practical implications of efficient nutrient management, the efficacy of measures implemented to enhance farm nutrient efficiency, and present and future risks associated with specific management measures (Aarts, 2000; Shepherd and Chambers, 2007; Le Gal et al., 2010). From the mid-1980s onwards, it was increasingly recognized that agronomy had to be reframed to facilitate the required transition by analyzing whole farm systems instead of their separate subcomponents. The whole farm research approach lacks the conventional experimental research modes including control- and treatment groups with sufficient replicates. To handle this alternative situation, a new research approach was introduced, i.e. 'prototyping', to explore the options for farm development, targeting multiple goals. This method, developed by Vereijken (1992, 1997), can be defined as: a participative method, (*i*) to explore in whole (integrated) farming systems the possibilities to realise pre-defined goals, (*ii*) considering agricultural, ecological and economic drivers and, (*iii*) to implement new designs or measures on the farm, (*iv*) according to a stepwise procedure of goal setting, designing, implementation, monitoring and evaluation. 'Prototype' refers to a newly designed farm layout that differs from common farm configurations. The performance of the alternative design is demonstrated to facilitate identification and adoption of beneficial measures by commercial farmers (Sterk et al., 2007; Duru, 2013).

5 METHODS USED FOR THIS THESIS; SYSTEM DEVELOPMENT 'DE MARKE'

The research performed in this study was conducted at experimental dairy farm 'De Marke' located in the eastern part of The Netherlands. 'De Marke' is a prototype dairy farm and defines, in addition to a location, also a research approach. 'De Marke' was established in 1989 to explore and demonstrate the possibilities to produce milk at an intensity of 12,000 kg milk per hectare on dry sandy soils under strict environmental standards. The farm provides the context of a common farm and the infrastructure to monitor flows of N and P within the farm and losses of N and P from the farm to the local environment. The experimental farm is located on light sandy soil that is extremely susceptible to nitrate leaching (Boumans et al., 2001). This selection was made purposefully to be able to explore efficient nutrient management under the most difficult circumstances and demonstrate its effectiveness, while showing that strict environmental standards can be attained (Aarts, 2000). The design of the farm was based on a prototype-model, a coherent scheme of the farm that was designed to meet strict environmental standards. Furthermore, the following restrictions had to be respected: (*i*) irrigation was restricted and was only applied under extreme conditions; (*ii*) manure or its products were not exported from the farm, and (*iii*) replacement stock was bred and raised on the farm. These conditions were established to prevent improvement of farm performance for N and P at the expense of other issues (e.g. efficient water use) or through solutions that transfer problems to beyond the farming boundaries (pollution swapping). The designed prototype was implemented in practice in 1989. From 1989 to 1992 farm lay out and research infrastructure were established and in 1992 'De Marke' started to function as an experimental farm.

Since the start, the system has been continuously developed to meet as close as possible the targets. To guide farm development, a structural cyclic procedure has been followed, consisting of: target setting (including translation of standards with respect to environmental quality in boundary conditions such as permissible surpluses that can be used to evaluate farm performance (Table 1)), design (based on targets and boundary conditions), implementation of the design, monitoring of the performance, analysis of the performance and evaluation (Figure 1). In all phases of system development, the primary target is to meet the set of boundary conditions presented in Table 1.

For various reasons, farm economics were beyond the scope of this study. System development at 'De Marke' may frequently bring the experimental farm in states that are not beneficial from an economic point of view. Innovations are tested deliberately at 'De Marke', instead of on commercial farms, because the risks to incur financial drawbacks are too high for commercial farmers. Implementation may require support of technical specialists, adjustments after implementation and sometimes it may take appreciable time to improve the innovation so that it performs in an acceptable way. The objective of the experimental farm is to explore the possibilities to meet future societal demands. Production methods that are not beneficial now, may become beneficial or acceptable in the future.

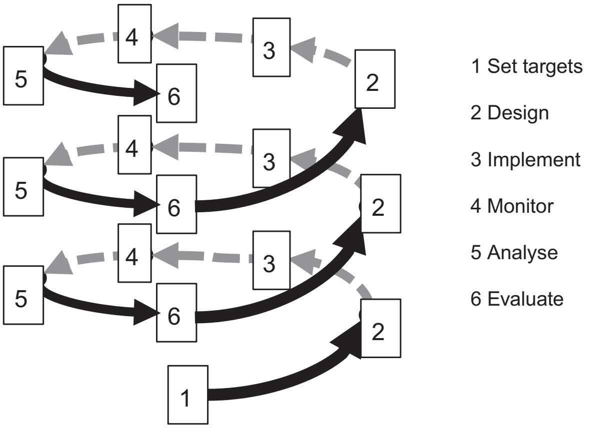

Figure 1: *Schematic representation of the system development procedure.*

Table 1: *Standard environmental conditions applied in the design of the dairy farming system of 'De Marke' (Aarts et al., 1992).*

Objective	Maximum value
Nitrate concentration in groundwater[1)]	50 mg l^{-1} in the upper meter of groundwater
Ammonia volatilization[2)]	30 kg N ha^{-1} yr^{-1} from manure
N surplus farm[3)]	128 kg N ha^{-1} yr^{-1} as farm inputs[4)] minus outputs assuming no accumulation in the soil
N surplus soil[5)]	79 kg N ha^{-1} yr^{-1} as soil inputs[6)] minus outputs with harvest
P concentration in groundwater[7)]	0.15 mg P l^{-1} in the upper meter of groundwater
P surplus[5)]	0.45 kg P ha^{-1} yr^{-1}

[1)] National and EU-standard according to the EU Nitrate Directive (EC, 1991).
[2)] Corresponding to national environmental plan to reduce emissions (VROM, 1989).
[3)] Calculated as the sum of acceptable losses, assuming no accumulation on the farm.
[4)] Including biological N fixation by clover and deposition.
[5)] Calculated on the basis of the maximum permissible concentration in the upper meter of groundwater.
[6)] Including chemical fertilizer, organic fertilizer, and biological N fixation by clover and deposition.
[7)] Corresponding to the Dutch surface water threshold value.

The performance of experimental farm 'De Marke' is studied as a whole. However, this thesis also focuses on the lower half of the scheme in Figure 2, i.e., the manure, soil, crop interaction and the transfer from soil to groundwater. The focus is primarily on the nutrients N and P and secondly on soil organic matter. This thesis covers the system development that was implemented since 2000. However, when relevant, data originating from 1989-1999 were also included in the analyses.

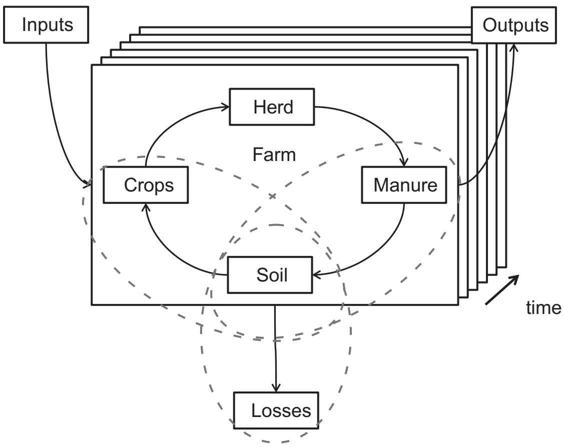

Figure 2: *Schematic representation of the flow of nutrients to, from and through the farm; the ovals of dotted lines indicate the main focus of this research.*

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CHAPTER 2

REDUCING NITRATE LEACHING TO GROUNDWATER IN AN INTENSIVE DAIRY FARMING SYSTEM

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ABSTRACT

Dairy farming is one of the main contributors to nitrate leaching to groundwater, particularly on soils that are susceptible to leaching, such as light well-drained sandy soils. In the Netherlands, as in many other European countries, these soils are predominantly used for dairy farming. A prototype dairy farming system that has been implemented in practice in 1989 has continuously been adapted since then to meet environmental standards (i.e. the EU-standard of 50 mg NO_3^- l^{-1}) without reducing milk production intensity (12,000 kg ha^{-1}). After an initial decline in nitrate concentration from 193 mg l^{-1} to 63 upon implementation, it subsequently 'stabilized' at a level higher than the environmental standard: 55 mg l^{-1}. The goal of this paper is to examine causes of excessive nitrate leaching. This was done by relating measured nitrate concentrations with management characteristics such as N balances, cropping patterns and grazing intensities. Special attention was paid to aspects that were supposed to be conducive for leaching: crop rotation of grass and maize and grazing. No evidence was found for enhanced nitrate leaching due to the rotation of grass with maize compared to permanent cultivation. This could be ascribed to the reduction in fertilization levels in first and second year maize with 90 and 45 kg N ha^{-1}, respectively to account for the expected N release from the ploughed-in grass sod. Triticale was found to lead to higher leaching than grass or maize which is attributed to its poor growth in the period that it should function as catch crop in maize. Grazing contributed to a nitrate increase of about 30 mg NO_3^- l^{-1} on grassland. As grazing management and intensity is already strictly optimized in order to restrict nitrate leaching, this result underpins the need to develop sustainable grazing methods on soil that is susceptible to nitrate leaching.

1 INTRODUCTION

1.1 Nitrate leaching and dairy farming

Nitrogen (N) and phosphorus (P) are indispensable external inputs in sustainable agricultural production systems, to replenish the nutrients exported in products and removed in unavoidable losses (Von Liebig, 1841). However, especially in high-intensity production systems, inputs of N and P from external sources have increased disproportionally in recent decades in comparison to their outputs. As a consequence, surpluses, defined as nutrient inputs minus nutrient outputs in products, have increased dramatically (Smaling et al., 1999; Van der Meer, 2000). Especially N-surpluses are associated with losses from the system through nitrate leaching, ammonia volatilisation and dissipation as N_2 or NO_x (Hilhorst et al., 2001; Delgado, 2002). The losses can negatively affect the quality of groundwater (Boumans et al., 2005), surface water (Collingwood, 1977; Jensen et al., 1991) and air (Monteny, 2000). Nitrate leaching to groundwater presents risks for drinking water supplies and is associated with eutrophication in streams and lakes (Hosper, 1997; Scheffer, 1998). According to the EU Nitrate Directive (Council Directive 91/676/EEC), European Union member States are obliged to establish action programs that guarantee nitrate concentrations in groundwater in leaching-sensitive areas not exceeding the standard of 50 mg l^{-1}.

In West European countries, agriculture is the major contributor to nitrate contamination of groundwater (De Walle and Sevenster, 1998; De Clercq et al., 2001), with dairy farming as the sector with the largest share. N losses are particularly high on soils that are susceptible to leaching in regions with high animal densities (Anon., 1993), as found in various countries in northern and western Europe and in some federal States of the USA (De Walle and Sevenster, 1998; Stout et al., 2000; Burkart and Stoner, 2002). The combination of intensive dairy farming, high N-inputs, low N-utilisation and high levels of nitrate leaching is particularly evident on the light sandy soils in the eastern and southern parts of the Netherlands (Fraters et al., 1998), where in the period 1992-1995, average nitrate concentrations exceeding 150 mg l^{-1} were monitored in the upper meter of the groundwater, associated with application of organic N doses of 320 kg ha^{-1} (Aarts et al., 1999a). In commercial farming, such high rates of manure application apparently result in excessive nitrate leaching. Reducing leaching by reducing the nitrogen application rate, i.e. transforming to low input and therefore extensive dairy farming, with the consequence of reduced milk production per ha (Aarts et al., 1999a), is economically unattractive. Therefore, a need exists to develop highly productive systems with enhanced utilisation of manure-N, reduced surpluses and thus reduced levels of nitrate leaching.

1.2 Integrated system research at 'De Marke'

In response to this need, the research project 'De Marke' was established, in which possibilities are being investigated to operate a dairy farming system that is both environmentally and economically sustainable (Aarts et al., 1992; Aarts et al., 2000a). The project uses an integrated, whole-farm approach, based on the many feedbacks within the system, i.e. the cluster of interactions between the sub-systems herd, manure, soil, crop and feeds. This research has a single replicate character, but this potential disadvantage is overcome by following a systematic procedure of modelling, system design, implementation in practice, evaluation and adjustment (Aarts et al., 1992).

Long-term EU and national environmental standards (Table 1) were set as boundary conditions for system design. In accordance with the EU Nitrate Directive, maximum permitted nitrate concentration in the upper meter of groundwater was set to 50 mg l^{-1}, corresponding to 34 kg N ha^{-1} leaching for the average annual precipitation surplus of 300 mm under Dutch conditions. Ammonia emissions from faeces and urine should not exceed 30 kg N ha^{-1}. Assuming no N accumulation in the soil, 14 kg N ha^{-1} yr^{-1} volatilization from crops and silage, and estimated 50 kg N ha^{-1} yr^{-1} losses by denitrification, the N surplus should not exceed 128 kg ha^{-1} yr^{-1} at farm level (Aarts et al., 2000b). Maximum P concentration in groundwater was set to the Dutch surface water threshold value (0.15 mg l^{-1}), corresponding to an annual P surplus of 0.45 kg ha^{-1}. These goals have to be realized at a milk production intensity of at least the national average of 11900 kg ha^{-1} in order to be representative for Dutch circumstances, and without exporting slurry to prevent shifting of problems involved in efficient use of slurry in intensive farming (Schröder et al., 2003).

Table 1: *Environmental objectives with regard to nutrients (Aarts et al., 1992).*

Objective	Maximum value
Nitrate leaching	50 mg l^{-1} in upper meter of groundwater
Ammonia volatilization	30 kg N ha^{-1} yr^{-1} from manure
N surplus	128 kg N ha^{-1} yr^{-1} as farm inputs[1] minus outputs assuming no accumulation on the soil
P leaching	0.15 mg P l^{-1} in upper meter of groundwater
P surplus	0.45 kg P ha^{-1} yr^{-1}

[1] Including deposition and fixation by clover.

In this paper, the effects of the specific 'De Marke-management' on nitrate leaching are evaluated to identify possible 'weak spots' in the system. The evaluation is based on nitrate concentration measurements. In the analysis, encompassing data of 1996-2003, the effects of the main crops, grass, maize and triticale, and of the crop rotations are examined, in relation to underlying management aspects, such as (*i*) the magnitude of the surpluses on the soil balance for specific crops; (*ii*) the management of nitrogen release from the ploughed-in grass sod; (*iii*) re-sowing of permanent pasture; (*iv*) grazing. The effect of maize receives special attention, because under current practice, maize presumably contributes to high nitrate concentrations, while at 'De Marke' maize occupies a large proportion (31%) of the land. Moreover, exploring whether the rotation enhances nitrate leaching is warranted, as some studies indicate that ploughing grassland for arable crops can lead to substantial increases in nitrate leaching (Whitmore et al., 1992; Hoffman, 1999). Possibilities for improvement of the system are discussed.

2 MATERIALS AND METHODS

2.1 Dairy farming system 'De Marke'

2.1.1 Site characteristics

'De Marke' is situated in the eastern part of the Netherlands on very light sandy soil. Its 55 ha of land was reclaimed from heather [*Calluna vulgaris* (L.) Hull] at the beginning of the 20th century. The soils at 'De Marke' are characterized by a 25-30 cm antropogenic upper layer, with an average organic matter content of 4.8%, overlying a layer of yellow sand, very low in organic matter and hardly penetrable by roots (Aarts et al., 2000b). The hydrological basis, extending to 70-120 meters below the surface, consists of various formations of which the oldest and deepest are of marine origin and those at 6-30 meter depth consist of fluviatile coarse sandy sediments. The upper layers, of eolian origin are finer-textured and low in organic matter and calcium contents (Van der Grift et al., 2002). Groundwater depth is 1 to 3 meter below soil surface, i.e. upward water transport from the saturated zone to the root zone is negligible in periods that potential evapotranspiration exceeds precipitation. Plant-available water holding capacity is below 50 mm on 50% and below 100 mm on 78% of the land (Dekkers, 1992), restricting unlimited crop transpiration to about 10 days during a dry period in summer. From 1996-2003 average annual precipitation was 757 mm. Water transport is predominantly vertical (Van der Grift et al., 2002), hence management of individual fields should be reflected in nitrate concentrations in the groundwater below the fields.

Before 1989 the land was in use by a number of commercial dairy farmers. In the period 1989-1991, when the 'De Marke-management' was under development, no cattle were present and only chemical fertilizer was used. From 1993 onwards, the integrated management system has been fully implemented.

2.1.2 System layout and land management

The farm area is divided into permanent grassland and two crop rotations (Table 2; Figure 1). Rotation I is implemented on fields close to the homestead that can be irrigated. Rotation II is implemented on fields situated further from the homestead that can not be irrigated. From 1996 till 1999, maize was the only arable crop, after which triticale was introduced as the last crop in the arable phase. On maize land, Italian ryegrass was sown as a catch crop between the rows in early summer. In early spring it was ploughed-in, some months before sowing the subsequent crop.

Table 2: *Farm plan of 'De Marke' (ha) and crop sequence of the two types of rotation.*

Parameter (unit)	Average 1996-2003
Total area	55
Permanent grassland	11
Ley	20
Maize	17
Triticale	7
Rotation I	3 years grassland -> 3 years arable crops -> etc.
Rotation II	3 years grassland -> 5 years arable crops -> etc.

Figure 1: *Plan of the experimental farming system 'De Marke'.*

N fertilization of crops aimed at complying with crop requirements (Table 3). In calculating N requirements of maize and grassland, incomplete availability of N from organic manure was taken into account. In calculating manure-N requirements for maize, a net release of N of 90 kg ha^{-1} in the first year and 45 kg in the second year from the ploughed-in grass sod was taken into account, resulting in rates in 1st and 2nd year maize that are lower than those in 3rd and 4th year maize. Differences in N surpluses for the various crops in different phases of the rotation reflect this fertilization strategy. The variability in surpluses, expressed as the standard deviation, Sd, for specific phases in the rotation is comparable for maize and grass (Table 3). On grassland, fertilizer application started on the 1st of March and ended in August. Fertilizer on maize was applied shortly after sowing (early May).

Table 3: *Nitrogen balances (kg ha^{-1}) for maize, triticale and grassland at 'De Marke'; means for 1996-2003.*

N-flow	Maize				Triticale	Grassland				
	1st yr	2nd yr	>2nd yr	All		1st yr	2nd yr	3rd yr	All	Permanent grassland
INPUTS										
Excretion during grazing	-	-	-	-	-	46	40	65	48	88
Organic manure[1)]	41	91	128	88	220	243	252	255	245	179
Chemical fertilizer	-	-	-	-	-	98	83	82	91	106
Deposition	49	49	49	49	49	49	49	49	49	49
Clover	-	-	-	-	24	31	36	22	26	12
TOTAL	90	140	177	137	293	467	459	475	458	432
OUTPUT (Harvest)	152	124	127	135	179	290	296	303	289	278
INPUT - OUTPUT	-62	-21	50	1	76	177	163	172	169	146
Sd	48	28	37	-	19	43	22	43	-	35

[1)] Slurry (faces + urine) following ammonia volatilization.

A total of 31 ha of grassland were available for grazing by 80 cows and young stock (Table 2). A rotational grazing system was applied, in which the cattle grazed for 5-7 days on individual plots of 1-3 ha, after which they were transferred to a next plot. In practice, grazing intensities were higher on the permanent grassland than on most of the leys, because permanent grassland is situated near the homestead and is thus easily accessible, while especially the leys of rotation system II are too far away for grazing. Grazing intensities, expressed as N excretion during grazing, are presented in Table 3. The dates on which grazing started and ended is specified per year in Table 4. Average seasonal grazing duration for the whole period, expressed as cow grazing days, was 103.

Table 4: *Onset and end of the grazing period in 1996-2003.*

Year	Onset of grazing season	End of grazing season
1996	May 3	October 1
1997	April 30	October 19
1998	April 6	October 1
1999	April 15	October 1
2000	April 19	September 16
2001	May 28	September 17
2002	May 30	September 21

2.2 Data collection

The total area of 'De Marke' is sub-divided in 30 plots of 1-3 ha, each, for which crop management, nutrient flows and yields were monitored separately (Figure 1). Relevant management data, i.e. rate, timing and method of manure application, harvesting/cutting, grazing, re-sowing and ploughing were monitored (Aarts et al., 1992). Organic mass flows (harvested crop material and applied slurry) were measured by weighing and the related nutrient flows were quantified on the basis of chemical analyses of representative samples. N-fixation by clover was estimated from its dry matter yield, assuming 50 kg N-fixation Mg^{-1} (Van der Meer and Baan Hofman, 1989; Elgersma and Hassink, 1997). Fresh grass consumption by cattle was calculated by estimating standing biomass just before and after grazing and correcting for growth during grazing. N in excreta during grazing was estimated from feed intake, by subtracting N output in milk and meat and in excretion indoors. N output in milk was quantified by monitoring protein levels (mg l^{-1}) and milk production (l). Milk production per cow was measured daily. Atmospheric deposition was derived from the national monitoring network. In late autumn, the upper meter of groundwater was sampled at 170 points. The sampling procedure is described in detail by Boumans et al. (2001).

2.3 Nitrate-related principles of system development

To restrict nitrate leaching, in the design of dairy farming system 'De Marke', two general strategies were followed:

1 limiting the intensity of N-fluxes within the system;
2 abatement of specific processes that induce nitrate leaching.

2.3.1 Limiting the intensity of internal N-fluxes

Within the farming system, the sub-systems feeds, cattle, manure, soil and crops are distinguished. In order to minimize losses, N transfers from manure to soil, from soil to crops, from crops to feeds, from feeds to cattle and finally, to exported products should be as complete as possible. Restricting the intensity of internal N-fluxes is required when N-contents of sub-systems exceed critical levels, above which the sub-systems become 'N-saturated'. High N-contents in (any of) the sub-systems make the whole system susceptible to N losses. In each of the sub-systems distinguished, the proportion of N transferred from that sub-system tends to decrease with increasing N-content in the sub-system. For example, intake of N in excess of the N (protein) requirements of cattle, results in 'N-saturation' of the animals and subsequently in increased N-excretion (Bannink et al., 1999; Kebreab et al., 2003). Excretion of urine-N on the soil is directly associated with nitrate leaching (Ryden, 1984), but above critical levels, also additions of total manure-N negatively affect the efficiency of nutrient cycling in the soil, leading to higher leaching losses (Kolenbrander, 1981; Ten Berge, 2002; Nevens and Reheul, 2003a). Similar 'saturation-type' effects take place in other sub-systems.

To restrict the internal N-flows, the *inputs* of N (in concentrates and chemical fertilizer) must be restricted. At a fixed milk production level, the input of concentrates can be restricted by covering the feed requirements of the herd as much as possible by homegrown crops, producing an optimum mix of energy and protein. On dry sandy soil this can be realized by growing maize for supply of energy (with low protein levels), in addition to grass for protein supply (Aarts et al., 1999b). N-fertilizer inputs can be restricted through efficient fertilization strategies. Moreover, N-use efficiency, expressed as the ratio of dry matter production to N input, can be maximized through optimum crop management. Introduction of these strategies in an integral system model suggested that at farm level at the average milk production intensity, an N-surplus of 128 kg N ha^{-1} yr^{-1} could be realized, associated with an anticipated N surplus on the soil balance of 78 kg N ha^{-1} yr^{-1} (Hilhorst et al., 2001).

2.3.2 Abatement of specific processes

N-excretion during grazing presents a risk for nitrate leaching to groundwater (Ryden, 1984, Van der Meer and Meeuwissen, 1989; Sauer and Harrach,1996). Urine N is deposited in mineral form in urine spots at rates estimated at 340-410 kg N ha^{-1} (Vertes et al., 1997; Vellinga et al., 2001) and is susceptible to leaching. Direct uptake is often hampered by damage caused by scorching and/or trampling. N-utilization from urine further declines with overlapping urine deposits (Haynes and Williams, 1993), of which the frequency increases with higher grazing intensities (Richards and Wolton, 1976; Vellinga et al., 2001). Therefore, rotational grazing is an effective strategy to restrict nitrate leaching, as it results in a large area with relatively low grazing-N rates, thus avoiding occurrence of spots with extremely high rates.

Slurry-N-application and urine N-excretion during grazing in late summer and early autumn enhance nitrate leaching (Titchen et al., 1993). At that time of the year, crop growth is slow, and so are water and N uptake rates, resulting in reduced utilisation of applied N (Sluismans and Kolenbrander, 1976; Cuttle and Bourne, 1993). Thus, excessive nitrate leaching can be prevented by avoiding N-applications and restricting grazing at the end of the growing season (Aarts et al., 2000b).

On sandy soils, nitrate leaching was found to be approximately twice as high on maize land than on grassland (Fraters et al., 1997). Hence, additional measures are required to prevent excessive nitrate leaching in maize. One of those is crop rotation. A crop rotation that includes a grassland phase in addition to an arable phase is preferred to continuous maize, as it may prevent the development of increasing susceptibility of the soil to nitrate leaching. Under continuous maize, the organic matter content in the top soil layer declines, due to the low return of organic matter compared to grassland, and to intensive soil tillage (Dam Kofoed,1982; Nevens and Reheul, 2003b). On dry sandy soils that are already low in organic matter, a further decline should be avoided to not jeopardize the water storage capacity of the rooting zone (Bell and van Keulen, 1995). However, upon transfer from the grassland phase to the arable phase, large quantities of mineral N are released, following ploughing in of the grass sod (Nevens and Reheul, 2002). This phase should be managed carefully, as it may lead to high rates of nitrate leaching (Whitmore et al., 1992; Francis, 1995; Hoffman, 1999; Goulding, 2000; Eriksen, 2001). To prevent mineral N levels exceeding the potential uptake capacity of maize, the N fertilizer dose for first year maize should be corrected for the expected release from the ploughed-in grass. Various studies have pointed out the positive role of catch crops in preventing leaching of residual mineral N from the profile after the harvest of maize (Schröder et al., 1992; Alvenas and Marstorp, 1993; Parente et al., 2003).

2.4 Identifying weak spots in the system with respect to nitrate leaching

System evaluation is required to examine whether system performance meets environmental and agronomic targets. Earlier studies have indicated the success of the management strategy implemented at 'De Marke' in reducing the levels of nitrate leaching (Boumans et al., 2001). However, the initial decline is not sufficient, as nitrate concentrations in the upper groundwater 'stabilized' around average levels of 55 mg l^{-1} (Aarts et al., 2001). As observed crop production was in good agreement with the levels predicted in system design (Hilhorst et al., 2001), and no indications of nutrient mining effects on crop productivity were found (Corré et al., 2004), a more specific focus on N-losses is warranted. Increased insight in the processes involved in nitrate leaching might result in identification of ways for further reduction.

Evaluation is carried out by relating plot management at 'De Marke' to measured nitrate concentrations in the groundwater, to examine whether correlations can be established with environmental conditions and related soil-Processes. Nitrate concentrations reflect the interacting effects of management practices, random variability of the system and weather conditions. These effects have to be disentangled for sound evaluation, which could be done by modelling the effects of random system variability. Boumans et al., (2001) concluded specifically for 'De Marke' that measured nitrate concentrations have to be corrected for variable dilution effects caused by the annual variability in precipitation surplus. Moreover, the effect of variability in groundwater level and water soluble organic matter content in groundwater (expressed as Dissolved Organic Carbon, DOC) should be taken into account, as denitrification is stimulated by the simultaneous occurrence of shallow groundwater levels and high contents of organic matter (Burford and Bremner 1975; Spalding and Exner 1993; Fraters et al., 2002).

2.5 Analysis

The analysis is focused upon the effects of crop rotation, grazing, N surplus and resowing on nitrate leaching. To account for known effects of system variability on nitrate leaching, possibly masking or confounding management effects, the effects of the precipitation surplus, groundwater level and dissolved carbon are also taken into account. Each year groundwater was sampled at the same location. During sampling the groundwater level was recorded and in each sample DOC was measured. In addition, for each sample a value for the precipitation surplus was calculated (Boumans et al., 2005).

The REML algorithm (method of residual maximum likelihood; Genstat 7.3 (2004)) was applied that estimates treatment effects and variance components in a linear mixed model and was developed especially for unbalanced datasets. Unbalanced datasets typically result from observational studies such as performed in the whole-farm approach followed at 'De Marke', in contrast to balanced datasets that follow from experimental designs where each treatment is randomly applied to a number of plots.

Linear mixed models (Table 5) are constructed by relating measured nitrate concentrations (dependent variable) to cropping history, grazing, N surplus, groundwater level, DOC concentration and precipitation surplus (fixed effects, Genstat 7.3). The unknown influence of sampling location is modeled as a random effect. To investigate whether rotation leads to higher nitrate concentrations, two models were developed (Table 5):

1 CROPS-model, consisting of four factors ($Crop_{-1}$, $Crop_{-2}$, $Crop_{-3}$, $Crop_{-4}$), referring to the four years before the year of nitrate sampling. Each factor can take either of two 'values', i.e. "grass" or "maize". Their effect in this model only depends on their temporal position (in years) with respect to the moment of nitrate sampling, irrespective of the rotation[1]. The CROPS-model allows predictions for every possible four-year sequence of grass and maize.
2 ROTATION-model, consisting of one factor (rotation phase), comprising 8 main sequences and 4 minor sequences of grass and maize which have been realized in 4 four-year periods. The ROTATION-model allows predictions for realized sequences only. Some realized crop sequences deviated from the normal rotation I or rotation II scheme. These were analysed as well, in addition to the 'normal' crop sequences. The natural factors were included in both the CROPS-model and the ROTATION-model as co-variables.

The differences between the results (95% confidence intervals) of the CROPS-model and the ROTATION-model were used to assess rotation effects. If rotation effects are significant, the CROPS-model will have larger and different confidence intervals than the ROTATION-model and modeled nitrate concentrations in first and second year maize will be higher than in subsequent maize years and in first and second year grassland higher than in subsequent grass years. No distinction was made between permanent grass and fourth year ley, as a preliminary analysis showed no effect (data not reported). To explore the effect of triticale, it was introduced as an additional crop type in the CROPS-model, and confidence limits for its contribution to the nitrate concentration in the four years following its cultivation were calculated.

1 $Crop_{-2}$ = Crop type two years before the year of sampling, either maize or grass (triticale). If it is grass, it refers to all plots that where used as grassland two years before sampling, irrespective of their position in the rotation. So, it includes permanent grassland as well as first, second and third year ley.

Table 5. *Models used to evaluate the influence of management variables.*

Model	Structure	
1: CROPS-model	Response:	Measured nitrate concentration
	Fixed variables:	Dissolved Organic Carbon + groundwater level + precipitation surplus + grazing* + N surplus[1)] + $Crop_{-1}$ + $Crop_{-2}$ + $Crop_{-3}$ + $Crop_{-4}$
	Random variable:	Sampling location
	$Crop_{-1}$:	Crop type 1 year before the year of sampling, maize or grass (triticale)
	$Crop_{-2}$:	Crop type 2 years before the year of sampling, maize or grass (triticale)
	Etc.	
2: ROTATION-model	Response:	Measured nitrate concentration
	Fixed variables:	Dissolved Organic Carbon + groundwater level + precipitation excess + rotation phase;
	Random variable:	Sampling location
	Rotation phase:	Realized crop sequence:
	Normal rotations	
	First year maize-1	Grass-Grass-Grass-Maize
	Second year maize-1	Grass-Grass-Maize-Maize
	Third year maize	Grass-Maize-Maize-Maize
	Fourth year maize	Maize-Maize-Maize-Maize
	First year grass-1	Maize-Maize-Maize-Grass
	Second year grass-1	Maize-Maize-Grass-Grass
	Third year grass	Maize-Grass-Grass-Grass
	Fourth year grass	Grass-Grass-Grass-Grass
	A-typical rotations	
	First year maize-2	Maize-Grass-Grass-Maize
	Second year maize-2	Grass-Grass-Maize-Grass
	First year grass-2	Grass-Maize-Maize-Grass
	Second year grass-2	Grass-Maize-Grass-Grass

[1)]*= added to the model separately to study effect.*

Three of the permanent grassland plots have been resown between 1996 and 2003. All observations of these three plots have also been used in the CROPS-model. For all model residuals, a percentile value was calculated. The influence of resowing on nitrate concentrations was assessed by comparing the percentile values of plot residuals before and after resowing.

The effects of grazing and N surpluses were investigated by introducing these variables jointly in the CROPS-model and by judging their effects on results (95% confidence limits) for grassland (grazing) and for maize and grass (N surplus).

Residuals appeared to be strongly non-Gaussian and bimodal, which makes the use of confidence limits and standard errors questionable. After transformation (power 1/3) and selection of responses (nitrate concentrations exceeding 1 mg/l), the residuals showed a more Gaussian distribution. Comparison of the results of the non-transformed and non-selected responses to those of the transformed and selected gave no reason to change the conclusions (data not reported).

3 RESULTS

3.1 Measured and modeled nitrate concentrations in grass and maize

Table 6 presents measured nitrate concentrations in different phases of the rotation and confidence intervals for means estimated by the CROPS-model and the ROTATION-model. Mean nitrate concentrations in groundwater were in general higher the year after maize than the year after grassland (Table 6, column 3). Disregarding the values that are represented by only 6 and 15 observations, the following pattern can be recognized (Table 6): The values for first, second and third year maize ($G_{-4}G_{-3}G_{-2}M_{-1}$, $G_{-4}G_{-3}M_{-2}M_{-1}$ and $G_{-4}M_{-3}M_{-2}M_{-1}$) are higher than for fourth year maize. The values for first and second year grass ($M_{-4}M_{-3}M_{-2}G_{-1}$ and $M_{-4}M_{-3}G_{-2}G_{-1}$) are lower than for fourth year grass. The mean nitrate concentration in groundwater was lower in the year after four years maize (45 mg NO_3^- l^{-1}) than after four years grass (63 mg NO_3^- l^{-1}).

The results of the CROPS-model (column 5) and the ROTATION-model (column 7) include corrections for effects of the natural factors, DOC, groundwater level and precipitation surplus, that may have influenced measured values according to the method summarized in Table 5. In the CROPS-model, the code G_{-n}, referring to grassland grown n years before the nitrate measurement, includes grassland in all positions in the rotation, i.e. first, second or third year ley, as well as permanent grass. The results of the ROTATION-model describe the combined effect of the crops in the four years prior to the nitrate measurement *and* rotation.

The results of the CROPS-model are rather similar to the measured nitrate concentrations except for G_{-4}+M_{-3}+M_{-2}+$M_{-1}$where model results are well below the mean of the measured values and for the a-typical rotations. The results of the ROTATION-model and the CROPS-model are similar, except for the a-typical rotations, and third ($G_{-4}M_{-3}M_{-2}M_{-1}$) and fourth ($M_{-4}M_{-3}M_{-2}M_{-1}$) year maize. For these rotations, the results of the ROTATION-model are in closer agreement with measured values than the results of the CROPS-model.

3.2 Effects of triticale, N-surpluses, grazing and resowing

Mean modeled nitrate concentrations and confidence intervals of the CROPS- and ROTATION-model for triticale are presented in Table 6 (section Rotations with triticale). The results of the CROPS- and ROTATION-model include correction for natural factors following the same method applied to maize and grass. Effects of triticale on mean measured nitrate concentrations in groundwater follow from comparison of rotations with triticale and rotations in which the position of triticale is occupied by maize (Table 6, column 3). Differences between rotations with triticale and rotations with maize are not consistent: (*i*) values for three years maize followed by triticale ($M_{-4}M_{-3}M_{-2}T_{-1}$) were higher than for four years maize ($M_{-4}M_{-3}M_{-2}M_{-1}$), (*ii*) values for two years grass, preceded by triticale ($M_{-4}T_{-3}G_{-2}G_{-1}$) were higher than those with maize on that position ($M_{-4}M_{-3}G_{-2}G_{-1}$), (*iii*) the difference between first year grass preceded by triticale ($M_{-4}M_{-3}T_{-2}G_{-1}$) and its maize equivalent ($M_{-4}M_{-3}M_{-2}G_{-1}$) was insignificant, (*iv*) values for two years maize followed by triticale ($G_{-4}M_{-3}M_{-2}T_{-1}$) were lower than for three years maize ($G_{-4}M_{-3}M_{-2}M_{-1}$). On average, the contribution of triticale to measured nitrate was somewhat higher than that of maize. However, the modeled concentrations from the CROPS-model are high compared to measured nitrate concentrations (measured values lie in the low range of the 95% confidence interval of the CROPS-model). Consequently, modeled concentrations from the CROPS-model for most rotations with triticale were much higher than those for their maize equivalents and the differences were larger than in measured values. Similarly to measured nitrate concentrations, the values of the CROPS-model for $G_{-4}M_{-3}M_{-2}T_{-1}$ were lower than for its maize equivalent. However, according to the CROPS-model the differences were marginal.

In the CROPS-model, the average effect of grazing in the period 1996-2003 at farm level was estimated for an average annual grazing intensity of 103 cow grazing days. Model results indicate that the nitrate concentration after four years grassland without grazing ($G_{-4}+G_{-3}+G_{-2}+G_{-1}$) would have been 23-49 mg l^{-1} (95% confidence limits) instead of 59-74 mg l^{-1} (Table 6, column 5).

Table 6: *Mean measured nitrate concentrations in different phases of the crop rotation*[1].

	Measured values		CROPS-model		ROTATION-model	
Rotation phase [2]	**Observations (number)**	**Means**	**Crop cultivated n years before nitrate measurement** [3]	**Estimated intervals for the mean**	**Rotation phase**[2]	**Estimated intervals for the mean**
Column number (1)	**(2)**	**(3)**	**(4)**	**(5)**	**(6)**	**(7)**
Normal rotations					*Normal rotations*	
$G_{-4}G_{-3}G_{-2}M_{-1}$	124	84	$G_{-4}+G_{-3}+G_{-2}+M_{-1}$	75-91	$G_{-4}G_{-3}G_{-2}M_{-1}$	68-87
$G_{-4}G_{-3}M_{-2}M_{-1}$	71	97	$G_{-4}+G_{-3}+M_{-2}+M_{-1}$	86-103	$G_{-4}G_{-3}M_{-2}M_{-1}$	81-104
$G_{-4}M_{-3}M_{-2}M_{-1}$	43	92	$G_{-4}+M_{-3}+M_{-2}+M_{-1}$	63-82	$G_{-4}M_{-3}M_{-2}M_{-1}$	77-107
$M_{-4}M_{-3}M_{-2}M_{-1}$	43	45	$M_{-4}+M_{-3}+M_{-2}+M_{-1}$	45-66	$M_{-4}M_{-3}M_{-2}M_{-1}$	36-54
$M_{-4}M_{-3}M_{-2}G_{-1}$	63	42	$M_{-4}+M_{-3}+M_{-2}+G_{-1}$	30-48	$M_{-4}M_{-3}M_{-2}G_{-1}$	28-53
$M_{-4}M_{-3}G_{-2}G_{-1}$	95	28	$M_{-4}+M_{-3}+G_{-2}+G_{-1}$	19-36	$M_{-4}M_{-3}G_{-2}G_{-1}$	10-31
$M_{-4}G_{-3}G_{-2}G_{-1}$	118	52	$M_{-4}+G_{-3}+G_{-2}+G_{-1}$	41-57	$M_{-4}G_{-3}G_{-2}G_{-1}$	43-62
$G_{-4}G_{-3}G_{-2}G_{-1}$	229	63	$G_{-4}+G_{-3}+G_{-2}+G_{-1}$	59-74	$G_{-4}G_{-3}G_{-2}G_{-1}$	61-79
			$G_{-4}+G_{-3}+G_{-2}+G_{-1}$(ung)[4]	23-49		
A-typical rotations					*A-typical rotations*	
$G_{-4}G_{-3}M_{-2}G_{-1}$	15	21	$G_{-4}+G_{-3}+M_{-2}+G_{-1}$	66-90	$G_{-4}G_{-3}M_{-2}G_{-1}$	10-62
$G_{-4}M_{-3}G_{-2}G_{-1}$	6	27	$G_{-4}+M_{-3}+G_{-2}+G_{-1}$	32-57	$G_{-4}M_{-3}G_{-2}G_{-1}$	0-56
$G_{-4}M_{-3}M_{-2}G_{-1}$	6	85	$G_{-4}+M_{-3}+M_{-2}+G_{-1}$	44-68	$G_{-4}M_{-3}M_{-2}G_{-1}$	45-125
$M_{-4}G_{-3}G_{-2}M_{-1}$	6	141	$M_{-4}+G_{-3}+G_{-2}+M_{-1}$	53-78	$M_{-4}G_{-3}G_{-2}M_{-1}$	89-170
Rotations with triticale						
$M_{-4}T_{-3}G_{-2}G_{-1}$	25	56	$M_{-4}+T_{-3}+G_{-2}+G_{-1}$	48-86	$M_{-4}T_{-3}G_{-2}G_{-1}$	47-85
$M_{-4}M_{-3}T_{-2}G_{-1}$	49	44	$M_{-4}+M_{-3}+T_{-2}+G_{-1}$	34-62	$M_{-4}M_{-3}T_{-2}G_{-1}$	34-61
$M_{-4}M_{-3}M_{-2}T_{-1}$	25	54	$M_{-4}+M_{-3}+M_{-2}+T_{-1}$	37-63	$M_{-4}M_{-3}M_{-2}T_{-1}$	35-72
$G_{-4}M_{-3}G_{-2}T_{-1}$	34	59	$G_{-4}+M_{-3}+G_{-2}+T_{-1}$	52-77	$G_{-4}M_{-3}G_{-2}T_{-1}$	44-76
A-typical rotations						
$T_{-4}G_{-3}G_{-2}M_{-1}$	6	82	$T_{-4}+G_{-3}+G_{-2}+M_{-1}$	41-118	$T_{-4}G_{-3}G_{-2}M_{-1}$	42-118

[1] Results of the CROPS-model: estimates of mean nitrate concentrations of maize and grass cropping on 'De Marke' related to the time interval (years) between crop cultivation and year of nitrate measurement (95% confidence limits); results of the ROTATION-model: estimates of the mean nitrate concentrations of maize and grass cropping on 'De Marke' related to the time interval (years) between crop cultivation and year of nitrate measurement *and* rotation (95% confidence limits) period 1996-2003

[2] The subscripts in the crop code represent the time interval (in years) between the moment of nitrate measurement and the time of cultivation. The sequence in the code represents the rotation. Thus, code $G_{-4}G_{-3}M_{-2}M_{-1}$ refers to nitrate measurements that were carried out one year after 'second year maize' was grown.

[3] The codes C_{-n} refer to the length of the time period n, in years, of cultivation of crop C before the nitrate measurement.

[4] Ungrazed grass.

Table 7 presents measured nitrate concentrations and modeled concentrations of the CROPS-model separately for categories of surpluses classified as 'low' and 'high', without distinguishing between grass and maize. However, grass is the dominant crop in the category with high surpluses and maize in the category with low surpluses (Table 3). Measured nitrate concentrations are higher for the category low surpluses than for high surpluses. The results of the CROPS-model also refer to grass and maize, but effects of the N surplus are corrected for crop effects (and for natural effects). Values of the CROPS-model are lower for low than for high surpluses.

Table 7: *Mean measured nitrate concentrations for plots with low and high N surpluses (maize and grass), results of the Crops-model: estimates of mean nitrate concentrations related to N surpluses (95% confidence limits).*

Category	Observations (number)	Measured (means)	Crops-model (estimated interval for the mean)
Low	382	70	33-50
High	388	54	51-68

The mean values of the percentiles of the residuals of modeled nitrate concentrations in the years before and after resowing permanent grassland plots (Table 8) show that for plot 8 resowing results in a clearly higher percentile value in the first year. The value of 82 in plot 14 does not indicate resowing effects as on this plot the percentiles are rather high in *all* years before and after resowing.

Table 8: *Mean of percentiles of the model residuals of measured nitrate concentrations in the years before resowing (-3, -2, -1), the year of resowing (0) and in the years following resowing (1 to 5).*

Time lapse from year of resowing	Plot number		
	K3 (autumn)	8 (spring)	14 (autumn)
-3	76	32	
-2	8	3	
-1	56	17	76
0	63	46	48
1	60	83	82
2	34	57	39
3	33		32
4			43
5			60

4 DISCUSSION

4.1 Crop-effects or rotation effects

The mean measured nitrate concentrations, the results of the CROPS-model and the results of the ROTATION-model (Table 6) show similar patterns: (*i*) nitrate concentrations are higher after four years grassland than after four years maize, (*ii*) nitrate concentrations are higher the first three years after grassland has been replaced by maize, (*iii*) concentrations are lower the first two years after maize has been replaced by grassland. These patterns thus refer to management effects, which are supposed to be disentangled from effects of natural factors.

The differences between the results of the CROPS-model and measured nitrate concentrations for a-typical rotations might be associated with the low number of observations and the resulting low accuracies. However, the differences also indicate that rotation effects indeed play a role when management deviates from the planned practices. In 'normal' rotations, the CROPS-model results deviate from measured nitrate concentrations in third-year ($G_{-4}M_{-3}M_{-2}M_{-1}$) and fourth-year ($M_{-4}M_{-3}M_{-2}M_{-1}$) maize, which would imply occurrence of rotation effects. However, the measured values and estimated means of the ROTATION-model for third-year maize are doubtful, as they are very high compared to $M_{-4}M_{-3}M_{-2}M_{-1}$. This difference can not be explained by rotation effects. Hence, this difference warrants further investigation to clarify the cause of the high nitrate values in third-year maize. This could reveal a 'weak spot' for nitrate leaching.

In general, the mean nitrate concentration for 'De Marke' is *not* significantly influenced by the rotation of maize and grassland, as the CROPS-model (Table 6) accurately estimates nitrate concentration for almost every possible sequence of maize and grassland. Hence, this mean nitrate concentration mainly depends on the areas of maize and grass, irrespective of their rotation. This conclusion is corroborated by the results of the ROTATION-model that estimates the lowest nitrate concentrations for $M_{-4}M_{-3}G_{-2}G_{-1}$ instead of for $G_{-4}G_{-3}G_{-2}G_{-1}$ or $M_{-4}M_{-3}M_{-2}M_{-1}$.

The observed nitrate concentrations are thus related to the specific management, as 'crops' in the models represent specific crop/management combinations. N fertilizer application to maize was reduced with 90 kg ha^{-1} in the first year and 45 kg in the second year to correct for N release from the ploughed-in grass sod. Apparently, this strategy effectively prevents nitrogen leaching in the grassland-maize transition. This is in agreement with results reported by Van Dijk (1999), who found that the N dose in the first and second year maize following grass could be reduced by 80-100 and 30-40 kg, respectively. Eriksen (2001) reported residual effects in first

year cereals following three year grassland, varying from 25 kg N ha^{-1} yr^{-1} for cut ryegrass, via 90-100 kg for grazed ryegrass to 115 kg for grass/clover mixtures. In the second year, residual effects were negative for cut grass, 40 kg N ha^{-1} yr^{-1} for grazed grass and 60 kg for grass/clover. The results of Nevens and Reheul (2002) point at somewhat higher residual effects, i.e. 124, 81 and 52 kg N in first, second and third year maize, respectively. Although the residual effects are variable, a zero N rate strategy in first year maize can prevent rotation-induced enhanced nitrate leaching.
Indirectly, crop rotation may be responsible for the relatively low levels of nitrate leaching in maize. Corré et al. (2004) concluded that inclusion of grass in the rotation prevents the rapid decline in soil organic matter content, associated with continuous maize, which leads to increased susceptibility of the soil for nitrate leaching (reduced buffering capacity for both water and nutrients).

4.2 Effects of maize and grass

The results of the CROPS-model suggest that maize results in lower nitrate concentrations than grassland, as can be deduced from Table 6 for sequences with maize in all years (-4, -3, -2, -1) before the year of nitrate measurement (45-66 mg NO_3^- l^{-1}) and sequences with grass in all years (59-74 mg l^{-1}). For permanent grassland without grazing, the CROPS-model with grazing added, estimated confidence limits (23-49 mg l^{-1}), below those for maize (45-66 mg l^{-1}).

The contribution of maize to nitrate leaching is smaller at 'De Marke' than in commercial farming in comparable conditions. The results of the CROPS-model, showing that the contribution of maize cropping to nitrate leaching is lower than that of grazed grassland, are not in agreement with observations in current practice (Fraters et al., 1997). In addition to the strategy of reduced fertilization in first and second year maize, this might be attributed to the strategy of growing catch crops. N-uptake in roots, stubble and leaves of Italian ryegrass at 'De Marke' is estimated at 108 kg ha^{-1} yr^{-1} (Aarts et al., 1994). This quantity would contribute to N-leaching, if catch crops were not grown. This is supported by results of the CROPS-model for triticale, showing a much larger contribution to nitrate leaching than of grass or maize. This is probably associated with its poor performance as a catch crop. Italian ryegrass as a catch crop is sown between the maize shortly after germination. As a result, its biomass in winter is better developed than that of triticale that is sown shortly after harvest.

The general perception of increased nitrate concentrations caused by replacing grass with maize neglects long-term effects. Modeled and measured nitrate concentrations are low when maize was present four and three years before nitrate measurement and higher when grassland was present in that period. Measurements indicate that nitrate concentrations are low in the year following grass and high in the year following maize. But according to the CROPS-model, high nitrate concentrations are associated with grass in the earlier years (3 and 4 years before the measurement) and low nitrate concentrations with maize in the earlier years (Table 8). Thus, when grass is followed by maize (as may happen on commercial farms, but also in field trials), leaching effects of maize cannot adequately be quantified by monitoring the groundwater one year after maize cropping. The 'memory effect' of the soil may cause delayed grass effects that may erroneously be attributed to maize.

4.3 Other effects

4.3.1 Grazing

According to the CROPS-model, grazing at the intensity practiced at 'De Marke' in the period 1996-2003 contributed about 30 mg l^{-1} of nitrate to the measured concentration in the upper groundwater of grassland. Assuming a linear relation between grazing intensity and nitrate concentration, this result suggests an urgent need to reduce grazing intensity. However, adaptation of the grazing system, leading to a reduction in nitrate leaching, while maintaining the grazing intensity, would be even more interesting. A further reduction in daily grazing time below the current six hours is practically not feasible, as the beneficial effects of grazing (assuming these to be proportional to length of the grazing period) would no longer warrant the required (unchanged) labor investments. A feasible way to reduce grazing-related leaching might be to stop grazing earlier in autumn. Currently, grazing continues until mid-September, the exact moment depending on grass production in late summer and autumn. This may be sub-optimal in terms of nitrate leaching, as utilization of nitrogen from urine deposits declines with later excretion (Deenen and Middelkoop, 1992; Cuttle and Bourne, 1993; Titchen et al., 1993; Lord, 1993; Van der Putten and Vellinga, 1996). Moreover, according to Vellinga and Hilhorst (2001), grazing management could be further optimized. They point out that the recommended yield for grazing can often not be realized in September and October, which leads to N-contents exceeding 30 g kg^{-1} in the grazed material. This observation supports the suggestion to end the grazing season earlier. Another aspect that deserves attention is the composition of the ration during grazing, which might be a major cause of excessive N deposition in grazed grassland (Van Vuuren and Meijs, 1987; Van Vuuren et al., 1993; Valk, 1994). Although the strategy at 'De Marke' is to prevent intake of protein in excess of the requirements, during grazing high protein intake levels may have occurred.

The ultimate abatement of grazing-related nitrate leaching is to stop grazing completely. Although in a production environment that aims at high production levels within the boundaries of environmental standards, this may appear an attractive option, for 'De Marke' it is the last resort. Grazing is appreciated for its contribution to an attractive rural landscape, while it enhances animal health and welfare, although the exact consequences of full time stabling are far from clear. Further research in this field is urgently needed, because if, in spite of further optimization of grazing management, full time stabling will appear inevitable, the consequences for a variety of aspects will have to be explored at farming system level.

4.3.2 N-surpluses

Measured nitrate concentrations were lower in grass and maize with high surpluses than in grass and maize with low surpluses. This can be attributed to the fact that grass (with lower measured nitrate concentrations than maize) is the dominant crop in the category with high surpluses, while maize (with higher measured nitrate concentrations) is dominant in the category with low surpluses. Hence, crop effects and surplus effects both affected measured nitrate concentrations. The difference in surpluses between grass and maize is caused by the crop-specific management with different input levels for grass and maize (Table 3). The results of the CROPS-model indicate the opposite, with higher modeled values for high surpluses than for low surpluses. In the CROPS-model, crop effects and effects of surplus are disentangled. It should be stressed that the variability in surpluses evaluated in the CROPS-model is not dependent on crop type and position in the rotation (Table 3). Instead, the variability is associated with variability in weather conditions, plot characteristics, such as phosphate availability or organic matter content (Corré et al., 2004) and sometimes pests and diseases. This variability in N-surpluses appeared a significant factor in nitrate leaching, that would not have been detected if effects of crop and of surplus would not have been disentangled. This is clearly illustrated by the contradiction between measured nitrate concentrations and results of the CROPS-model (Table 7). Lack of data prevented separate modelling of the specific effect of the surpluses in maize and grass. However, extensive evaluation of field experiments (Ten Berge, 2002; Bobe et al., 2004) indicated that the risk of nitrate leaching at a given level of exceedence of the critical N-rate (defined here as the fertilizer application rate necessary to bring available N at the level of potential crop uptake) is much higher in maize than in grassland, which is attributed to the much higher denitrification rates in grassland (Colbourn and Dowdell, 1984; Paustian et al., 1990). Therefore, restricting incidences of exceeding critical N-rates in maize, which may have occurred in $G_{-4}M_{-3}M_{-2}M_{-1}$, might lead to further reduced nitrate leaching. An option might be to improve plot-specific estimates of potential N-uptake rates, as a basis for plot-specific fertilizer recommendations. This option will have to be explored in further research.

4.3.3 Resowing

The number of resowings (3) is too low to allow firm conclusions from the results. However, in two of the three situations, where resowing took place in autumn, nitrate concentrations in groundwater in the years after resowing were not markedly affected. The third resowing, in spring, resulted in high nitrate levels one year after resowing. These results are in disagreement with the general perception, supported by experimental evidence, that resowing in spring has no effect, whereas resowing in autumn increases nitrate leaching (Francis, 1995; Velthof and Hoving, 2004; Seidel et al., 2004).

4.4 Implications for N-management at farm level

To reduce nitrate leaching, it is advisable to replace triticale by another crop and to abate grazing-related contributions, for instance by earlier ending the grazing season. A next step could be to reduce the share of maize in roughage production, as nitrate leaching in maize is higher than in non-grazed grassland. Such an adaptation will have a strong impact on the flows of energy and nutrients within the production system. Therefore, for sound judgment of this option, a thorough (re-)analysis is required of changes in productivity of roughage, N requirements of the crops, N inputs through purchase of concentrates and changes in crop transpiration. Results of that analysis will be reported in a follow-up paper.

5 CONCLUSIONS

No indications were found for enhanced nitrate concentrations due to rotation of grass and maize. Nitrate concentrations caused by triticale were higher than those caused by maize. This high value was attributed to the poor growth of the crop in the period that it should function as a catch crop after harvest of maize. Grazing contributes up to 30 mg NO_3^- l^{-1} on grassland. As grazing management and intensity have been optimized already to restrict nitrate leaching, this result underpins the need to develop sustainable grazing methods on soils susceptible to nitrate leaching.

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CHAPTER 3

EFFECTS OF LAND USE AND MANAGEMENT ON SOIL-N MINERALIZATION IN AN INTENSIVE DAIRY PRODUCTION SYSTEM

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ABSTRACT

This paper describes the dynamics of soil-N mineralization in intensive dairy farming system 'De Marke' on dry sandy soil in the Netherlands. We hypothesized that knowledge of the effects of crop rotation on soil-N mineralization and of the spatial and temporal variability of soil-N mineralization in a farming system can be used to better synchronize N application with crop-N requirements, and hence to increase the recovery of applied N and to reduce N losses. Soil-N mineralization was recorded continuously in the soil layer 0 to 0.30 m, from 1992 to 2005, using a sequential *in situ* coring technique on 6 observation plots, of which 2 were located in permanent grassland and 4 in crop rotations with a 3 year grassland phase and an arable phase of 3 or 5 years, dominated by maize. Average annual soil-N mineralization was highest under permanent grassland: 381 kg ha^{-1} and lowest under 3^{rd} or more years arable crops: 184 kg ha^{-1}. In ley, soil-N mineralization increased in the order: 1^{st} year, 2^{nd} year, 3^{rd} year grassland and in arable crops after grassland mineralization decreased in the order: 1^{st} year, 2^{nd} year, 3^{rd} or more years. Total mineral N input, i.e. the sum of N mineralization and mineral N supply to soil, exceeded crop-N requirements in 1^{st} year maize and did not meet the requirements in 1^{st} year ley. Changing N fertilization from 1^{st} year arable crops to ley would improve synchronization of N availability and requirements. Mineralization in winter, outside the growing season, was 77 kg N per ha in maize and 60 kg N per ha in grassland. This points at the importance of a suitable catch crop to reduce the susceptibility to N leaching. Temporal and spatial variability of soil-N mineralization was high and could not be related to known field conditions. This limits the extent to which N fertilization can be adjusted to soil-N mineralization. Variability increases with the magnitude of soil-N mineralization. Hence, situations with high soil-N mineralization may be associated with high risks for N losses and to reduce these risks a strong build-up of soil organic N should be avoided. This might be realized, for instance, by fermenting slurry before application on farmland to enhance the fraction mineral N in slurry at the expense of organic N.

1 INTRODUCTION

Over the past decades, inputs of nitrogen (N) in intensive dairy farming systems have increased more than their outputs, leading to high surpluses. Particularly on light sandy soils, the associated N losses to the environment in the form of nitrate leaching, have increased (Aarts et al., 2000; Carton and Jarvis, 2001; Schröder et al., 2005), threatening water quality (Hosper, 1997; Boumans et al., 2005; Schröder et al., 2009). To reduce N surpluses and nitrate leaching, management strategies for dairy farming systems should aim at enhanced N utilisation.

Soil-N mineralization, i.e. the transformation of N immobilized in organic components in the soil (SON) to mobile soil mineral N compounds (SMN), NO_3^- or NH_4^+, plays an important role in the partitioning of system N between crop uptake (contributing to N utilisation) and losses (i.e. denitrification and leaching). Mineralization in the soil is highly variable in time and in space (Vinther, 1994) under the influence of soil characteristics (Hassink, 1992), environmental and management factors, such as temperature, soil water content, cropping pattern and present and past soil management (Gill et al., 1995). When soil-N mineralization is not synchronized with crop-N requirements, high SMN contents may be the result, increasing the risks for denitrification and/or leaching (Pollmer et al., 1979; Steffens and Vetter, 1984; Nevens and Reheul, 2002; Kayser et al., 2011). Insight in the temporal and spatial patterns of soil-N mineralization in dairy farming systems allows identification of periods and places in the system when and where release of SMN is excessive. This knowledge could indicate required management adjustments to anticipate mineralization. Earlier work on soil-N mineralization focused on effects of soil characteristics on mineralization rates, the design of methods to measure N mineralization in the field (see below) and effects of crop rotation (e.g. Vinther, 1994). Fewer studies focus on the effects of weather conditions and land management, whereas year-round observations that relate N release to crop uptake capacity at farm level and in specific parts of the soil/crop system are scarce (Crohn, 2006).

The objective of this study was to measure and analyze soil-N mineralization dynamics in a whole dairy farming system and to discuss implications for N management aiming at minimizing N losses to the environment on dairy farm 'De Marke' but also more generally, on other dairy farms on dry sandy soils. The focus was on total annual soil-N mineralization and its intra-annual dynamics. The main questions were: (*i*) what is the effect of crop rotation on soil-N mineralization? (*ii*) what is the spatial and temporal variability of soil-N mineralization? (*iii*) to what extent are the dynamics of soil-N mineralization synchronized with crop-N uptake? In order to answer these questions, data on net soil-N mineralization (that is mobilization minus immobilization), recorded continuously on six 'permanent' plots at experimental dairy farm 'De Marke' between 1992 and 2005, were analysed every month and every year. Soil-N mineralization was estimated using a sequential in situ coring technique (Raison et al., 1987) which is further justified in the next section.

2 MEASUREMENT OF SOIL-N MINERALIZATION

Various methods have been proposed to estimate soil-N mineralization. To characterize soils in terms of their mineralization capacity, incubation techniques under standardized laboratory (*ex-situ*) conditions and chemical tests have been applied (Velthof, 2003; Bushong et al., 2007; Ros et al., 2011). However, to explore the effect of field conditions, *in situ* techniques are required. Of the available *in situ* incubation techniques, the following have been used quite frequently: the buried-bag technique (Subler et al., 1995; Cusick et al., 2006), the coring technique (Raison et al., 1987), the resin trap technique (Hanselman et al., 2004) and incubation containers (Jarvis et al., 2001). A common characteristic of the methods is that net soil-N mineralization is estimated from the increase in mineral N in the incubated soil. The soil is more or less isolated from the surrounding soil to prevent N uptake by plants. In the coring technique, the incubation tube is covered with a lid to prevent atmospheric inputs and leaching. In contrast, in the resin trap technique, tubes are not covered and leached N is captured. Consequently, with the resin trap technique inputs of atmospheric N can not be distinguished from N released by mineralization, which is a constraint for its use in regions with high atmospheric N deposition, such as the Netherlands. A major question for all the methods is whether temperature, moisture and aeration in the incubated soil fluctuate similarly to those in untreated soil (Hatch et al., 2000; Jarvis et al., 2001; Abril et al., 2001; Isaac and Timmer, 2007). The buried-bag technique cannot be carried out without perturbation of the incubated soil which may affect conditions and, hence, its soil-N mineralization (Hassink, 1992). Particularly in poorly drained soils it has been reported that soil water content in incubated soil may be higher than under undisturbed field conditions. Part of the mineralized N may escape in gaseous forms by denitrification, again, particularly under wet conditions that are conducive to denitrification (Hatch et al., 1998). In deep-drained soils these problems are less severe (Jarvis et al., 2001). Moreover, they can be minimized by restricting the length of the incubation period of tubes to one month (Hook and Burke, 1995) and by allowing ventilation of free air through holes in the side of the tubes, just above the soil surface. Both measures minimize differences in soil water content inside and outside the tubes. Finally, upon inserting the cores, plant roots may be severed and these dead roots may affect N mineralization rates in the core (Binkley and Hart, 1989). Sieving out the roots from the soil is inappropriate as that affects soil texture, and hence mineralization (Hassink, 1992; Whitehead, 1995). The on-going methodological discussions on mineralization indicate that each of the methods has its limitation(s) and that different methods may be optimal under different circumstances. On dry sandy soil the coring technique appears most adequate, as (*i*) conditions inside the tubes are quite similar to those in the surrounding soil, provided the tubes are replaced frequently to minimize effects of their divergent soil water contents and (*ii*) soil structure and texture are hardly affected by perturbation. Still, the problem of severed plant roots cannot be completely resolved and therefore critical evaluation of results remains vital.

3 MATERIALS AND METHODS

3.1 Experimental farm 'De Marke'

3.1.1 Site characteristics

'De Marke' is situated in the eastern part of the Netherlands on very light sandy soil, characterized by a 0.30 m anthropogenic upper layer, with an average organic matter content of 4.4% and a bulk density of 1300 kg m^{-3}, overlying a layer of yellow sand, very low in organic matter and hardly penetrable by roots (Aarts, 2000). Soil water content is usually at field capacity in spring (0.18 g per g oven dry soil) and decreases to 0.12 in mid-summer, or even to 0.10-0.05 in dry summers. Groundwater depth is 1 to 3 meter below soil surface, i.e. upward water transport from the saturated zone to the root zone is negligible in periods that potential evapotranspiration exceeds precipitation. In the experimental period (1993-2005), precipitation (mean annual 792 mm) was on average evenly distributed over the year (Table 1). However, in specific years the distribution fluctuated between 47 mm in April and 82 mm in July. In the October-March period, average temperature was 4.7 °C and in April-September 14.1 °C. December was the coldest month with an average temperature of 2.3 °C and July the warmest with 17.3 °C (Table 1).

Table 1: *Monthly temperature and precipitation, means and standard deviations (stdev) of 1993-2005.*

	Temperature (˚C)		Precipitation (mm)	
Month	**Mean**	**Stdev**	**Mean**	**Stdev**
January	2.5	2.1	66	37
February	3.2	2.8	54	28
March	5.7	1.6	51	31
April	8.7	1.7	49	21
May	12.5	1.5	72	35
June	15.1	1.2	70	44
July	17.3	1.8	82	40
August	17.0	1.6	72	33
September	13.8	2.1	87	62
October	9.7	2.2	66	43
November	5.2	1.9	60	32
December	2.3	2.1	71	36

3.1.2 Farm layout

As farm layout is described in detail in Verloop et al. (2006), a summary is given here. The farm area (55 ha) is divided into permanent grassland (11 ha) and two crop rotations: ROTI and ROTII (Table 2). The rotation scheme of the arable phase was modified slightly in the course of time. From 1993 until 1996, fodder beet was the first arable crop to succeed the grassland phase. From 1996 till 1999, maize was the only arable crop, after which triticale and spring barley were introduced as the last crop in the arable phase. On maize land, Italian ryegrass was sown as a catch crop between the rows in June and ploughed-in in the first week of March. A total of 31 ha of grassland were available for grazing by 80 milking cows and young stock. A rotational grazing system was applied, in which the cattle grazed for 5-7 days on individual plots of 1-3 ha, after which they were transferred to a next plot.

Fertilizer management aimed at satisfying crop requirements. Target crop-N uptake was set to the value that can be realized on dry sandy soils without violating threshold values for acceptable losses of N (Aarts et al., 1992). In calculating manure-N requirements for maize, a net release of N from the ploughed-in grass sod of 90 kg ha^{-1} in the first and 45 kg in the second year was taken into account (Aarts et al., 1992), resulting in rates in 1st and 2nd year maize that are lower than in 3rd and 4th year maize (Table 3). On grassland, cattle slurry application started on

March 15th and ended August 1st. On maize land, cattle slurry was applied at the end of April, just before sowing. Since 2003, slurry was fermented before application. From 1993 to 2003, 49% of the N in slurry was in organic form, which decreased to 40% in 2004 and 2005.

Table 2: *Crop sequences in the two types of rotation and in permanent grassland.*

Soil use	Area (ha) 1993-2005	Scheme
Permanent grassland	11	grassland -> etc.
ROTI	27	3 years grassland -> 3 years arable crops, etc.
ROTII	17	3 years grassland -> 5 years arable crops, etc.
Total	*55*	

Table 3: *Area (ha) of permanent grassland and distinct phases of crop rotation at 'De Marke' and N inputs (kg ha^{-1}, means for 1993-2005).*

Crops	Area (ha)	Manure[4)]	Mineral fertilizer	Clover	Deposition
1st yr arable crops[1)]	6	43	-	0	41
2nd yr arable crops[2)]	6	92	-	0	41
3rd -5th yr arable crops[3)]	11	129	-	0	41
1st yr ley	8	307	80	56	41
2nd yr ley	7	307	80	56	41
3rd yr ley	7	319	80	52	41
Permanent grassland	11	273	104	28	41

1) Maize (less frequently: fodder beet).
2) Maize.
3) Maize (less frequently triticale, spring barley).
4) 49% in the form of organic N.

3.2 Data collection

3.2.1 Observation plots

The total area of 'De Marke' is divided in 30 parcels of 1-3 ha each. Each of six 'permanent' observation plots for mineralization is located in a different parcel, such that in combination they are as representative as possible for soil characteristics and land use at 'De Marke'. Their soil characteristics (Table 4) were similar to those of the average farmland, however, the rotation schemes of the plots (Table 5) were somewhat different from those of ROTI and ROTII (Table 2). In 1998 and 1999 mineralization was recorded on permanent grassland only (Table 5). The observation plots on grassland were accessible to cattle, hence they were grazed during grazing periods. Grazing frequency was governed by the rotational grazing system.

Table 4: *Characteristics of the soil: observation plots (1-6) and farm average (average across the entire farm), layer 0-0.20 m.*

Plot	1	2	3	4	5	6	Farm
pH-KCl[1]	5.3	5.0	5.0	4.9	5.4	5.6	5.2
Organic matter content (mass %)[1]	3.1	3.3	3.1	3.6	3.7	3.4	4.4
N content (g kg^{-1} dry soil)[1]	1.6	1.2	1.1	1.2	1.2	1.5	1.3
Soil water content spring (mass %)[2]	18	20	14	14	19	18	-
Soil water content summer (mass %)[2]	13	12	10	10	15	14	-
Soil water content autumn (mass %)[2]	17	17	13	14	18	17	-

[1] 1993.
[2] Means for 1993-2005.

Table 5: *Land use and cultivated crops on the six 'mineralization observation plots' at 'De Marke'; underlined codes: no observation.*

Year \ Plot	1	2	3	4	5	6
Land use	PG	PG	ROTI	ROTI	ROTII	ROTII
1993	PG	PG	M^{3+}	FB^{1}	M^{2}	M^{3+}
1994	PG	PG	M^{3+}	M^{2}	M^{3+}	M^{3+}
1995	PG	PG	M^{3+}	M^{3+}	TG^{1}	TG^{1}
1996	PG	PG	TG^{1}	TG^{1}	TG^{2}	TG^{2}
1997	PG	PG	TG^{2}	TG^{2}	TG^{+}	M^{1}
1998	PG	$\underline{\text{PG}}$	TG^{3}	TG^{3}	$\underline{M}^{1}$	$\underline{M}^{2}$
1999	PG	$\underline{\text{PG}}$	M^{1}	M^{1}	$\underline{M}^{2}$	$\underline{M}^{3+}$
2000	PG	PG	M^{2}	M^{2}	T^{3+}	M^{3+}
2001	PG	PG	M^{3+}	T^{3+}	TG^{1}	M^{3+}
2002	PG	PG	T^{3+}	TG^{1}	TG^{2}	M^{3+}
2003	PG	PG	TG^{1}	TG^{2}	M^{1}	M^{3+}
2004	PG	PG	TG^{2}	TG^{3}	M^{2}	B^{3+}
2005	PG	PG	TG^{3}	M^{1}	TG^{1}	TG^{1}

Rot: rotation (see text for explanation).
PG: permanent grassland.
TG^{x}: ley, x^{th} year (x =1, 2 or 3).
M^{x}: maize, x^{th} year (x = 1, 2 or 3 and more).
FB^{1}: fodder beet, 1st year.
T^{3+}: triticale, 3rd year and more.
B^{3+}: spring barley, 3rd year and more.

3.2.2 Monitoring

N-mineralization was measured in plots of 20 m × 20 m, using the method as described by Raison et al. (1987). At the start of an incubation period, in each plot 18 tubes with a diameter of 2.5×10^{-2} m were inserted in the soil to a depth of 0.2 m in grassland and 0.3 m in arable land. Each tube was closed with a lid to prevent N leaching during rainfall and N deposition. Aeration was provided through perforations in the side of the tube above ground level. Directly following insertion of the tubes, three soil samples were taken, as close as possible to the tube, to determine SMN. After an incubation period (1 month), the tubes were removed from the soil. To determine SMN, the soil was extracted with a $CaCl_2$ solution, filtered and analysed with a photometer. Net N mineralization was calculated as the difference in SMN between the incubated soil and the soil sampled at the start of the incubation. Of three replicates per plot, each consisting of 6 cores of which the soil was mixed after incubation, the replicates with the lowest and highest value were excluded from the analysis. Soil water content was measured each month, when the cores were replaced. The new tubes were placed close to the location of the preceding tube, however, successive tubes could be about a meter apart. Daily weather conditions were recorded at a weather station at the farm.

3.3 Statistical analysis

The dynamics of soil-N mineralization were analysed through evaluation of (*i*) mineralization in permanent grassland and distinct phases of the crop rotation (see below for a more detailed description), (*ii*) intra-annual dynamics in ley, permanent grassland and the most important crop (maize) and (*iii*) temporal and spatial variability.

3.3.1 Effects of crop rotation

Results of observation plots were related to the phase of the plots in the crop rotation. Management factors such as fertilization and tillage are strongly associated with crop rotation. Therefore, in the analysis, the effect of the combination of crop sequence and the associated management (i.e. treatment) was investigated. We distinguished 1st, 2nd and 3rd year ley and 1st, 2nd and 3rd year or older arable crops. No distinction was made among the different arable crops growing in the same arable phase. Hence, 1st year maize and 1st year fodder beet were both categorized as 1st year arable crop and 3rd year or older maize, 3rd year or older triticale and 3rd year or older spring barley where all categorized as 3rd year and older arable crop (Table 5).

The REML (residual maximum likelihood; VSN International 2012) was used to analyze the data, with the factors ‘year’ and ‘observation plot within year’ as the random model and variables treatment as the fixed model.

3.3.2 Intra-annual dynamics

Intra-annual dynamics were analysed for permanent grassland, ley and maize, using the REML procedure, according to a similar procedure as described in Sub-section 3.3.1, with the factors 'year' and 'observation plot within year' as the random model and variables treatment as the fixed model. Crops other than maize and grass were not included, because they were not grown frequently enough.

The data were analysed for 1^{st} year maize land and 2^{nd} and 3^{rd+} year maize land separately, by comparing the means of all available data for the two categories: 4 observations for 1^{st} year maize land and 22 for older maize land. Similarly, the data were analysed separately for 1^{st} year grassland and 2^{nd} and 3^{rd} year grassland by comparing the means of all available data for the two categories: 9 observations for 1^{st} year grassland and 11 for older grassland.

3.3.3 Spatial and temporal variability

Spatial variability was analysed by establishing the standard deviations of the mineralization measured in different plots with the same crop in the same year (sd_{plot}). Temporal variability was analysed by establishing the standard deviations of the mineralization measured in the same plot, for the same crop in different years (sd_{year}). Regression analysis was used to explore the extent to which spatial and/or temporal variability was related to soil and weather conditions: (*i*) soil water content, (*ii*) temperature sum, i.e. the sum of temperatures (means of 24 hours) above 0 °C in a period (month, year), (*iii*) precipitation and (*iv*) their interactions.

4 RESULTS

4.1 Effects of crop rotation

Mean soil-N mineralization was highest on permanent grassland (Table 6). On ley, mean soil-N mineralization increased with age, i.e. it increased in the order 1st year grassland, 2nd year grassland, 3rd year grassland. In contrast, on arable land, soil-N mineralization decreased with increasing age, i.e. it decreased in the order 1st year arable crop, 2nd year arable crop, 3^{rd+} year arable crop. However, differences were significant in only a few situations (Table 6). The standard deviations of mean mineralization quantify the high variability.

Table 6: *Mean mineralization (kg N ha^{-1} yr^{-1}) in permanent grassland and in distinct phases of the crop rotation.*

Land use[1)]	Number of observations	Mean	Standard deviation	Homogeneous groups[2)]		
				low	int.	high
PG	24	381	133			c
TG^1	9	252	93	a	b	
TG^2	7	329	87		b	c
TG^3	3	360	74		b	c
AR^1	5	318	93		b	c
AR^2	7	205	97	a	b	
AR^{3+}	17	184	84	a		
Average		*290*	*94*			

[1)] AR^x: Arable crops, for meaning of other acronyms, see Table 5.

[2)] Indicated as non-significant items according to t-test with $P<0.05$; int.: intermediate.

The spatial variability (variability among plots) was high compared to absolute annual soil-N mineralization, as can be seen by comparing the variability in Table 7 with the absolute values in Table 6. The same was observed for temporal variability (year-to-year variability). No significant relationships were found between annual soil-N mineralization and: (*i*) soil water content (data not shown), (*ii*) precipitation (data not shown), (*iii*) temperature and (*iv*) their interactions. This is illustrated for temperature in Figure 1, showing no significant relation between annual soil-N mineralization under permanent grassland and Tsum.

Table 7: *Standard deviation of mineralization (kg ha^{-1} yr^{-1}) in plots with the same treatment (crops, rotation phase and year), sd$_{plot}$ (means for 1993-2005) as a measure of spatial variation and standard deviations of the mineralization in different years, sd$_{year}$ as a measure of temporal variation (for meaning of acronyms, see Table 5).*

Rotation	Sd_{plot}	Sd_{year}
PG	112	133
TG[1]	-	93
TG[2]	90	97
TG[3]	46	84
AR[1]	105	93
AR[2]	88	87
AR[3+]	-	128
Pooled average	*96*	*120*

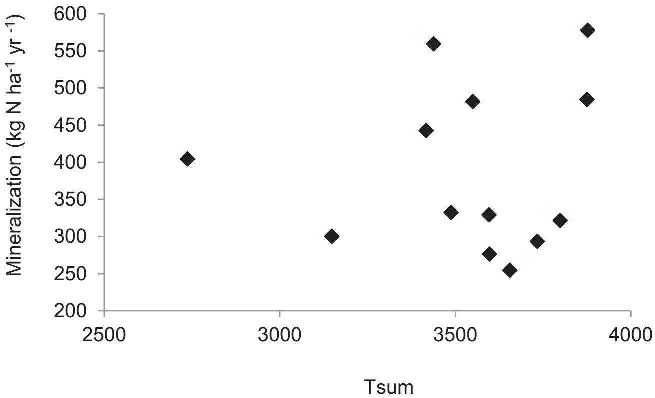

Figure 1: *Annual mineralization under permanent grassland as a function of the annual sum of daily temperatures above zero (Tsum, degree days).*

4.2 Intra-annual dynamics

Soil-N mineralization on grassland strongly increased from winter until mid-summer, and attained its highest values, i.e. 40-60 kg N ha^{-1} $month^{-1}$ in July and August (Figure 2). On maize land on the other hand, soil-N mineralization was approximately constant throughout summer with relatively high values in spring and low values in late summer, autumn and winter. Although soil-N mineralization on grassland was consistently higher than on maize land, differences were most pronounced from May until October.

On maize land, soil-N mineralization in the 1st year following grassland was higher early in the season than in later-year maize land, while later in the season, mineralization in 1st and later years maize land followed a similar pattern (Figure 3). The opposite was observed on ley, where soil-N mineralization under 1st year grassland started to increase later in the year than under later-year grassland, with significant differences in April and May (Figure 4).

No significant relationships were found between soil-N mineralization in specific months and (*i*) soil water content (data not shown), (*ii*) precipitation (data not shown) and (*iii*) temperature. This is illustrated for temperature in Figure 5, using the results of permanent grassland. Tsum in February ranged from about 20 to 200 d°C and soil-N mineralization ranged from 0 to over 30 kg per ha. Tsum and mineralization, however, were not clearly related in this month, and neither in March, April, September, October and November.

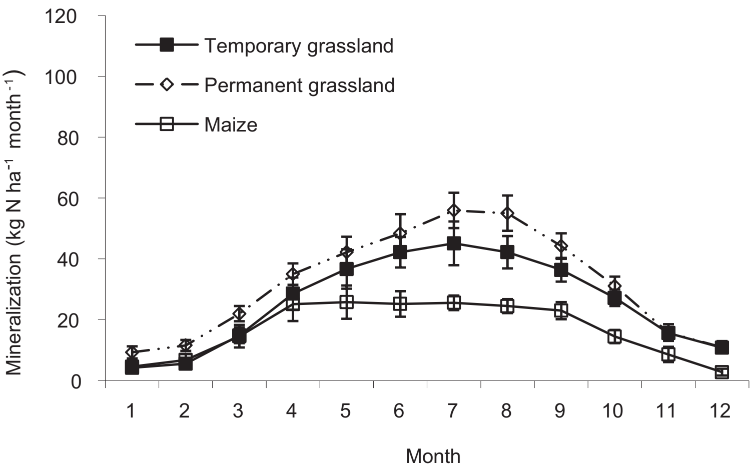

Figure 2: *Time course of soil-N mineralization (means of 1993-2005) during a growing season in permanent grassland, ley and maize. Vertical bars indicate standard errors of the means.*

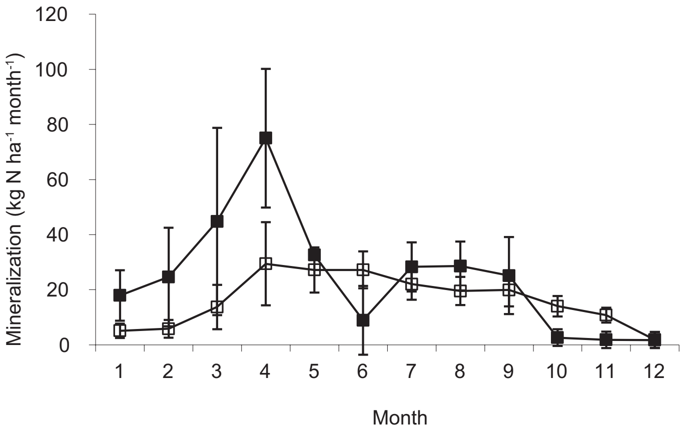

Figure 3: *Time course of soil-N mineralization (means of 1993-2005) during a growing season in first year maize (black squares, n = 4) and subsequent-year maize (averages of 2nd and 3^{rd+} year, open squares, n = 22). Vertical bars indicate standard errors of the means.*

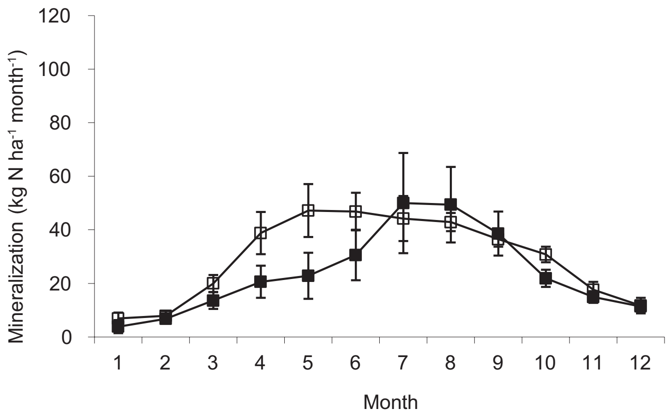

Figure 4: *The dynamics of N mineralization (means of 1993-2005) within a growing season in first year ley (black squares, n = 9) and later-years ley (open squares, n = 11). Vertical bars indicate standard errors of the means.*

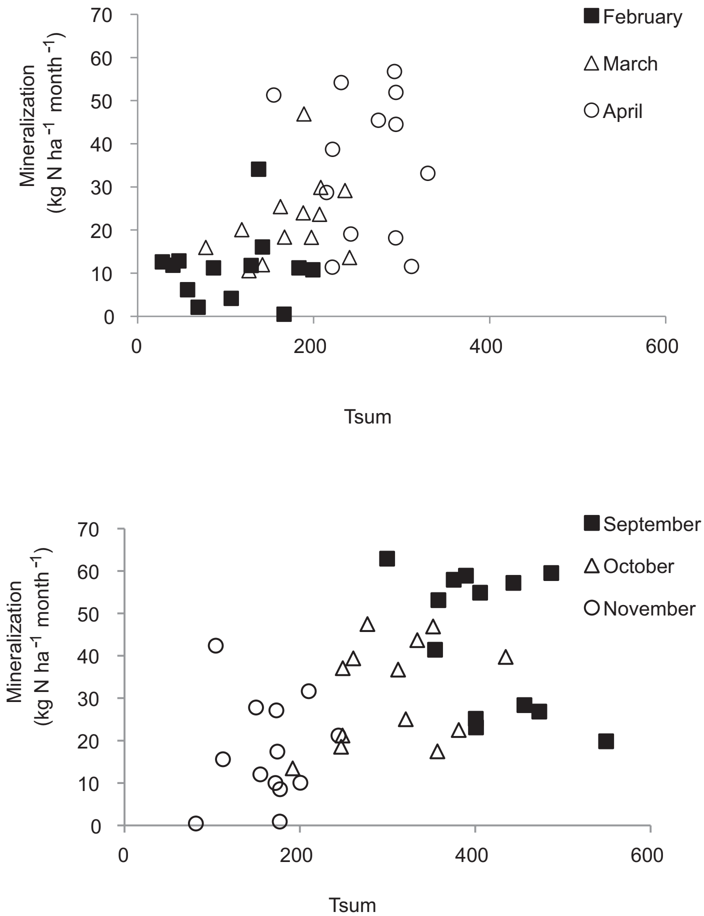

Figure 5: *Mineralization in permanent grassland in distinct months as a function of the sum of daily temperatures above zero in these months (Tsum, degree days). Each symbol presents the results of one year.*

5 DISCUSSION

5.1 Long-term average for all treatments

Long-term (13 year) average soil-N mineralization of the six observation plots of 290 kg ha^{-1} yr^{-1} (Table 6) can be considered representative for farming system 'De Marke', as the plots were evenly distributed over the total area and land use and management were representative for the system. Estimated soil-N mineralization at farm level (corrected for land cover of crops, Table 6) amounts to 288 kg ha^{-1} yr^{-1}. This is somewhat high, but not extreme in comparison with the range reported in literature for agricultural soils using the field incubation method. Vinther (1994) reported values of 73-93 kg N ha^{-1} yr^{-1} under barley on a coarse sandy soil in Denmark at a fertilizer dose of 120 kg N ha^{-1}, with and without catch crops, Gill et al. (1995) values from 135 to 376 kg ha^{-1} yr^{-1} under pastures on a silty clay loam with fertilizer doses of zero and 200 kg N ha^{-1}, respectively, and Hatch et al. (1991) values of 310 kg N ha^{-1} per growing season under permanent pasture, consisting of a grass-clover mixture and 415 kg per growing season on grassland receiving a fertilizer dose of 420 kg N ha^{-1}.

5.2 Effects of phase of crop rotation

Annual soil-N mineralization was significantly higher following grass than following another crop (Table 6). This can be explained by the input of substrate. On grassland, the flow of plant material in dead leaves and roots is much higher than on arable land (Johnston, 1986; Tyson et al., 1990; Vertès and Mary, 2004), while also the use of organic manure is higher (Table 3). Thus, the stock of substrate will be higher in years following grassland, than in years following arable cropping. Assuming a Norg input originating from crop residues of 250 kg N ha^{-1} in grassland (Whitehead, 1986; Hassink, 1996; Corré and Conijn, 2004) and 87 kg N ha^{-1} in maize land, including a catch crop, that is completely ploughed-in in spring (Schröder et al., 1996; Bolinder et al., 1999), the input of Norg in crop residues can be estimated for each distinct phase of the crop rotation. Total input of substrate into the soil is estimated as the sum of Norg inputs from crop residues and from organic manures (Table 3). Soil-N mineralization and Norg inputs are depicted in Figure 6 for permanent grassland and the different phases of the crop rotation. Soil-N mineralization is strongly correlated with substrate added in the preceding year, as indicated by an R^2 value of 0.90.

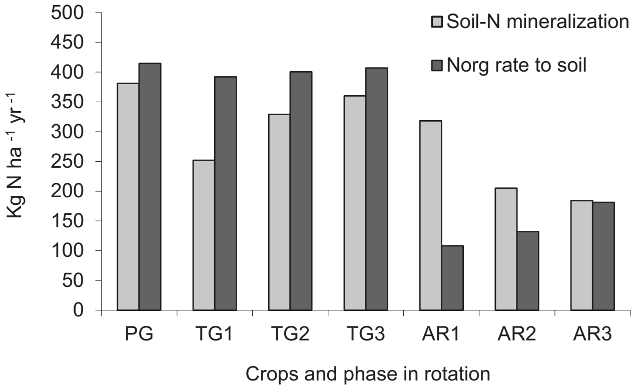

Figure 6: *Mean mineralization and input of Norg in permanent grassland and in the arable crop rotation.*

5.3 Intra-annual dynamics

Figure 2 indicates that differences in N release under different crops mainly occur in summer. The increase in soil-N mineralization in spring and early summer under grassland is associated with increasing temperatures (compare Figure 2 with Table 1) and the difference between maize and grass is most probably associated with the lower availability of substrate on maize land, as a result of lower organic matter inputs in the preceding year (Section 5.2). The intra-annual dynamics in the specific rotation phases of maize and ley provide more insight. The higher soil-N mineralization in spring under first-year maize than under later year maize (Figure 3) points at effects associated with the destruction of the grass sod in February, well before maize is sown in the beginning of May. The amount of N released between grass destruction and sowing of 1st year maize is about 100 kg ha^{-1}. The intra-annual constancy of N mineralization in later-year maize (Figure 3) is somewhat surprising, because of the absence of any effect of ploughing-in Italian ryegrass. In contrast, in 1st year ley, soil-N mineralization is low in spring (Figure 4), presumably associated with the lower total labile SON stock following the arable phase. In 1st year grass, the equilibrium between input and decomposition has to be restored, before ley-specific mineralization rates are re-established.

The positive effect of temperature on N mineralization (Figure 2 and Table 1) is not surprising and is in agreement with basic insights, established and confirmed in many studies (Conant et al., 2011), but it is quite remarkable that this temperature effect appears not significant in Figure 1 and Figure 5. It might be expected that annual mineralization in a 'warm' year would be higher than in a 'cold' year (Figure 1) and the same holds for mineralization in specific months that are relatively 'warm' or relatively 'cold'. Probably, this 'temperature effect' is not significant because it is masked by variation in other factors that affect mineralization. Of course, our observations do not refute the basic insights in the influence of factors like climate and soil characteristics on mineralization, but they indicate the extent to which these effects are significant in the context of a farming system.

5.4 Implications for N management

Comparison of the total mineral N input to soil, i.e. the sum of observed soil-N mineralization and applied mineral N rates, with mineral N requirements of the soil/crop system may reveal where and/or when mineral N input is in excess or limiting. The required N input to the soil/crop system includes both, the target crop-N uptake and the associated N losses. Note that at 'De Marke', target crop-N uptake was established as crop-N uptake that can be realized without exceeding the threshold values for acceptable losses, i.e. 79 kg N ha^{-1} (Aarts et al., 1992). Target crop-N uptake is the total nitrogen accumulated in plant tissue, i.e. the sum of N in economic crop products and N in crop residues. This approach is illustrated in Table 8 for 'De Marke', presenting for permanent grassland and distinct phases of the crop rotation, soil-N mineralization, mineral N supply to the soil (consisting of mineral N in manure, chemical fertilizer-N, atmospheric deposition and/or N fixation by legumes), target crop-N uptake, associated (permissible) N losses and the resulting excess mineral N supply. In 1st year maize, excessive N supply is highest, followed by 3rd year ley and permanent grassland. N supply is insufficient in 1st year ley. The annual excess of N supply at farm level (weighted average) amounts to only 7 kg ha^{-1}, indicating that improved N management would require reallocation within the farm, rather than a reduction at farm level. The rotation phases in which excess mineral N supply is high, may be used as indicators for high risks for N losses. To reduce these risks, supply of manure-N in the 1st year arable crop should be abolished. Instead, the manure-N could be added to 1st year ley. An additional measure could be to reduce the fertilizer-N rate in 3rd year ley to reduce soil-N mineralization in 1st year maize following ploughing-in.

Table 8: *Fate of mineral N (kg ha^{-1}) in permanent grassland and distinct phases of crop rotation (input data: means of 1993-2005). (For meaning of abbreviations, see Table 5).*

	Mineral N input		N requirement soil/crops		Excess N input
	Soil-N mineralization	Mineral N supply	Crop-N uptake[1)]	Permissible losses	
PG	381	266	556	79	12
TG[1]	252	334	565	79	-58
TG[2]	329	334	565	79	19
TG[3]	360	335	575	79	41
M[1]	318	63	241	79	61
M[2]	205	88	212	79	2
M[3+]	184	107	216	79	-4

[1)] Based on data of crop-N uptake (mean yields (means of 1993-2005) and estimates of N in crop residues).

Soil-N mineralization outside the growing season can be calculated from Figure 2, taking into account differences in the length of the growing season of grass (end of March until end of October) and maize (end of April until end of September). Out-of-season soil-N mineralization on maize land is 77 kg ha^{-1} yr^{-1} (38% of the annual total) and on grassland 60 kg (17% of the annual total). These data highlight the importance of catch crops after maize and after arable crops in general. However, it is questionable whether the uptake capacity of catch crops (such as Italian ryegrass) and grassland in winter is sufficient to utilize all released N. Reported values of crop uptake vary from 25 kg N ha^{-1} in the whole crop (including roots) in the UK (Shepherd, 1999), and 40 kg (Schröder et al., 1996) to 108 kg (Aarts et al., 1994) in Italian ryegrass in the Netherlands. The latter value was established in rather warm winters and may not be representative for normal winter weather conditions. Therefore, 40-50 kg N ha^{-1} seems reasonable under normal weather conditions.

The values of sd_{plot} and sd_{year} indicate that soil-N mineralization is highly variable. The magnitude of spatial and temporal variability is comparable. This variability forms a constraint for the aim to further synchronize N release with crop demand, unless the variability can be attributed to a known factor. An N-dose targeted to meeting soil/crop requirements under 'average' soil-N mineralization rate, will be excessive in years and on plots with high soil-N mineralization rates and insufficient on plots and in years with low soil-N mineralization rates. The possibilities to anticipate weather conditions are limited, because in the period that manure is applied (from early spring until early summer), the weather for the major part of the growing season is unknown. Therefore, exploration of spatial variability may be of more practical use. In a dairy production system on dry sandy soil such as 'De Marke', it seems plausible that soil-N mineralization is correlated with soil water content in the parcels. However, such correlations could not be confirmed in our research. Hence, no reliable indicators could be identified for prediction of the variability in soil-N mineralization.

The variability in N mineralization increases with its absolute values. It is uncertain to what extent peaks in soil-N mineralization in the growing season can be adequately captured. As long as variability in soil-N mineralization cannot be predicted adequately, high soil-N mineralization, resulting from long-term accumulation of substrate, entails a high risk of N-leaching. Thus, to restrict N losses to groundwater, the variability in soil-N mineralization might be restricted by reducing substrate accumulation in soil. Similarly to phosphorus, N may reach levels of saturation in soil as discussed by Schröder and Van Keulen (1997), Schipper et al. (2004) and Kayser et al. (2011). Earlier work on 'De Marke' indicated that nitrate leaching was higher on permanent grassland than on ley (Verloop et al., 2006). This may be attributed to the higher soil-N mineralization in permanent grassland, caused by a higher availability of substrate. This shows the importance of aiming at a high N recovery to restrict N surpluses, also in crops that seem less susceptible to losses. In the research period, average soil-N mineralization and Norg inputs appear to be well-balanced for permanent grassland and for arable rotations, although in the latter, phases of buildup of Norg are alternated with phases of decline (Figure 6). At farm level, annual Norg input is 298 kg ha^{-1}, indicating an average build-up of substrate of 10 kg N ha^{-1}, as soil-N mineralization was estimated at 288 kg ha^{-1} (Section 5.1). In order to prevent saturation of the soil with SON, inputs of substrate and mineralization should be balanced. Hence, Norg inputs should be reduced by 10 kg ha^{-1}. In order to further reduce the risks of N losses, it might be necessary to realize this balance at lower levels of Norg inputs and N mineralization. An option could be application of fermented N that is lower in Norg and higher in mineral N.

6 CONCLUSIONS

The results of the current analysis indicate that soil-N mineralization is strongly affected by crop rotation. Observations in permanent grassland and crop rotations with ley followed by arable crops (predominantly maize) indicate that soil-N mineralization is highest (318-381 kg ha^{-1}) under permanent grassland, 2^{nd} and 3^{rd} year ley and 1^{st} year arable crops. Soil-N mineralization is lowest (184-252 kg ha^{-1}) under 2^{nd} and 3^{rd} and more years arable crops and 1^{st} year ley. Soil-N mineralization strongly correlates (R^2: 0.90) with inputs of Norg in manure and crop residues in the preceding year. Temporal and spatial variability of soil-N mineralization is high, particularly in the growing season, which can not be explained by variations in soil water content, temperature and precipitation. Variability increases with the magnitude of soil-N mineralization. Therefore, in situations with high soil-N mineralization, associated with high inputs of substrate, the risks for N losses are high. To reduce these risks, situations leading to strong build-up of substrate should be avoided. A practice, limiting organic matter build-up would be fermenting slurry before application on farmland to enhance the fraction mineral N in slurry at the expense of organic N.

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CHAPTER

4

UTILISATION OF MANURE-N BY CROPS IN AN OPTIMIZED DAIRY FARMING SYSTEM

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ABSTRACT

Recycling of nutrients in manures deserves due attention because it helps livestock farmers to reduce expenses on mineral fertilizers and saves the environment. The utilisation of nutrients, nitrogen (N) in particular, is strongly affected by manure management, as demonstrated in field trials showing that utilisation can benefit from a better spatial distribution, anaerobic digestion, and slurry separation. We suggest it is useful and necessary to test whether such measures could also enhance the N fertilizer replacement value (NFRV) of manures at the whole farm level, if alone because farmers that are supposed to improve their manure management may better recognize themselves in results derived at whole farm scale. Experimental dairy farm 'De Marke' was used as a test case. 'De Marke' is located on a dry sandy soil where grassland and silage maize are the major feed crops. We developed a procedure by which the apparent NFRV of manure can be deduced from farm records that were collected from 1994 to 2010. We applied this procedure to each period in which manure management of 'De Marke' was constant and to different crops. The calculated long term mean NFRV of manure was 0.70 at the whole farm level. The estimates for permanent (0.71) and temporary grassland (0.63) match reasonably well with the ranges that were expected based on field trials (0.47-0.79). The estimate for maize (0.97) was high compared to results found in field trials (0.12-0.75). However, the uncertainty of $NFRV_{manure}$ (indicated for grassland by lower and upper quartiles of 0.42 and 0.80, respectively) was considerable due to uncertainties of the underlying assumptions. Therefore, we also evaluated effects of manure on the basis of the dynamics of N use efficiency (NUE). NUE in grassland showed some increase over the whole research period, despite the reduction (2000) and abolishment (2004) of mineral fertilizer. This result was analysed in relation to weather effects, net soil-N mobilisation and N inputs to soil other than manure-N, suggesting a possible increase of manure-N utilisation due to (a combination of) measures, such as reduction of grazing hours, anaerobic digestion of manure and row application of slurry. The improvement could not be attributed to specific measures.

1 INTRODUCTION

On dairy farms adequate management of animal manure is crucial to benefit from its potential value as a source of nitrogen (N) for grassland and forage crop production and to minimize losses to the environment. Proper management of manure reduces the need for supplementary mineral fertilizer-N inputs (Aarts, 2000; Oenema et al., 2001; Schröder, 2005). Even if managed adequately, farmers may underestimate the N fertilizer replacement value (NFRV) of manures and apply more mineral fertilizer-N than necessary (Oenema et al., 2010; Gourley et al., 2011). Therefore, there is an urgent need to demonstrate the NFRV of manures. Field experiments have shown that NFRV's of manures can be as high as 0.80 (i.e. 100 kg of applied manure-N is equivalent to 80 kg fertilizer-N) through proper timing, positioning and dosing (Aarts et al., 2000a; Sørensen et al., 2003; Schröder, 2005; Lalor et al., 2011; Webb et al., 2013). Farmers may question the applicability of the outcomes to their own situations, because field experiments are usually executed under ideal circumstances in terms of soil quality and sometimes even weather conditions. At the level of a commercial farm this will generally not be so, if alone because each individual field cannot always be addressed at the most suitable moment. From this perspective it is highly relevant to also assess the NFRV of manure at the whole-farm level of a commercial farm. When analysing the NFRV of farming systems, due attention must be given to their N dynamics, as the intensity (N input level) may change over time affecting residual N effects (Nevens and Reheul, 2005), and new, allegedly more efficient techniques and practices may be adopted in the course of time. Such changes can have an impact on NFRV of manure (Gutser et al., 2005; Schröder, 2005).

Alternatively to the NFRV of manure, the ratio of crop-N yields and N inputs (i.e., N use efficiency, NUE) could be analysed to detect changes of manure-N utilisation. This approach is simple because the NUE can be established on the basis of N input and output data, but it must be recognized that manure-N utilisation is not the only factor that influences NUE. Variability of other N sources than manure and of weather conditions may affect N uptake and these factors are not kept constant on farms (Schröder, 2005). However, when other N-sources and weather are known their impact may be quantified or assessed and changes in NUE may be evaluated and cautiously related to manure-N utilisation and manure management.

The aim of this paper was to: (*i*) assess the NFRV of manure and analyse NUE at a dairy farm to quantify the effects of changing manure management, and (*ii*) present a method to estimate $NFRV_{manure}$ at whole-farm level and discuss its strengths and limitations. Experimental dairy farm 'De Marke' in The Netherlands was used as a test case.

2 MATERIALS AND METHODS

2.1 Experimental farm 'De Marke'

2.1.1 Site characteristics

'De Marke' is situated in the eastern part of The Netherlands on very light sandy soil, characterized by a 0.30 m anthropogenic upper layer, with an average organic matter content of 4.8% and a bulk density of 1300 kg m^{-3}, overlying a layer of yellow sand, very low in organic matter and hardly penetrable by roots (Dekkers, 1992; Aarts, 2000). Soil water content is usually at field capacity in spring (0.18 g per g oven dry soil) and decreases to 0.12 in mid-summer, or even to 0.10-0.05 in dry summers. Groundwater depth is 1 to 3 meter below soil surface and upward water transport in dry periods from the saturated zone to the root zone is hence negligible. In the experimental period (1994 -2010), precipitation (mean annual 792 mm) was on average evenly distributed over the year. In the October-March period, average temperature was 4.7 °C and in the April-September period it was 14.1 °C. December was the coldest month with an average temperature of 2.3 °C and July the warmest with 17.3 °C (Figure 1).

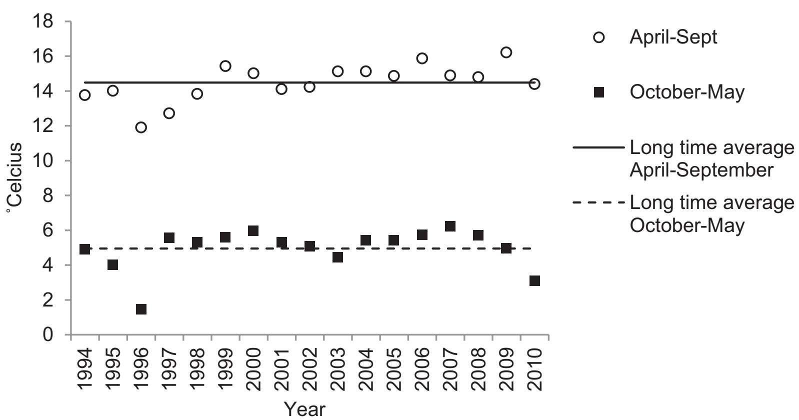

Figure 1: *Mean temperature per year from April-September and from October-May.*

2.1.2 Farm layout

The farm area (55 ha) is divided into permanent grassland (11 ha) and two crop rotations in which grass was alternated with arable fodder crops: ROTI and ROTII, each phase being present every year (Table 1). The rotation scheme of the arable phase was modified slightly in the course of time. From 1994 until 1996, fodder beet was the first arable crop to succeed the grassland phase. From 1996 till 1999, maize, mainly grown for whole crop silage, was the only arable crop, after which triticale and spring barley were introduced as the last crop in the arable phase from 2000 to 2005 and from 2005 to 2010, respectively (Table 1). In maize, Italian ryegrass was sown as a catch crop between the rows in June to create a soil cover directly after the harvest of maize in September. Catch crops were ploughed-in in the first week of March. A total of 31 ha of grassland were available for grazing by 80 milking cows and, on average, 55 young stocks. This herd excreted on average 273 kg total N ha^{-1} yr^{-1}, including dung and urine while grazing. Ploughing depth was 0.25 m in crop rotations and occasionally in permanent grassland when it was reseeded about once in six years.

Table 1: *Farm plan of 'De Marke' (ha) and crop sequence[1)] in the two types of rotation; underlined: crops included in the analysis.*

Period	**Land use (total 55 ha):**		
	Permanent grassland (11 ha)	**Rotation I (23 ha)**	**Rotation II (21 ha)**
1994-1995	<u>GGGG</u>	GG<u>G</u>FM<u>M</u>	GGGFM<u>MMM</u>
1996-1999	<u>GGGG</u>	GG<u>G</u>MM<u>M</u>	GGGMM<u>MMM</u>
2000-2003	<u>GGGG</u>	GG<u>G</u>MMT	GGGMM<u>MM</u>T
2004-2010	<u>GGGG</u>	GG<u>G</u>MMB	GGGMM<u>MM</u>B

[1)] G = perennial ryegrass, F = fodder beet, M = maize, T = triticale, B = spring barley.

2.1.3 Fertilizer and manure management

N application rates were tuned to targeted N yields which were, in turn, based on production levels that were considered feasible on dry sandy soils without violating environmentally acceptable N losses (Aarts et al., 1992).

Manure management was adjusted three times over the studied period (Table 2). From 1994-1999 manure and urine excreted indoors were collected as mixed slurry and stored in a covered silo until application to grassland and arable land. A rotational grazing system was applied according to which the herd grazed for 5-7 days on individual grassland plots of 1-3 ha, after which it was moved to a next plot. The duration of the grazing season was limited (usually from March to September) and cows were allowed in the pasture 8 hours a day, so that about 77% of the N excretion was collected indoors on an annual basis. Collection and storage took place in the form of slurry (System A, Table 2). In the subsequent period (2000-2003) grazing hours per day were reduced to 5 hours so that 89% of the N excretion was collected (System B, Table 2), reasoning that urine and dung droppings are less efficient sources of N than evenly spread and well-timed manure that has been collected and stored (Sauer and Harrach, 1996; Wachendorf et al., 2004). From 2004 onwards, slurry was anaerobically digested before application, assuming that the utilisation of slurry-N benefits from a smaller share of organically bound N (Figure 2). Moreover, to maize slurry was no longer applied evenly but in rows next to where the maize seeds were to be positioned. This placement was expected to improve the match between the supply and demand of N and P (Schröder et al., 1997) (System C, Table 2). In 2009 and 2010, 30% of the aerobically digested slurry was separated into a liquid and solid fraction with supposedly varying contents of mineral N, organically bound N and P (Hjorth et al., 2010). Note that separation did not produce a liquid fraction with a substantially higher share of mineral N (Figure 2), unlike prior expectations. Allocation of the different products was tuned to the N and P requirements of specific needs of plots and crops (System D, Table 2). The periods of manure application was the same for all systems (A-D). On maize, manure was applied in May. On grassland slurry application started, on average, in mid-March and ended in the beginning of August and grazing started in the beginning of March and ended in September.

The targeted levels of available N from manure and mineral fertilizer amounted to 250 and 150 kg N ha^{-1} for grassland and maize, respectively. For P, the fertilization level was set to the expected withdrawal of P in harvested crop products to realize P equilibrium fertilization (Aarts et al., 2000b; Verloop et al., 2010). Calculations of the required rates of mineral fertilizer-N took account of the partial availability of N in manures according to default values. In addition to that, a net release of N from the ploughed-in grass sod of 90 kg ha^{-1} in the first and 45 kg in the second year was factored in for the 1st and 2nd year maize in ROTI and ROTII (Aarts et al., 1992). Moreover, a release of N from the ploughed-in catch crops was subtracted from the remaining N requirements whenever maize was preceded by such a crop.

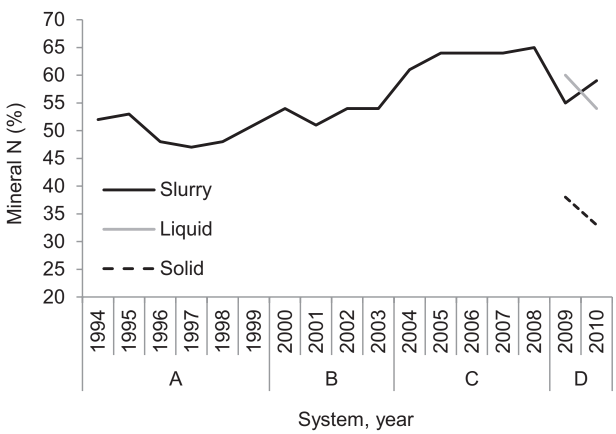

Figure 2: *Development of mineral N (percentage of total N) in untreated slurry (system A and B), digested slurry (system C) and digested slurry and its separation products (system D).*

Table 2: *Manure management at 'De Marke' from 1994-2010.*

Period	System	N grazing fraction[1]	Manure products applied	Manure-N rate[2] (kg ha^{-1})	Amount[3] %	Application method: Grassland	Application method: Maize	Application method: Other arable crops
1994-1999	A	0.23	Slurry	223	100	Injection	Injection after ploughing	Injection after ploughing
2000-2003	B	0.11	Slurry	211	100	Injection	Injection after ploughing	Injection after ploughing
2004-2008	C	0.10	Digestate	203	100	Injection	Injection after plouging, but in rows[4]	Injection after ploughing
2009-2010	D	0.07	Unseparated digestate	136	70	Injection	Injection after plouging, but in rows[4]	Injection after ploughing
			Liquid fraction digestate	49	25	Injection	Injection after plouging, but in rows[4]	Injection after ploughing
			Solid fraction of digestate	10	5	Broadcast	Broadcast followed by immediate incorporation	-

[1] The fraction of N annually excreted during grazing.
[2] Means across the entire farm.
[3] Percentage of total manure-N (kg) collected at the farm.
[4] Row spacing coinciding with the position of maize rows.

2.2 Data

2.2.1 Collection

The total area of 'De Marke' is sub-divided in 30 parcels of 1-3 ha each, for which nutrient flows and yields were monitored separately and each parcel was sub-divided in two or three plots of 0.5-1 ha, yielding 59 plots in total. For each of the 30 parcels relevant management information, i.e. timing and method of manure application, dates of harvests, grazing, re-sowing and ploughing were recorded (Aarts, 2000). N-fixation by clover was estimated from its dry matter yield, assuming 0.05 kg N-fixation per kg dry matter (Elgersma and Hassink, 1997; Van der Meer and Baan Hofman, 1989). The dry matter yield of clover was established by the dry matter yield of the grass/clover mixture and visual estimates of the share of clover in the total dry matter yield in each parcel for each harvest. This procedure was followed for each parcel and for each harvest by cutting or grazing. Fresh grass/clover consumption by cattle was calculated by estimating yields just before and after grazing and correcting for regrowth during grazing. Amounts of N excreted during grazing were estimated from feed intake by subtracting N output in milk and meat and the N excreted indoors. Atmospheric N deposition was derived from the national monitoring network (Velders et al., 2010).

Net change of the soil-N pool was estimated from soil samples collected in the 59 plots. Samples of the 0-0.2 m layer were taken annually from 1994 to 2010 and in the few years preceding that period when 'De Marke' project was initiated. In each plot, 40 cores were pooled and mixed into a composite sample. Samples from the 0.2-0.4 m layer, the lower layer of the rooting zone (lower topsoil; 31 plots) and the 0.4-0.6 m layer, the layer below the rooting zone (subsoil; 6 plots) were selected randomly from all plots. These deeper layers were sampled occasionally in the course of time but not annually. The mass fraction of soil-N was determined in all samples by Near Infrared Spectrometry. It was assumed that the N mass fractions determined in the 0-0.2 m layer were representative for the 0-0.25 layer, the depth over which the soil is ploughed and homogenized since the start of 'De Marke'. N mass fractions were multiplied by the soil bulk density (ρ, kg m^{-3}) to arrive at masses per unit area (kg N ha^{-1}). The bulk density was assumed to depend on organic matter (SOM, mass %) content according to $\rho = 1.728\text{-}0.271 \times ((0.45 \times SOM)^{0.5})$, after Whitmore et al. (1992). This procedure was executed for permanent grassland and for land in rotations.

N-mineralization was measured in tubes that were inserted in the soil to a depth of 0.2 m in grassland and 0.3 m on arable land on 6 locations measuring 20 m × 20 m. Each of them was situated in a different parcel, thus representing the soil characteristics and land use of 'De Marke' as good as possible. More details on the method are described by Raison et al. (1987) and details regarding the monitoring and elaboration procedures applied on 'De Marke' are described in Verloop et al. (submitted).

2.2.2 Selection

The selection of crops (Table 1) included in the analysis, represent the complete crop production system. The means for permanent grassland, temporary grassland and maize were based on measurements in 6, 3 and 3 parcels, respectively. Only crops were selected in which manure-soil-crop-interactions were more or less stabilized, i.e., crops that were grown at least two years after the transition of grassland to arable land and vice versa (>2nd year maize and >2nd year temporary grassland). Maize >2nd year was absent in 2004-2006. Consequently, data for these years are lacking.

2.3 Analyses

The apparent availability of soil mineral N was derived from the harvested N yields (NYH) by dividing these by their crop specific apparent recoveries ($NREC_i$):

$NYH_i = NA_i \times NREC_i$ or $NYH_i / NREC_i = NA_i$

where:
NYH_i = harvested crop i by machine or grazing
$NREC_i$ = N recovery of crop i
NA_i = amount of N available for crop uptake
i = grassland, maize

The literature provides many values for the NREC's in grassland (e.g. Prins, 1980; Vellinga and André, 1999; Schils and Kok, 2003) and maize (Schröder et al., 1998; Ten Berge et al., 2007). However, the reported numbers range from values as low as 50% (Verloop and Hilhorst, 2011) or less up to values of 100% (Ten Berge et al., 2007). Instead of using fixed values from literature, we modelled the N recovery of grassland (NRECg) and maize (NRECm, Table 3) for each year to account for weather effects. The results of NRECg were assumed to represent both permanent and temporary grassland. For this purpose we used CNGRAS (Conijn and Henstra, 2003; Conijn, 2005) for grassland. We calibrated CNGRAS on the mean (over the entire research period) N input and N yield at 'De Marke'. Next, the annual fluctuations of NREC were established with CNGRAS using daily weather data as input and the average N input, resulting in year-specific yields (and thus NREC). For maize we followed the same procedure with an extended version of the crop rotation model ROTASK, which is based on WOFOST and a soil organic matter model for maize (Jongschaap, 1996; Wolf and Van Diepen, 1995; Verberne *et al.*, 1990), but we calibrated to an average default value for NRECm of 70% corresponding to what was found in several field trials executed within 'De Marke' (Baan Hofman and Ten Holte, 1995; Schröder et al., 1998).

The amounts of available N calculated accordingly, were supposed to originate from either atmospheric deposition (N_{dep}), biological fixation ($N_{biolfix}$), mineral fertilizer ($N_{minfert}$), or manure-N (N_{manure}) comprising grazing excreta-N, slurry-N, digested slurry-N, liquid fraction-N and solid fraction-N. Each source of manure has a specific NFRV, but these were lumped to one value. The amounts of available N were corrected for the net N mobilisation from soil organic matter ($N_{minsoil}$, negative value in case of net immobilisation):

$$NA_i = \sum(NSOURCE_j \times NFRV_j) \qquad \text{Equation 1}$$

with:

$j = N_{dep}, N_{biolfix}, N_{minsoil}, N_{minfert}, N_{manure}$, so:

$$NA_i = (NFRV_{dep} \times N_{dep}) + (NFRV_{biolfix} \times N_{biolfix}) + (NFRV_{minsoil} \times N_{minsoil}) + (NFRV_{minfert} \times N_{minfert}) + (NFRV_{manure} \times N_{manure}) \qquad \text{Equation 2}$$

$NFRV_{manure}$ can be solved from the above equation as:

$$NFRV_{manure} = (NA_i – ((NFRV_{dep} \times N_{dep}) + (NFRV_{biolfix} \times N_{biolfix}) + (NFRV_{minsoil} \times N_{minsoil}) +(NFRV_{minfert} \times N_{minfert})) \;/\; (N_{manure}) \qquad \text{Equation 3}$$

The NUE efficiency was defined as:

$$NUE = \sum(NYHi \;/\; \sum NSOURCEij) \qquad \text{Equation 4}$$

for i crops and j types of N source, including the N that has become available through the background net N mobilisation by soil.

We adopted values for $NFRV_{dep}$ and $NFRV_{minsoil}$ of 0.75 for both grassland and maize, as opposed to Schröder et al (2007) who proposed values of 0.75 and 0.60 for grassland and maize, respectively. However, as maize on ‘De Marke’ is grown in combination with cover crops, we assumed recovery of N_{dep} and $N_{minsoil}$ to be equal for maize and grassland.

For permanent grassland and rotations the long term net N mobilisation (negative in case of immobilisation) was estimated from the decline of the soil-N content in the 0-0.25 m layer over the whole research period (long term N change), established by regression of soil-N content over time, resulting in values of long term mean $N_{minsoil}$ of -8 kg ha^{-1} yr^{-1} for permanent grassland and 7 kg ha^{-1} yr^{-1} for crop rotations. For maize and temporary grassland rotation effects were superimposed on the long term development assuming an N immobilisation of 27 kg ha^{-1} in temporary grassland and an N mobilisation of 38 kg ha^{-1} in maize, resulting in mean long term $N_{minsoil}$ values for each crop as presented in Table 3. We established relative year-specific deviations from the above average net N mobilisation by applying a year-specific index, based on the mineralisation measured in the inserted tubes, assuming that they reflected the annually fluctuating effects of mobilisation of residual N built up in preceding years and temperature.

Table 3: *Estimates for each year of the nitrogen recovery (NREC, %) and net nitrogen mobilisation ($N_{minsoil}$, kg ha^{-1} yr^{-1}) in grassland and maize on 'De Marke'.*

Year	NREC		$N_{minsoil}$		
	Grassland	Maize[1)]	Permanent grassland	Temporary grassland	Maize[1)]
1994	81	53	-34	-16	27
1995	71	66	20	-8	30
1996	96	84	77	-14	57
1997	103	65	127	-15	88
1998	96	61	93	-4	73
1999	113	78	-13	-6	34
2000	108	65	114	-15	88
2001	88	73	-37	-18	22
2002	99	88	-85	-3	39
2003	78	54	-26	9	20
2004	104	-	-69	1	-
2005	111	-	-13	0	-
2006	56	-	-35	3	-
2007	115	73	-83	-4	47
2008	110	73	-90	5	27
2009	88	71	-4	12	52
2010	72	77	-82	11	22
Average 1994-2010	*93*	*70*	*-8*	*-20*	*45*

[1)] Data 2004-2006 not available.

In addition to this, an uncertainty and sensitivity analysis of $NFRV_{manure}$ and NUE (i.e. equations 3 and 4) was performed with a Monte Carlo procedure, applied to the input parameters (Appendix I; Jansen et al., 1994). The uncertainty of selected input data was estimated by assigning a probability density function (PFD, mean and variance) to each input factor (Appendix I). An input matrix consisting of 5000 samples was generated using a random sampling procedure for all combinations of the model inputs and parameters. Model output was calculated for each sample and the variance of the model output was specified and analysed as a measure of the uncertainty. To analyse the sensitivity, for each input parameter, the Top Marginal Variance (TMV) was calculated. The TMV of an input is the variance reduction that would occur if the input would be fully known (Jansen et al., 1994).

The observed $NFRV_{manure}$ was compared to the expected $NFRV_{manure}$ which was based on the NFRV values of the components of manure-N (grazing excreta N, slurry-N, digested slurry-N, digested liquid fraction N and digested solid fraction N, estimated by long-term empirical data and expert judgement (www.bemestingsadvies.nl), weighed by their application rates.

3 RESULTS

The inputs of N originating from the various sources are presented in Figure 3. The annual atmospheric deposition gradually declined over time from 48 to 31 kg ha^{-1} yr^{-1}. In grassland, the N-fixation by clover was on average 25 kg ha^{-1} yr^{-1} in permanent grassland, and around 47 kg ha^{-1} yr^{-1} in temporary grassland, but with large fluctuations, in particular in temporary grassland. From 2000 to 2004 the mineral fertilizer-N rate was gradually reduced to zero. In both permanent and temporary grassland the amounts of N excreted during grazing were reduced from 1999. Instead, this 'saved' N was returned to the fields in the form of slurry-N. In 2009 and 2010, the liquid fraction was applied to grassland. On permanent grassland 70% of slurry-N was applied in the form of liquid fraction and on temporary grassland 20% of slurry-N was applied in the form of liquid fraction (not shown separately in Figure 3), The solid fraction was not applied to grassland. The N inputs to maize consisted of atmospheric N and manure-N only. The manure-N rate was 123 kg ha^{-1} yr^{-1} on average but declined over time (Figure 3). In 2009 and 2010 20% of the slurry-N was applied in the form of solid fraction (not shown separately in Figure 3). The liquid fraction was not applied to maize.

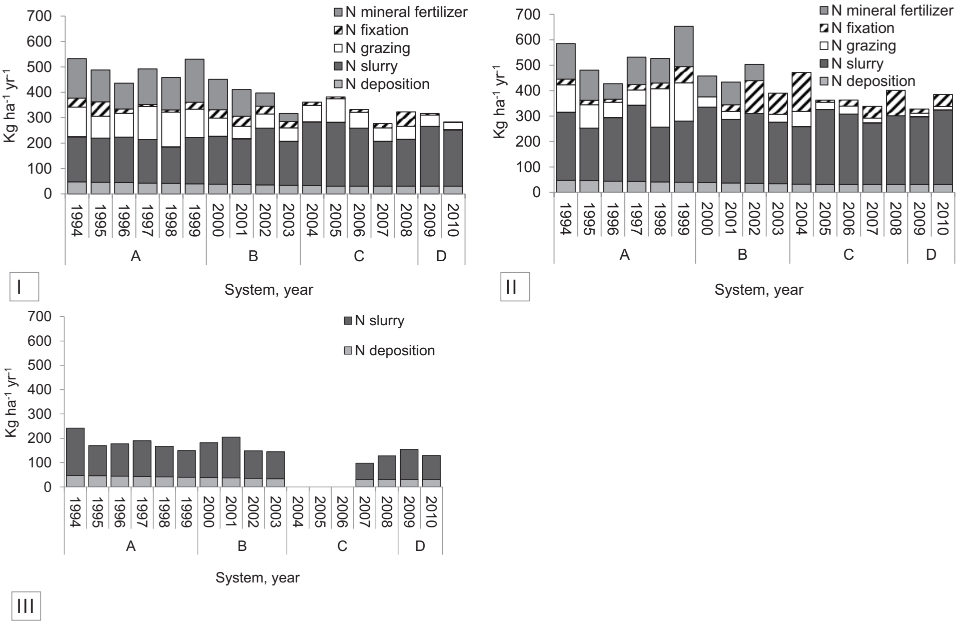

Figure 3: *N inputs in permanent grassland (I), >2nd year temporary grassland (II) and >2nd year maize from (III) 1994-2010 (no data in 2004-2006 for maize), under systems A, B, C, D; see Section 2.1.3 for more explanation.*

Permanent grassland yielded, on average, 265 kg N ha^{-1} yr^{-1} with considerable year-to-year fluctuations (Figure 4) but a significant decline over the studied period (Table 4). The dynamics of the N yield shows a drop in 2002 followed by a fairly constant level until 2007 and a further decline in 2008-2010 (Figure 4). N yields also declined significantly in temporary grassland (Table 4), in particular in the second half of the studied period and 2009-10 (Figure 4). The N yield of maize did not show a significant trend (Table 4); mean N yield was 127 N kg ha^{-1} yr^{-1} with substantial year-to-year fluctuations. The dynamics of dry matter yields (Figure 4) followed those of the N yields as also indicated by strong correlations between both variables, i.e., 87%, 84% and 78% for permanent grassland, temporary grassland and maize, respectively. Dry matter yields did not show a significant linear trend over time.

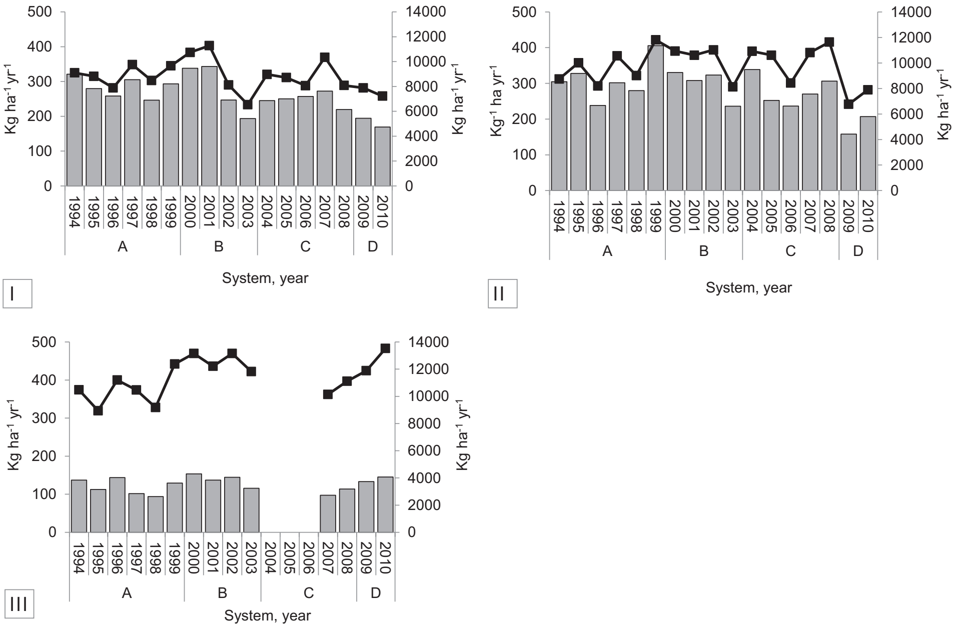

Figure 4: *N yields (bars) and dry matter yields (lines) in permanent grassland (I), >2nd year temporary grassland (II) and >2nd year maize (III), under systems A, B, C, D.*

Table 4: *Regression coefficients (rc) of N yield (NYH, kg ha^{-1}), dry matter yield (DMYH, kg ha^{-1}), NUE (%) and NFRV$_{manure}$ (dimensionles) against time (yr), established by linear regression over the research period (1994-2010).*

	NYH		**DMYH**		**NUE (%)**		**NFRV$_{manure}$**	
	rc	**R^2**	**rc**	**R^2**	**rc**	**R^2**	**rc**	**R^2**
Permanent grassland	-2.0*	0.42	-74	0.09	1.0*	0.39	0.030**	0.14
Temporary grassland[a]	-5.9*	0.26	-39	0.02	0.2	0.02	0.028	0.21
Maize[b]	0.13	0.13	120	0.20	2.0*	0.48	0.034*	0.29

[a] >2nd year temporary grassland.
[b] >2nd year maize.
* Significant at P < 0.10.
** Significant at P < 0.05.

NUE for permanent grassland and for maize increased significantly by 1% and 2% yr^{-1}, respectively (Figure 5, Table 4). In temporary grassland NUE tended to increase slightly from 1994 to 2008, but dropped in 2009 and 2010, resulting in a non-significant change over the entire period (Figure 5, Table 4). It is known that weather conditions may strongly affect the recovery of N and indirectly NUE. However, for grassland, correlations between NUE and N recovery for corresponding years were low, i.e., (R^2 0.02) for permanent and (R^2 0.015) for temporary grassland. For maize, the correlation between NUE and recovery was higher (R^2 0.39), but high and low recoveries were evenly distributed over the research period.

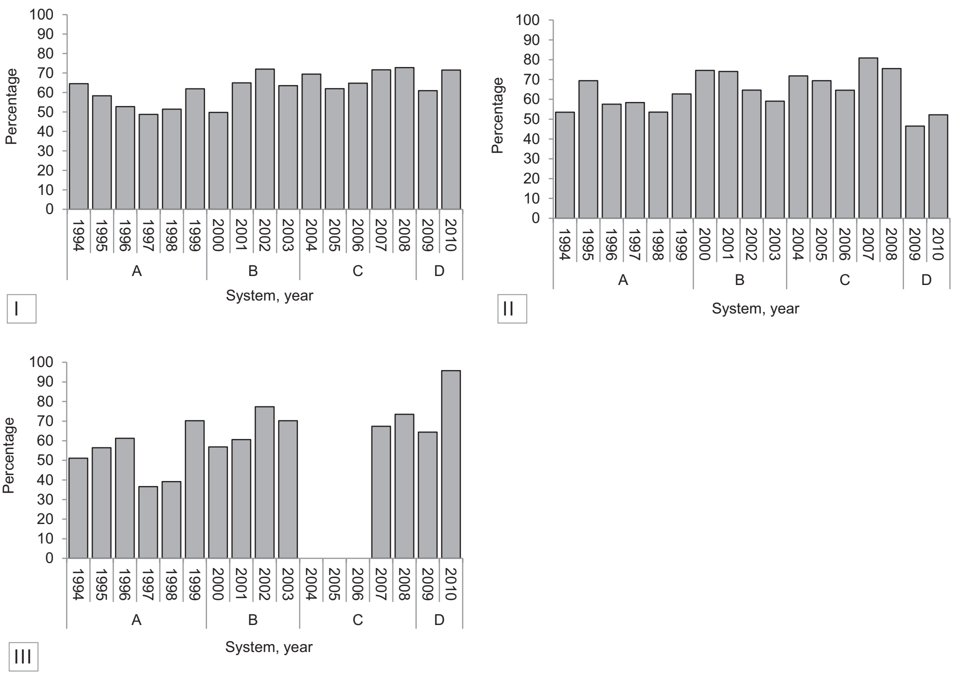

Figure 5: *NUE in permanent grassland (I), >2nd year temporary grassland (II) and >2nd year maize (III), under manure management systems A, B, C, D.*

Averaged over the whole research period and the entire farm estimated $NFRV_{manure}$ amounted to 0.70, with averages for permanent grassland, temporary grassland and maize of 0.71, 0.63, and 0.97 respectively (Figure 6). The estimates of $NFRV_{manure}$ for permanent grassland increased significantly with time (Table 4), but showed great fluctuations with low values in 1997-2000 and the increase seemed mainly caused by an exceptionally high value in 2006 (Figure 7). The $NFRV_{manure}$ in temporary grassland followed the pattern observed in permanent grassland. The fluctuations were, however, stronger than in permanent grassland with remarkably low values in 1997, 1998 and 1999 (Figure 7). In maize, mean $NFRV_{manure}$ over the whole period was very high (0.97), but varied substantially over the years; low and high values did not occur in the same years as for grassland.

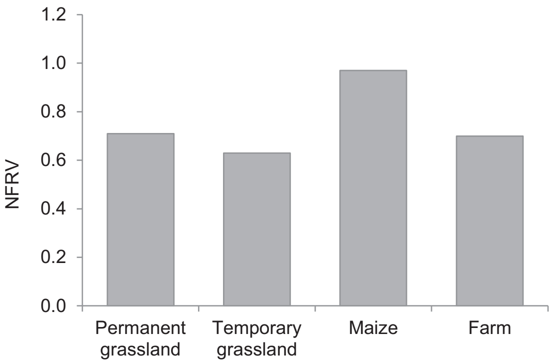

Figure 6: *Estimates of $NFRV_{manure}$ (dimensionless) in: permanent, temporary grassland >2nd year, maize >2nd year and the entire farm (means of 1994-2010).*

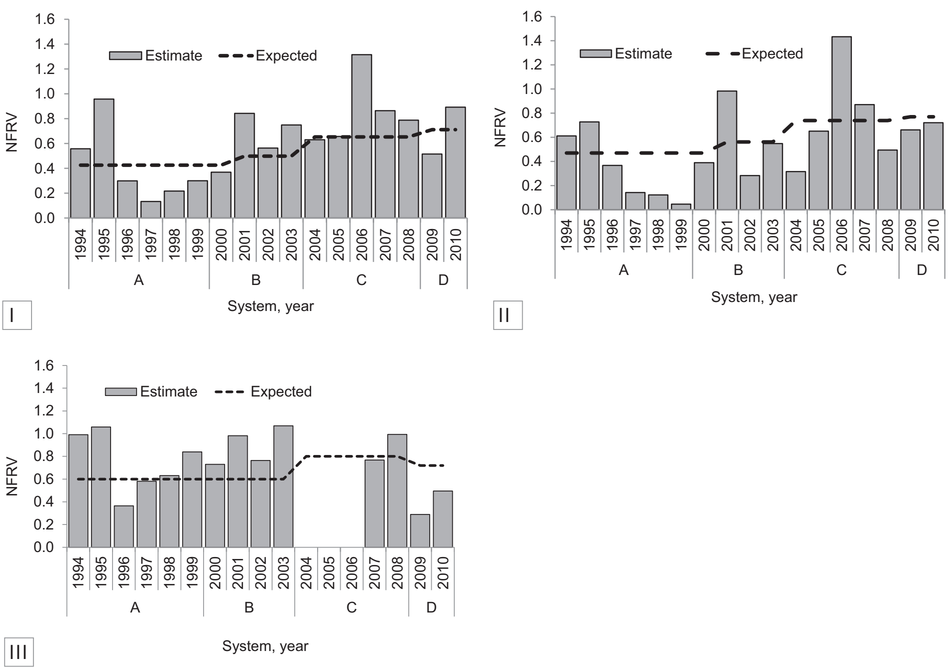

Figure 7: *Estimates of $NFRV_{manure}$ and expected $NFRV_{manure}$ (dimensionless), in (I) permanent, (II) temporary grassland >2nd year, and (III) maize >2nd year, at 'De Marke' for different manure management systems (A, B, C, D).*

Table 5 presents the NUE, $NFRV_{manure}$, N yields and dry matter yields for each crop type and each of the consecutive management systems A, B, C and D. The NUE of temporary grassland increased from A to C, but fell back drastically under system D. In maize and permanent grassland differences between the NUE of the management systems were not significant. The estimates of $NFRV_{manure}$ in permanent grassland, temporary grassland and maize were not significantly different for any of the management systems, although $NFRV_{manure}$ tended to increase in all crops from system A to C. This increase was followed by a drop in grassland (temporary and permanent) when shifting to system D. This drop was associated with reduced N yields in grassland, whereas N yields remained the same in maize.

Table 5: *NUE, $NFRV_{manure}$, N yield (NYH) and dry matter yield (DMYH) in different crops and manure management systems (A, B, C, D); means with no shared characters (a, b or c) are significantly different (P<0.05), according to Bonferroni-test for multiple comparisons.*

	Permanent grassland	Temporary grassland[1)]	Maize[2)]
NUE (%)			
Management A	56 a	59 ab	52 a
Management B	63 a	68 bc	66 a
Management C	68 a	72 c	70 a
Management D	66 a	49 a	80 a
$NFRV_{manure}$ (-)			
Management A	0.49 a	0.40 a	0.77 a
Management B	0.72 a	0.63 a	1.0 a
Management C	0.94 a	0.83 a	1.12 a
Management D	0.78 a	0.76 a	1.24 a
NYH (kg ha^{-1})			
Management A	292 a	310 a	120 a
Management B	261 ab	300 ab	138 a
Management C	249 ab	281 ab	106 a
Management D	182 b	183 b	139 a
DMYH (Mg ha^{-1})			
Management A	9.2 a	9.7 a	10.4 a
Management B	8.6 a	10.2 a	12.6 a
Management C	8.8 a	10.5 a	10.6 a
Management D	7.5 a	7.3 a	12.7 a

1) >2nd year temporary grassland.
2) >2nd year maize.

Estimates of $NFRV_{manure}$ were very sensitive for uncertainties of the input parameters (Appendix I) as indicated by a variance of 0.08 and quartiles of 0.42 and 0.80 around the mean of 0.61. NREC and $N_{minsoil}$ contributed by far the most to the uncertainty, as indicated by high TMVs (Table 6). The parameters 'aboveground clover biomass' and 'biological N fixation by clover', both used to establish $N_{biolfix}$, also contributed to the uncertainty of the $NFRV_{manure}$. Estimates of NUE (mean 0.73) were less sensitive for uncertainties of input variables. The variance was 0.04 and 50% of the estimates were in the range of 0.58 to 0.81. Uncertainties related to the net N mobilisation from soil ($N_{minsoil}$) contributed most to the uncertainties of the NUE (Table 6).

Table 6: *Relative contribution (Top Marginal Variances, TMV) of input variables to the uncertainty of FNRV and NUE for grassland.*

		TMV (%)	
Input variable	**Unit**	**$NFRV_{manure}$**	**NUE**
N mass fraction manure[1)]	kg m^{-3}	0.4	1.1
Manure rate[1)]	m^3 ha^{-1}	0.4	0.9
N grazing excreta	kg ha^{-1}	1.2	2.4
N mass fraction min fertilizer[2)]	kg kg^{-1}	0	0
Mineral fertilizer rate[2)]	kg ha^{-1}	0	0
Aboveground clover biomass[3)]	Mg ha^{-1}	5	2.3
Biological N fixation by clover[3)]	kg Mg^{-1}	4.7	2.4
N deposition	kg ha^{-1}	0.3	0.4
Nminsoil	kg ha^{-1}	49.4	70.4
Dry matter yield crops[4)]	kg ha^{-1}	2.8	3.2
N content in crops[4)]	kg kg^{-1}	2.8	3.1
$NFRV_{biolfix}$	-	1.2	-
$NFRV_{dep}$	-	0.1	-
$NFRV_{minsoil}$	-	0.0	-
NREC	%	30.2	-

[1)] Parameters used to assess the rate of N originating from manure products, grazing excluded.
[2)] Parameters used to asses $N_{minfert}$.
[3)] Parameters used to assess $N_{biolfix}$.
[4)] Parameters used to assess NYH.

4 DISCUSSION

4.1 Estimated NFRV of manure at farm level and effects of management

The first aim of this study was to estimate $NFRV_{manure}$ using farm data. To evaluate the method the estimates may be compared to values that were presented in literature on field trials. Our estimates of $NFRV_{manure}$ for grassland (0.71 in permanent grassland and 0.63 in temporary grassland) are in line with the range of 0.47-0.79 found in field trials in grassland and reported by Schils and Kok (2003), Reijs et al. (2007), Schröder et al. (2007), Verloop and Hilhorst (2011) and Sieling et al. (2013). $NFRV_{manure}$ in maize was high (0.97) as compared to the range of 0.12-0.75 obtained in other field trails with maize (Motavalli, 1989; Muňoz, 2004; Sieling et al., 2013; Schröder et al., 2013). Together these estimates for specific crops result in the farm average $NFRV_{manure}$ (0.70) for which we have no reference, as to our knowledge this is the first attempt in literature to estimate NFRV at farm level. The farm $NFRV_{manure}$ of 0.70 seems rather high but realistic for a farming system with optimized manure management and restricted grazing intensity. However, the remaining difference between $NFRV_{manure}$ and $NFRV_{minfert}$ (0.30) suggests that further optimization could lead to a further improvement of N utilisation.

To discern effects of manure management on utilisation of manure-N, the dynamics of $NFRV_{manure}$, NUE and N yields over time were evaluated and their respective values in systems A, B, C, and D were compared. The dynamics of $NFRV_{manure}$ in grassland and maize (Figure 7) showed strong fluctuations with seemingly unrealistically high and low values. These fluctuations urge to be cautious while interpreting trends. The increase of $NFRV_{manure}$ observed in permanent grassland and maize can, though significant (Table 4), only be interpreted as a weak indication. The means of $NFRV_{manure}$ for management systems (A-D) were not significantly different for all crops (Table 5). This must be attributed to the high year-to-year fluctuations of $NFRV_{manure}$ values within management systems, considering that for instance in permanent grassland even the $NFRV_{manure}$ of 0.49 in system A is not significantly different from 0.94 in system C.

The year-to-year fluctuations of NUE are less extreme, at least in grassland. For permanent grassland NUE seemed to increase somewhat (Table 4, Figure 5) and for temporary grassland NUE did not change significantly. Hence, NUE did not respond negatively to the decrease of $N_{minfert}$ inputs in grassland since 2000. That is remarkable, i.e., a decline of NUE might be expected because $N_{minfert}$ is considered to be the N source, besides N fixed by clover, with the highest N fertilization value and, consequently contributes strongly to available N (NA, eq. 2). However, NUE may also have been affected by weather which should then be reflected by high correlations between NREC (modelled for the specific years on the basis of weather data, Table 3) and NUE. However, the correlations between NREC and NUE for grassland were absent (Section 3). Thus it seems likely, that the reduction of $N_{minfert}$ was compensated for by other sources. Atmospheric deposition was approximately constant and net mobilisation from soil showed no substantial increase since 2000 (Table 3). In temporary grassland higher N fixation was observed in 2002-2004 but not in later years and in permanent grassland N fixation was more or less constant. Hence, it is plausible that the remaining source, manure-N, was better utilized and thus prevented a decline of NUE when $N_{minfert}$ was withheld. For temporary grassland also N fixation may have played a role in 2002-2004. Similar to permanent grassland, for maize some increase in NUE was observed (Figure 5), high N recoveries (Table 3) were evenly distributed over the research period and N mobilisation from soil did not increase over time (Table 3). Hence, also the results of maize suggest an increase of the part of N in manure that was available for crop uptake. However, the mean NUE for specific crops and management systems (A-D, Table 5) do not allow extending the above reasoning concerning NUE and manure-N utilisation to specific management systems or measures, because only in temporary grassland significant differences were observed between NUE in the distinct management systems.

The N yields in grassland declined gradually over time since 2002 (permanent grassland) and 2000 (temporary grassland, Figure 4, Table 4). Probably the decline was caused by the reduced mineral fertilizer-N inputs since 2000. This implies that changes that were made in the management of manure were not sufficiently effective to make chemical fertilizer-N inputs redundant. This should not be interpreted as a failure of the measures that were implemented to increase $NFRV_{manure}$ but merely as a too ambitious cut down of mineral N fertilizer use in the system. The total amount of available N in grassland (NA, eq. 2) would have been maintained at the targeted level by a more careful adjustment of the $N_{minfert}$.

4.2 Estimating $NFRV_{manure}$ based on farm data

For specific years $NFRV_{manure}$ shows great fluctuations over time with occasionally unrealistically high and low values (Figure 7). This might (partly) be explained by accumulation of uncertainties underlying the calculation of $NFRV_{manure}$. The uncertainty analysis performed for grassland indicates that the error terms associated with input variables result in a rather wide range of possible outcomes of $NFRV_{manure}$. Because of these uncertainties and the (partly) associated fluctuation in estimates $NFRV_{manure}$, even substantial changes in the $NFRV_{manure}$ were not statistically significant. NREC, $N_{minsoil}$ and, to a lesser extent $N_{biolfix}$, contribute most to the uncertainties as indicated by their Top Marginal Variances (Table 6). Consequently, the estimation of $NFRV_{manure}$ could be improved, by enhancing the precision of NREC and $N_{minsoil}$ in particular. More reliable estimates of annual NREC could be obtained when establishing a few mineral fertilizer-N trials including control plots across the farm. The use of the method of in situ measurement of N mineralization in tubes to estimate annual fluctuations of net N mobilisation might be reconsidered as a problem of this method is the high spatial variability (Verloop et al, submitted), which makes it hard to interpret the mineralization data and assess the representativeness of these data. The uncertainties related to $N_{biolfix}$ could be avoided easily by further testing the proposed method to estimate $NFRV_{manure}$ in fields without clover. If more N is released from ploughed-in grassland in later years (i.e. >2nd year maize) than we assumed, the $NFRV_{manure}$ in maize will be overestimated. Such a phenomenon may have played a role in our analysis. If so, it also implies that this additional N must have been invested in the grassland phase. Consequently, an overestimation of the $NFRV_{manure}$ in maize is then linked with an underestimation of the $NFRV_{manure}$ in grassland. Hence, the followed method appears more suitable and robust for analyses at the level of rotation and of farms as a whole. Further development of the method is important because the method facilitates on-farm explorations, to convince farmers of the fertilizer value of their manures, and thus contributing to lower N surpluses and fewer emissions to the environment.

5 CONCLUSIONS

A method providing estimates of the N fertilizer replacement value of manure ($NFRV_{manure}$) on the basis of farm data was tested on experimental dairy farming 'De Marke'. This study showed that:

1. Estimates of $NFRV_{manure}$ on farm scale (0.70), temporary grassland (0.63) and permanent grassland (0.71) were in the range of data found in literature for field experiments, but $NFRV_{manure}$ for maize (0.97) was high.
2. Estimates of $NFRV_{manure}$ for specific years and crops show strong fluctuations which are caused partly by accumulation of uncertainties of input parameters, in particular of N recovery, soil-N mobilisation and to a lesser extent of parameters related biological N fixation.
3. The proposed method to determine $NFRV_{manure}$ in a whole farming system should be further developed and tested, to enhance the reliability of the estimates.
4. The estimates of $NFRV_{manure}$ and NUE in permanent grassland and maize showed some increase over the research period; for temporary grassland there was no significant change. We have indications that the increase can at least partly be attributed to a change in manure management.

Appendix I: *Description of the input data[1] used for uncertainty analyses of $NFRV_{manure}$ and NUE in grassland.*

		Characterisation of input		Contribution to uncertainty (%)
Input variable	**Unit**	**Mean**	**Variance**	**Distribution**
N mass fraction manure[2]	kg m^{-3}	4	0.05	lognormal
Manure rate[2]	m^3 ha^{-1}	48	0.05	lognormal
N grazing excreta	kg ha^{-1}	79	0.2	lognormal
N mass fraction min fertilizer[3]	kg kg^{-1}	0.27	0.025	lognormal
Mineral fertilizer rate[3]	kg ha^{-1}	174	0.025	lognormal
Aboveground clover biomass[4]	Mg ha^{-1}	1.83	0.3	lognormal
Biological N fixation by clover[4]	kg Mg^{-1}	30	0.3	lognormal
N deposition	kg ha^{-1}	38	0.17	lognormal
$N_{minsoil}$	kg ha^{-1}	-20	4.08	normal
Dry matter yield crops[5]	kg ha^{-1}	8882	0.05	lognormal
N content in crops[5]	kg kg^{-1}	0.03	0.05	lognormal
$NFRV_{biolfix}$	-	1.00	0.15	lognormal
$NFRV_{dep}$	-	0.75	0.15	lognormal
$NFRV_{minsoil}$	-	0.75	0.075	lognormal
NREC	%	0.75	0.075	lognormal

[1] Variance and mean of NREC were taken from Ten Berge et al. (2007); other data were taken from Oenema et al. (submitted).
[2] Parameters used to assess the rate of N originating from manure products, grazing excluded.
[3] Parameters used to assess $N_{minfert}$.
[4] Parameters used to assess $N_{biolfix}$.
[5] Parameters used to assess NYH.

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CHAPTER

5

P-EQUILIBRIUM FERTILIZATION IN AN INTENSIVE DAIRY FARMING SYSTEM: EFFECTS ON SOIL-P STATUS, CROP YIELD AND P LEACHING

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ABSTRACT

In the coming decade, European dairy farms are obliged to realize a balance between phosphor (P) inputs to their farmland (in inorganic fertilizers and manure) and outputs (in crop products), the so-called P-equilibrium fertilization. The objective of the present study is to analyze the long-term effects of P-equilibrium fertilization on soil-P status (total soil-P and available soil-P), crop yield and P leaching on dry sandy soil, using data from experimental dairy farm 'De Marke', where P-equilibrium fertilization has been applied since 1989. For grassland, P availability is expressed in P-Al and for arable land in Pw. Total and available P status were monitored in the upper topsoil (layer 0-0.2 m). Total soil-P was also monitored in the lower topsoil (layer 0.2-0.4 m) and in the subsoil (0.4-0.6 m). From 1989 to 2006, Pw and P-Al (means of all farmland) decreased by 26 and 25%, respectively. In the same period, mean total P content of the farmland decreased by 16%. There was a large variation in initial P status (1989) of the various plots. The rate of decline in all soil-P indicators was positively correlated to their initial values. In plots with the lowest initial values, P status did not change, while in plots with high initial values it tended to stabilize at lower levels. At equilibrium-P fertilization, Pw is estimated to stabilize at 20. This is lower than the recommended P status of Dutch soils used for maize cropping. P-Al is estimated to stabilize at 30-40, which corresponds to the current recommendations for grassland. The data show that at P-equilibrium fertilization, soil available-P status is higher in a maize-ley rotation than in permanent grassland. The decline in total P and available P did not affect crop yield, nor did it affect the P concentration in groundwater, but at 'De Marke', P emission to groundwater is generally low. The results obtained suggest that P-equilibrium fertilization can be compatible with efficient crop production.

1 INTRODUCTION

At dairy farms, phosphor (P) is an indispensable external input to replenish the P exported in products and the P emitted to the environment (Von Liebig, 1841). During recent decades, however, P inputs to dairy farms in purchased feed and fertilizer have increased disproportionally in comparison to outputs in milk and meat (Edwards and Withers, 1998; Halberg, 1999; Aarts et al., 2000; Watson et al., 2002; Pfimlin et al., 2006). Currently, P inputs into commercial dairy farms in the Netherlands amount to 25 kg ha^{-1}, while output with products is only 14 kg ha^{-1} (Aarts et al., 2009). About 20 kg P ha^{-1} is imported in feeds, purchased to meet animal feed requirements, in addition to home-grown crops. The remaining 5 kg ha^{-1} is in inorganic fertilizer-P (Aarts et al., 2009). The difference between P inputs and outputs, i.e. the P surplus, may vary in Europe from 10 to 72 kg ha^{-1} (Pfimlin et al., 2006). The high P input in external feeds results in a high manure-P production, most of which is applied to the farmland, often in combination with fertilizer-P application. This may lead to higher P inputs to the farmland than crops are able to absorb and thus to a soil-P surplus (Aarts et al., 2000).

A soil-P surplus contributes to the build-up of P in agricultural soils and may ultimately lead to P saturation of the soil, resulting in an increased risk of leaching to groundwater (Van der Zee and Van Riemsdijk, 1988; Hooda et al., 2001; Kronvang et al., 2007). In areas with steep slopes also surface runoff contributes to P transport to aquatic systems (Haygarth et al., 1998). Enrichment of water bodies with P causes eutrophication and thus negatively affects the quality of aquatic ecosystems (Tunney et al., 1997; Hosper, 1997; Ulén et al., 2007). The European Water Framework Directive (WFD) aims at preserving water quality by reducing pollution (Anonymous, 2000). With respect to P, WFD urges EU-member states to implement regulations preventing P accumulation in agricultural soils, notably in regions with intensive animal production (Anonymous, 2000; Neeteson et al., 2006). Therefore, intensive dairy farms are obliged to balance the P input to farmland through manure and inorganic fertilizer with P output in crop products, i.e. by applying P-equilibrium fertilization. The P concentration in manure, and hence the P application rates to farmland, can be reduced through feeding the cattle with concentrates low in P, without affecting herd performance (Morse et al., 1992; Valk and Šebek, 1999). When this strategy does not reduce manure-P inputs sufficiently, the surplus manure has to be exported from the farm to be able to comply with P-equilibrium fertilization.

However, great uncertainty exists about the long-term effects of P-equilibrium fertilization on plant available soil-P. The level of plant-available P might ultimately decrease to such an extent that the P requirement of crops may not be fully satisfied anymore (e.g. Leigh and Johnston, 1986). To maintain an agronomically acceptable level of available P, farmers' practice and fertilizer recommendations suggest that a supply of 10-30 kg P ha^{-1} yr^{-1} is required to compensate for the continuous transfer of plant available soil-P to non-available P through leaching and fixation (Aarts et al., 2000; Anonymous, 2002; Neeteson et al., 2006). The observation in a number of field experiments that for optimum crop production P input should exceed P output may support the suggestion that part of supplied P cannot be utilized by the crop, because it has been transferred to the non-available P pool (Gallet et al., 2003, Neeteson et al., 2006, Syers et al., 2008). Syers et al. (2008), however, argue that the importance of P fixation is often overestimated, due to biased interpretation of the results of field trials. They point at (1) investigations in which a weak yield response to P application was interpreted as an effect of low availability of supplied P, whereas it was probably caused by saturation of crop needs, (2) the short response time of experiments that leads to underestimation of the long-term recovery of P by crops, and (3) insufficient attention for residual effects of earlier supplies of P. Deliberately applying more P than is required for maximum crop production to maintain an adequate level of available soil-P, may thus not always be necessary, as is often presumed.

It should be noted that when P-equilibrium fertilization is applied to soils with a high P status, P may still move to the subsoil or leach to groundwater (Van der Zee and Van Riemsdijk, 1988; Schippers et al., 2006). If this is the case, P mining, i.e. deficit P fertilization, may be required to restrict P leaching (Koopmans et al., 2004). For soils with different P status, subjected to P-equilibrium fertilization, data on P losses to the subsoil (layers below the root zone) and groundwater are scarce (Koopmans, 2004).

Uncertainties thus still exist on the extent to which P surplus applications are needed and on how soil-P status and crop yields will respond to long-term P-equilibrium fertilization. The aim of the current study is to evaluate the effects of P-equilibrium fertilization in a dairy production system on: (1) the development of the soil-P status, (2) crop yields, and (3) P losses to subsoil and groundwater. The evaluation is based on data that have been collected at experimental dairy farm 'De Marke', where P-equilibrium fertilization is tested at farm scale.

2 MATERIALS AND METHODS

2.1 Experimental farm 'De Marke'

2.1.1 Site characteristics

'De Marke', established in 1989 and fully operational since 1992 (Aarts et al., 2000), is situated in the eastern part of the Netherlands on very light sandy soil, characterized by a 0.25-0.30 m anthropogenic upper layer, with an average organic matter content of 4.8% and a bulk density of 1300 kg m^{-3}, overlying a layer of yellow sand, very low in organic matter and hardly penetrable by roots (Aarts et al., 2000). At the start of 'De Marke', the soil was ploughed to a depth of 0.3 m to homogenize the topsoil. Soil water content is usually at field capacity in spring (0.18 g per g oven-dry soil), decreasing in mid-summer to 0.12 g per g oven-dry soil under normal rainfall conditions and to 0.10-0.05 g per g oven-dry soil in dry summers. Groundwater depth is 1-3 m below soil surface during summer and 0-1 m during winter. In the research period, precipitation (mean annual 788 mm) was on average rather evenly distributed throughout the year, with a minimum of 47 mm in April and a maximum of 82 mm in July. As the groundwater level is relatively deep and the water holding capacity of the topsoil is low, available water is often limiting plant growth during the summer period.

2.1.2 Farm layout

The experimental farm was established to test an environmentally and economically sustainable dairy production system on a dry sandy soil, with a milk production intensity of at least 12,000 kg ha^{-1} (a common level in the Netherlands). The objective was to meet environmental standards for N and P. Environmental standards were translated into corresponding surpluses that were used as boundary conditions for farm performance. For P, the maximum surplus was calculated to be 0.45 kg P ha^{-1} yr^{-1} corresponding to the Dutch surface water threshold value: 0.15 mg P l^{-1}. To realize this, a P-equilibrium fertilization strategy was applied (Aarts et al., 1992). To ensure that a zero P surplus was realized, P inputs were adjusted to the expected P output in milk and meat. To satisfy feed (energy and protein) requirements, with minimum use of concentrates, production of home-grown feeds was maximized through optimal fertilizer management.

The farm area (55 ha) is divided into permanent grassland (11 ha) and two grassland-arable crop rotations (ROTI and ROTII, Table 1). The rotation scheme of the arable phase was slightly modified in the course of time. From 1993 until 1996, fodder beet was the first arable crop to succeed the grassland phase. From 1996 until 1999, maize was the only arable crop, after which triticale and spring barley were introduced as the last crop in the arable phase. During the cultivation of maize, Italian ryegrass was sown in early summer as a catch crop between the maize rows. A total of 31 ha of grassland was available for grazing by 80 cows and young stock. To prevent problem swapping, environmental goals for the farming system were to be met without exporting slurry. See Verloop et al. (2006) for further details on the farm layout.

Table 1: *Farm plan of 'De Marke' (ha) and crop sequence of the two rotation regimes.*

Component	Average
Total area	55
Permanent grassland	11
Ley	20
Maize	17
Triticale, spring barley	7
Rotation I	3 years grassland → 3 years arable crops → etc.
Rotation II	3 years grassland → 5 years arable crops → etc.

2.1.3 Farm fertilizer strategy

Fertilizer management aimed at matching supply and crop demand. No inorganic P fertilizer was used, because it was hypothesized that crop-P requirements could be met with manure-P only.

Annual manure-P supply was related to the phase of the rotation (Table 2). Manure-N supply was adjusted to meet N requirements in all phases of the rotation, which was high in ley and low in maize. Consequently, in the crop rotations, P supply to ley exceeded crop P uptake, resulting in a positive P balance during the ley phase, while it was lower than crop P uptake in the arable phase, resulting in a negative P balance. The overall result of manure-P supply in the crop rotations was a zero P surplus. The N requirement of permanent grassland was constant over time and, consequently, so was the N-manure supply and P-manure supply, resulting in a lower average annual P supply than in ley. P input in atmospheric deposition was estimated to be 1 kg ha^{-1} yr^{-1} (Aarts, 2000).

On grassland, fertilizer application was restricted to the period March 15 until August 1. Catch crops, grown in the arable phase, were ploughed-in during the first week of March. On maize, fertilizer was applied at the end of April, just before sowing. From early 2003 onwards, slurry was fermented before application.

Table 2: *Phosphorus supply (kg ha^{-1}) to maize, triticale and grassland at 'De Marke'; means for 1993-2006.*

Land use	Arable phase			Grassland phase			Continuous
	1st yr	2nd yr	>2nd yr	1st yr	2nd yr	>2nd yr	
Rotation I	5	12	22	42	46	45	-
Rotation II	12	15	21	41	40	36	-
Permanent Grassland	-	-	-	-	-	-	35

2.2 Data collection

The total area of 'De Marke' was divided in 24 fields and each field was sub-divided in two or three plots of about 1 ha (resulting in 59 plots). For each of the 24 fields, management, nutrient flows and yields were monitored since 1993. Rate, timing and method of manure application, and dates of harvesting/cutting, grazing, re-sowing and ploughing were monitored. Organic matter flows (harvested crop material and applied slurry) were measured by weighing and the related nutrient flows were quantified on the basis of chemical analyses of representative samples. Fresh grass consumption by grazing cattle was calculated by estimating standing biomass just before and after grazing and correcting for growth during grazing. P in excreta voided during grazing was estimated from feed intake, by subtracting (*i*) P output in milk and meat and (*ii*) P excretion indoors. P output in milk was derived from chemical analyses of milk and monitoring of milk production.

Soil samples of the upper layer of the rooting zone (0.2 m upper topsoil) were taken in each of the 59 plots, at the start of 'De Marke' in 1989 and annually from 1994 to 2006 (Aarts et al., 2000). In each plot, 40 cores were mixed into a composite sample. Total soil-P was determined, using destruction with Fleischman acid (Murphy and Riley, 1962). Plant-available P was expressed as Pw (Table 3) for arable crops (Van der Paauw, 1971) and P-Al (Table 3) for grassland (Van der Paauw, 1956) in accordance with their use as indicators in Dutch fertilizer recommendations. Pw is an indicator for P that is readily available for arable crops (maize) and P-Al is an indicator for P that comes available during the growing season for grassland, respectively (Van der Paauw, 1956, 1971; Neyroud and Lisscher, 2003). Samples from the 0.2-0.4 m layer, the lower layer of the rooting zone (lower topsoil; 31 plots) and the 0.4-0.6 m layer, the layer below the rooting zone (subsoil; six plots), selected randomly from all plots, were collected at several occasions between 1991 and 2004. In 1989 and from 1994 to 2006 at the end of each year, the upper groundwater was sampled at 170 points, distributed proportionally per field on the basis of the area. Boreholes were constructed up to 0.8 m below the groundwater table. Water, sampled with a well-screen, using a suction pump, was directly filtered and acidified with sulphuric acid. P concentration was determined colorimetrically (Boumans et al., 2001).

Table 3: *Soil analyses for P.*

P-test	Method	Unit	References
Pw	1:60 (volume/volume), extraction with water, 22 h incubation, 1 h shaking	mg P_2O_5 per litre soil	Van der Paauw (1971)
P-Al	1:20 (dry weight/volume), extraction with 0.1 ammonium lactate + 0.02 m hydrochloric acid, pH 3.7, 1.5 h shaking	mg P_2O_5 per 100 g dry soil[a]	Van der Paauw (1956)
Total soil-P	Destruction with Fleischman acid; spectrophotometric determination, following coloring with ammonium molybdate, antimone and ascorbic acid	mg P_2O_5 per 100 g dry soil[a]	Murphy and Riley (1962)

[a] 1 mg P_2O_5 per 100 g dry soil equals 4.37 mg P kg^{-1} dry soil.

2.3 Data analyses

We analysed effects of equilibrium fertilization by evaluation of (*i*) development of soil-P status (upper topsoil), (*ii*) crop yields and (*iii*) losses to subsoil and groundwater. For that purpose, we first analysed the variability in initial soil-P status: P_{init}, Pw_{init} and $P\text{-}Al_{init}$.

2.3.1 Development of soil-P status

The development of the P status in the upper topsoil was analysed at three scales: (*i*) farm, (*ii*) land use: permanent grassland, ROTI and ROTII and (*iii*) plots. Effects at farm scale were analysed by assessment of total soil-P and plant-available P (means of all plots). Effects at land use scale (all uses subjected to P-equilibrium fertilization) were analysed by comparing the P status as observed since the year 2000. We compared mean total soil-P and plant-available P of plots in permanent grassland, with means of plots in ROTI and means of plots in ROTII. The effects of land use type on P status were corrected for (rather small) differences in initial P status (values in 1989).

At the start of 'De Marke', soil-P status varied from rather low to very high, depending on manure application rates in the period of commercial exploitation before its establishment. This allows analysis of effects of P-equilibrium fertilization on plots with different initial soil-P status. For total P, Pw and P-Al, the relationship between initial status and their development was explored for all individual plots, using the measurements in the course of time. Eight of the 59 plots were excluded from the analysis, since they were not part of the 'De Marke' system during the entire research period (1989-2006). P dynamics were explored for each plot by linear regression analysis. This resulted in estimates of the initial P status and the rates of decline (dP/dt, dPw/dt and dP-Al/dt) for each individual plot. We expected the rate of decline in plots with high initial status to level off in the course of time. To test this hypothesis we also fitted the dynamics with an exponential model (non-linear regression). Next, the relationship between estimates of the initial status and estimates of the rate of decline was analysed by linear regression.

2.3.2 Crop yields

Effects of P-equilibrium fertilization on crop yields were analysed at two scales: (*i*) farm and (*ii*) field. Effects at farm scale were analysed by assessment of P uptake and dry matter yield (means of all fields) in the course of time. At the field scale, the relation was explored between P status in the upper topsoil and yield for the main crops: grass and maize. The effect of P status on maize production for the period 1994 to 2006 was explored at the field scale by linear regression of average Pw values (arithmetic average of Pw values of the plots belonging to the field) on maize P uptake and dry matter yield. The same procedure was followed for P-Al in grassland. Effects of temperature, radiation and moisture availability and their interactions may result in different production potentials each year. Consequently, temporal variability in weather conditions may lead to differences in crop yields that could interfere with the analysis of crop response to P status. To take this variability into account, three weather types were defined with their corresponding production potential: weather type 1 (WT1) with low production potential, weather type 2 (WT2) with intermediate production potential and weather type 3 (WT3) with high production potential. For each year, potential annual grass production was simulated with the weather-based model CNGRASS (Conijn, 2005), and potential maize production with the model WOFOST (Supit et al., 1994), based on daily precipitation, radiation and temperature during the growing seasons. On the basis of calculated potential production, years were categorized as WT1, WT2 or WT3. Effects of P status on crop yield were analysed for years with the three weather types separately.

2.3.3 Losses to subsoil and groundwater

Effects on P losses to the lower topsoil and the subsoil were analysed at two scales: (*i*) farm and (*ii*) plot. Effects at farm scale were analysed by comparing the change in total soil-P in the upper topsoil (means of all plots) over time with the change in the lower topsoil and the subsoil (means of plots). At the plot scale the same procedure was followed for individual plots. Effects on groundwater were analysed at two scales: (*i*) farm and (*ii*) field. Effects at the farm scale were analysed by assessment of the P concentration in groundwater (means of groundwater samples). Effects at field scale were analysed by exploring the relationship between soil-P status and P concentration in groundwater for the years 1994-2006. P concentration in groundwater was calculated as the arithmetic mean of the values observed in the (3-6) boreholes per field. Mean soil-P status was calculated as the arithmetic mean of the values of the plots of each individual field.

3 RESULTS

3.1 Development of soil-P status

For total soil-P, Pw and P-Al, the initial P status in the upper topsoil was 169 (range 116-289), 57 (range 26-141) and 75 (range 38-142), respectively. High and low values were evenly distributed over Permanent Grassland, ROTI and ROTII. The distributions of P_{init}, Pw_{init} and $P\text{-}Al_{init}$ were all skewed (Fig. 1). P_{init}, Pw_{init} and $P\text{-}Al_{init}$ were highly correlated, with correlation coefficients of 0.81 for P_{init} and Pw_{init}, 0.92 for P_{init} and $P\text{-}Al_{init}$ and 0.86 for Pw_{init} and $P\text{-}Al_{init}$. The P status of the plots can thus be characterised in terms of either total soil-P, Pw or P-Al. In the layers 0.2-0.4 m (n = 31) and 0.4-0.6 m (n = 6), P_{init} (1991) was 94 ± 27.2 and 28 ± 5.9 (mean ± standard deviation), respectively, i.e. appreciably lower than in the top 0.2 m.

At the farm scale, annual P surplus of the soil varied from -7 to 6 kg ha^{-1}, with a long-term average of -1 kg ha^{-1} (Table 4). Since the application of P equilibrium fertilization, total soil-P, Pw and P-Al all decreased: Pw by 26% (the value in 2006 as compared to the initial value), P-Al by 25% and total soil-P by 16%. Hence, plant-available P decreased more strongly than total soil-P.

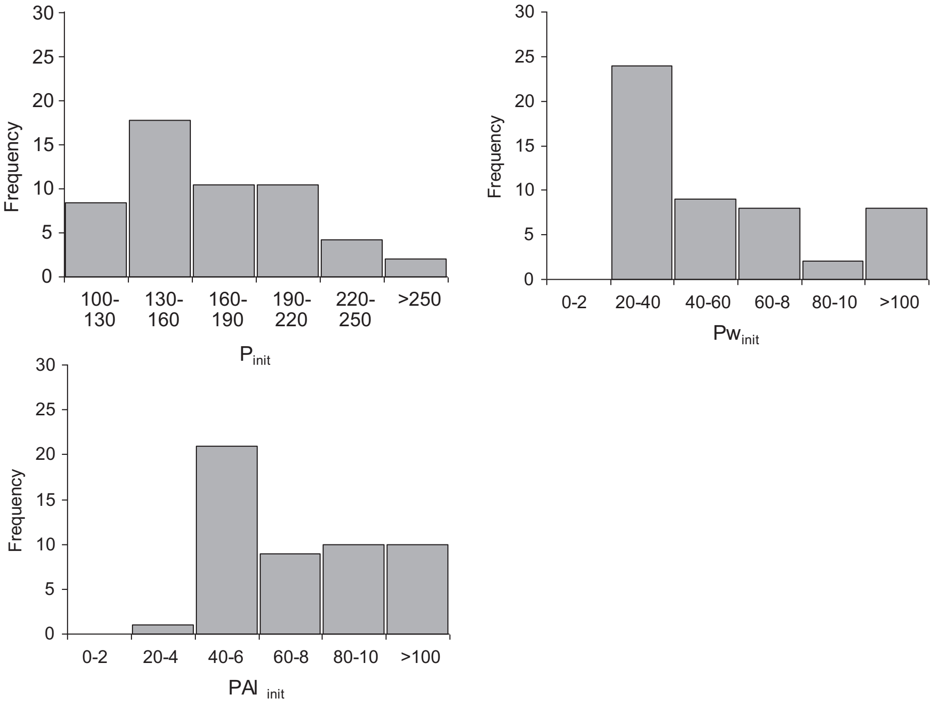

Figure 1: *Distribution of total soil-P and plant-available P in the layer 0-0.2 m, observed in 1989 (P_{init}, Pw_{init} and P-Al_{init}).*

Table 4: *Development over time of P surplus (kg ha^{-1} yr^{-1}), total soil-P, Pw, P-Al, crop P uptake and dry matter (DM) yield (Mg ha^{-1}) and P concentration in groundwater (mg l^{-1}) at 'De Marke'; means for the whole farm.*

Year	Surplus	P status			Crop uptake/yield		P groundwater
		P	Pw	P-Al	P	DM	
1989	-	169	57	75	-	-	0.04
1993	0	-	-		33	11.1	0.01
1994	6	165	49	69	34	9.6	0.01
1995	0	166	42	64	28	9.1	0.02
1996	1	162	44	65	26	9.4	0.02
1997	1	164	44	65	29	10.6	0.10
1998	1	160	48	61	29	9.2	0.02
1999	4	161	45	65	32	10.6	0.03
2000	-4	162	38	57	36	11.9	0.02
2001	-7	158	46	63	36	11.9	0.04
2002	-2	151	47	63	34	11.7	0.02
2003	-4	160	46	60	28	9.8	0.04
2004	-4	159	41	62	33	10.9	0.02
2005	1	154	38	61	33	11.1	0.03
2006	-2	142	42	56	28	9.4	0.02

Table 5 shows the P status in the upper topsoil for the different land use types (each subjected to P-equilibrium fertilization). The results refer only to the period 2000-2006, because land use effects were expected to become significant some years after the start of P-equilibrium fertilization, while the results of the earlier years would mainly reflect the initial soil-P status. Pw was lower under permanent grassland than under ROTI and ROTII. Trends for P-Al and total soil-P were similar, but differences were not significant. The average rate of decline (units per year) in total soil-P, Pw and P-Al in the various plots decreased with decreasing initial values (Fig. 2). Total soil-P hardly changed in plots with low initial values (P_{init} range 100-130). Similarly, Pw and P-Al did not tend to decrease in plots with Pw_{init} of 20-30 and $P\text{-}Al_{init}$ of 30-40, respectively (Fig. 2). The rates of decline in total soil-P, Pw and P-Al can accurately be described as simple linear functions of their initial values. For most plots, the dynamics of total soil-P could be accurately described as a linear function of time. Non-linear regression did not result in a better fit. For most plots with a high initial-P status, however, the dynamics of Pw and P-Al could be described more accurately by an exponential function, indicating that the rate of decline in those plots tended to level off.

Table 5: *Estimates of Pw, P-Al and total soil-P in permanent grassland (Pg), ROTI and ROTII; means of 2000-2006 of plots corrected for Pw_{init}, $P\text{-}Al_{init}$, P_{init} (see text for explanation).*

	Pg		ROTI		ROTII	
Pw	38	(2.2)	43	(1.2)	45	(1.8)
P-Al	60	(2.5)	63	(1.4)	62	(2.0)
Total soil-P	153	(4.8)	157	(2.8)	152	(3.9)

Numbers in brackets are standard errors.

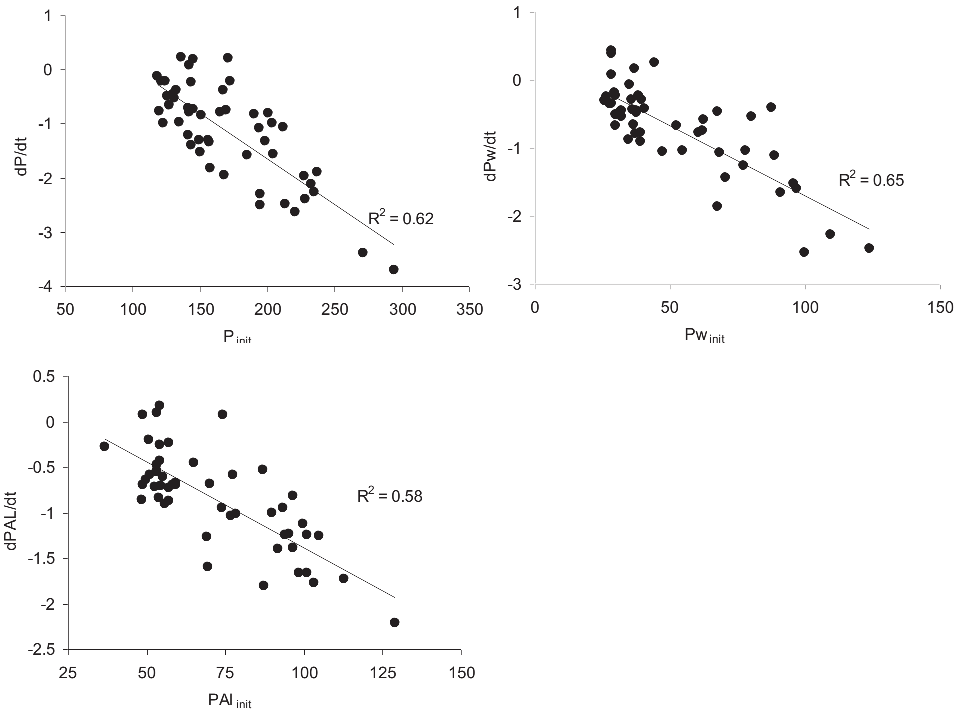

Figure 2: *Average rate of decline (units per year) in total soil-P, Pw and P-Al over the period 1989-2006, as a function of their initial value.*

3.2 Crop yield

At the farm scale, crop P uptake varied from 26 to 36 kg ha^{-1}, but P uptake was not correlated with the development of P status and did not show any trend over the research period (Table 4). The same was observed for dry matter yield. Dry matter yield was positively correlated with P uptake (Table 4).

In Fig. 3, dry matter yields of maize and ley are plotted as a function of the P status of the upper topsoil (Pw for maize and P-Al for ley). The yields are clustered according to weather types with low (WT1), normal (WT2) and high production (WT3) potential. Dry matter yields of maize and ley did not show a significant correlation with the P status of the upper topsoil. This was neither the case when the distinction between weather types was not made. Dry matter yield of maize, however, tended to be positively correlated with Pw in WT2 and WT3 years, and negatively in WT1 years (Fig. 3). Dry matter yields of ley were highly variable, irrespective of the value of P-Al, and tended to be positively correlated with P-Al in WT3 years, not correlated in WT2 years and negatively correlated in WT1 years (Fig. 3). Correlations between dry matter yield and P-Al in permanent grassland (not shown) were similar to those in ley.

In Fig. 4, P uptake by maize and ley is plotted as a function of the P status of the upper topsoil (Pw for maize and P-Al for ley). P uptake by maize and ley was not correlated with soil-P status when no distinction was made between weather types. There was, however, a positive correlation between P uptake by maize and Pw in WT2 years. P uptake by ley correlated positively with P-Al in WT2 and WT3 years (Fig. 4). There was no significant correlation between P uptake by permanent grassland and P-Al (data not shown). The relations between N uptake by maize and grass and soil-P status (data not shown) were similar to those between dry matter yields and soil-P status. The N/P ratios in maize and grass dry matter were on average 7.1 (with a variation from 4.0 to 10.4). The ratios did not show any correlation with the P status of the topsoil.

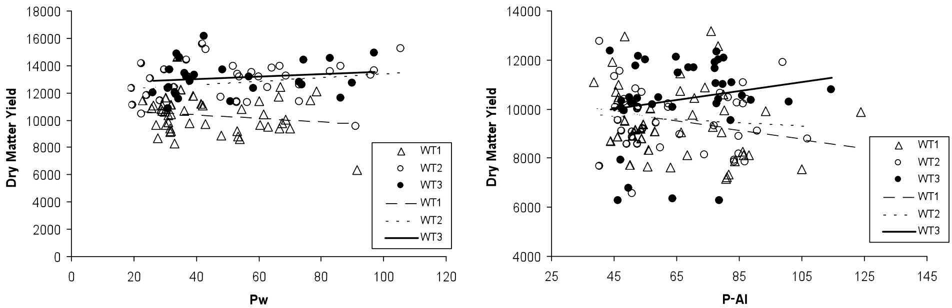

Figure 3: *Relation between Pw and dry matter yield (kg ha^{-1}) of silage maize and relation between P-Al and dry matter yield of ley in low (WT1), normal (WT2) and high (WT3) production potential years.*

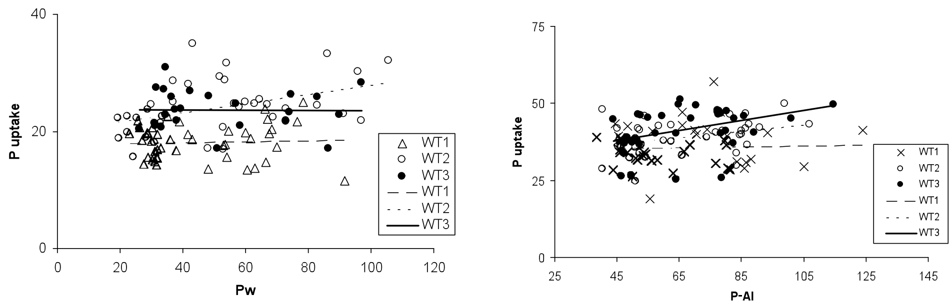

Figure 4: *Relation between Pw and P uptake (kg ha^{-1}) by silage maize and relation between P-Al and P uptake by ley in years with low (WT1), average (WT2) and high (WT3) production potential conditions, respectively.*

3.3 P losses to the lower topsoil, subsoil and groundwater

For the 31 plots selected for multi-layer sampling, total soil-P in the upper topsoil declined by 1.5 units (17 kg P ha^{-1}) per year (at a bulk density of 1,300 kg m^{-3}, 1 unit of total soil-P is equal to 11,400 kg P in a soil layer of 0.2 m). In the lower topsoil, total soil-P increased by 1.4 units (16 kg P ha^{-1}) per year, and in the subsoil it decreased by 0.3 units (3.4 kg P ha^{-1}) per year. Apparently, the decrease in P in the top layer practically equals the accumulation in the 0.2-0.4 m layer (Fig. 5). For individual plots, no significant correlation was observed between the change in total soil-P in the upper topsoil and the change in the lower topsoil and the subsoil (Fig. 6). For most plots, a decline in total soil-P in the upper topsoil coincided with an increase in the lower topsoil. However, several plots showed a decline in both, the upper and the lower topsoil, as well as in the subsoil.

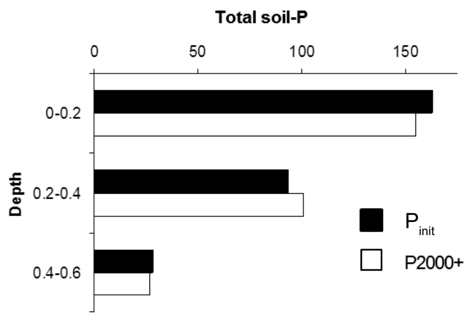

Figure 5: *Initial total soil-P (P_{init}) and means of 2000-2004 (P2000+) in the layers 0-0.2, 0.2-0.4 and 0.4-0.6 m.*

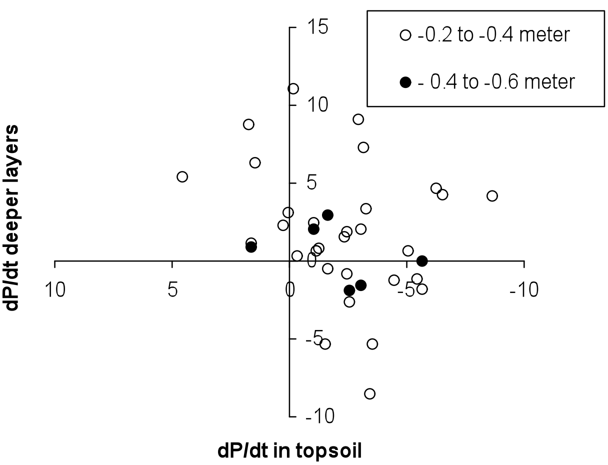

Figure 6: *Relation between the average annual rate of change in total soil-P in the topsoil (0-0.2 m) and that in the lower topsoil (0.2-0.4), respectively the sub-soil (0.2-0.6 m) for individual plots during the period 1991-2004.*

At the farm scale, the P concentration in groundwater (1-3 m below soil surface) fluctuated over time, but did not show a clear trend over the research period. Consequently, the P concentration in the groundwater was not correlated to the gradually declining P status of the soil (Table 4). It should be noted that the P concentration was well below the target level of 0.15 mg l^{-1}. In Fig. 7, the P concentration in the upper meter of groundwater is plotted as a function of Pw. Since the groundwater level varied in time and space, and the groundwater samples were collected from the upper meter of groundwater, sampling depth corresponds roughly to the groundwater level at the moment of sampling. To prevent entanglement of effects of groundwater depth and Pw, the data in Fig. 7 are presented for different groundwater level classes. P concentration in the upper meter of groundwater was not correlated to Pw in the upper topsoil at any of the groundwater levels. Occasionally, higher P concentrations were found when the groundwater level was shallower than 2 m below the soil surface, but even in those situations, P concentrations in the groundwater were not correlated to Pw in the upper topsoil. The combination of shallow groundwater depth and relatively high P concentrations was observed particularly in 1998 (Fig. 7), when precipitation in early spring was extremely high. Results for total soil-P and P-Al were similar to those presented for Pw.

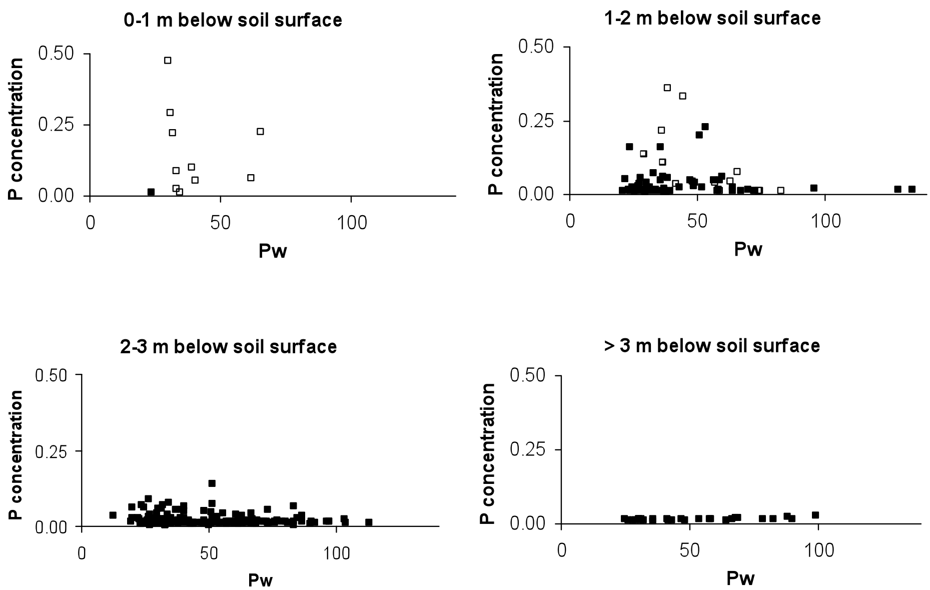

Figure 7: *P concentration in the upper meter of groundwater (mg P l^{-1}) as a function of Pw in the upper 0.2 m of the soil (both, means per field), for different groundwater depths. Open squares refer to 1998, black squares to all other years.*

4 DISCUSSION

4.1 Development of soil-P status

From Fig. 2 it can be concluded that P-equilibrium fertilization leads to a gradual decline in total soil-P, Pw and P-Al values in the upper topsoil, with average rates of decline proportional to their initial values. Consequently, in the upper topsoil, total soil-P, Pw and P-Al were more or less stable in the fields with low initial status, while these low values were approached in fields with high initial values. These values, therefore, provide an indication of the equilibrium-P status in the topsoil that will be attained after long-term P-equilibrium fertilization. Thus, we may anticipate an equilibrium total soil-P of about 120, an equilibrium Pw of about 20 and an equilibrium P-Al of about 45. These values of plant available P are somewhat higher than those predicted by Del Campillo et al. (1999), based on modelling. They explored the dynamics of Pw on farmland in a scenario where excessive P fertilization was followed by P-equilibrium fertilization. In that scenario, Pw declined to 13 for soils with a normal P-binding capacity and lower for soils with a low P-binding capacity. The difference between our results and those of Del Campillo et al. (1999) may be associated with: (*i*) a higher P-binding capacity at 'De Marke' than assumed in the scenarios (a high binding capacity is associated with a high equilibrium level of available P; Van der Zee and Van Riemsdijk, 1988), (*ii*) effects of crop rotation that are not accounted for in the chemical models used (as will be discussed below).

The decrease in Pw in permanent grassland was stronger than in ROTI and ROTII and the same tendency was found for P-Al (Table 5). Apparently, in the crop rotations a larger proportion of the total P pool was available for plant uptake. In the literature, stimulating effects of land use and soil management on P availability have been ascribed to (*i*) the release of root exudates by crops, e.g. lupine (Kahm et al., 1999; Horst et al., 2001; El Dessougi et al., 2003) and rape (Hoffland et al., 1992), (*ii*) effects of soil fauna (Breure et al., 2004) such as bioturbation or mycorhizal activity, often associated with minimum tillage management, (*iii*) the use of deep rooting crops, such as tree species that can acquire P from deep soil layers (Breman and Kessler, 1995) and (*iv*) the return of plant residues to build up a P pool that is bound to organic matter (Kabengi et al., 2003; El Dessougi et al., 2003). None of these explanations logically applies to our situation. In our cropping scheme, the stimulating root exudates should have been produced by maize and to a lesser extent small grains (triticale, spring barley), but notably maize is not renowned for stimulating P mobilization to a larger extent than grass (Kahm et al., 2002). Moreover, significant effects of root exudates have been observed particularly in extensive cropping systems with low soil-P levels and low P inputs. Concerning (*ii*), at 'De Marke', the activity of soil fauna is lower in ROTI and ROTII than in permanent grassland. With respect to (*iii*), at 'De Marke', all crops have a similar rooting depth due to the sharp boundary between the upper, relatively fertile, rooting zone of 0.3-0.4 m and the underlying layer of yellow sand. Concerning (*iv*), calculations using crop-specific default values indicate that the return flow of plant organic matter is higher in permanent grassland than in ROTI and ROTII (Verloop and Van Keulen, 2007). Presumably, in the current study, the stimulating effect of crop rotation on P availability is the result of ploughing. Perturbation of the topsoil through ploughing may mobilize P that is bound to organic matter and/or metal oxides. This implies that ploughing, associated with crop rotation, could be an important practical measure to maintain soil-P fertility on farms that have to apply P-equilibrium fertilization. This, however, holds only in flat areas. In hilly areas, ploughing may induce P runoff and hence loss of available P from the soil, particularly in the period between ploughing and the start of growth of the subsequent crops when the soil is bare (Heathwaite et al.,1999).

4.2 Crop yields

At farm scale, P uptake and dry matter yields were positively correlated. In 'normal' field trials, with P supply as the variable factor, this might be interpreted as an effect of P on crop development. In the current study, however, the opposite might be the case: an effect of crop development on P uptake, because (*i*) we hardly observe an effect of P supply (as indicated by soil-P status) on P uptake and (*ii*) N:P ratios in crops do not suggest any P limitation.

At farm scale, no significant effects were observed of soil-P status on crop P and N uptake and crop dry matter yield (Table 4). This was neither the case when the data were differentiated in low, normal and high production potential weather types (Figs. 3, 4). The temporal variability in crop nutrient uptake and crop yield is high, particularly for grass, but this is likely to be related to other factors than P availability, such as intensity of grazing, age of the sward and/or variations in cutting frequency. Moreover, the observed tendencies in the relations between soil-P status on the one hand and P uptake and crop yield on the other, were not consistent. We expected that a low soil-P status would limit yields, particularly when the production potential is high, but this occurred only in the ley. Apparently, at 'De Marke', P limitation was almost absent. This conclusion is supported by the low N:P ratio in the crops (Penning de Vries et al., 1980; Koerselman and Meuleman, 1996). The ratio of the N and P contents in plant tissue is an indication for the limitation of plant growth by either N or P: at N:P ratios >16, P is the main limiting nutrient and at N:P ratios <16, it is N (Koerselman and Meuleman, 1996). In the current study, the N:P ratio did not exceed 11, indicating that N is the main limiting nutrient for all fields and in all weather types. This is plausible, since at 'De Marke', N is restrictively applied to prevent N losses. Hence, also in fields with a soil-P status at or close to the equilibrium value, P limitation is not likely. This finding is in agreement with insights formalized in Dutch fertilizer recommendations for grassland, since the equilibrium P-Al value derived from our research corresponds to the level considered sufficient for optimum grassland production (Anonymous, 2002). For maize, however, the equilibrium Pw value (20) is 10 units below the level considered necessary for optimum production (Van Dijk, 2003).

4.3 P losses to lower topsoil, subsoil and groundwater

From 1989 to 2006, total soil-P in the upper topsoil declined by 27 units (Table 4) which is equivalent to 17 kg P ha^{-1} yr^{-1}. With a net withdrawal from the fields (supply minus export in crop products) of 1 kg P ha^{-1} yr^{-1}, average 'loss' of P would be 16 kg ha^{-1} yr^{-1}. Results of groundwater monitoring indicate that leaching to groundwater (depth 1-3 m below the soil surface) was negligible. In general, P concentrations in groundwater were far below 0.15 mg l^{-1}, at an annual precipitation surplus of 300 mm, equivalent to a P flux of about 0.5 kg ha^{-1} yr^{-1}. It can thus be concluded that P must have moved from the upper topsoil to the lower topsoil and the subsoil, but not deeper than 0.6 m below soil surface. This is illustrated by the developments in P contents in the different soil layers (difference between the initial contents and the means of 2000-2004; Fig. 5): P accumulation in the deeper layers practically equalled P depletion in the upper topsoil. The lack of significant correlation between P depletion in the upper topsoil and P accumulation in the lower topsoil and subsoil per plot (Fig. 6) may be associated with the inherent soil heterogeneity. The anticipated decrease in total soil-P in the upper topsoil from the present level (142) to the equilibrium level (120) is equivalent to a P 'loss' from the top layer of about 250 kg ha^{-1}. This amount will be transported to deeper layers after the introduction of P-equilibrium fertilization.

The analysis of P leaching in situations with high and low groundwater levels does not suggest any effect of soil-P status on P leaching. High P concentrations were observed more frequently in situations with a shallow groundwater table than in situations with a deep groundwater table. The risk of P leaching on free draining sandy soils seems to be determined by groundwater depth rather than by soil-P status. This is in agreement with the known mechanisms of P binding to soil. Both, in the top layer and in the subsoil, P can be immobilized by Fe- and Al-(hydr)oxides (Freese et al., 1992). In the situation of shallow groundwater levels, however, these (hydr)oxides are (partially) dissolved, leading to release of P. Hence, transport of P to the subsoil may result in an increased risk of P leaching to groundwater. Soil-P mining by applying less P than P-equilibrium fertilization may be an appropriate strategy to reduce risks in the long term, but management of groundwater levels seems crucial to reduce risks of P leaching in the short term.

5 CONCLUSIONS

It is to be expected that future legislation on nutrient management will force farmers to apply P-equilibrium fertilization. The results obtained on dry sandy soil suggest that P-equilibrium fertilization can be compatible with efficient crop production. Particularly on P-rich soils, P-equilibrium fertilization will lead to a decline in total P content in the upper 0.2 m of soil. However, continuation of P-equilibrium fertilization will result in an equilibrium soil-P status with a total soil-P content of 120 mg P_2O_5 per 100 g dry soil, a Pw value of 20 mg P_2O_5 per dm^3 soil and a P-Al value of 45 mg P_2O_5 per 100 g dry soil. A crop rotation with maize, ley and spring barley/triticale resulted in higher values for crop-available P in the soil than permanent grassland, which is probably caused by ploughing. Ploughing, associated with crop rotation, might thus be an adequate strategy to prevent a strong decline in soil-P status when P-equilibrium fertilization has to be applied. Soil-P availability did not affect dry matter production and P and N uptake by maize and grass over the full range of values, including those corresponding to the equilibrium status. Hence, P-equilibrium fertilization is not likely to affect crop production as is often presumed, provided that appropriate crop and nutrient management practices are implemented. Negligible P concentrations were detected in the upper groundwater (at depths of 1-3 m), so the observed loss of P from the upper topsoil must have resulted in immobilization in the lower topsoil or the subsoil. P concentration in groundwater was not correlated to soil-P status. Actual risks of P leaching to groundwater seem to be determined by groundwater level rather than by soil-P status. Mining of P from P-enriched soils may be required to prevent P leaching to groundwater in the long term.

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CHAPTER 6

ORGANIC MATTER DYNAMICS IN AN INTENSIVE DAIRY PRODUCTION SYSTEM ON DRY SANDY SOIL

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ABSTRACT

In many studies, possibilities are being explored to adjust farm management to increase or maintain soil organic matter (SOM) contents in agricultural soils. Some options may be conflicting with efficient use of nutrients (N and P). This study explores effects of efficient nutrient management on SOM dynamics on an experimental dairy farm with permanent grassland and a crop rotation in which grass and arable crops are alternated. The farm is located on light sandy soil in the Netherlands. The study was based on: (*i*) trend analyses based on data of SOM mass percentage from 1989-2010, and (*ii*) simulations of long-term (50 years) developments of SOM mass % for management without manure digestion, with mild (conventional) and strong manure digestion and with and without application of catch crops after maize. The trend analyses showed that soil organic matter percentage of the 0-0.2 m layer was approximately stable in permanent grassland. In the crop rotation SOM mass percentage decreased on average by 0.04 per year and the decline did not slow down over time. SOM decline tended to be higher on plots with high than on plots with low initial SOM mass %. Decomposition was described using a mono-component model with a time dependent relative decomposition rate. Decomposition rates in the crop rotation were not higher than in permanent grassland indicating that tillage did not affect decomposition rate and that SOM development was dominated by substrate input. Estimated decomposition model parameters for 'De Marke' were in agreement with values found in literature. Simulations indicated that in the long term decline of SOM must be expected in plots in crop rotation and in permanent grassland. Even if only grassland would be applied and manure would not be digested, according to the simulations, the OM input rate was too low to stabilize SOM in the long term. Our results indicate that strong manure digestion puts a pressure on future SOM mass percentage and that the contribution of a catch crop to long term SOM is marginal.

1 INTRODUCTION

There is general concern about decreasing soil organic matter (SOM) contents in agricultural soils. In soils low in SOM, the retention capacity for water and plant nutrients is relatively low, thus restricting their availability for crops and possibly resulting in drought stress and low crop yields (Bell and Van Keulen, 1995; Mayr and Jarvis, 1999; Vereecken et al., 1989). These soils are susceptible to leaching of nitrate to ground- and surface water (Boumans et al., 2001; Boumans et al., 2005; Wösten, and Van Der Zee, 1993). Moreover, decomposition of SOM contributes to atmospheric CO_2, whereas sequestration of C in soils is promoted to mitigate emissions of greenhouse gasses (Lal and Kimble, 1998; Lal, 2011). For these reasons, management strategies are explored for agricultural production systems to sequester C (Paustian et al., 2000; Freibauer et al., 2006; Hopkins and Del Prado, 2007) or, at least, maintain SOM levels.

Since 1992, in research project 'De Marke' we have operated an experimental farm on sandy soil that was designed to produce 12,000 kg milk per ha annually within the boundary conditions of Dutch and European environmental standards with respect to nitrate leaching, ammonia emission, phosphorus leaching and phosphorus accumulation in soil, both in the short and long term (Table 1, Aarts et al., 1992). The project uses an integrated, whole-farm approach, based on the many feedbacks within the system, i.e. the interactions between the sub-systems herd, manure, soil, crop and feeds. The study has a single replicate character, but this potential disadvantage is overcome by following a systematic procedure of modelling, system design, implementation in practice, evaluation and adjustment (Aarts et al., 1992) and intensive monitoring. The combination of farm structure and farm management results in low losses of nutrients from the farming system and in efficient use of N and P imported into the farm (Aarts et al., 2000a; Aarts et al., 2000b). In the design and in the practical implementation it was recognized that soil quality must be maintained or, if possible, improved to support efficient crop production and meet the preset targets. Thus anticipated SOM dynamics were an important criterion in farm design and were one of the reasons to incorporate grass in the crop rotation rather than relying on continuous maize and arable crops. In earlier studies ample attention was paid to N and P flows on the farm (Aarts et al., 2000a; Aarts et al., 2000b; Verloop et al., 2006; Verloop et al., 2010); the present study focuses on SOM dynamics.

The objective of this study is to evaluate the impacts of farm structure and management on SOM dynamics. In particular the following questions are addressed:

1 What have been the dynamics of SOM in the topsoil (layer 0-0.2 m of the soil), the lower topsoil (layer 0.2-0.4 m) and the subsoil (0.4-0.6 m below soil surface), in the period from 1989 to 2010?
2 What were the effects of soil and crop management and of soil characteristics on these dynamics?
3 What SOM levels are to be expected in the long term, if current farm management continues?
4 What changes in management would be necessary to maintain SOM levels or to enhance C sequestration in soil? Do they conflict with environmental and other targets?

This study was based on trend analyses of SOM mass percentage in the research period (1989-2010). A mono-component model proposed by Yang and Janssen (2000) was parameterized for 'De Marke' and was applied to estimate long-term developments of SOM concentrations corresponding to management alternatives varying in OM inputs to the farmland.

Table 1: *Standard environmental conditions applied in the design of the dairy farming system of 'De Marke' (Aarts et al., 1992).*

Objective	Maximum value
Nitrate concentration in groundwater	50 mg l^{-1} in upper meter of groundwater
Ammonia volatilization	30 kg N ha^{-1} yr^{-1} from manure
N surplus farm	128 kg N ha^{-1} yr^{-1} as farm inputs[1)] minus outputs assuming no accumulation in the soil
N surplus soil	79 kg N ha^{-1} yr^{-1} as soil inputs[2)] minus outputs with harvest
P concentration in groundwater	0.15 mg P l^{-1} in upper meter of groundwater
P surplus soil	0.45 kg P ha^{-1} yr^{-1}

[1)] Including biological N fixation by clover and deposition.
[2)] Including chemical fertilizer, organic fertilizer, biological N fixation by clover and deposition.

2 MATERIALS AND METHODS

2.1 Experimental farm 'De Marke'

2.1.1 Site characteristics

'De Marke' is situated in the eastern part of the Netherlands on very light sandy soil. Before establishment of 'De Marke' soil characteristics of the farmland were determined by intensive sampling (3 points per ha) up to a depth of 2 to 3.5 m below the soil surface (Dekkers, 1992). The soil was characterized by a 0.25-0.50 m anthropogenic layer, with an average organic matter mass percentage of 4.8%, and a bulk density of 1300 kg m^{-3}, overlying a layer of yellow sand which is very low in organic matter and hardly penetrable by roots (Aarts et al., 2000c). The area comprising the farm was reclaimed from heather at the beginning of the 19th century (Kleinsman et al., 1973). From 1950 until 1989, parts of the farmland received high loads of cattle slurry, equivalent to estimated manure OM rates up to 6.1 Mg ha^{-1} yr^{-1} (Verloop and Van Keulen, 2007), while other fields received lower, agriculturally 'normal' doses of manure that were more or less corresponding to crop requirements. At the establishment of the experimental farm in 1989, the upper 0.25 m of all farmland was homogenized by ploughing.

Soil moisture usually is at field capacity in spring (0.18 g per g oven dry soil) and decreases to 0.12 in mid–summer, or even to 0.10-0.05 in dry summers, corresponding to pF values of 1.8 to higher than 4.2. Groundwater depth is 1 to 3 meter below soil surface. In the experimental period (1989-2010), precipitation (mean annual 792 mm) was on average evenly distributed over the year, and average daily temperature fluctuated from 2.3 °C in December to 17.3 °C in July. Silt mass percentage is on average 12, ranging from 8 to 20. Base saturation is low and pH tends to drop below 5, but is kept within the range of 5 to 6 by liming.

2.1.2 Farm layout

The farm area (55 ha) is divided in permanent grassland (11 ha) and two crop rotations: ROTI and ROTII (Table 2). Implementation of the farm lay out started immediately after establishment of the farm (1989) and was finalized in 1993 after which monitoring started off. The rotation scheme of the arable phase was modified slightly in the course of time (Verloop et al., 2006). From 1993 until 1996, fodder beet was the first arable crop after grassland. From 1996 till 1999, maize was the only arable crop. Thereafter triticale (2000 to 2005) and spring barley (2005 to 2010) were introduced as the last crop in the arable phase. On maize land, Italian ryegrass was sown as a catch crop between the rows in June to create a soil cover directly after the harvest of maize in September. Catch crops were ploughed-in in the first week of March. Permanent grassland was resown approximately once in six years.

To prepare the new seed bed, the grass sod was destroyed and ploughed to a depth of 0.25 m. A total of 31 ha of grassland were available for grazing by 80 milking cows and, on average, 55 young stocks. In the arable phase of ROTI and ROTII the land was ploughed in spring to a depth of 0.25 m.

Table 2: *Farm plan of 'De Marke' (ha), crop sequence[1] and the ratio of arable crops/grass in the two types of rotation.*

Land use	Average area, ha
Permanent grassland	11
Rotation I	23
Rotation II	21
Crop sequence rotation I	3 years grassland -> 3 years arable crops
1993-1996	G,G,G -> Fb,M,M
1996-2000	G,G,G -> M,M,M
2000-2003	G,G,G -> M,M,Tr
2003-2010	G,G,G -> M,M,B
Crop sequence rotation II	3 years grassland -> 5 years arable crops
1993-1996	G,G,G -> Fb,M,M,M,M
1996-2000	G,G,G -> M,M,M,M,M
2000-2003	G,G,G -> M,M,M,M,Tr
2003-2010	G,G,G -> M,M,M,M,B

[1] G = perennial ryegrass, Fb = fodder beet, M = maize, Tr = triticale, B = spring barley.

2.1.3 Fertilizer and manure management

Fertilizer management aimed at complying with crop requirements, defined as the crop uptake that can be realized on dry sandy soils and application rates that do not violate threshold values for acceptable losses of N and P (Table 1, Aarts et al., 1992).

Manure and urine excreted indoor were collected as mixed slurry and stored in a covered silo until application to grassland and arable land. Manure management was adjusted in 2004 and in 2009 (Table 3). From 2004 onwards, slurry was anaerobically digested before application. In 2009 and 2010, 30% of the digested slurry was separated into a liquid and solid fraction. On grassland, manure application started in mid-March. On maize land, manure was applied in May. Application methods are described in Table 3. A rotational grazing system was applied, in which the herd grazed for 5-7 days on individual grassland plots of 1-3 ha, after which it was moved to a next plot.

The rates of N and P application through manure products (slurry, liquid and solid fraction) were tuned to the target N and P fertilization schemes. These schemes and the N and P mass fractions in the organic fertilizers, determined the volumes applied (m^3 ha^{-1}), which in combination with the OM mass fractions in the products determined the OM inputs through manure (Figure 1). The target N fertilization level for grass was 250 kg ha^{-1} and 150 kg ha^{-1} for maize. Fertilizer-P was applied at a rate corresponding with P-equilibrium fertilization (Aarts er al., 2000a, Verloop et al., 2010).

Table 3: *Manure management at 'De Marke' from 1993-2010.*

			Application method		
Period	**Manure products applied**	**Amount[1) %]**	**Grassland**	**Maize land**	**Other arable crops**
1993-2003	Slurry	100	Injection	Injection after ploughing	Injection after ploughing
2004-2010	Digestate	100	Injection	Injection in row	Injection after ploughing
2009-2010	Digestate	70	Injection	Injection in row	Injection after ploughing
	Liquid fraction digestate	26	Injection	Injection in row	Injection after ploughing
	Solid fraction digestate	4	Broadcast	Broadcast followed by immediate incorporation	-

[1)] Percentage of the total amount of manure produced at the farm.

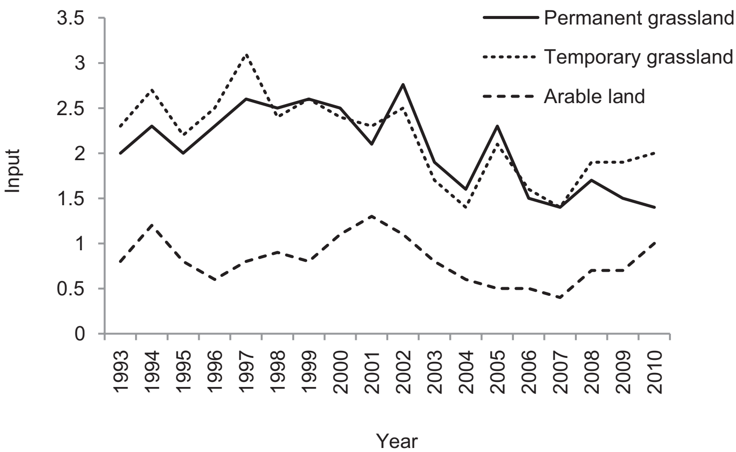

Figure 1: *Manure-OM application (10^3 kg ha^{-1}) to permanent grassland, temporary grassland and arable land from 1993-2010 (including grazing excreta).*

2.2 Data collection

The total area of 'De Marke' was sub-divided in 30 parcels of 1-3 ha each and each parcel was sub-divided in two or three plots of 0.5-1 ha, resulting in a total of 59 plots. For each of the 30 parcels relevant management data, i.e. timing and method of manure application, harvesting/cutting, grazing, re-sowing, ploughing, nutrient flows and yields were monitored (Aarts et al., 1992). Organic mass flows of harvested crop material and applied slurry were measured by weighing and the related nutrient flows were quantified on the basis of chemical analyses of representative samples (the procedure is described in detail in Verloop et al., 2006). OM from crop residues comprising roots, stubbles, harvest losses and grazing losses, was estimated as a fraction of dry matter yields (harvest) or as fixed values (Table 4) according to Ten Berge et al., 2000, Whitehead et al. (1986) and supported by data of Van Dijk et al., (1996). Residues of catch crops were assumed to be equal to the total biomass because they are not harvested but ploughed in.

Samples of the 0-0.2 m soil layer were taken in each of the 59 plots, at the start of 'De Marke' in 1989 and annually from 1994 to 2010 (Table 5). The 0.2-0.4 m and 0.4-0.6 m layers were sampled in only a selection of plots (details are specified in Table 5). SOM mass percentage was determined in all samples as loss on ignition (NEN 5754, 2005). In 1989, pH, soil mass percentages of SOM, soil texture and depth of the ground water table were assessed for all plots (Dekkers, 1992, see also Subsection 2.1.1). From 1993 until 2010, depth and pH of ground water were measured annually as described in Verloop et al. (2010).

Table 4: *Estimates of organic matter input to soil with crop residues (kg ha^{-1}).*

Crops	Estimate of crop residue[1)]	Remarks
Grass ungrazed	1.3 × harvest	Roots, stubbles and harvest loss
Grass grazed	1.4 × harvest	Roots, stubbles and grazing loss
Barley, Triticale	0.25 × harvest	Roots and stubbles. Straw is removed from the fields
Maize	2050	Roots and stubbles
Fodder beet	0.25 × harvest	Roots and stubbles. Leaves are removed off from the fields
Catch crops	Total biomass production	Total plant (Italian rye grass) ploughed in, not harvested

[1)] Units of harvested and total biomass production: kg dry matter ha^{-1}.

Table 5: *Collection of soil data.*

Data	Soil layer (m)	Method of sampling	Years
Texture, SOM, structure of soil profile	0-3.5	Soil survey based on 306 sampling points	1989
SOM, pH, N, P, K	0-0.2	Composite sample per plot, 40 soil cores	1993-2010
	0.2-0.4	Composite sample per plot, 40 soil cores. Each year 6-36 (average 16) plots were selected at random.	1991-2004
	0.4-0.6	Composite sample per plot, 40 soil cores. Per year 1-6 (average 3) plots were selected at random.	1991, 1998, 2000, 2001, 2002, 2004
Depth of ground water table	0.8 m below ground water table	3 boreholes per plot, annually, in spring.	1993-2010

2.3 Analysis

2.3.1 SOM development

At the start of 'De Marke', SOM mass percentage varied strongly, depending on land use and management in the preceding period of commercial exploitation. This allowed analysis of development of SOM mass percentage on plots with different initial organic matter status, which was done for all but eight plots. These eight were excluded, since they were not part of the 'De Marke' system during the entire research period (1989-2010). SOM dynamics were explored for each plot by regression analysis. We expected the decline of SOM to level off in the course of time. To test this hypothesis we fitted the data to linear and power functions. Next, the relationship between estimates of the initial status and estimates of the rate of decline was analyzed by linear regression. This procedure was also followed for the 0.2-0.4 m (26 plots) and 0.4-0.6 m (6 plots) layers, but for the period until 2004 only, due to lack of available data from later years (Table 5).

2.3.2 Organic matter decomposition

SOM dynamics is the result of annual OM inputs, and of OM outputs in the form of CO_2 produced by microbial OM decomposition. To better understand SOM dynamics we considered SOM to consist of: (*i*) a portion remaining from SOM present at the start of the experimental period, denoted by OSOM, (SOM), (*ii*) a portion originating from repetitive OM inputs by manure products (slurry, its separation products and grazing excreta), denoted by MANSOM, (*iii*) portions originating from inputs by residues of the main crops (MCSOM) and of catch crops

(CCSOM). SOM mass was calculated for each year as the sum of OSOM, MANSOM, MCSOM and CCSOM.

It is generally assumed that decomposition follows first order kinetics. The quantity of a substrate remaining at time t can then be calculated by:

$Y_t = Y_0 \times \exp(-K \times t)$ Eq. 1

where:

K = average relative decomposition rate between time 0 and time t

Y_0, Y_t = the amount of substrate (SOM) at time 0 and t, respectively.

It was found that K is not a constant (Yang and Janssen, 2000, 2002) but changes over time as:

$K = R_9 \times (f \times t)^{1-S}$ Eq. 2

where:

t = time (years)

R_9 = average decomposition rate between t = 0 and t = 1 (dimension: t^{1-S}) at a temperature of 9°C

S = aging factor (dimensionless)

f = temperature correction factor (dimensionless); $f = 0$ for $T < -1$, $f = 0.1 \times (T + 1)$ for $-1 < T < 9$, $f = 2^{(T-9)/9}$ for $9 < T < 27$, where T (°C) is the average temperature during the period considered (Yang, 1996).

After substitution of Eq. 2 in Eq. 1 the quantity of a substrate remaining at time t can be calculated by:

$Y_t = Y_0 \times \exp[-R \times (f \times t)^{1-S}]$ Eq. 3

The product of (f × t), denoted by decomposition time (Dt), was calculated to account for effects of temperature on decomposition. The dynamics of daily temperatures were summarized in an 'average weather year' for the whole research period. The daily values of f × t were added to provide the total annual decomposition time for the average weather year, which was 1.09 per calendar year.

Eq. 3 describes the amount remaining after *one* application of a source of organic matter (manure, roots, etc). Hence, the total amount of organic matter remaining at time (n+1) after n annual applications of e.g. manure (MANSOM) is calculated as:

$MANSOM = Y_1 + Y_2 + Y_3 + \ldots + Y_{(n+1)}$ Eq. 4

The values of the model parameters R and S are characteristics of the organic material considered (Yang and Janssen, 2000). However, R and S might, for a given organic material, be different in soils with different texture, moisture regime and pH.

First, the model was run with standard values of R and S, reported by Yang and Janssen (1997), for soil organic matter, manure, roots of the main crops and catch crops (Table 6). For manure, R and S were adjusted to values found in the Netherlands, and for crop residues R and S were calculated as the mean of standard values for roots and straw (Yang and Janssen, 1997). Next, we improved the fit of predicted and observed SOM for each plot by adjusting R and S of all organic matter sources with one adjustment factor for R ($R_{adjusted}/R_{standard}$), and one for S ($S_{adjusted}/S_{standard}$). The adjustment factors were found for each plot through an iterative procedure. First, OSOM, MANSOM, MCSOM and CCSOM were calculated for each year of the research period, using Eq. 2 with the standard values of R and S (Table 6). The sum of OSOM, MANSOM, MCSOM and CCSOM was compared with the observed values of SOM and the differences of the estimates and observed values were added across the whole research period. The adjustment factors with the best fit were found with the Solver function of Excel, as the value of R and S that gave the lowest cumulative error.

Table 6: *Standard values for decomposition rates ($R_{standard}$) and aging factor ($S_{standard}$) for old organic matter (OSOM), manure, residues of main crops, and ploughed in catch crops used to estimate SOM dynamics. Values are reported in Yang and Janssen (1997).*

Substrate	$R_{standard}$	$S_{standard}$
OSOM	0.057	0.46
Manure	0.82 (0.6)[1)]	0.49 (0.4)
Main crop	0.79 (0.96)[2)]	0.67
Catch crop	1.39	0.64

[1)] Adjusted to Dutch circumstances.
[2)] Mean of standard values for roots and straw.

We assumed that OM inputs and OSOM were distributed homogeneously over the 0-0.25 m soil layer that was homogenized at the start of the research period and has been ploughed since, and hence that the SOM mass percentage in 0-0.25 m was the same as in the 0-0.2 m layer. To convert SOM mass percentage into kg per ha, the percentage must be multiplied by soil mass per ha. The soil mass per ha for the 0-0.25 m layer is equal to its volume, 2.5×10^3 m^3, times soil bulk density, ρ. Soil bulk density decreases with increasing SOM mass percentage because soil porosity increases and organic matter has a lower specific density than the mineral component of the soil. Therefore, soil bulk density was corrected for SOM mass percentage, using ρ (kg m^{-3}) = $1.728\text{-}0.271 \times ((0.45 \times \text{SOM})^{0.5})$ according to Whitmore et al., (1992). Effects of land use and of the soil characteristics ground water depth, pH and silt (Hassink.,1997) on decomposition rates were explored by regression analysis with the estimated adjustment factors of R and S. Lutum was absent in the upper top layer, thus it was not included in the analysis. The depth of ground water was used as a proxy of the time that the soil is wet enough for microbial activity in soil organic matter decomposition (Leirós et al., 1999; Rey et al., 2008).

2.3.3 Long-term simulations

The mono-component decomposition model, parameterized for 'De Marke', was used to estimate long-term developments of SOM in the upper 0.25 m of the soil. A decomposition time of 50 years was selected to explore effects of management alternatives (A, B, C, D) with corresponding OM inputs specified in Table 7 for permanent grassland and ROTII, the land uses included in the simulation. Management A (Table 7) corresponds to the farming system between 1993-2003. Manure was not treated by digestion. Management B corresponds to the farming system between 2004-2010. Farm slurry was digested by a conventional system using farm slurry as the only substrate without additives and imported biomass. In this system 35% of the OM in slurry was transformed into methane. Management C corresponds to the farming system adopted since 2011. Slurry is digested in a more controlled way to realize a higher turnover of organic matter into mineral components. We assume that in this system 70% of the organic matter is transformed into methane, slightly lower than the maximum turnover values reported by Zeeman (1994). Hence, only 30% of the OM originally present in untreated manure will be supplied to the farmland. Management D is similar to management A except that no catch crops are grown. The alternatives were applied to permanent grassland and ROTII.

Table 7: *Characteristics of management alternatives (A, B, C, D) and corresponding OM application rates (Mg ha^{-1}) for the two land use types that are included in simulations of long term dynamics[1)], i.e. permanent grassland (PG) and crop rotation (ROTII).*

Management alternative		**A**	**B**	**C**	**D**
Manure digestion		No	Conventional	Strong	No
Land use	OM source				
PG	Manure	2.9	2.5	0.7[3)]	2.9
	Main crop[2)]	10.8	10.8	10.8	10.8
ROTII	Manure	2.5	2.0	0.6[3)]	2.5
	Main crop[2)]	5.8	5.8	5.8	5.8
	Catch crop	2.1	2.1	2.1	-

[1)] Application rates were averaged across all years.

[2)] OM originating from main crops (perennial ryegrass in permanent grassland and arable crops (mainly maize) in ROTII).

[3)] Low values are caused by the low level of residual OM after strong digestion.

3 RESULTS

3.1 Initial SOM status

At farm scale, initial average SOM percentage in the upper topsoil was 4.8 (ranging from 3.0 to 6.4, but slightly skewed - Figure 2). High and low values occurred in all land uses, although the distribution in permanent grassland (3.1-6.2; average 4.6) was different from that in ROTI (3.0-6.0; average 4.9) and ROTII (3.6-6.4; average 4.6; Figure 2). In the lower topsoil initial SOM mass percentage was 3.9 (range 1.9 to 5.9). Initial SOM mass percentage of permanent grassland, ROTI and ROTII, was 4.3, 3.6 and 3.7, respectively, in the lower top layer and 2.5, 3.6 and 1.2 in the subsoil.

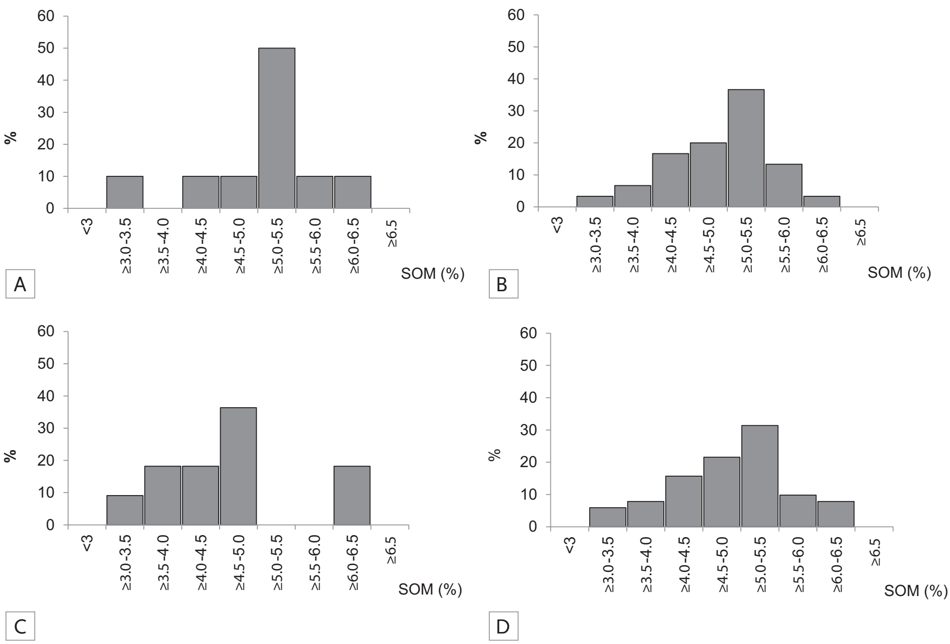

Figure 2: *Distribution of initial (1989) SOM mass percentage (layer 0-0.2 m); A: permanent grassland (10 plots), B: ROTI (30 plots), C: ROTII (11 plots), D: Entire farm (51 plots).*

3.2 OM rates to farmland

The yearly OM application rate (means for 1993-2010) was 13.3 Mg ha^{-1} in permanent grassland, 11.0 Mg ha^{-1} in ROTI and 11.1 Mg ha^{-1} in ROTII (Figure 3) with annual fluctuations in each type of land use. Catch crops contributed 18 and 15% to the OM application rate in ROTI and ROTII, respectively. At farm scale, the estimated OM application rate amounted to 11.5 Mg ha^{-1} with annual fluctuations from 9.4 to 13.5 Mg ha^{-1} (Figure 3). The effect of anaerobic manure digestion since 2004 (Table 3) on the OM inputs was most pronounced in permanent grassland.

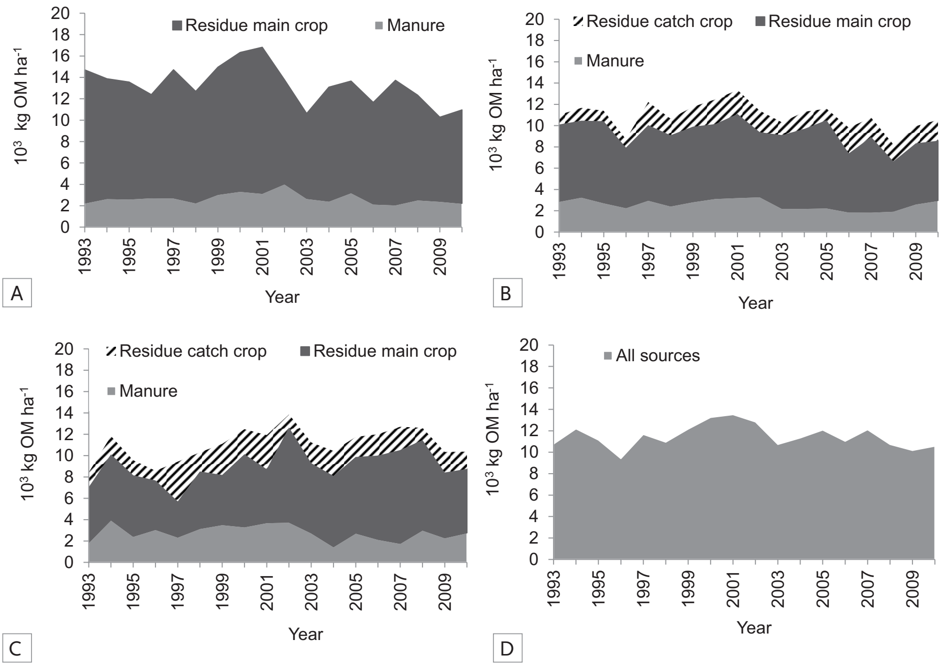

Figure 3: *OM inputs (10^3 kg ha^{-1}) to: (A) permanent grassland, (B) ROTI and (C) ROTII and (D) the entire farm (weighted averages across grassland and rotations).*

3.3 Development of SOM in the upper topsoil

SOM mass percentage at farm scale (aggregate of grassland and arable land) showed strong annual fluctuations but declined by 0.03 yr^{-1} (Figure 4A: $P < 0.05$) over the research period. In permanent grassland SOM mass percentage was constant over the whole research period (the regression coefficient being not significantly different from 0), however, year-to-year variability was high (Figure 4B). In ROTI and ROTII, SOM mass percentage decreased significantly over the research period ($P < 0.05$), again with high year-to-year variability (Figure 4B). In all categories of observations (farm scale, permanent grassland, ROTI and ROTII) SOM mass percentage was remarkably low in 2006 and 2007 and high in 2009.

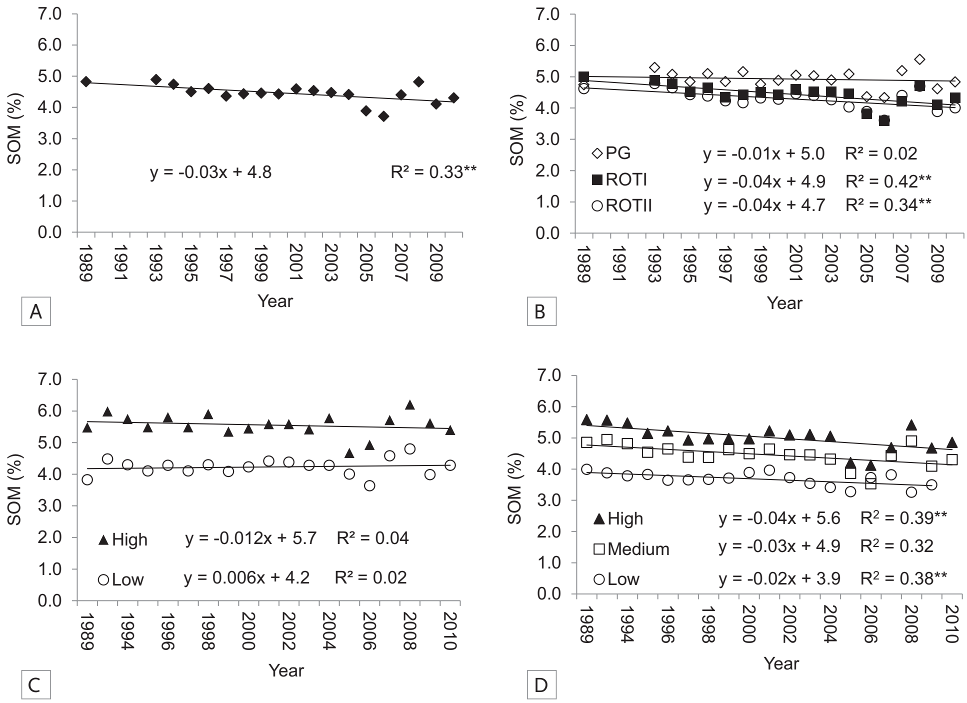

Figure 4: *Development of SOM mass percentage (0-0.2 m), from 1989-2010. in the layer 0-0,2 m; (A) Entire farm (mean of all plots); (B) permanent grassland, ROTI and ROTII; (C) permanent grassland clustered in high and low initial SOM%; (D) ROTI and ROTII clustered in high, medium and low initial SOM%; * significant at P<0.1; ** significant at P<0.05.*

In permanent grassland, SOM mass percentage responded differently in plots with high and low initial values: it tended to increase in plots with low initial values and to decrease in plots with high initial values, but the difference in regression coefficients between these categories was not significant (Figure 4C). In the crop rotations (ROTI and ROTII) similar patterns were observed, but, again, the difference between slopes was not significant (Figure 4D). In 21 out of 51 plots the reduction in SOM mass percentage slightly, but insignificantly, decreased over time. Power functions did not better describe the change of SOM over time than linear functions. In the remaining 30 plots, linear functions accounted for a similar percentage of the observed variation ($R^2 = 0.23$) as power functions ($R^2 = 0.22$).

Figure 5 presents the relationship between (Y-axis) the regression coefficient of linear regression of SOM percentage against time, determined per plot, and (X-axis) initial SOM percentage. All effects mentioned here, were significant at P<0.05. For each land use, the regression coefficient decreased with increasing initial SOM percentage. However, the effect of land use was also significant, as was also shown in Figure 4. Initial SOM mass percentage and arable crops/grass ratio combined accounted for 53% of the variation in SOM change.

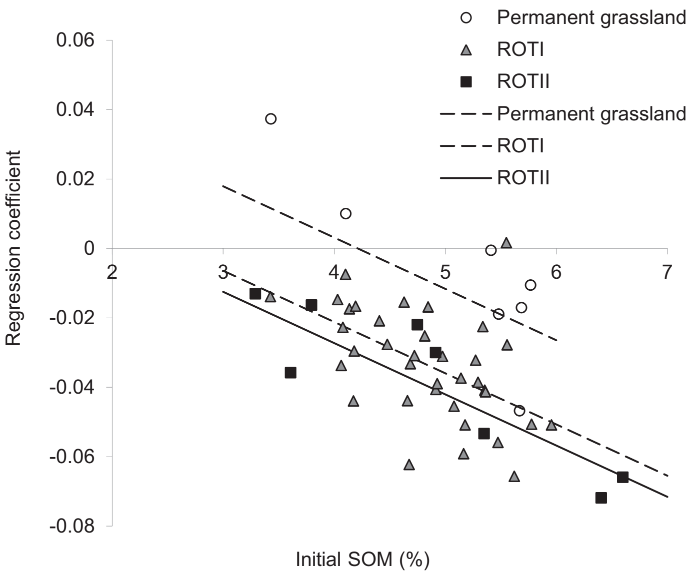

Figure 5: *Regression coefficients of the relations between SOM mass % (0-0.2 m) and time (1989-2010) plotted versus the initial SOM mass%, for permanent grassland, ROTI and ROTII for the 0-0.2 m layer as a function of the initial SOM%. Each point represents a plot.*

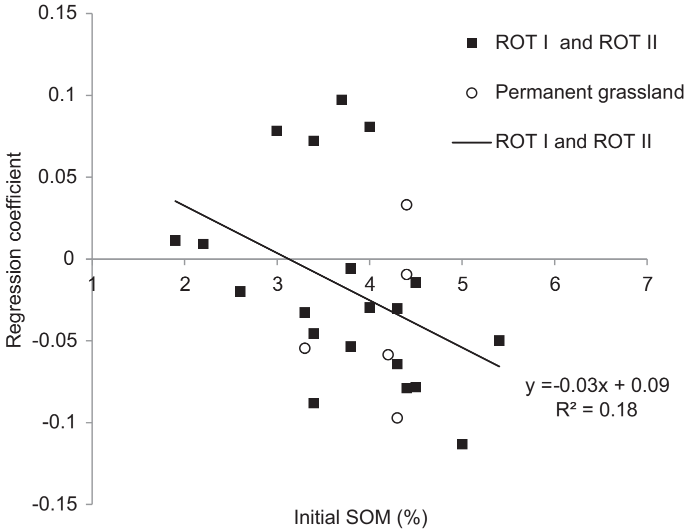

Figure 6: *Regression coefficients of the relations between SOM mass % (0.2-0.4 m) and time (1989-2004) plotted versus the initial SOM mass % (0.2-0.4 m), for permanent grassland, ROTI and ROTII (combined). Each point represents a plot.*

3.4 Development of SOM in deeper layers

Regression analysis indicated that in deeper layers, as in the upper topsoil, SOM mass percentage in ROTI and ROTII declined at a constant rate. At farm scale, SOM mass percentage in the layer 0.2-0.4 m declined with 0.03 per year (average of available plots on the farm, n = 26; Figure 6) and in the layer 0.4-0.6 m SOM mass percentage declined with 0.07 per year (n = 6; data not shown), Note that the results are based on the period until 2004 only, as data for later years are lacking. In the 0.2-0.4 m layer in ROTI and ROTII, the effect of the initial SOM mass percentage on the rate of SOM decline was similar to the pattern observed in the upper topsoil (Figure 5) though the relationship was weaker: the initial value accounted for only 18% of the observed variation in rate of decline. In permanent grassland no significant correlation was observed between decline rate and initial SOM mass percentage.

3.5 Organic matter decomposition rates

For each plot SOM development during 17 years was simulated initially with standard values of R and S (Table 6) and compared with observed values. The correlation between model results and observed values was poor as indicated by R^2 for permanent grassland (0.12), ROTI (0.008) and ROTII (0.003). After adjusting the parameters R and S, individually on each plot, simulated SOM development was more in agreement with observations as indicated by higher (R^2) between simulated and observed values for permanent grassland (0.76), ROTI (0.99) and ROTII (0.81) and only 3 outliers on a total of 51 plots. On average the adjustment factors for both R and S were close to 1 and higher for ROTI than for permanent grassland and ROTII (Figure 7, Table 8), but the differences between the land use types were not significant. Consequently, the averages of adjusted R and S did not differ from the standard values.

Regression analysis showed no significant relations between the adjustment factors of R and S and the soil characteristics, pH, silt content and depth of the groundwater table. Moreover, a fast decline of SOM in the upper top layer, resulting in a high adjustment factor for R, could be caused by transport of SOM to deeper layers than the homogenized soil layer of 0-0.25 m (the ploughing depth). This is possible when ploughing depth exceeds the planned depth. To examine this possibility, we explored whether high values of the adjustment factors for R were associated with abnormal dynamics in SOM percentage in the lower top layer. We found no indications of ploughing in SOM in plots with high values of the adjustment factors (data not shown).

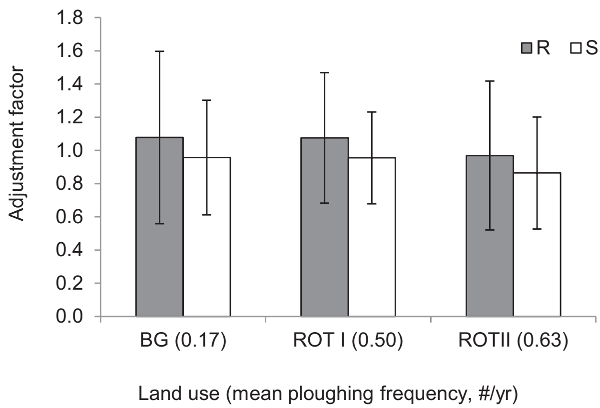

Figure 7: *Average adjustment factors of R and S found by fitting of simulated and observed SOM dynamics for three land uses with different frequency of ploughing; error bars show standard deviations.*

Table 8: *Estimates of decomposition rates (R) and aging factor (S) of OSOM, manure, roots of main crops, and ploughed in catch crops; means and standard deviations (standard deviations between brackets) for permanent grassland, ROTI and ROTII.*

	Permanent grassland		ROTI		ROTII	
	R	**S**	**R**	**S**	**R**	**S**
OSOM	0.05 (0.02)	0.44 (0.16)	0.06 (0.02)	0.44 (0.15)	0.06 (0.03)	0.40 (0.23)
Manure	0.58 (0.21)	0.47 (0.17)	0.65 (0.24)	0.47 (0.16)	0.58 (0.27)	0.42 (0.25)
Main crop	1.06 (0.39)	0.63 (0.22)	1.18 (0.43)	0.63 (0.22)	1.07 (0.49)	0.57 (0.33)
Catch crop	-	-	1.49 (0.55)	0.61 (0.21)	1.35 (0.62)	0.55 (0.32)

3.6 Long-term developments

Figure 8 presents the dynamics of SOM and OSOM, simulated for permanent grassland and ROTII using the model parameters, R and S, specified in Table 8 and with an initial SOM of 4%. According to the simulations, SOM percentage at 'De Marke' declined gradually in the crop rotation and on permanent grassland (Figure 8, left hand side). Dynamics of OSOM was practically equal in permanent grassland and ROTII and was, therefore, presented by the same curve. SOM percentage in permanent grassland was slightly increased until year 17 after which it declined, it became lower than the initial value after year 26. SOM percentage in ROTII declined immediately. The difference in SOM dynamics between permanent grassland and ROTII were caused by MANSOM, MCSOM and CCSOM as dynamics of OSOM was equal for the both land uses. Figure 8 (right hand side) shows the simulated impact of management alternatives on dynamics of SOM for permanent grassland and the crop rotation. The effects of manure digestion were rather small for permanent grassland and ROTII (compare management alternatives A and B); however effects of 'strong manure digestion' were significant (compare A and C). The effect of catch crops was negligible (compare options A and D).

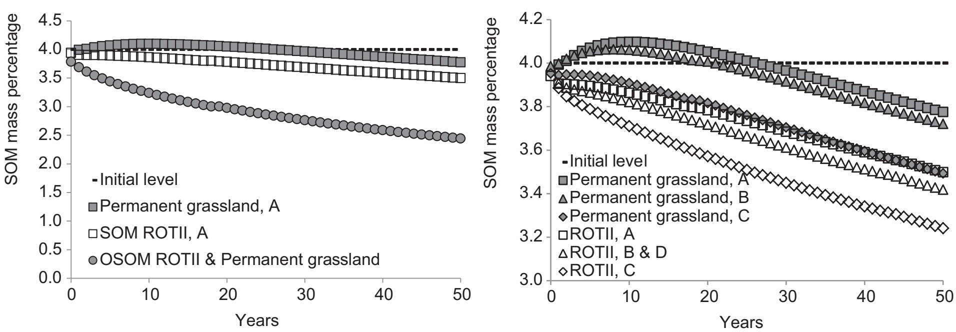

Figure 8: *Simulated long-term dynamics in the upper topsoil in relation to land use and management alternatives; left-hand side: OSOM and SOM in permanent grassland and in ROTII under management alternative A; right-hand side: SOM in permanent grassland and ROT II under management alternatives A, B, C, and (for ROTII) D. Note that the scales of the Y-axis are different for left and right.*

4 DISCUSSION

4.1 SOM dynamics from 1989-2010

The decline in SOM content on 'De Marke' as a whole was the result of the decline in ROTI and ROTII. Clearly, arable cropping results in a decrease in SOM content, as has been observed before (Johnston, 1986; Tyson et al., 1990; Gregorich et al., 2001; Vertès and Mary, 2007). The positive correlation between the rate of decline and the ratio of arable years to grassland years in the rotation (Figure 5) is in agreement with this observation. Hence, incorporation of grassland in the crop rotation contributes to the maintenance of SOM (Reeves, 1997). However, apparently, at 'De Marke', systematic application of this system was not sufficient to stabilize the SOM level. The fact that in permanent grassland at 'De Marke' SOM is stable, although it tends to decline in plots with high initial levels, indicates that equilibrium between inputs and decomposition can be reached. Equilibrium was also reported by Hoogerkamp (1973) for old grassland. A constant SOM level or decline (dependent on the initial level) has also been reported in other studies (Lettens et al., 2005; Freisinger et al., 2007; Reijneveld et al., 2009).

Observed year-to-year variability in SOM is very large (Figure 4). Some variations must be considered unrealistic. For instance, the decrease in Fig 4A from 4.4 in 2005 to 3.9 in 2006 corresponds to a loss of about 15 Mg SOM per ha. Under stable (weather) conditions and management, such losses are very unlikely. Also the increase from a SOM mass percentage of 4.4 in 2008 to 4.8 in 2009 is hard to explain, if only because this increase, corresponding to more than 11 Mg SOM per ha, exceeds the sum of total OM inputs in 2008 (Figure 3). Hence, part of the year-to-year variation is most probably due to uncertainties related to sampling and chemical analysis of the samples.

In most plots, the decrease in total SOM mass percentage did not slow down over time. SOM decomposition rate usually slows down over time (Yang, 1996), because the components that are most susceptible to decomposition are decomposed first, resulting in an increase in the share of more persistent components that decompose more slowly. On 'De Marke', this pattern is masked by the fact that new OM is added continuously. Hence, at any moment during the whole research period, the measured SOM mass percentage does not present the remainder of SOM present at the start of the study, but a mixture of fresh material added recently and the remainders of initial SOM.

In the lower topsoil (0.2-0.4 m), just as in the upper topsoil (0-0.2 m), SOM mass percentage declined with time. SOM development in the lower topsoil was correlated with that in the upper topsoil (not shown). This could be expected as part of this lower topsoil (0.2-0.25 m) belongs to the homogeneous plough layer (0-0.25 m). A decrease in SOM in the plough layer will also be found in the 0.2-0.25 m layer and affects the SOM mass percentage in the entire lower top soil (0.2-0.4 m). The results for the subsoil (layer 0.4-0.6 m) indicate that the rate of decomposition of SOM, probably mainly consisting of material that was incorporated in the soil at the time the land was reclaimed from heather, exceeds the inputs. This is a concern on dry sandy soils, as it may affect rooting in subsoil. Incorporation of deep rooting crops such as alfalfa (Bell et al., 2012) could be considered to increase OM inputs to subsoil.

4.2 Decomposition rates

The wide ranges of adjusted R and S of individual plots (Table 8) suggest that plot characteristics strongly affected the decomposition process, although a part of the variability of the estimates may have been caused by variations in sampling and analysis (Section 4.1). Moreover, the calculation of decomposition rates relies in part on assumptions on the OM inputs with crop residues, and it must be kept in mind that coefficients used (Table 4) will not be representative for each plot in each year. The failure to find correlations between soil characteristics and adjusted R and S urges to be careful to draw strong conclusions on the results of plots with high and low decomposition rates and to focus on the means.

The R and S adjustment factors, and consequently decomposition rates, were not significantly different for (*i*): plots with different initial SOM mass % and (*ii*): crop rotations and permanent grassland (Figure 7). This suggests that the effect of initial SOM mass % on the magnitude of SOM decline (Figure 5) was caused by substrate quantity and that tillage did not accelerate decomposition at ‘De Marke’, which is in contrast to what is found in other studies (Strebel et al., 1988; Whitehead et al., 1990; Studdert and Echeverría, 2000).

4.3 Simulations

The simulated course of SOM for permanent grassland (Figure 8) during the first 15 years is in agreement with the observed stable SOM mass percentage in permanent grassland (Figure 4) and with literature data (e.g. Reijneveld et al., 2009). However, from Figure 5 it might be concluded that in permanent grassland SOM mass % will converge to ca. 4% for which the regression coefficient is 0, whereas the simulations indicate a decline to somewhat lower levels (Figure 8). This difference can be explained by the fact that the extrapolation according to Figure 5 does not account for the increase in decomposition rate that is to be expected because of substrate quality. The model takes into account that part of the original, slowly decomposing soil organic matter (OSOM) is gradually replaced by substrates that decompose faster (MANSOM, MCSOM and CCSOM) resulting in an increase of decomposition rate of total SOM. The small impact of catch crops in rotations compared to fallow in the winter period (Alternative D - Figure 8) is due to the combined effect of fast decomposition (cf. the values of Radj and Sadj for CCSOM in Table 8) and the low application rates of OM through catch crops. The relatively low impact of conventional anaerobic manure digestion (Alternative B - Figure 8) compared to untreated manure (Alternative A) is caused by the fact that conventional digestion does not strongly decrease OM inputs to soil. The relatively high impact of strong manure digestion (Alternative C-Figure 8) is associated with the much lower OM inputs to soil (Table 7). The results for conventional digestion are in agreement with conclusions of Thomsen et al. (2013).

4.4 Implications for system development

The simulations presented in Figure 8 suggest that in future a gradual decrease of SOM is to be expected, which will proceed faster in rotations than in permanent grassland, and faster when manure is applied after strong rather than to conventional digestion, or when manure is not digested. Hence, according to our results, SOM dynamics are affected by OM input rates, and OM inputs rates in their turn are determined by land use and manure management (Table 7). We found no indications that, in our system, the ploughing frequency affects SOM dynamics. For SOM maintenance it is necessary to increase OM inputs and to take into account the drawbacks of strong anaerobic digestion.

One of the options to increase SOM is to convert crop rotations to permanent grassland (eg. Freibauer et al., 2004; Vellinga et al., 2004). However, this change would have serious drawbacks for the N use efficiency of the farming system. For efficient use of resources, feeds for the cattle should as much as possible be produced on the farm (Aarts et al., 1992). Because total crop energy production is less in a farming system with only grassland than in a system with integrated maize and grass production, an increase of feed import would be imperative to supply sufficient energy to the cattle. Besides, in a grassland based system there would be a relative excess of protein in the roughage that could impair efficient use of N by the herd (Bannink et al., 1999). Moreover, N utilisation efficiency of maize is higher than of grassland (Aarts et al., 2002) and a larger share of grassland would strongly increase N surpluses in the soil (Verloop et al., 2006). Hence, without a significant share of maize in land use it seems not feasible to produce milk at an intensity of 12,000 kg ha-^{1} within the environmental boundary conditions mentioned in Table 1. A less drastic option is to dedicate the complete farmland to crop rotation and thus, abolish permanent grassland. This would lead to a more even distribution of OM over the farmland than in the present crop plan with high OM input rates on permanent grassland and low OM input rates on rotations. In the rotations the ratio of arable crop to grassland would decrease and the OM input rates would increase. However, in the long term SOM even declines under grassland when manure is not digested. Possibly, to some extent a further decline of SOM must be accepted as an autonomous development. The potential to sustain efficient crop production at lower SOM levels should be evaluated by modelling as was done by Aarts et al. (2000c).

5 CONCLUSIONS

This study has shown for a dairy farming system on light sandy soil that:

1. Soil organic matter percentage of the 0-0.2 m layer declined in crop rotation in which grass and arable crops are alternated. On permanent grassland soil organic matter percentage could be maintained at 4% during the experimental period of 20 years.
2. Differences between the magnitude of SOM decline rate in grassland and arable crops were mainly caused by differences in the quantity of applied organic material. Relative decomposition rates of the same organic matter source were not significantly different between permanent grassland and rotations with temporary grassland and arable crops, suggesting no or small effects of tillage.
3. Organic matter input to subsoil is insufficient to compensate for decomposition of subsoil SOM.
4. Simulations show that strong manure digestion negatively affects future SOM% and suggest a trade-off between use of manure for bio energy and SOM%.
5. Simulations show that the effects of catch crops on long-term SOM dynamics are marginal.

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CHAPTER

GENERAL DISCUSSION

1 INTRODUCTION

This thesis investigates the limits of effective nutrient management in dairy farming on dry sandy soil, with emphasis on the following key questions:

1. How can nitrate leaching be restricted to less than 50 mg l^{-1} in groundwater?
2. To what extent can nutrient use efficiency at soil level and whole farm level, be enhanced by management with the techniques that are available at present?
3. What are the effects of efficient N and P management on long term soil fertility?

To answer these questions this research focussed on five specific issues:

1. What are the causes of high nitrate leaching on the farm?
2. Is N mineralization well synchronized with crop uptake capacity and crop requirements in the crop plan of 'De Marke', and if not, can mismatches between N mineralization and crop-N uptake be diminished?
3. Is the utilisation of manure-N by crops enhanced as a result of manure management?
4. What are the effects of P-equilibrium fertilization on long term soil-P-status and crop yields?
5. What are the effects of efficient nutrient management on SOM dynamics in the short and in the long term?

This research was conducted on experimental dairy farm 'De Marke' and covers the system development that was carried out at 'De Marke' since 2000. A historical overview of the research performed at 'De Marke' shows the system development (Table 1). From 1993-1999 research at 'De Marke' focussed on exploring the possibilities to realize an intensive production on light sandy soil without violating environmental standards (Aarts, 2000). The first results were promising as the environmental performance of the farm, established in 1992, was much better than that of average commercial farms of the mid-1980s (Aarts, 2000). An evaluation, carried out in 2000, indicated that many environmental standards were met. However, the nitrate leaching to groundwater and N surpluses on soil and farm scale exceeded permitted levels. Therefore, adjustments were proposed to further reduce nitrate leaching and N surpluses, primarily by maximising the N use efficiency. Since then research was more focused on the limits of the effectiveness of efficient nutrient management in the production system. The farming system of the experimental farm was further developed to explore maximum attainable efficiency. However, it was recognized that efficient nutrient management should not be a short term success and impair the fitness of the system to sustain efficient dairy production in the future (EC, 2012). Therefore attention was paid to the impact of efficient nutrient management on long term soil fertility.

Table 1: *Historical overview of the experimental farming systems at 'De Marke'[1)].*

Period	System	Main characteristics
1993-1999	A[2)]	Rotational grazing Crop rotation with mainly maize and grass Application of undersow Italian rye in maize as a catch crop Optimized timing and distribution of manure Fertilization based on raw manure and mineral fertilizer-N No mineral fertilizer P
2000-2003	B[3)]	Lower grazing intensity Lower inputs of mineral fertilizer-N Adoption of triticale as last arable crop
2004-2008	C[3)]	Anaerobic slurry digestion Adoption of spring barley as last arable crop Slurry injection after ploughing in rows of maize No inputs of mineral fertilizer-N
2009-2010	D[3)]	Separation of digestate into a solid and a liquid fraction.

[1)] This overview only refers to management of manure, soil and crops and does not comprise the complete farm management.
[2)] Original design implemented in practice with main characteristics.
[3)] Modifications relative to the previous system.

The major findings of the research are addressed in Section 2. Next, the effects of system development on environmental performance (Section 3) are discussed. In Section 4 the implications for further research and for practical farmers are discussed, and in Section 5 experiences with prototyping as a research method are described. Finally, conclusions are presented in Section 6.

2 RESEARCH ON N, P AND SOM TO SUPPORT SYSTEM DEVELOPMENT

2.1 Reducing nitrate leaching

To explore options to restrict nitrate leaching by modifications of the farm management, the causes of high nitrate leaching were examined by relating nitrate concentrations measured in the upper meter of groundwater, to N balances, cropping patterns and grazing intensities.

There is much evidence that grazing induces nitrate leaching (e.g. Hack-Ten Broeke, 2001; Wachendorf, 2004; Nevens and Reheul, 2003; Oenema et al., 2010), particularly at high grazing intensities (Ledgard et al., 1999; Anger et al., 2002) where poaching and trampling damage the grass sod and urine spots are unevenly distributed over the land (Richards and Wolton, 1976). To avoid these problems, at 'De Marke' a rotational grazing system was applied to more evenly distribute urine spots and to enhance N utilisation of the grazed grassland (Vellinga and Hilhorst, 2001). However, in spite of this careful management, grazing at 'De Marke' was associated with higher nitrate leaching, i.e. 23-49 mg l^{-1} in non-grazed grass and 59-74 mg l^{-1} in grazed grass ($P<0.05$). This indicates that even in a grazing system with even distribution of limited amounts of N excreta (88 kg per ha on permanent grassland and 48 kg per ha on temporary grassland) longer confinement of cattle might be required to realize the maximum allowed nitrate concentration in groundwater.

Maize is often associated with substantially higher nitrate leaching than grass (Nevens and Reheul, 2005; Oenema et al., 2010). This was not so clear at 'De Marke'. The effects of crops and of crops in rotation were analysed separately in a 'CROPS-model' and a 'ROTATION-model'. The 'CROPS-model' indicated both short and longer term (up to 4 years) effects on nitrate concentration. Nitrate concentrations in groundwater were higher under maize (45-66 mg l^{-1}) than under ungrazed grass (23-49 mg l^{-1}) but lower than under grazed grass: (59-74 mg l^{-1}). This relatively good performance of maize with respect to nitrate leaching, can be attributed to (*i*) the application of catch crops and, (*ii*) tuning of fertilizer-N to the crop requirements after correction for N release from soil, in particular directly after ploughing the grass sod (Schröder, 1998). It should be emphasized that the means of observed nitrate concentration *did* indicate higher nitrate concentrations in 1st, 2nd and 3rd year maize than in 4th year maize indicating rotation effects. However, the mean nitrate concentration in 1st year maize, where the highest values might be expected due to effects of N release from the ploughed grass sod, were not higher than in 2nd and 3rd year maize. So, although rotation potentially increases nitrate leaching in maize, adjusting fertilization in 1st year maize seems to be an effective counter measure. To further evaluate rotation effects, also the results for grassland should be taken into account. The nitrate concentration in groundwater under grass in rotation was substantially lower than in permanent grassland. Hence, reduced nitrate leaching in temporary grassland might be considered as a side-effect of maize in rotation. The overall nitrate leaching of rotations (means over the complete rotation including grassland and mainly maize) and grazed permanent grassland were similar, i.e. 59 and 63 mg l^{-1}. Hence, in monitoring programs it might be considered for dairy farms to evaluate observed nitrate leaching on the level of crop rotations and not per crop. It was found that triticale was associated with high nitrate leaching. Triticale was implemented in 2000 to replace maize as the last arable crop in rotation. This change was to optimize the transition of the arable phase to the grassland phase. Maize was problematic as last arable crops because harvest of maize, in mid-September, was too late to realize a good establishment of the new grass sod, sown after maize harvest. The emergence and development of the grass was impaired by sub-optimal temperatures in autumn (Problem 1 in Figure 1). In harsh winters it tended to completely disappear after which it was necessary to re-sow in spring. Triticale was sown after the 2nd maize crop, to function during winter as a catch crop instead of Italian ryegrass. It was assumed to develop faster in winter than perennial ryegrass. After winter, it was harvested in mid-summer in the last arable year, after which grass was sown and could develop more strongly before the onset of winter. The high nitrate concentrations found under triticale were explained as an effect of a mediocre development of triticale as catch crop after maize (Problem 2 in Figure 1). This observation led to replacement of triticale by spring barley in combination with Italian ryegrass as catch crop in the preceding maize. The barley was harvested in summer after which grass was sown (Solution in Figure 1).

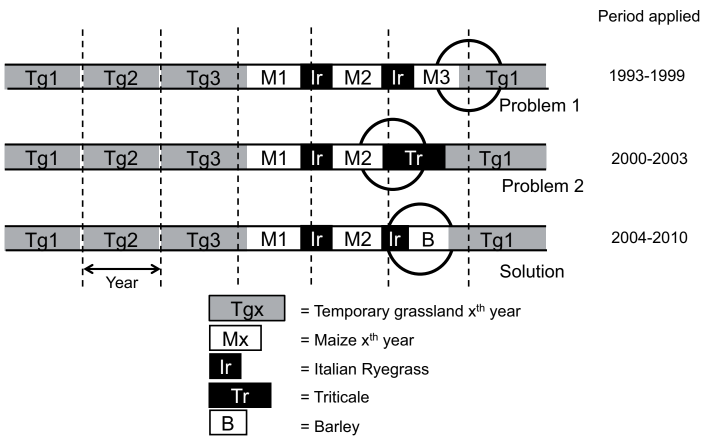


Figure 1: *Crops applied in the transition of the arable phase to the grassland phase in rotation at 'De Marke' in different periods.*

It can be concluded that the overall performance in terms of nitrate leaching of a crop rotation, including grass and maize is not very different from permanent grassland. This might partly be attributed to management of maize. Two management options were detected to reduce nitrate leaching: (*i*) reducing grazing intensity and (*ii*) adjusting the transition of the arable to grassland phase in rotation by adopting spring barley as last arable crop. Both options were implemented at 'De Marke'.

2.2 Dynamics of N mineralization

N mineralization was explored at 'De Marke' on six observation spots to gain insight in the temporal and spatial pattern of soil-N mineralization and to detect where and when total mineral N supply is excessive compared to N crop requirement, i.e., mineral N supply + N mineralization > N crop requirement. These insights were to improve the synchronization of N application and crop-N requirements. There were systematic differences between N mineralization in the different crops. Average annual soil-N mineralization was highest under permanent grassland. In temporary grassland soil-N mineralization increased in the order 1st year < 2nd year < 3rd year grassland and in arable crops after grassland mineralization decreased in the order 1st year > 2nd year > 3rd or more years. For all crops, the mean difference between total mineral N supply and N uptake (mineral N surplus) was 89 kg N per ha, slightly higher than the maximum permissible soil surplus of N (79 kg N per ha). The mineral N surplus in 1st year maize is highest and may enhance nitrate leaching, although this is not confirmed by the data of nitrate concentrations. For 1st year maize but also for other crop species, mineral N surplus and nitrate leaching were hardly correlated (R^2 = 0.03, Figure 2); note that the data of nitrate concentrations and mineral N surplus do not refer to exactly the same research periods; hence the data do not allow definite conclusions. The pattern of mineral N surplus suggests to redistribute fertilizer-N, in particular, to lower the N application rate in 1st year maize and increase the N application rate in 1st year temporary grassland. It should be emphasized that in the research period (1993-2005) N fertilization rates in 1st year maize (41 kg ha^{-1}) were already much lower than in 2nd and in particular >2nd year maize (91 and 128 kg ha^{-1}, respectively). N fertilization in 1st year maize was already stopped when the study was performed (2006). This study confirmed that it was a good decision to stop fertilizing 1st year maize, because even low rates of N fertilization in 1st year maize are redundant. Additionally, measures might be considered in the 3rd year temporary grassland to reduce N mineralization from the ploughed grass sod. N mineralization in 1st year maize might drop when 3rd year temporary grassland is not grazed (Clement and Back, 1969; Davies et al., 1997) or when N fertilization rates in 3rd year temporary grassland are reduced. Another option is to adopt crops with a higher N uptake capacity, such as fodder beet, in the first year of the arable phase (Nevens and Reheul, 2002).

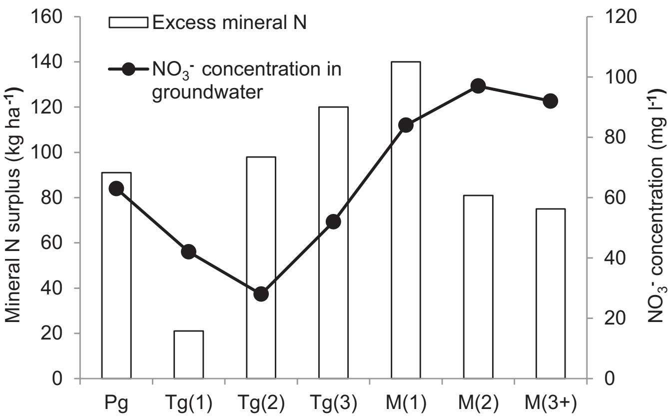


Figure 2: *Mineral N surplus, defined as the sum of N mineralization and mineral N supply minus crop-N requirements (1993-2005) and nitrate concentrations (1996-2003) for specific crops at 'De Marke'; Pg = permanent grassland; Tg(n) = nth year temporary grassland, M(n) = nth year maize.*

Mineralization in winter, outside the growing season, was 77 kg N per ha in maize. It can be assumed that the N released in winter will leach unless it is taken up by crops. This underpins the importance of a suitable catch crop to reduce potential N leaching in winter. The mean N uptake by Italian ryegrass applied as undersow at ‘De Marke’, calculated according to Schröder (1998), is 50 kg N per ha (mean of 1993-2010). This suggests that a higher N uptake is needed to further reduce N leaching. However, it is uncertain if this can be achieved with Italian ryegrass or alternatives (Schröder et al., 1992; Hoek et al., 2006; Hilhorst and Verloop, 2009).

2.3 Utilisation efficiency of manure-N by crops

Since 2000 several modifications of manure management were implemented (Table 1) to enhance soil-N use efficiency (soil-NUE). Manure management comprises more than manure treatment that is carried out to change the chemical characteristics of manure before application; it also refers to the grazing method and intensity (Ledgard et al., 1999; Vellinga and Hilhorst, 2001) or the adoption of new application techniques (Lalor et al., 2010; Aarts, 2000; Schröder, 2005). Many field trials focus on the fertilizer quality of the manure. However, given the significance of the whole chain of activities concerning manure, there is a need for insight in the fertilization value at the farm level.

Various indicators can be used to evaluate utilisation of manure-N by crops. The most straightforward indicator is the N fertilizer replacement value ($NFRV_{manure}$), which indicates the N uptake from manures as compared to a reference N fertilizer. A procedure that was developed to deduce the apparent $NFRV_{manure}$ from farm data was tested at ‘De Marke’. The estimated $NFRV_{manure}$ of the entire farm for the period of 1994-2010 (0.70) was credible, though rather high. The result indicates that manure-N is well utilized at ‘De Marke’, but that further enhancement of N utilisation is possible. However, the estimates of $NFRV_{manure}$ appeared to be very sensitive to uncertainties of input parameters. Therefore, it was not possible to evaluate effects of manure management on the basis of dynamics of $NFRV_{manure}$ over time. Instead, this analysis was conducted on the basis of NUE, i.e. the ratio of N yields with harvested crops and N inputs to the soil. However, NUE is not only the result of availability of manure-N for crops, but also of (*i*) variability of other sources, such as net N supply from soil (Schröder, 2005; Nevens and Reheul, 2005) and (*ii*) variability of crop-N recovery as affected by weather. These variables were taken into account in the analysis to prevent biased interpretations.

Interestingly, the NUE in grassland did not decline over time even in the period after 2004 when N mineral fertilizer was abolished (shown for permanent grassland in Figure 3). This is remarkable because, together with N fixed by legumes, N mineral fertilizer is considered the N source with the highest N fertilizer value. The maintenance of NUE in temporary grassland and increase in NUE in permanent grassland indicates that the cut down of mineral fertilizer has been compensated by an increase of N availability from other sources. An analysis of the dynamics of annual N recovery and net N mineralisation from soil, indicated that the absence of a decline in NUE in grassland was not caused by weather or net soil-N mineralization.

Simultaneously with stopping mineral fertilizer-N use since 2004, N yields and to a lesser extent dry matter and P yields in grassland declined. This seems conflicting with the maintenance of NUE and the suggested higher N utilisation from manure in later years, but the reduction of yields is a result of a reduced total N fertilization level. A simultaneous increase $NFRV_{manure}$ and decrease of the N fertilization level is possible as the N fertilization level is the sum of N rate times NFRV for each source. In grassland the N fertilization level was reduced by a, in hindsight, too ambitious cut down of mineral fertilizer-N.

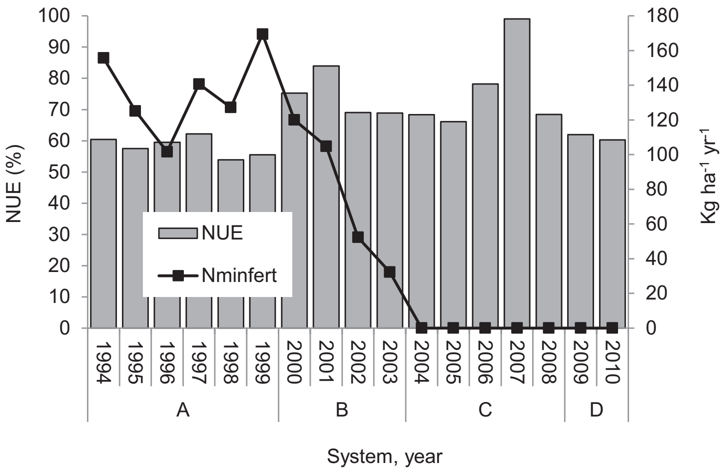


Figure 3: *Application rates of mineral fertilizer-N (Nminfert) and dynamics of NUE in permanent grassland, from 1994-2010.*

2.4 Impact of P equilibrium fertilization on soil-P status and crop yields

Plants need P to grow and not all soil-P is available for plants. For many years agricultural advisory services emphasized the need to supply more P to farmland than the withdrawal with crop harvest to compensate for fixation and losses. Therefore, there was, and still is, a general concern on the effects of a fertilization strategy based on equilibrium between P input rates and P uptake rates with harvested crops (P-equilibrium fertilization). The research on effects of P-equilibrium fertilization on soil-P status, crop yields and P leaching, showed that P-equilibrium fertilization can be compatible with efficient crop production, in the short and in the long term (Chapter 5). Since 1989 fertilization schemes at 'De Marke' have been based on P-equilibrium. Due to historical land use, there was a large variation in initial (1989) P status, expressed as total P, P-AL and Pw (Van der Paauw, 1956, 1971; Murphey and Rhiley, 1962) of the various plots. From 1989-2006 Pw, P-Al and P-content (means of all farmland) decreased. However, in plots with the lowest initial values, P status did not change, while in plots with high initial values it tended to stabilize at lower levels. On parcels with high initial P content, P was transported to deeper layers. The decrease in P in the 0-0.2 m layer was equal to the increase of P in the 0.2-0.4 and 0.4-0.6 layer. Crop P and dry matter yields in plots with high and low P status were similar. N/P ratios in crops were lower than 11, indicating that N was more limiting than P (Koerselman and Meuleman, 1996), while earlier work indicated that at 'De Marke' water availability limits the potential production (Aarts et al., 2000). Hence, at 'De Marke', P is not a dominant limiting factor for grass and maize yields. The weak response of crops to P status may, besides limitation by N and water, be explained as follows (*i*) maize with its relatively long growing season tends to catch up with growth retardation that might occur during a short period when a root system has to be developed in the juvenile stage of the plant (Neeteson et al., 2006), (*ii*) manure application in the rows of maize brings P closer to the roots developed by the young plant (Schröder et al., 1997), (*iii*) P may be released by decomposition of organic matter, and (*iv*) binding to soil seems not to be very strong implying that P in the soil is well available for crop uptake. Under circumstances that are different from 'De Marke', e.g. on soils with higher production potential, water availability and N application rates, or higher P binding capacity, the impact of P-equilibrium on P status and crop yields may be different than observed at 'De Marke' (Van Middelkoop et al., 2007).

To abate P losses from parcels with a high P status, these parcels could be mined by applying lower P rates than withdrawal with crop harvest. Also Koopmans (2004) suggested that mining soils that are rich in P is effective to reduce leaching of phosphorus. The P that is saved on those parcels might be applied on parcels with a low or moderate P status. This could be realized on the basis of organic fertilizer by separating slurry in a solid fraction with low and a liquid fraction with high N/P ratios (Verloop et al., 2007[a]).

2.5 Dynamics of soil organic matter

An analysis of SOM dynamics based on observed SOM percentage in different layers and simulations with a mono-component decomposition model (Yang and Janssen, 2000), showed that (*i*) over the period 1989-2010 SOM was more or less stable in permanent grassland (observed and simulated), but in the long term a decline in permanent grassland is to be expected (simulated), (*ii*) SOM declined in crop rotation over the experimental period unless grass was incorporated in rotation (observed and simulated), (*iii*) SOM decomposition rates were the same in permanent grassland and rotation (simulated).

The simulations show that in the long term a decline in permanent grassland is expected while SOM in permanent grassland has been constant over a period of 1989-2010. This can be explained by a gradual replacement of persistent SOM (residues of heather ploughed under when the land was reclaimed) by less persistent SOM (residues of agricultural crops and manure) that has a higher decomposition rate. It might be expected that replacement of old, persistent organic matter by easily degradable agronomic crop residues, coheres with a decrease of C/N quotient. However, the C/N-quotient in the soil at 'De Marke' shows no significant decline (Figure 4).

The similarity of decomposition rates of the same types of organic matter in rotation and permanent grassland implies that the frequency of ploughing does not influence decomposition. Hence, the difference between observed SOM dynamics in crop rotation and permanent grassland must be attributed to differences between organic matter input rates. This suggests that tillage is not a relevant factor in thinking of ways to maintain SOM at 'De Marke', but that adjusting the total rate of organic matter is. Calculations showed that to maintain SOM at the present level, organic matter inputs are required up to more than twice the present input. It is beyond doubt that when this input would be realized using crop residues and organic manure, this would lead to substantial increase in N and P losses and it would be detrimental for N and P use efficiency.

From this perspective measures that do not put pressure on nutrient efficiency *and* increase OM input rates to soil should be welcomed. Catch crops (already applied since 1992) were thought to be functional in this respect, but their contribution to SOM in the long term appears to be low, which is confirmed by results of Van Schooten et al. (2006). This is caused by their high decomposition rate. It might be considered to distribute SOM more evenly over the farmland by allocating the arable years with relative low organic matter inputs evenly over the farmland. This could be done by incorporating the land that is currently used as permanent grassland in crop rotation. Then, a rotation could be constructed with a lower ratio of arable/grassland years. Finally, the study urges to reconsider the application of strong manure digestion as this would reduce the organic matter input substantially. The effect of the anaerobic digestion practiced since 2004 on SOM are marginal. Possibly, a further decline of SOM must be accepted as an autonomous development. The consequences of this decline should be evaluated by modelling the performance of the farm on the basis of the expected lower SOM levels in soil as was also done by Aarts et al. (2000). Altogether, in the long term maintenance of SOM might be a more critical factor then was recognized so far.

So far, particularly long term SOM dynamics have been discussed. However, SOM that is decomposed shortly after incorporation in the soil might also contribute to conditions in soil that are favourable for crop production. In a review Loveland and Webb (2003) conclude that 'active soil organic matter', standing for easily decomposable substrates, might enhance aggregate stability. This was confirmed, particularly for the residues of cover crops in a study of Jokela et al. (2009). This indicates that it might be too strong to conclude that catch crops do not contribute to soil fertility, other than by preventing N from leaching (Section 2.2). In studies on the effects of SOM decline, it should also be reconsidered that different substrates that are part of the complete SOM pool, may not have the same effect on water retention capacity (Tisdall and Oades, 1982).

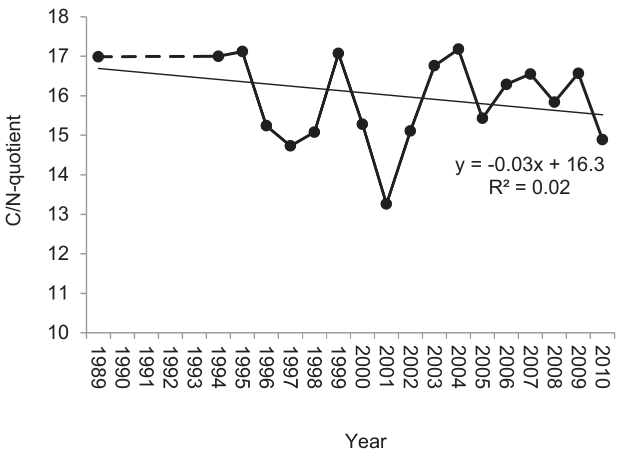


Figure 4: *Dynamics of C/N-quotient (average across all farmland) at 'De Marke' from 1989-2010.*

3 ENVIRONMENTAL PERFORMANCE

3.1 System development and environmental results

Table 2 presents the environmental performance concerning N and P in the different stages of system development. The farming system implemented in 1992, system A, met most environmental targets (Aarts, 2000; Hilhorst et al., 2001). Ammonia emissions were consistently lower than the allowed maximum, which may be explained as an effect of the strategy to limit protein levels in the ration of the cattle resulting in relatively low ammonium concentrations in manure. Also the concentration of P in groundwater was consistently low. Under the regime of system B with lower grazing intensity, nitrate concentrations in groundwater, which were excessive in system A, declined and in the succeeding systems this decline proceeded till 44 mg l^{-1} in system D. It is plausible that this decline is a result of the various system adjustments: reduced grazing intensity in system B, changes in crop rotation and application of anaerobic manure digestion in system C, and reduction of mineral fertilizer inputs in systems B and C. But, the decline in nitrate concentration is rather small and weather conditions may also have affected nitrate dynamics at 'De Marke' (Rozemeijer et al., 2009). The drop in N surpluses at soil and farm scale, which similarly to nitrate leaching were excessive in system A, was more pronounced than that of nitrate and was clearly induced by manure management.

Table 2: *Standard environmental conditions applied in the design of the dairy farming system of 'De Marke' (Aarts et al., 1992); for the description of systems A-D, see Section 1.*

Objective	Maximum value	System A 1993-1999	System B 2000-2003	System C 2004-2008	System D 2009-2010
Nitrate leaching[1)]	50	59	55	52	44
Ammonia volatilization[2)]	30	21	19	21	20
N surplus farm[3)]	128	165	133	98	156
N surplus soil[4)]	79	129	86	58	87
P leaching[1)]	0.15	0.03	0.03	0.03	0.03
P surplus[3)]	0.45	2	0	2	6

[1)] Concentration in the upper meter of groundwater (mg l^{-1}).
[2)] From manure, calculated (kg ha^{-1} yr^{-1}, Aarts, 2000).
[3)] As farm inputs, including biological N fixation by clover (for N) and deposition, minus outputs assuming no accumulation in the soil (kg ha^{-1} yr^{-1}).
[4)] As soil inputs, including chemical fertilizer, organic fertilizer, biological N fixation by clover and deposition, minus outputs with harvest (kg ha^{-1} yr^{-1}).

The analysis of the efficacy of manure management suggests that NFRV of manure has increased since 2000 and that this offered the opportunity to reduce the use of mineral fertilizer-N. However, it was also shown that N fertilizer levels were sub-optimal since 2004 and put a pressure on grass production (Chapter 4) resulting in a decline of N and P yields. This is reflected by an increase in the N and P surplus, in particular, in system D, in which slurry separation was adopted (Table 2). Hence, it is difficult, to specifically judge the effects of manure separation at this stage because delayed effects of reduced fertilizer inputs on N and P yields may have affected the results of system D.

3.2 Development of farm N and P use efficiency

Trends of farm nutrient use efficiency (presented in Table 4) are the result of changes in the subsystems soil, cattle, manure, soil, crops and feed. To understand farm nutrient use efficiency, first developments in the soil subsystem and interactions between the performance of the soil subsystem and feeding are discussed.

The soil-N use efficiency increased from 65% in system A to 81% in system C, after which it decreased in system D (Table 3). The soil-P use efficiency increased to levels higher than 100% in systems B and C (harvested yields higher than inputs) and decreased again in system D (Table 3). N and P yields were similar in systems A, B and C but dropped in system D. Apparently, on the scale of the entire farm, dry matter, N and P yields of roughage could be maintained at a rather constant level in system B, when mineral fertilizer-N rates were reduced, and in system C when mineral fertilizer-N was withheld. The data suggest that system C could have been continued as the effects of the gradual decline of grass N yield on the crop-N yields of the whole soil system are not very distinct (Table 3). However, the decline of grass N yields had implications for feeding.

To supplement the perceived shortcomings of roughage quality and quantity, more feeds (mainly concentrates), and with it, feed N, was imported and provided to the herd (Figure 5). Hence at farm scale a transition occurred of N inputs with mineral N to N inputs with feed N. At 'De Marke', not only for N but also for P the boundary conditions (Table 2) limit the space for increasing inputs with feeds. Particularly for P 'De Marke' is close to the maximum permissible P input, which is equal to the realized P export in products (13-16 kg ha^{-1}, Table 4). Therefore, concentrates were purchased with P concentrations as low as 4.0-4.5 g per kg, which is indicated somewhat by the constant level of the inputs of P with concentrates as compared to that of N; the amount of concentrates-N offered to the herd increased significantly whereas the amount of concentrates-P offered to the herd showed no significant trend (Figure 5). The ratio of (concentrates-N)/(concentrates-P) increased from 5.5 in 1993 to 9.4 in 2010. The purchase of feeds with low P concentrations is possible without unacceptable risks of P deficiencies of cattle as P requirements of the cattle are more than met (Valk and Šebek, 1999; Šebek et al., 2007). Despite careful selection of feeds low in P, the results of Tables 2 and 4 indicate that P inputs and P outputs of the soil system were not always in balance.

Table 3: *Inputs of N and P to the soil, outputs with harvested corps, dry matter yields of crops (kg ha^{-1}) and N and P use efficiency[1)] (%) as prognosed and realized in systems A, B, C, D.*

	Prognosis	System A 1993-1999	System B 2000-2003	System C 2004-2008	System D 2009-2010
Nitrogen					
Inputs	355	368	345	293	281
Crop yields	276	240	259	235	194
Efficiency	78	65	75	81	69
Phosphorus					
Inputs	37	35	32	30	33
Crop yields	37	32	37	34	28
Efficiency	100	94	115	115	85
Dry matter					
Crop yields	-	9991	11167	10521	9638

[1)] Outputs with harvested crops as percentage of inputs including mineral fertilizer, manure products, biological N fixation by clover and deposition.

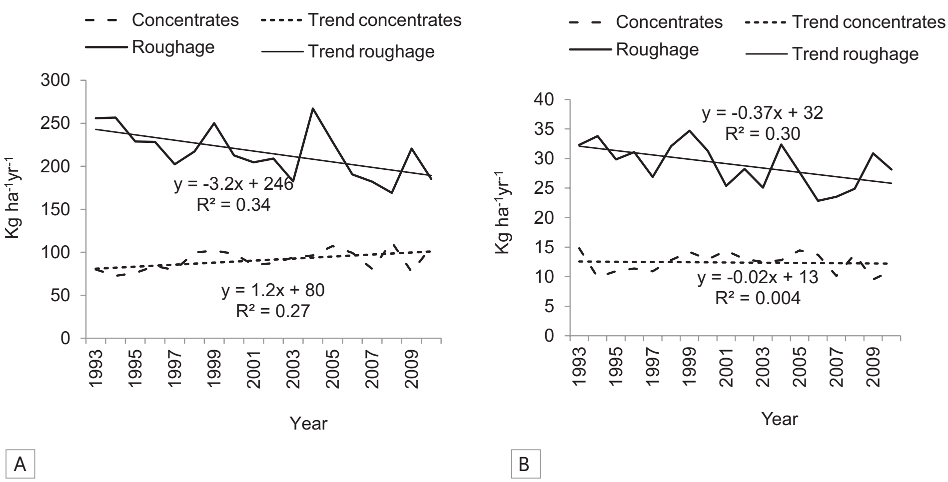


Figure 5: *N and P offered to the herd with purchased feeds (concentrates) and with crops grown on the farm (silaged and grazed grass and arable products - roughage), kg ha^{-1}; (A): nitrogen, and (B): phosphorus.*

At farm scale, the system development resulted in a substantial increase of the N use efficiency (Table 4 – compare system C with systems A and B). In system B, C and D, the total N input was lower than in system A (Table 4), in spite of the increased import of feed N (Figure 5). N output was more or less maintained at the same level. The prognosed farm-N use efficiency was realized in system C and D, while the realized farm-N use efficiency in systems B, C and D was higher than in system A.

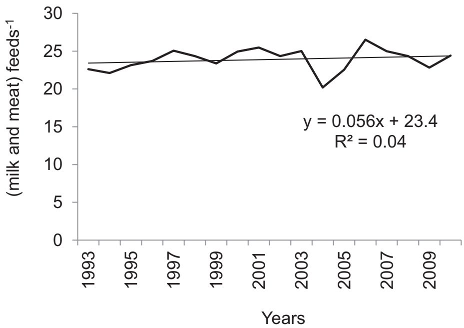
This can be explained, at least partly, by the replacement of N inputs with mineral fertilizer-N by feed N. A higher fraction of feed-N inputs will be transferred into farm products than of mineral fertilizer-N, because only a part (78% according to the prognosis) of fertilizer-N applied to soil is utilized by crops and, after silaging of crops, again only a part (about 92%) of crop-N will be transferred into feeds offered to the herd. Thus per kg mineral fertilizer-N only 0.72 kg is offered to the herd, against 1 kg per kg of purchased feed-N. However, replacement of fertilizer-N inputs by feed-N inputs cannot explain the complete increase in farm N use efficiency. The changed manure management and changed transition from the arable phase in crop rotation to the grassland phase, probably accounted for the major part of the increase in N use efficiency.

Table 4: *Inputs, outputs in products (milk and meat) (kg ha^{-1}) and N and P use efficiency[1)] (%) at 'De Marke' as prognosed and realized in systems A, B, C, D.*

	Prognosis	System A 1993-1999	System B 2000-2003	System C 2004-2008	System D 2009-2010
Nitrogen					
Inputs	195	238	217	178	212
Outputs	70	78	72	74	75
Change of stocks	0	-5	11	7	-19
Efficiency	36	33	35	43	37
Phosphorus					
Inputs	13	16	14	15	16
Outputs	13	14	12	12	14
Change of stocks	0	-1	2	1	-5
Efficiency	100	87	103	91	69

[1)] Milk and meat produced as percentage of input and corrected for change of stocks.

The tight N fertilization schemes have led to a decrease of N concentrations in grass silage of permanent and temporary grassland from 35 to 23 and from 35 to 26 g per kg, respectively. This gave rise to concern about the roughage quality (Verloop et al., 2007[b]) and it is argued that the digestibility of the grass has dropped (Šebek en Bannink, unpublished data). A drop in digestibility of feeds affects the nutrient efficiency of the herd and would bring the farm into a position that the production relies more on external inputs, not only to compensate for lower yields but also to compensate for lower utilisation of the energy and protein in roughage. Until now the N and P utilisation of the herd was maintained, although with fluctuations (N and P utilisation do not show a significant trend at $P < 0.05$ - Figure 6). However, the issue of digestibility should be explored carefully, because a decline of the efficiency of the herd has a high impact on farm nutrient use efficiency.



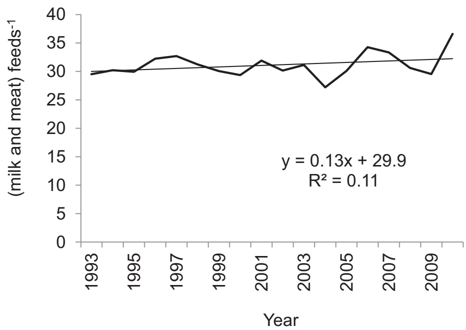


Figure 6: *Transfer of N and P offered to the herd with feeds into products (milk and meat), %; (A): nitrogen, and (B): phosphorus.*

4 IMPLICATIONS

4.1 Further research

In this research maximum attainable nutrient use efficiency was explored since 2000. System modifications within the farming system were accompanied with reductions of inputs of mineral fertilizer-N. Milk production at an intensity of 12000 kg ha^{-1} could be realized without using fertilizer P and N. But N limitation in grassland has become more severe. Hence, it might be concluded that the techniques of manure management do not allow leaving out mineral fertilizer without putting yields of grassland under pressure. Probably, continuation of the current system D will in the near future *not* result in the best attainable farm performance. Hence, to optimize farm performance (the primary target), it might be appropriate to re-introduce mineral fertilizer-N although the rates can be restricted to lower levels than usual before 2000 (114 kg N ha^{-1} in grassland). However, it might be more informative to further explore the problems that arise, through continuation of the current farming system with its tight fertilization schemes. The relevance of this direction is high because farmers will be asked in future to increase nutrient use efficiency in systems that rely as far as fertilization is concerned, as much as possible on the farm manure excreted by the herd. In this development farmers could be supported by focussing further research on:

1. the enhancement of N utilisation from manure;
2. a correct assessment of the nutrient replacement value of $NFRV_{manure}$;
3. maintenance or increase of yields and quality of grass and maize under the regime of tight fertilization;
4. enhancement of digestibility of grass silage by the herd by mechanical or chemical treatment of silage through additives.

Meanwhile assessments could be made of the effects on farm performance of re-introducing the use of small (optimum) amounts of mineral fertilizer-N.

The explorations on future soil fertility showed that phosphorus availability will probably not limit crop production under a continued strategy of P-equilibrium fertilization. The analysis of SOM percentage in the farmland indicated that future dynamics are controlled mainly by the organic matter inputs to soil. Moreover, it was shown that a decline of SOM will likely take place, not only in crop rotations but also in permanent grassland. The results seem to contradict with other studies (Reijneveld et al., 2009) and the results are therefore remarkable, but of great importance. Concerning soil fertility the following issues are relevant:

1. The effects of P-equilibrium fertilization may be different for different soil types and hydrological conditions (e.g. wet soils). Therefore, it might be explored to what extent the results obtained at 'De Marke' are representative for other regions and soil types.
2. It seems justified to conduct a comprehensive analysis of the simulations on long term SOM dynamics to confirm or adjust the indicated trends.
3. The effects could be explored of decline of SOM on light sandy soil on water retention capacity and on crop productivity and, next on the attainable milk production intensity under the boundary condition of environmental standards.

During this research the issue of greenhouse gas (GHG) emissions became increasingly important. At present many studies are conducted on the possibilities to reduce GHG emissions (e.g. Schils et al., 2006; Schils et al., 2013). GHG emission has already been adopted as a research item at 'De Marke'. This could lead to additional system development. For this process again clear boundary conditions and targets are required and these targets should be combined with the present standards for N and P to form a coherent starting point of system design and implementation of measures. The present standards for N and P should be considered as the playing field within which innovations can be implemented to reduce GHG emissions.

4.2 Practical implications

As mentioned before there is a need for further development of efficient management. Strategies to increase farm nutrient use efficiency by improved farm mineral management as explored at 'De Marke', demand more and new skills from farmers. Therefore, farmers will have to be prepared to invest in their knowledge and skills. In return, farmers obtain the capacity to systematically develop the farm and to avoid problems by management.

Future management has to take into account societal demands such as those referring to grazing. At 'De Marke' more nitrate leaching was found on grazed grass than on ungrazed grass in spite of the application of an optimized grazing method. At present, Dutch farmers that apply grazing are rewarded by a higher price for milk than farmers that keep their cattle indoors. Therefore, the challenge is to combine efficient farming with substantial grazing. A part of this dilemma may be solved by increasing management skills on grazing. Many extension and training programmes in and outside the Netherlands are dedicated to grazing and these may result in better grazing practices that can be reconciled better with efficient management.

One of the most daring steps in efficient nutrient management, is to anticipate higher efficiency by reducing inputs. For instance, the 25-50 kg of N taken up by a catch crop, will not be lost by leaching and will, instead, contribute to N that is available for the next maize crop. The next step should then be to lower the N fertilization with 25-50 kg or its fertilizer equivalent to prevent over fertilization in the maize and to prevent that the gain of catch crops would be lost, after all, in maize. This principle of reconsidering inputs to follow up on measures to increase efficiency, holds for any part of the production system.

Strategies and measures at 'De Marke' with potential for practical implementation by dairy farmers are:

- Understanding nutrient flows using schemes of the transfer of nutrients from the cattle to products and manure, to soil, crops and feeds. A tool to facilitate this, the 'Annual Nutrient Cycling Assessment (ANCA)', is being developed in collaboration with the project Cows & Opportunities.
- Application of catch crop as undersow between the maize rows to reduce N losses. The spared N losses can be subtracted from fertilization. The saved N fertilization in maize can be re-assigned to grassland.
- Application of systematic crop rotations with benefits for P availability and the even distribution of organic matter over the farmland and reduced nitrate leaching.

- Fine tuning the application of N in 1st and 2nd year maize and fine tuning the transition from arable to grassland in the crop rotation to prevent bare soil or weak vegetation in winter and the associated N losses.
- Mining P on 'leaky' plots with high P status, in order to accelerate transition of the plots to a normal P status with lower P losses. The saved P could be re-assigned to plots with low P status.

5 ON PROTOTYPING

The research that is conducted at experimental farm 'De Marke' is a typical example of prototyping. The method, introduced by Vereijken (1992, 1997) and further developed for dairy production by Aarts et al. (1992), contributes strongly to knowledge development in dairy system research on experimental and commercial farms, for instance in the projects 'Cows & Opportunities' (Oenema et al., 2001), 'Proeftuin Natura 2000 Overijssel' (*www.proeftuinnatura2000.nl*), and DAIRYMAN (*www.interregdairyman.eu*). Prototyping has been described in several publications (Vereijken, 1992; Aarts, 2000; Oenema et al., 2001), but there is no concise definition of the method. It may be characterized as: a participative method (*i*) to explore on whole (integrated) farming systems the possibilities to realise pre-defined goals, (*ii*) considering agricultural, ecological and economic drivers and, (*iii*) structured by a stepwise procedure of goal setting, designing, implementation, monitoring and evaluation. During the conduction of prototyping research, frequently decisions have to be made on monitoring, system development and *ad hoc* interventions to anticipate unforeseen conditions (for instance extreme drought). This is different from most, more classical experiments. This difference justifies reflection and analysis on the process of prototyping. Some of the experiences and lessons learned in the research of 'De Marke' are discussed here.

A prototype dairy farm is a complex research object. In agronomic research usually experiments are designed to reduce the complexity of the real world by keeping many factors constant (Popper, 1959). It is beyond doubt that this reduction of complexity is invaluable in research. However, a drawback is that experiments may become so different from the real world that their results are hard to relate to entire systems and, consequently are insufficiently relevant. In prototyping the aim is to maximize both comprehensiveness and relevance. Thus, the prototype farm should provide the relevant context and the conditions to discern impacts of measures (Aarts, 2000). The definition of the context should be based on research objectives and research questions. In the research at 'De Marke', objectives were well defined and the context was formalized by pre-defined boundary conditions and targets (Table 2). This appears to be crucial, because this pre-defined context is helpful to adhere the main line of system development and avoid losing track by distracting tendencies, such as:

1 adjusting the farm management and structure to common practice to be more representative;
2 acting like a ‘good farmer’ would do;
3 adopting fancy novelties that are not in line with targets.

These tendencies are conflicting with the task of a prototype farm, because (*i*) A farm with the task to explore and demonstrate alternative management options cannot be ultimately representative for common practice. The relevant context of the farm is the context that is under exploration, thus the designed alternative, and not common practice; (*ii*) In explorations it may be relevant to know risks that a ‘good farmer’ would avoid. If so, the prototype farm should take that risk and cannot be governed in the way a ‘good farmer’ would; (*iii*) Integrated farming research is complex. Any addition to the complexity by adopting research aspects that are of minor importance should be avoided, because it may disturb the main line of the research.

This seems like a straightforward guideline for prototyping research, but in reality, there are grey shades in decision making. To make work on the experimental farm enjoyable and to keep the farm interesting and vivid for a broad public (farmers, farm advisors, teachers, scientists and policy makers) it is important to develop and innovate. From 2005 to 2007 the developments on the farm were reduced to the minimum. This may have been beneficial for scientific analysis, but at that time, it harmed the satisfaction of the people involved in the research and the charisma of the farm. Hence, an experimental farm cannot be run as if it were a long term field trial.

In the process of system development, often the urge was felt to develop and to innovate faster in order to make progress. But, implementation of multiple measures in a short timeframe will certainly complicate analyses as it will become harder to disentangle the effects of individual measures. In some situations this was considered acceptable. Therefore, in 2000 it was decided to reduce grazing, adjust rotation and reduce the number of young stock in one year to enforce a fast reduction of the soil-N surplus (Aarts and Van Keulen, 2000). Beyond doubt this gave a strong impulse to the prototype and it can be acceptable that later the combination of measures can be evaluated rather than any individual measure. To find the optimum between progression and consistency it is helpful to work with a team where guards of consistency and strong innovators cooperate.

Prototype farm 'De Marke' is an experiment without replication. Thus, standard statistical tests cannot be applied at farm level and the statistical reliability of an observed parameter, e.g. the farm N surplus, is undefined. It is practically not feasible to establish a second prototype, with the same structure, that is explored in exactly the same way and is subjected to the same or similar geophysical circumstances. This problem can be coped with in different ways. Data of different years, together give an impression of uncertainties of parameters. A network of farms such as the project 'Cows and Opportunities' (Oenema et al., 2001) allows analysis of data of different farms. Furthermore, the 'one replicate problem' is less severe at the level of sub systems, because within sub systems more observational units are available: the farmland comprises 23 parcels and 59 plots. On these plots 170 nitrate bore holes are present and there are 6 fixed observation plots for detailed analyses. These are no 'pure replicates' meaning that their locations are not assigned at random and they are not evenly distributed over strata (e.g. soil characteristics and land use). But still, the large number of observational units makes it possible to get information on the reliability of an estimate and to analyse responses using elaborate datasets (Aarts, 2000; Verloop et al., 2006; Verloop et al., 2010). From the perspective of farm management it might not be optimal to adhere to the division of the farmland in 23 parcels, but for the analysis of the farm it is crucial. The establishment of balances of nutrient inputs and outputs and the records of management and crop yields for each parcel forms the basis of many analyses.

The farming context facilitates the discovery of problems and solutions. The starting point of the implementation of slurry separation was a problem: 'How to meet N/P requirements per parcel without using mineral fertilizers when only slurry with fixed N/P ratio is available?' (Verloop et al., 2007). Also improvements in crop rotation, in particular on the transition of grassland to arable land and *vice versa*, were developed in response of observed problems. Hence, the indication of practical problems in a real farm context is an important function of integrated system research. In short: in prototype research the innovation does not only drive system development and the new farming context, the context also drives the innovation.

The challenge of prototyping is to reconcile an innovative and dynamic attitude with options for analysis. There is only little scientific exchange on the method and the majority of publications on systems research are focussed on extension (Langeveld, 2005) tools for farm development (Sterk et al., 2007; Le Gall, 2011) or design and modelling (Ten Berge et al., 2000; Van de Ven et al., 2003). A more intensive exchange between researchers involved in prototype research, as made possible in the project DAIRYMAN but also in scientific working groups might contribute to further development and successful application of the method.

6 CONCLUSIONS

Main scientific and applied conclusions of this thesis include:

1. Even in a rotational grazing system, optimized to distribute restricted amounts of grazing-N excreta (50-80 kg ha^{-1}) evenly over farmland, longer confinement of cattle might be required to realize the maximum permissible nitrate concentration in groundwater.
2. Nitrate leaching from a maize/grassland crop rotation is similar to the leaching from permanent grassland.
3. Maize in rotation reduces nitrate leaching in temporary grassland and, consequently an evaluation of nitrate leaching based on only crop level results in a biased judgement of the effect of maize. Therefore, in monitoring programs nitrate leaching should be evaluated on the level of crop rotations and not per crop.
4. Milk production at an intensity of 12000 l ha^{-1} is possible on light sandy soil without violating strict environmental standards related to N and P
5. The techniques that are currently available to process manure applied at rates of 225 kg N ha^{-1} do not allow leaving out mineral fertilizer without jeopardizing yields of grassland.
6. At 'De Marke' phosphorus availability is not likely to limit crop production under a continued regime of P-equilibrium fertilization.
7. Strong anaerobic manure digestion may be desirable for production of renewable energy, is conflicting with maintenance of SOM levels in the long term.
8. Tillage at 'De Marke' does not significantly accelerate SOM decomposition.

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SUMMARY

Nitrogen (N) and phosphorus (P) are indispensable external inputs in agricultural production systems to replenish the nutrients exported in sold products and the nutrients removed through unavoidable losses. High production levels of dairy farms in the Netherlands are associated with high inputs and high environmental pressure. Therefore, the Dutch government and the dairy sector cooperate in stimulating farmers to reduce losses through more efficient use of inputs, by improving their management skills and through the introduction of management tools and innovations. To support the process of change in dairy farming, knowledge is required on the effects of nutrient management on farm performance. This thesis deals with the development of sustainable dairy production, with a focus on efficient use of nutrients (N and P).

The objectives of this study are (*i*) to explore the possibilities for development of a dairy production system on dry sandy soil, to meet strict environmental standards related to N and P, (*ii*) to maximise the N and P use efficiencies at the level of the farm and of the soil, and (*iii*) to quantify the potential risks of efficient nutrient management on soil fertility. The research was conducted on experimental dairy farm 'De Marke' in the eastern part of The Netherlands.

'DE MARKE'

'De Marke' was established in 1989 to explore and demonstrate the possibilities to produce milk at an intensity of 12,000 kg per hectare, without violating strict environmental standards on a dry sandy soil that is extremely susceptible to nitrate leaching. The farm provides the context of a common, commercial farm and the infrastructure to monitor N and P flows and other relevant parameters. Since the start, the farming system has been continuously developed in order to meet the targets, according to a method called prototyping. Prototyping consists of a cyclic procedure of target setting, establishing criteria to judge farm performance, design, implementation of the design, monitoring, analysis of data and evaluation. This thesis covers system development since 2000, but data originating from 1989-1999 are also included in the analyses.

NITRATE LEACHING

After implementation of the original farming system the nitrate concentration in the groundwater decreased from 193 mg l^{-1} in 1990 to 63 mg l^{-1} in 1993 (Chapter 2). However, in 1994 it 'stabilized' at 55 mg l^{-1}, a level exceeding the environmental standard of

50 mg l^{-1}. The causes of excessive nitrate leaching were examined by relating measured nitrate concentrations to soil and crop management. The effects of precipitation surplus, groundwater level and dissolved carbon were taken into account to prevent confusion of their variability with management effects. The study indicated that grazing leads to higher nitrate leaching, with nitrate concentrations of 23-49 mg l^{-1} in non-grazed grass and 59-74 mg l^{-1} in grazed grass. This substantial effect was observed in spite of careful grazing management with rotational grazing to distribute grazing excreta evenly over the grazing area. It is concluded that even in an optimized grazing system, longer confinement of cattle is required to restrict the nitrate concentration in the groundwater to the maximum allowed value. Nitrate concentrations under maize (45-66 mg l^{-1}), were between those under non-grazed and grazed grass. The nitrate concentration in grass in rotation was substantially lower than in permanent grassland, which can be interpreted as an effect of (maize in) rotation. The overall nitrate leaching under rotations (means over the complete rotation) and under permanent grassland was similar: 59 and 63 mg l^{-1}. Bare soil in winter forms a high risk for nitrate leaching. Therefore, effective functioning of catch crops between two maize years is crucial, as well as the transition of the arable phase to grassland. On 'De Marke' the optimal last arable crop in the rotation is spring barley. It is harvested in mid-summer after which new grass can be sown and develop well before winter.

NITROGEN MINERALIZATION

Nitrogen mineralization is determined on six observation spots (Chapter 3). Insights in spatial and temporal patterns of N mineralization are necessary to improve the synchronization of N application and crop-N requirements. Average annual soil-N mineralization was highest under permanent grassland. In temporary grassland, soil-N mineralization gradually increased (1^{st} year < 2^{nd} year < 3^{rd} year) and in arable crops after grassland, mineralization gradually decreased (1^{st} year > 2^{nd} year > 3^{rd} or more years). Mineralized N, together with N applied through fertilizers, exceeded crop uptake in 1^{st} year maize, so fertilizing 1^{st} year maize can be stopped. Mineralization in winter, outside the growing season, was 77 kg N per ha in maize. This underpins the importance of a suitable catch crop to reduce potential N leaching in winter. Italian ryegrass, undersown in maize at 'De Marke', yielded 50 kg N per ha during winter. This suggests that a higher N uptake is needed to reduce N leaching. However, it is uncertain whether this can be achieved with Italian ryegrass or alternative crops. Temporal and spatial variability of soil-N mineralization was high and showed no relations with known field conditions.

N UTILISATION FROM MANURE

On 'De Marke' four systems of manure management can be distinguished in the course of its development. In the system applied from 1994-1999, manure and urine excreted indoors was collected as mixed slurry and stored in a covered silo until application to grassland and arable land. A rotational restricted grazing system was applied, so that nearly 80% of the N excretion was collected indoors and grazing excreta were relatively evenly distributed over the grazed parcels. From 2000-2003 grazing time per day was reduced to 5 hours so that nearly 90% of the N excretion was collected indoors. From 2004 onwards, slurry was anaerobically digested before application, to increase the efficiency of utilisation of slurry-N through a smaller share of N in organic form. Moreover, to maize, slurry was no longer applied evenly over land before sowing, but in rows next to where the maize seeds were positioned. In 2009 and 2010, 30% of the digested slurry was separated in a liquid and a solid fraction. To correct for the anticipated increase in N utilisation, the supply of mineral fertilizer-N was stopped since 2004.

Analysis of the dynamics of (*i*) N use efficiency (NUE: (harvested N crop yield)/(sum of N inputs to soil)); (*ii*) the N replacement value of manure-N ($NFRV_{manure}$), and (*iii*) the yields of N and dry matter, gives an impression of the performance of manure management (Chapter 4). Results of the analyses suggest that N utilisation of manure has increased over the research period, although this cannot be attributed directly to specific measures. The estimate of $NFRV_{manure}$ for the entire farm over the research period of 1994-2010 (0.70) indicates a rather high utilisation of manure-N by crops. The estimates for permanent (0.71) and temporary grassland (0.63) match reasonably well with the range derived from field trials (0.47-0.79). The estimate for maize (0.97) was high compared to results from field trials (0.12-0.75). The study suggests that a substantial portion of the mineral fertilizer-N that was used in grassland in the period from 1994-1999 could have been abolished as a result of modifications in manure management. However, the possible increase in $NFRV_{manure}$ is not sufficient to make mineral fertilizer-N completely redundant. This explains the decline in N and dry matter yields in grass since 2004, when application of mineral fertilizer-N was stopped.

P-EQUILIBRIUM FERTILIZATION

The results of the research on effects of P-equilibrium fertilization on soil-P status, crop yields and P leaching, show that P-equilibrium fertilization can be compatible with efficient crop production, in the short and in the long term (Chapter 5). Initially, when 'De Marke' was established, there was a strong variation in P status, expressed as either total P, P-Al and Pw of the various plots. Since 1989, fertilization schemes on 'De Marke' have been based on P-equilibrium. Since then, in plots with the lowest initial values, P status did not change, while in plots with high initial values it tended to stabilize at lower levels. Crop P and dry matter yields in parcels with high and low P status were similar and there are no indications of P limitation. On parcels with high initial P content, P was transported to deeper layers, but was not found in the groundwater.

DYNAMICS OF SOIL ORGANIC MATTER

Currently, soil organic matter (SOM) percentage in the topsoil of 'De Marke' decreases on average by 0.03 yr^{-1}. Over a research period of 20 years, the decline did not significantly slow down. The decline occurred in crop rotations, while SOM was more or less constant in permanent grassland. The decrease in SOM in crop rotations is a concern and a motivation for the analysis of possible causes. The dynamics of SOM is influenced by two factors: (*i*) the annual input of substrates, and (*ii*) the decomposition rate of available substrates. An analysis of SOM dynamics based on observed SOM percentage in different layers and simulations with a mono single-component decomposition model, showed that decomposition rates of SOM are similar in permanent grassland and in crop rotations. This suggests that tillage does not accelerate the decomposition rate of SOM, and thus that the limited annual input of organic matter is the cause of the current decline in SOM in crop rotations. Hence, management alternatives that lead to increased organic matter inputs to soil will contribute to the maintenance of SOM levels. An analysis of effects of management alternatives showed that conventional (mild) anaerobic digestion hardly affected future SOM levels. However, strong anaerobic digestion puts future SOM levels under pressure. Application of a catch crop in maize hardly affected future SOM levels. Simulations of SOM levels over a period of 50 years show that in the long term a decline in SOM may be expected, even in permanent grassland. This seems surprising, as SOM in permanent grassland remained constant over the period of 1989-2010. This can be explained by gradual replacement of persistent SOM (residues of heather) by less persistent SOM (residues of agricultural crops and manure) with a higher decomposition rate. The consequence is an acceleration of SOM decomposition over time. Measures to change this trend seem conflicting with efficient nutrient use. However, a decline in SOM would make the soil more susceptible to leaching of nitrate. Possibly, a further decline in SOM must be accepted as an autonomous development and the potential to sustain efficient crop production at lower SOM levels should be evaluated by modelling. The study strongly suggests to reconsider the application of strongly digested manure as this substantially reduces the organic matter input.

GENERAL DISCUSSION

The general discussion reveals the impact of system development on farm performance. It shows that at 'De Marke' N and P use efficiencies can be increased to levels higher than those realized from 1993-1999 (33 and 87% for N and P, respectively). Farm-N use efficiency amounted to 35% for 2000-2003, 43% for 2004-2008 and 37% for 2009-2010. Farm-P use efficiency in these periods amounted to 103, 91 and 69%, respectively. Farm performance in terms of farm- and soil-nutrient use efficiency and N and P yields declined in 2009-2010, which is interpreted as a delayed effect of N limitation resulting from the abolishment of fertilizer-N. The development of the farm performance for N and P is clearly linked to N and P flows in the soil system with an increase in soil N use efficiency from 65 (1993-1999) to 81% (2004-2008) and an increase in P use efficiency of 94 (1993-1999) to 115% (2004-2008). The corresponding soil N surplus amounts to 58 kg ha^{-1} and the surplus is negative for P, however, the corresponding nitrate concentration in the groundwater amounts to 52 mg l^{-1}, similar to the target, but not lower. The systems with the most favourable performances are all characterized by reduced grazing, implying that the fraction of N that is excreted on pasture is reduced from 0.23 to 0.11, and by lower application rates of mineral fertilizer-N. The adoption of anaerobic digestion, application of manure in the rows of maize and optimization of the transition of the arable to the grassland phase brings the farm into the status of the most favourable performance for N. It was not possible to attribute this development to specific measures.

A decline since 2004 of N yields in grassland is probably caused by increasing N limitation. This N limitation also resulted in lower digestibility of grass silage. In response to these developments, more N had to be imported in concentrates. P import with feeds can be restricted by selecting concentrates that are low in P content. Continuation of this system might change the farm into a production system with increased nutrient efficiency, but that is somewhat less self-sufficient for feed. In the long term P is not expected to become a major limiting factor. However, SOM decline seems to proceed, which may be critical for efficient crop production in the long term.

Based on these observations and analyses it is suggested to further explore the current farming system with its tight fertilization schemes. This might reveal new pathways for farmers looking for possibilities to safely increase nutrient use efficiency on their farm. The focus should be on:

1 improvement of N utilisation from manure and a reliable on-farm assessment of the nutrient replacement value of the manure, $NFRV_{manure}$;
2 improvement of digestibility of roughage

It seems justified to conduct a comprehensive analysis of the simulations on long-term SOM dynamics to confirm or adjust the indicated trends and to explore effects of a decline in SOM on light sandy soil.

PRACTICAL IMPLICATIONS

The knowledge gained in this thesis can be applied in practical dairy farming and will contribute to the following issues:

1 It is recommended to stimulate application of catch crop undersown between the maize rows to reduce N losses. Next, it is important to reduce the chemical fertilizer application in maize to account for the saved N losses and assign the saved N fertilization in maize to grassland.
2 Application of a systematic crop rotation has many advantages. It enhances P availability in soil, results in an even distribution of organic matter over the farmland and reduces nitrate leaching.
3 Practical farmers should be more informed on the advantages of stopping the N supply in 1^{st} year maize after grassland and to reduce N supply in 2^{nd} year maize.
4 It should be considered in monitoring programs to not evaluate specific crops in the rotation, but to evaluate the complete rotation.
5 To restrict the loss of P to deeper layers, it might be effective to mine P on plots with high P status. The saved P could be assigned to plots with low P status.

SAMENVATTING

Intensieve productie in de melkveehouderij in Nederland gaat gepaard met een hoge aanvoer van de nutriënten stikstof (N) en fosfor (P) naar de bedrijven, een betrekkelijk lage benutting van de nutriënten en hoge verliezen naar het milieu. De Nederlandse overheid en de zuivelsector stimuleren veehouders om nutriënten efficiënt te gebruiken om zo verliezen te beperken. Dit gebeurt door verbetering van het nutriëntenbeheer op de bedrijven. Om dit proces te ondersteunen, is inzicht nodig in de effectiviteit van innovaties en in de gevolgen van efficiënt nutriëntenbeheer voor het functioneren van het melkveebedrijf. Dit proefschrift gaat over de mogelijkheden en consequenties van efficiënt gebruik van nutriënten (N en P) in bedrijfsverband.

Het doel van deze studie is om te verkennen: (*i*) of een melkproductiesysteem op droogtegevoelige, lichte zandgrond zo ontwikkeld kan worden dat het voldoet aan strikte milieueisen met betrekking tot N en P, (*ii*) welke N en P-efficiëntie op het niveau van het bedrijf en de bodem maximaal haalbaar is in een praktische bedrijfscontext en (*iii*) wat de mogelijke consequenties zijn van efficiënt nutriëntenbeheer voor de bodemvruchtbaarheid.

Het onderzoek werd uitgevoerd op het proefbedrijf voor duurzame melkveehouderij 'De Marke', gelegen in Hengelo (Gelderland) op een droogtegevoelige zandgrond. Op het bedrijf kunnen N en P stromen en andere relevante parameters gemeten worden. Het bedrijf is ontworpen om te voldoen aan strenge milieunormen, maar het ontwerp is sinds de implementatie (in 1989) steeds verder verbeterd. Hierbij wordt een werkwijze gevolgd die in het bedrijfssysteemonderzoek bekend staat als prototyping. Bij prototyping wordt een bedrijfsontwerp ontwikkeld door stappen in een vaste volgorde steeds opnieuw te doorlopen: doelen stellen voor het bedrijf, vaststellen van criteria om de bedrijfsprestatie te beoordelen, ontwerpen van het bedrijfssysteem, implementatie van het ontwerp, monitoren, analyse en evaluatie van de bedrijfsprestaties. In dit onderzoek is de bedrijfsontwikkeling die heeft plaatsgevonden sinds 2000 geëvalueerd; resultaten uit 1989-1999 werden bij de analyses betrokken om inzicht te geven in lange termijn ontwikkelingen. De evaluatie is gericht op nitraatuitspoeling, mineralisatie van stikstof, N benutting uit mest, P-evenwichtsbemesting en ontwikkeling van het organisch stofgehalte in de bodem.

NITRAATUITSPOELING

Nadat het bedrijfssysteem ‘De Marke’ was geïmplementeerd nam de nitraatconcentratie in het grondwater af van 193 mg l^{-1} in 1990 tot 63 mg l^{-1} (Hoofdstuk 2). Echter, in 1994 stabiliseerde de concentratie zich op een niveau rond 55 mg l^{-1}, dat is hoger dan de nitraatnorm. De oorzaken van deze te hoge nitraatuitspoeling werden onderzocht door de meetwaarden van de nitraatconcentratie te relateren aan bodem- en gewasbeheer. In het onderzoek werd rekening gehouden met omgevingsfactoren, te weten: het neerslagoverschot, de grondwaterstand en de concentratie van opgeloste organische stof in het grondwater, opdat hun invloed niet met managementeffecten zou worden verward.

Beweiding deed de nitraatconcentratie in het grondwater toenemen van 23-49 mg l^{-1} in onbeweid grasland tot 59-74 mg l^{-1} in beweid grasland. Dit aanzienlijke effect trad op ondanks een nauwkeurige uitvoering van de beweiding, gebaseerd op een systeem van omweiden, waarbij mestflatten en urine gelijkmatig over het grasland werden verdeeld. Zelfs bij een dergelijk beweidingssysteem kan langer opstallen van vee dus nodig zijn om de nitraatnorm in het grondwater te halen. De nitraatconcentratie onder maïs (45-66 mg l^{-1}) was lager dan in beweid gras en hoger dan in onbeweid gras. De nitraatconcentratie in gras dat wordt afgewisseld met akkerbouwgewassen (tijdelijk grasland) was duidelijk lager dan in blijvend grasland, wat geïnterpreteerd kan worden als een effect van (maïs in de) vruchtwisseling. De nitraatuitspoeling in de volledige vruchtwisseling en in blijvend grasland waren gelijk: 59 en 63 mg l^{-1}. Een onbedekte bodem in de winter verhoogt de kans op nitraatuitspoeling. Daarom is het functioneren van een vanggewas tussen twee maïsjaren cruciaal en is een ‘groene’ overgang van de akkerbouwfase naar grasland van belang. Op ‘De Marke’ is zomergerst het meest geschikte laatste akkerbouwgewas in de vruchtwisseling. Het wordt ruim binnen het groeiseizoen geoogst, waarna er nog voldoende tijd is om nieuw gras in te zaaien dat zich voor de winter tot een volledige nieuwe graszode kan ontwikkelen.

MINERALISATIE VAN STIKSTOF

Op zes vaste waarnemingsplekken op ‘De Marke’ werd de N mineralisatie gemeten (Hoofdstuk 3). Inzicht in de verdeling in ruimte en tijd van N mineralisatie kan bijdragen aan een betere synchronisatie van de (kunst)mest-N dosering en de N behoefte van gewassen. De jaargemiddelde N mineralisatie in de bodem was het hoogst onder blijvend grasland. In tijdelijk grasland nam de N mineralisatie toe in de volgorde: 1^e jaars gras < 2^e jaars gras < 3^e jaars gras en in akkerbouwgewassen volgend op tijdelijk gras nam de mineralisatie af volgens: 1^e jaars gewas > 2^e jaars gewas > 3^e of meerjaars gewas. De som van de N die vrijkwam uit mineralisatie en minerale N die werd aangewend met mest overschreed de gewasopname in 1^e

jaars maïs. Bemesting van 1e jaars maïs is dus onnodig. Mineralisatie in de winter, buiten het groeiseizoen, was 77 kg N per ha in maïs. Dit onderschrijft het belang van een goed ontwikkeld vanggewas om potentiële N verliezen door uitspoeling te beperken. Italiaans raaigras dat werd toegepast als onderzaai in maïs bracht op 'De Marke' 50 kg N per ha op. Dit suggereert dat een hogere N opname nodig is om de N uitspoeling verder terug te dringen. Echter, het is de vraag of dit haalbaar is met Italiaans raaigras of een ander vanggewas. De variabiliteit van N mineralisatie in ruimte en tijd was hoog en vertoonde geen significante relaties met bodemomstandigheden.

N BENUTTING UIT MEST

De systeemontwikkeling op 'De Marke' heeft geleid tot 4 wezenlijk verschillende systemen van mestbeheer. Van 1994-1999 werd op de stal uitgescheiden mest en urine verzameld als drijfmest en opgeslagen in een afgedekte silo totdat het werd aangewend op grasland en bouwland. De beweiding was beperkt, zodat 80% van de mest-N werd uitgescheiden en verzameld op de stal en het systeem van omweiden zorgde voor een gelijkmatige verdeling van weidemest-N over het begraasde weiland. Van 2000-2003 werd de beweidingsduur per dag teruggebracht tot 5 uur zodat bijna 90% van de uitgescheiden mest-N werd verzameld op de stal. Sinds 2004 wordt drijfmest vergist om het aandeel organisch gebonden N in de drijfmest te verlagen en zo de benutting van drijfmest-N te verhogen. Bovendien vond toen de overstap plaats van volvelds bemesten van maïs naar plaatsing van de mest in de rij, nabij het maïszaad. In 2009 en 2010 werd 30% van de vergiste mest gescheiden in een dunne en een dikke fractie. Vanaf 2004 werd geen kunstmest-N meer gebruikt als reactie op de veronderstelde toename van de mest-N benutting door gras en gewassen door het veranderde mestbeheer. De effecten van het veranderde mestbeheer werden geëvalueerd (Hoofdstuk 4) aan de hand van de volgende indicatoren:

i) de N-efficiëntie op de bodembalans (NUE); dit is de N opbrengst in geoogst gewas gedeeld door de som van N aangevoerd uit verschillende bronnen naar de bodem;
ii) de N vervangingswaarde van mest-N, $NFRV_{manure}$, en
iii) de gewasopbrengst uitgedrukt in N en droge stof in kg ha^{-1}.

Uit de analyse volgde dat de N benutting uit mest toegenomen is in de loop van het onderzoek. Dit kan echter niet overtuigend worden toegeschreven aan afzonderlijke maatregelen. De '$NFRV_{manure}$', geschat als gemiddelde voor het hele bedrijf over de periode 1994-2010, was 0,70. Dit wijst op een nogal hoge benutting van N uit mest door gewassen. De schattingen voor blijvend (0,71) en tijdelijk grasland (0,63) komen vrij goed overeen met de bandbreedte die te verwachten is op basis van veldproeven uit de literatuur (0,47-0,79). De schatting voor maïs (0,97) was hoog vergeleken met de resultaten van veldproeven (0,12-0,75). Het onderzoek geeft aan dat een substantieel deel van de kunstmest-N dat werd gebruikt op grasland in de periode van 1994-2004 weggelaten kan worden door de aanpassingen in het mestbeheer. Echter, de kennelijke toename van de ‘$NFRV_{manure}$’ is niet voldoende om kunstmest-N volledig overbodig te maken. Dit verklaart de afname van de opbrengsten van N en droge stof in grasland sinds 2004, het jaar waarin het gebruik van kunstmest-N werd beëindigd.

P-EVENWICHTSBEMESTING

Dit onderzoek ging over de effecten van P-evenwichtsbemesting op de fosfaattoestand van de bodem, de gewasopbrengst en de uitspoeling van P. Het onderzoek liet zien dat P-evenwichtsbemesting, zowel op de korte als op de lange termijn goed te verenigen is met efficiënte gewasproductie (Hoofdstuk 5). Toen ‘De Marke’ werd gesticht, verschilde de fosfaattoestand, uitgedrukt als P, P-Al en Pw, sterk in verschillende blokken (dit zijn onderdelen van percelen van ongeveer 1 ha groot). Sinds 1989 is bemest volgens het schema van P-evenwichtsbemesting. In blokken met de laagste initiële waardes bleef de fosfaattoestand gelijk, terwijl de fosfaattoestand in blokken met een hoge initiële waarde eerst afnam, waarna de toestand leek te stabiliseren op lagere niveaus. De gewasopbrengst van P en droge stof was in percelen met hoge en lage fosfaattoestand gelijk en er waren geen indicaties dat P de belangrijkste opbrengstbeperkende factor was. Op percelen met een hoog initieel fosfaatgehalte verplaatste een deel van de P zich naar diepere lagen, maar het bereikte het grondwater niet.

DE DYNAMIEK VAN ORGANISCHE STOF IN DE BODEM

Het organisch stofgehalte (SOM) in de bovenste laag van ‘De Marke’ nam gemiddeld over het bedrijf af met ongeveer 0,03 procent jr^{-1}. Gedurende de onderzoeksperiode van 20 jaar vertraagde de afnamesnelheid niet aantoonbaar. De afname vond plaats in de bodem die in gebruik was onder vruchtwisseling, terwijl SOM min of meer constant bleef in blijvend grasland. De ontwikkeling van SOM is het gevolg van: (*i*) de jaarlijkse aanvoersnelheid (kg ha^{-1} jr^{-1}) van organische stof naar de bodem, en (*ii*) de afbraaksnelheid (kg $ha^{-1}jr^{-1}$) van de organische stof in de bodem. In dit onderzoek werd geen significant verschil gevonden tussen de afbraaksnelheid van organische stof onder blijvend grasland en onder vruchtwisseling. Dit suggereert dat de mechanische bodembewerking de afbraak niet versnelt en dus dat de jaarlijkse aanvoer van organische stof (lager in de vruchtwisselingspercelen) de oorzaak is van de afname van SOM in de vruchtwisselingspercelen. Alternatieve beheerssystemen, die resulteren in een hogere organische aanvoer dragen dus bij aan de stabilisatie van de SOM gehalten. Een analyse van de gevolgen van diverse alternatieve beheerssystemen liet zien dat conventionele (milde) anaerobe vergisting nauwelijks effect heeft op toekomstige SOM gehalten. Sterke vergisting zet SOM echter onder druk. Dit komt doordat bij sterke vergisting duidelijk minder organische stof uit mest naar de bodem wordt aangevoerd, terwijl milde vergisting een veel geringer effect heeft op de organische stofaanvoer naar de bodem. Toepassing van een vanggewas heeft nauwelijks invloed op de dynamiek van SOM.

Simulaties over een termijn van 50 jaar laten zien dat SOM nog verder zal afnemen, ook in blijvend grasland. Dit laatste lijkt in strijd met het stabiele SOM gehalte in blijvend grasland in de periode van 1989-2010. De te verwachten afname kan echter worden verklaard door geleidelijke vervanging van persistente componenten van SOM afkomstig van heide die bij de ontginning werd ingewerkt door minder persistente, sneller afbreekbare SOM componenten afkomstig van de actuele landbouwgewassen en mest. Deze vervanging veroorzaakt per saldo een versnelling van de afbraaksnelheid van SOM in de loop van de tijd. Maatregelen die deze trend zouden kunnen ombuigen lijken strijdig met efficiënt nutriëntengebruik. Anderzijds, ook een verdere afname van SOM kan efficiënt nutriëntenbeheer bemoeilijken, doordat de bodem door afname van SOM gevoeliger wordt voor nitraatuitspoeling. Mogelijk moet een verdere afname van SOM geaccepteerd worden als een autonome ontwikkeling. Met behulp van modellen kunnen de mogelijkheden verkend worden om bij lagere SOM gehalten toch nog een efficiënte gewasproductie te realiseren. De implementatie van sterke vergisting van mest verdient heroverweging omdat deze maatregel de organische stof aanvoer aanzienlijk beperkt.

ALGEMENE DISCUSSIE

De N en P-efficiënties op 'De Marke' namen voor het gehele bedrijf en voor het bedrijfsonderdeel bodem toe tot niveaus die hoger waren dan in de periode 1993-1999. De hoogste N en P-efficiënties werden bereikt toen het bedrijfssysteem over was gegaan op een verdere beperking van de beweiding, mestvergisting, aanwending van drijfmest in de rij in maïs, aanpassing van de overgang van de akkerbouwfase naar de graslandfase en een drastische vermindering van het kunstmest-N gebruik. Het bleek niet mogelijk om deze ontwikkeling toe te schrijven aan afzonderlijke maatregelen.

De N-efficiëntie op bedrijfsniveau bedroeg 35% in 2000-2003, 43% in 2004-2008 en 37% in 2009-2010. De P-efficiëntie op bedrijfsniveau bedroeg in deze periodes respectievelijk 103, 91 en 69%. De efficiënties van N en P op bedrijfsniveau lopen min of meer parallel met die van de bodem. De N-efficiëntie in de bodem nam toe van 65 (1993-1999) tot 81% (2004-2008) en de P-efficiëntie van 94 (1993-1999) tot

115% (2004-2008). Een terugval van de efficiënties van N en P op het gehele bedrijf en in de bodem in 2009-2010 werd veroorzaakt door een vermindering van de N en P opbrengsten van gewassen in 2009-2010. Dit laatste is op zijn beurt te verklaren als een uitgesteld effect van de krappe N bemesting door het weglaten van kunstmest-N sinds 2004. Het laagste N-overschot op de bodembalans werd gerealiseerd in 2004-2008, maar ondanks het relatief lage overschot (58 kg ha^{-1}) was de nitraatconcentratie in het grondwater in deze periode met 52 mg l^{-1} niet lager dan de norm. Dat was wel het geval in 2009-2010.

De afname van de N opbrengst in grasland sinds 2004 is waarschijnlijk veroorzaakt door een afnemende N bemesting. Het resulteerde in een N tekort in grasland met als gevolg een lagere verteerbaarheid van kuilgras. Om de melkproductie op peil te houden, moest meer N geïmporteerd worden met krachtvoer. Aanvoer van P met voer kan beperkt worden door krachtvoer te kiezen met een laag P gehalte. Voortzetting van de bedrijfsvoering met een krappe N voorziening in gras kan resulteren in een bedrijfssysteem dat functioneert met hoge N en P-efficiënties, maar dat minder in de eigen voerbehoefte kan voorzien. Op lange termijn is het niet te verwachten dat P een belangrijke opbrengst-beperkende factor wordt. Echter, de afname van SOM lijkt door te gaan, wat de mogelijkheid van efficiënte gewasproductie op den duur wel kan bemoeilijken.

Op grond van deze waarnemingen en analyses kan het huidige bedrijfssysteem met de krappe N voorziening verder ontwikkeld worden. Hierdoor kunnen nieuwe manieren ontdekt worden voor melkveehouders die zoeken naar de mogelijkheid om de nutriëntengebruiksefficiëntie op hun bedrijf veilig te verhogen. De volgende onderwerpen verdienen daarbij bijzondere aandacht:

1. verdere verhoging van de benutting van de mest-N door het gewas en het verder ontwikkelen van een betrouwbare schattingsmethode van de stikstofvervangingswaarde van mest, de ‘$NFRV_{manure}$’;
2. verhoging van de verteerbaarheid van ruwvoer.

PRAKTISCHE BETEKENIS

De volgende inzichten die gebaseerd zijn op het onderzoek beschreven in dit proefschrift kunnen van praktische betekenis zijn voor de melkveehouderij:

1. Het is raadzaam om het gebruik van vanggewassen die als onderzaai tussen de maïsrijen kunnen worden toegepast, te bevorderen, om N verliezen te beperken. Vervolgens is het belangrijk om rekening te houden met de nalevering van N uit het vanggewas door de bemesting van N in het volgende maïsgewas te verlagen. De uitgespaarde N kan ten goede komen aan grasland, of aan het verminderen van het N-overschot.
2. Toepassing van een structurele vruchtwisseling heeft veel voordelen. Vruchtwisseling bevordert de beschikbaarheid van P in de bodem, resulteert in een evenredige verdeling van organische stof over het bedrijfsareaal en beperkt nitraatuitspoeling.
3. Voor een efficiënt gebruik van N en P is het beter om in eerste jaars maïs volgend op gras geen bemesting toe te passen.
4. Evaluatie van nitraatuitspoeling op basis van volledige vruchtwisselingen van gras en maïs geeft een beter beeld van de bijdrage van maïs aan de nitraatuitspoeling, dan evaluatie van de afzonderlijke gewassen. Daar zou bij monitoring en bij de interpretatie van meetgegevens rekening mee gehouden moeten worden.
5. Om P-verliezen uit landbouwgronden naar grotere diepten te beperken, is het effectief om P uit te mijnen op percelen met een hoge fosfaattoestand. De bespaarde P kan dan gebruikt worden op percelen met een lage fosfaattoestand.

CURRICULUM VITAE

Koos Verloop was born in 1967 in Edam, a small city in a dairy region, located some 30 km north of Amsterdam, The Netherlands. In 1976 Koos moved to Leerdam and attended high school in Culemborg from 1979 to 1985. In 1985 he moved to Wageningen to study 'Soil protection and soil quality management'. During his study he opted for specialisation in the working field of Soil Chemistry. Following graduation in 1991 he started his professional career as assistant secretary of the Technical soil protection committee (TCB). The task of the TCB is to provide the national government with recommendations on technical and scientific aspects of soil policy and soil protection. The working field of the TCB is rather elaborate and Koos, being a 'Wageninger' was asked to take care of issues concerning agriculture, one of which was mineral and manure management in agriculture. In this function he learned how scientific knowledge can contribute to policy making. In his position, mineral management was mainly approached from the perspective of regulation and standard setting. However, Koos became increasingly interested in explorations of the development and design of sustainable agricultural practices. As a result, his network started to shift from the 'soil pollution and soil protection world' to 'agronomy'. He expressed his preference to return to research and in 2001 he started as a researcher at the 'Agrosystem Research' unit of Plant Research International, Wageningen. In that position his main focus was on mineral management in dairy farming. He conducted research at experimental farm 'De Marke' and was strongly involved in system development on that farm. Furthermore he was involved in research in 'Cows and Opportunities', DAIRYMAN and Cantogether. He put a lot of effort in establishing connections between nutrient management research and agricultural schools, which resulted in long lasting collaborations.

PE&RC PhD Training Certificate

With the educational activities listed below the PhD candidate has complied with the educational requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)

Review of literature (5 ECTS)
- Significance of integrated , whole system research for resource management in intensive dairy farming systems

Writing of project proposal (3 ECTS)
- Nutrient management in intensive dairy farming systems on light sandy soil (2007)

Post-graduate courses (3 ECTS)
- PHLO Course applied statistics; WUR (2004)
- Uncertainty and sensitivity analysis for model development; Biometris, WUR (2007)

Laboratory training and working visits (1.2 ECTS)
- Mineral management in France; Institute l' Elevage Brittany, France (2005)

Competence strengthening / skills courses (4.5 ECTS)
- Projectmatig werken; Licht & Gadella (1995)
- Presentatietechnieken; Licht & Gadella (1996)
- Personal development seminar; Kok & deVos (1998)
- Techniques for writing and presenting a scientific paper; WGS (2007)
- Projectmanagement; VROM in house (2012)

PE&RC Annual meetings, seminars and the PE&RC weekend (1.8 ECTS)
- Seminar on the waterframework directive (2004)
- Seminar on open access science (2011)

Discussion groups / local seminars / other scientific meetings (7 ECTS)
- NVWV The Dutch society for grassland and forage science; member of the board from 2005-2008 (2004-)
- EGF Working group on grassland systems (2004-)
- Dairyman: general meetings and progress meetings on collaboration between Knowledge transfer Centres (2009-)
- European Grassland Federation (EGF) Working group on grazing (2009-)

International symposia, workshops and conferences (7.8 ECTS)
- Alice, Agricultural Land Improvement through in Europe, initiated by Environment Agency; UK (2007)
- Manure workshop; poster and oral presentation (2010)
- Kick off meeting Dairyman; oral presentation; the Netherlands (2010)
- Meeting of the VCM; poster presentation; Gent Flanders (2011)
- Dairyman meeting; oral presentation; Ireland (2011)

Lecturing / supervision of practical's/ tutorials; (3 ECTS)
- Nutrient equilibria on experimental dairy farm De Marke (2005-)
- Lectures held in the framework "Kansen voor koeien" (2009)
- Improved mineral management (2010)