Impressions of Interactions

Land as a Dynamic Result of Co-Production between Man and Nature

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Land as a Dynamic Result of Co-Production between Man and Nature

M.P.W. Sonneveld

Proefschrift

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“De bodemonderzoeker ... dient inzicht te hebben in de architectuur van het landschap. Niet slechts de ‘geschapen grond’ dient hij te kennen naar de werken des Scheppers of der scheppende krachten, doch ook de herscheppingen dient hij te zien.”

W.A.J. Oosting, 1936

“Where can wisdom be found? Where can we learn to understand?”

Job 28

Cover

The Friesian Woodlands study area as indicated on 1:50 000 scale topographic maps (1999 and 1897) and on the 1:50 000 scale soil map.

Voorwoord

Dit proefschrift is het resultaat van een vierjarig onderzoek wat verricht werd als onderdeel van het onderzoeksprogramma Agrinovim. Het multi-disciplinaire karakter van dit programma alsmede de internationale opzet heb ik als bijzonder inspirerend ervaren voor het uitvoeren van mijn onderzoek. Allereerst wil ik mijn beide promotoren bedanken voor de mogelijkheid die zij mij gaven om dit onderzoek te doen. Johan Bouma, bedankt voor je enthousiasmerende rol. Jouw brede wetenschappelijke kennis en maatschappelijke betrokkenheid heb ik als bijzonder stimulerend ervaren. Tom Veldkamp, bedankt voor de ruime inzichten die je met me deelde betreffende bodem- en landschapsprocessen. De leden van het Agrinovim-team wil ik bedanken voor de leerzame gezamenlijke bijeenkomsten. Jan Douwe van der Ploeg van Wageningen Universiteit, Arie Rip van de Universiteit Twente, Jaap van Bruchem van Wageningen Universiteit, Frits Rijkenberg van de Universiteit van Natal (Zuid-Afrika) en Flaminia Ventura van de Universiteit van Perugia (Italië) hebben ieder een eigen inhoudelijke bijdrage geleverd die mij verder gescherpt hebben in het denken over de landbouw. Han Wiskerke en Dirk Roep bedank ik beide voor hun inhoudelijke en organisatorische aandeel. Als AIO's van Agrinovim, verder bestaande uit Samantha Adey, Pierluigi Milone, Joan Reijs en Marian Stuiver, hadden we de bijzondere mogelijkheid om de pieken en dalen van het promotie-onderzoek met elkaar te delen. Me dunkt, de regelmatige werkweken in de afzonderlijke onderzoeksgebieden in Nederland, Italië en Zuid-Afrika zorgden voor een verrijking in de ruimste zin van het woord.

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Chapter 1

Introduction

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1.1 Background

This thesis is about relationships between people and land in agriculture. More specifically, it is about interactions between farmers and the land they use. But what do we mean when we talk about land? It involves a lot of things, which are somehow connected. Land is something that has different colours in different parts of the year because of seasonal variation, involving different stages of plant growth. It's something with a certain level of accessibility that is relevant for its use. It provides habitat to animals and receives invading seed from other pieces of land. Land has topography, which determines how easy it is to get from one part of the land to another part. Land may feel hard when you walk on it, may stick to machinery in wet seasons or may lead to serious injuries for the careless walker when there are many stones at its surface.

Obviously, land is more than soil. Land refers to all elements of the earth's terrestrial surface that affect potential land use and environmental management including not only soil but also landforms, climate, hydrology (surface- and groundwater), vegetation and fauna (FAO, 1976; Pieri *et al.*, 1995). The soil itself is defined as the unconsolidated material on the immediate surface of the earth that serves as a natural medium for the growth of plants (SSSA, 1987). After decades of modernization in many developed western countries, the status of the land is not highly appreciated. In fact, modern agriculture is said to have left dying land as its legacy (Pretty, 1998). Pollution of the environment and land degradation are, for example, signs that this type of agriculture is out of balance. New questions are raised and new answers are demanded to deal with this. The role of soil scientists in dealing with agro-environmental issues has also changed (Tinker, 1985), partly because of the problems that occur in agriculture. In western society, they are no longer in the central position to pave the way for agricultural development as they had done so in the past (Bouma, 2001).

If both the demands from the agricultural and environmental communities have changed as well as the position of soil scientists in society, a different perspective on both land and soils might also be considered. Much research has so far been carried out on the chemical and physical processes that occur in the soil system but less emphasis has been put on the interaction between soils and their socio-economic environment. Given the important role that the socio-economic system plays in determining the physical and chemical fate of soils, this is surprising (Dent *et al.*, 1996). According to Pretty (1995), agricultural scientists should take another look at agriculture, a wider perspective so to say on the interaction between people and their land. In the domains of soil science as well as other scientific disciplines, there are some valuable notions that can be taken up to express this perspective.

1.2 The position of farmers; a Dutch perspective

1.2.1 Trends in soil science

In the Netherlands, W.A.J. Oosting is considered to be one of the founding fathers of soil science. Studying geology, geomorphology, vegetation and soil forming processes in their mutual coherence was typical for his approach (Oosting, 1936). He realized that landscape structures human activities and developments and that, in turn, man moulds and shapes the land to fulfill his needs. Therefore, to “understand the state of the soil, one needs to know its past” with all the natural processes and human influences that have taken place. Oosting’s approach to study the soil in its landscape context was further developed by C.H. Edelman. In 1949 he published his book on ‘social and economic soil science’ in which he specifically addressed the social aspects of land (Edelman, 1949). Emerging examples of severe land degradation, for example documented by Jacks and Whyte (1939), showed that man was capable of destroying the land to the extent that agricultural activities were no longer viable. This was posing a threat to agricultural productivity, with both social and economic impacts. Edelman suggested that soil scientists were to use their knowledge of soil- and landscape processes to assist in adjusting land use practices in agriculture to the specific characteristics of the land and prevent such degradation.

To improve the efficiency of land use, rapid developments took place in the domains of soil classification and soil survey, which are both strongly connected to each other in The Netherlands (De Bakker, 1970; Schelling, 1970). In 1966, this resulted in the Dutch soil classification system (De Bakker and Schelling, 1966). By that time, it was already widely acknowledged that man could be considered as one of the soil-forming factors, following Jenny’s classical book on soil formation from 1941 (Jenny, 1941). Cultivation practices, fertilization and cropping had been shown to have a significant effect on soil properties. However, in the Dutch classification system, only considerable alterations of the soil profile were recognized, such as plaggen epipedons. The effects of different types of management over a period of decades within one soil series were not taken into account. So only more or less fixed properties were used as criteria to distinguish between different soil groups and subgroups. Other soil taxonomic systems such the international FAO system, the American Soil

Taxonomy (Soil Survey Staff, 1998) or the South African Bionomial System (Soil Classification Working Group, 1991) do not explicitly include effects of land use practices on soil properties either. At the time of the first publication of the Dutch soil classification system though, a need was felt to include this.

In the 1950s several regional soil survey studies were performed (e.g. Sonneveld, 1958) which included separate sections on land evaluation for agricultural productivity (Van Diepen, 1995). In its earliest stages, the empirical forms of land evaluation were referred to as “systematized farmers’ experience” because farmers’ knowledge was used to rank soil types according to several factors as e.g. trafficability and yield level. Moreover, farmers were still considered being essential to really appreciating the true value of the land. Experience and local knowledge of farmers were believed to be at the core of soil suitability classification. Soil suitability became defined as “the degree of success with which a crop or range of crops can be regularly grown on a certain soil, within the existing type of farming, under good management, and under good conditions of parcellation and accessibility”. The addition “within the existing type of farming” though not explicitly defined, was considered to be essential “as one would otherwise become too much removed from reality” (Vink, 1963). But, so much was clear, even among farmers that were considered to be good farmers because of good management, a large variation in yields could be observed because of know-how, experience and even luck.

Increasingly, emphasis was put on defining scientifically sound methods to characterize agricultural possibilities for Dutch soils. This development was further stimulated by the production of 1:50 000 scale soil maps at a nationwide scale level because an applicable system was needed at this level (Vink, 1963; Vink and Van Zuilen, 1974). Soil suitability systems were developed which included qualitative assessments of suitabilities and limitations for agricultural land use, mainly based on expert judgment and field trials (Diels *et al.*, 1996; Haans, 1979). The land evaluation procedure consisted basically of a matching exercise involving two steps (Bouma, 1989). The first step consisted of defining diagnostic factors, which can either be land characteristics or land qualities. The second step consisted of combining these factors to arrive at a specific suitability judgment given the requirements of a particular crop. The Dutch suitability classification system included, for example, for Dutch grasslands, moisture supply capacity, drainage status and trafficability in the first step yielding a total of 28 possible suitability classes, which have been included in many soil survey reports. At international level, the more or less similar FAO land evaluation framework from 1976 (FAO, 1976), introducing terms as ‘land use system’, ‘land unit’ and ‘land utilization type’, was widely accepted among scientists (Beek, 1978; Bouma, 1997a).

The 1980s showed a strong increase in the development and use of computer simulation models. It was observed that separate diagnostic functions were often not independent and that the inflexibility of the classical systems prohibited dynamic and more quantitative evaluations of land. The 1980s was also the decade that environmental pollution raised considerable concerns in society (e.g. Ryden *et al.*, 1984). Simulation models of hydrological processes in soils and crop uptake of solutes were therefore combined with algorithms on the behavior of chemicals to evaluate the impacts of land use and land management on the environment (e.g. Diels *et al.*, 1996; Hutson, 1996). Ideally, optimal application levels of fertilizers, herbicides and pesticides should be defined at the local level. The scope of soil science broadened from only considering production aspects towards also taking account of environmental impacts (e.g. Bouma, 1997a; Hack-ten Broeke *et al.*, 1993). The development of GIS technology stimulated the development of precision agriculture at field and farm level (e.g. Verhagen, 1997; Van Alphen, 2002) whereas at regional and country level predictive and exploratory approaches were adopted (e.g. Veldkamp and Fresco, 1996; Verburg *et al.* 2002). In addition, the expansion of land evaluation tools made clear that different spatial scale levels could be involved in the land evaluation process. Boundaries of soil systems could be defined at different spatial levels and there was a need for more explicit recognition of these different scale levels (Bouma *et al.*, 1998; Fresco and Kroonenberg, 1992; Wagenet, 1998).

1.2.2 Farmers' knowledge

In the history of soil science, the study of local people’s knowledge of their natural resources has received little attention. Only recently, the field of ethnopedology has emerged as a distinct field of research (Sillitoe, 1998; WinklerPrins and Sandor, 2003). Ethnopedology, the study of local soil knowledge, aims to document and understand the local approaches to soil perception, classification, appraisal, use and management (Barrera-Bassols and Zinck, 2003).

Although this thin history of the study of local soil knowledge is in strong contrast with the history of soil classification, local knowledge is still at the root of contemporary scientific classification systems. In the 19th century, the Russian school of Dokuchaev made use of folk names for soils such as *Podzol* and *Solonetz* (Yaalon, 1997), names that are still widely used in the realms of soil science. Also in The Netherlands, more than half a century later, the Dutch soil classification system made extensive use of local names (Siderius and De Bakker, 2003). The soil names that were introduced at the lowest level of the classification system, formed a combination of scientific names with the name of the locality where the specific soil was well represented. Soil conditions but also land use and reclamation history strongly influenced these local fieldnames (Schönfeld, 1950). On the farm, the naming of land was similar to the custom of naming cattle, as the farmer knew his plots in great detail (Siderius and De Bakker, 2003). However, the process of modernization in the Netherlands, including cadastral mapping and land consolidation to be able to farm bigger fields, has led to a substantial loss of local soil knowledge. It has now been largely replaced by scientific soil knowledge, basically a more universal system of soil classification (WinklerPrins and Sandor, 2003).

But does it matter when the local name of a soil or land unit is replaced by a different name, one that fits into a larger system? Both local people and scientists see the same soil, one could argue. But, as Sillitoe points out, while we see the same soil "out there", we may think about it in quite different ways (Sillitoe, 1998). The perception of the farmer of soils and soil behaviour is different from that of the researcher (Bouma, 1993; Garlynd *et al.*, 1994; Harris and Bezdicek, 1994). Given local names are often expressions of a more holistic approach of local farmers compared to scientists. This approach has been described as "art de la localité" (Van der Ploeg, 1991), or "art of the specific". It involves a degree of craftsmanship, the ability to combine the specific elements of a farm such as animals, soils, crops and technology into a "working whole" (Roep, 2001)[1]. Local knowledge forms in essence the vehicle to integrate and coordinate the various elements that exist and farmer's labor acts here as the linking agent, coordinating the various farm components and balancing them in relation to each other (Van der Ploeg, 1991). Agricultural enterprises are in reality characterized by a unique integration of natural phenomena and human activities and are transformed into a working agro-ecosystem. Local people may think of a soil, not so much as 'something out there' but more of 'something inside'. Mendras (1970) reported that "the farmer felt as if he 'made' his field and knew it as the creator knows his creation, since the soil was the product of his constant care; plowing, fertilizing, rotating crops, maintenance of fallow ground and so on". At farm level, the assemblage of fields becomes a properly working whole, namely the farm, because of the decisions and activities of the farmer. By his selection of fields for various types of land use and by his 'freedom' to apply different types of management at different fields and through time, the farm becomes a unique configuration of characteristic land units and land use. In its specificity, the combination of social, material and natural elements and the interrelationships between them, constitutes a farming style (Van der Ploeg, 1999). It appears as the expression of a coherent set of strategic notions about the way in which farming should be practised. This

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[1] It is fascinating to see that soil scientists also regarded exactly those farmers who combined a high degree of craftsmanship with an extraordinary feeling to manage their farm as "artists" (e.g. Vink, 1958).

also implies that, for local people, soil knowledge forms an element of a wider knowledge domain. It is contextual, locally embedded within a cultural repertoire.

1.2.3 Reappraisal of disciplinary roots

In the 1990s, interaction with stakeholders, and in the first place farmers, regained new attention (Bouma, 1997b; Dent *et al.*, 1996; Hurni, 1998). It had become apparent that the whole classical system of land evaluation had become pedocentric, supply-oriented, top-down and virtually irrelevant (Bouma, 1998; Dumanski, 1997; Sillitoe, 1998). Evidence for this did not only come from within the discipline itself. There were also influences from other publications as e.g. Chambers *et al.* (1989) and Scoones and Thompson (1994) which criticized the Transfer-of-Technology (TOT) model, with scientists deciding research priorities, generating technology and passing it to extension agents to transfer to farmers.

Despite earlier reservations on the procedure of land evaluation (e.g. Beek, 1978), the impression had arisen that it merely consisted of the preparation of ‘cook book recipes’ on how to use the land, leaving the role of the individual farmer to begin where the land evaluation procedure stopped. In its most extreme form, the role of farmers would be narrowed to that of industrial managers with computers taking over all management aspects of farming. Though this can be described as the most advanced and scientific approach to farming (Mermut and Eswaran, 2001), it is strongly contested whether this still captures the essence of farming (e.g. Frouws and Van der Ploeg, 1988). It points to the scientification of agriculture; the on-going reorganization of management following a scientific design (Van der Ploeg, 1987).

Often, the implicit idea behind the land evaluation procedure can be regarded as, what used to be called, a ‘two-stage’ approach (FAO, 1976). A physical land evaluation procedure is first executed, predicting the performance of specific land use systems as conditioned by the constraining influence of physical land conditions (Beek, 1978) and this is followed by a socio-economic analysis. However, land as such is not a static physical natural resource. In the past, it was known that farmers can and do adapt soils to their needs (Oosting, 1936; Vink, 1959). Famous are the Anthrosols such as the Plaggen soils for Western Europe (Conry, 1974; Edelman, 1952; Pape, 1970) and the Black Earth soils in South America’s Amazone (German, 2003) but many more examples exist (De Visser, 1958). Also interactions through land use in a ‘narrower sense’ regarding soil structure, soil fertility or flow patterns, were known to exist and were described (e.g. Bouma, 1969). With descriptions of soil systems, emphasis was put on various properties since it was known that one property does often not change irrespective of other properties. For example, increasing the organic matter content of the soil affects the soil structure and often flow patterns too. In the Netherlands, this coherent pattern of soil properties used to be called the *correlative complex*. But, in general, these changes have later often been ignored in land evaluation, mostly for purposes of simplification (Beek, 1978).

In short, it appears that something has been lost in time which made up one of the key-elements of soil science in its early stages: the broad relationship between people and their land which expressed itself in knowledge, including an experimental capacity, and a specific configuration of land- and soil properties.

1.3 Co-production as a new framework for research

To better understand relationships between farmers and their natural environment, I will introduce the term co-production. Co-production is a concept coming from the field of rural sociology (Gerritsen, 2002; Renting and Van der Ploeg, 2001; Roep, 2001; Van der Ploeg, 1999). In a general sense, it has been regarded as the on-going interaction between and mutual transformation of farmers and living nature. Co-production influences the characteristics of farming and natural resource management and of living nature (Roep, 2001). More specifically, 'land' is both the result of an interaction between natural processes with land use practices and it also influences future land use decisions and biophysical processes.

In contrast with other components of agriculture such as technology, crops and animals that can also be considered from a co-production perspective (Van der Ploeg, 1999), land is evidently non-transferable. It is at the roots of the locality, being either field or farm or regional level. It influences farming, a.o. the specificities of technology, crops and management practices, and is at the same time influenced by it. Specific landscapes, roughly defined as the outward appearance of land, can for example be regarded as outcomes of co-production (Faber *et al.*, 2000; Hendriks and Stobbelaar, 2003; Van der Ploeg, 1999), as results of a continuing encounter between and mutual transformation of man and nature. Both land and landscape are not merely the physical backdrop of human activities, but they involve a whole set of complex connections (Cheng *et al.*, 2003; Hendriks and Stobbelaar, 2003). Especially a lot of ancient cultures did not see the land as solely a capital or a means of production. In many cases, spiritual values were attached to the land and a deep sense of connection was felt and expressed (see e.g. Hillel, 1991; Yaalon, 2000).

People can be connected to the landscape in which they grew up – it can provide identity and a feeling of belonging (Van der Ploeg, 1999). Röling and Maarleveld (1999) refer to this as the ‘soft side’ of land, involving terms as organisation, religious beliefs, cultural practices and so on. As such, these specific outcomes reflect past interactions between people and land. But in the future, they will influence both the decisions that are taken in society with respect to the further development of these landscapes as well as the hydrological, nutrient and matter fluxes that occur.

But also in a more restricted sense, co-production is a part of agriculture. Land use practices influence land properties and these changing properties also have their influence on the knowledge and behaviour of land users. Land use is not simply a set of technical operations and artefacts. Land use is both an emergent property of the society that lives off it (Röling and Maarleveld, 1999) and of the land itself. Initially, the land will limit land use because some processes work on a geological time scale, but not to the extent as to eliminate all human creativity and ingenuity (WinklerPrins and Sandor, 2003). The ‘hard’ terms in which land and land use have been usually thought of are not so rigid at all. There is space to diversify or, more poetically expressed, to *unfold*. It is possible to create specific expressions of the land and the soil. One farmer, again cited by Mendras (1970), puts it this way: “To know one’s land, to improve it, takes a long time! The more you know it, the more you become attached to it”.

Izac and Swift (1994) regarded the unfolding material outcome of the soil as a *by-product*, apart from the general variety of *products* of agriculture, such as animals, crops, fruits and medicines. This conceptualisation however does not stress enough that such a by-product is also re-used within the farming system. Specific soils are the output and at the same time the input for agriculture. In (semi) closed farming systems *all* products will become input within the farming system through e.g. breeding with animals or producing seeds with plants. In other words, there will be continuous production and reproduction. Droogers and Bouma (1997) suggested to define genophorms, the taxonomically defined soil series, and phenophorms, the results of different types of management or land use, as operational concepts. These authors basically showed that similar soils could follow different land use trajectories leading to relevant results from an environmental point of view. In the follow-up survey conducted by Pulleman *et al.* (2000) attention was paid to a whole range of land use trajectories that exist in practice. In combination with more theoretical considerations by Van der Ploeg (1999) and Roep (2001), these findings suggest that co-production is promising as a new framework for research.

1.4 Contents of the thesis

1.4.1 Main objective

The main objective of this thesis is to describe soil systems in two case study areas along the lines of co-production. This regards the resulting outcomes, their development as well as the conditions under which these can be fitted into the existing land use practices given the present environmental objectives. The research has been executed in two case study areas. One is located in the northern Friesian Woodlands in Friesland, The Netherlands and the other one is located in the Okhombe catchment which is part of the Upper Tugela catchment in KwaZulu-Natal, South Africa.

1.4.2 Case study: Friesian Woodlands

Veenenbos (1949) probably wrote one of the earliest publications on soil survey in the Friesian Woodlands. His study was performed on request of the Dutch Ministry of Agriculture and dealt with the ‘problem of intensification of small farms’. The soil map, which would be made on the basis of a detailed soil- and landscape survey, should indicate which soils would be suitable as arable land, as grassland and as rotational land. A landscape description and a soil map were produced later as final products, yielding insights in the structure of the landscape and the distribution of soils (Veenenbos, 1954; Veenenbos, 1964). After the survey of Veenenbos (1949), Van der Schans and Vleeshouwer (1956) also performed a specific soil survey as a basis for improving the hydrology of the northern part (Achtkarspelen). Additionally, they also provided information on the suitability of the mapped units for grassland. Because the descriptions of Veenenbos were actually mainly related to the southern parts of the Friesian Woodlands, Cnossen and Heijink (1958) made a more detailed description of the northern part. For this area, the soil survey of 1972–1978 resulted in the 1:50 000 scale soil map and additional reports (StiBoKa, 1981). More detailed surveys in 1986 (Kiestra and Rutten, 1986) and 1991 (Makken, 1991) were performed for other specific purposes.

Currently, dairy farming is the dominant type of land use in the area. This agricultural sector has received an increasing pressure from society to reduce nutrient losses to the environment. Especially nitrogen (N) losses to atmosphere, ground- and surface water have received considerable attention. Therefore, there is a strong need to develop more sustainable dairy farming systems. Within the Friesian Woodlands, a mineral project called the VEL & VANLA project was initiated in 1998 in which researchers from Wageningen University and almost 60 farmers participated. This was extended with a 3-year follow-up project in 2001. The research in the Friesian Woodlands described in this thesis has been executed within the context of this mineral project.

1.4.3 Case study: Okhombe catchment

The Okhombe catchment belongs to the Tugela Basin that covers a large part of the present KwaZulu-Natal province. In 1951, the Natal Town and Regional Planning Commission began to devote special attention to the Tugela area as it was considered to have a great potential for development. Among various other studies and surveys, also a soil survey was planned for the entire catchment. Four years later, the leading soil scientist from Natal consulted prof. C.H. Edelman in the Netherlands because his ideas and techniques were considered very useful for undertaking the survey. In 1969, the survey was completed yielding soil maps with a scale of 1:100 000 and an extensive survey report (Van der Eyk *et al.*, 1969). After the soil survey had been completed, it was believed that the prospect for “applying the ink of scientific planning to a nearly blank page” was an exciting challenge. The Tugela Basin soil classification system was met with considerable success as it gave planners, soil scientists and agronomists a good understanding of the soils in the area (Macvicar, 1978). This finally led to the construction of the Binomial System, which is still in use today on a nationwide scale level (Soil Classification Working Group, 1991).

Nowadays, signs of land degradation are common in the communal areas of the Tugela catchment and increasing emphasis is put on sustainable natural resource management. Within the Okhombe catchment, a three-year Landcare project was launched in 1999. Landcare is a community-based and government supported approach to the sustainable management and use of agricultural resources. The research in the Okhombe catchment as described in this thesis has been executed within the context of this Landcare project.

1.4.4 Outline

Most part of this thesis will be related to the case study area of the Friesian Woodlands since most of the research that has been performed was related to this area. Chapter 2 introduces the study area in the Friesian Woodlands and describes for a dominant soil series in the area how soil properties are related to land use history. Chapter 3 describes how for the same soil series, also the soil water regime is affected by land use history. In chapter 4, a model simulation study is presented in which the effects of different levels of N application on nitrate leaching are investigated. In chapter 5, attention is paid to institutional issues in terms of N policies in The Netherlands and the role for research in meeting groundwater quality objectives set out for dairy farming. In chapter 6, the study area in the Okhombe catchment is introduced. In this chapter the dynamics and drivers of land degradation in communal grazing land are investigated. Finally, chapter 7 will contain a synthesis of the various issues that have been raised in the previous chapters. From the material presented in chapters 1 – 6, key elements of a co-production framework are derived. In this chapter, suggestions for a future perspective for soil science will be presented.

All chapters can be read separately. The chapters 2, 3, 4 and 5 are based on papers that have already been published in scientific journals whereas chapter 6 is based on a paper that has been submitted to a scientific journal. Literature references are combined and included in the *references* section at the end of the thesis.

Chapter 2

Refining soil survey information for a Dutch soil series using land use history

Abstract

Differences in land-use history within soil series, although not influencing soil classification, lead to variability of non-diagnostic soil properties in soil databases. Regional studies that use soil databases are confronted with this considerable variability. This has, for example, been reported in regional studies focused on nitrate leaching from agricultural land. Such findings have a direct impact on regional assessments of nitrate leaching from dairy farms on sandy soils, a major environmental issue in the Netherlands. There is thus a need to deal with this variability in soil properties.

We were able to relate soil organic nitrogen, soil organic carbon and its dynamics to land use history for a Dutch sandy soil series. Within one soil series, three different land use histories were identified: old grassland, reseeded grassland and grassland converted from continuous cropping with silage maize. The addition of landscape characteristics significantly improved the regression models based on land use only. Once established for any given soil series, such relationships can significantly improve soil survey input into dynamic models of soil behaviour such as regional nitrate leaching studies.

Based on: Sonneveld, M.P.W., Bouma, J. and Veldkamp, A., 2002.
Refining soil survey information for a Dutch soil series using land use history.
Soil Use and Management, 18: 157-163.

2.1 Introduction

Soil maps and soil survey reports generally interpret map units in terms of soil suitability and its limitations for certain land uses (Bouma *et al.*, 1996; Diels *et al.*, 1996). However, this interpretation may not adequately describe the dynamic relations between soil and land use. It is known that identical taxonomic units may behave differently in response to different land management (Bouma, 1994). The Dutch system of soil classification (De Bakker and Schelling, 1966), which served as a basis for soil survey (De Bakker, 1970), deliberately left information concerning management effects out of the classification system, though a need was expressed to include this. To fill this gap, Droogers and Bouma (1997) defined and described phenoforms resulting from different types of soil management within one soil series, which they called the genoform. Pulleman *et al*. (2000) more specifically used functions to relate soil organic matter contents of soils belonging to the same soil series with land use history.

The search for more sustainable agricultural systems may profit from this refined soil survey information (Droogers and Bouma, 1997). For example that described by Aarts *et al*. (1999) for Dutch dairy farming on sandy soils because of concerns of nitrate leaching to the groundwater. Most of the sandy soils in the Netherlands are used as pasture for dairy cattle but may have differences in land use history due to changes in the dairy farming sector over the last decades. The total area for growing silage maize (*Zea mays* L.) in the Netherlands increased from negligible before the 1970s to 10.5% of the total agricultural land in 2000 (CBS 2000a). This crop is often grown continuously. Conversions of these fields to pasture do occur however especially because of long term adverse impacts on the soil (Vellinga *et al.*, 2000).

Soil tillage in dairy farming is not only restricted to the conversion of pasture to arable cropping. A proportion of pasture is regularly ploughed, generally to a depth of 20–25 cm, and reseeded, mainly to improve the botanical composition of the grass sward. In 1999, this area was estimated to be 8% of the total pasture area (CBS 2000b). The ploughing out of old pastures may contribute significantly to nitrate leaching (Whitmore *et al.*, 1992) and is, also because of other environmental issues, discouraged (Vellinga *et al.*, 2000).

So far, most literature that deals with the effects of these land use dynamics on soil organic matter and nitrate leaching processes do mention soil survey information in their study area description (e.g. Sparling *et al.*, 1992, Haynes and Tregurtha, 1999; Shepherd *et al.*, 2001) but the implications for future use of that same soil survey information are hardly ever given. A discussion on these implications is considered to be relevant as more and more of the standard (digital) soil survey information is being used for regional environmental assessments. Hansen *et al.* (1999) and also Finke *et al.* (1996) however noted already that this use can lead to great uncertainties in model output. They found this was mainly due to the variability in soil properties either present in soil databases or derived from other sources. Some recent nitrate leaching studies have simply ignored this variability under the assumption of ‘homogeneous soil units’.

This chapter addresses the issue of how the variability in soil properties within soil databases can be related to spatially explicit land use histories for the Netherlands. The objective is to determine whether effects of differences in land use history can be found in a Dutch sandy soil series under grassland and to evaluate the consequences for future use in regional nitrate leaching studies. Attention will be paid to soil organic carbon and soil organic nitrogen because of their importance in grassland soils (Hoogerkamp, 1984). As differences in land use are often expressed in soil organic matter decomposition rates (Jenkinson, 1988), soil respiration was also studied.

2.2 Materials and methods

2.2.1 Study area

The study area (Figure 2.1) is located in the north of the Netherlands, in the province of Friesland, and covers some 140 km^2. Current land use is mainly grassland (80%), arable land (about 4%) and 16% is covered by buildings. The cHn23 soil series was selected for this study as it is the dominant soil series, covering more than 40% of the area (1:50 000 scale soil map, StiBoKa, 1981). These soils are classified as coarse loamy, siliceous, mesic Plagganthreptic Alorthods (Soil Survey Staff, 1998) and are excellent soils for agricultural purposes. The soil database (De Vries, 1999) indicates that the organic matter content of this soil series under grassland varies from 3% to 10%.

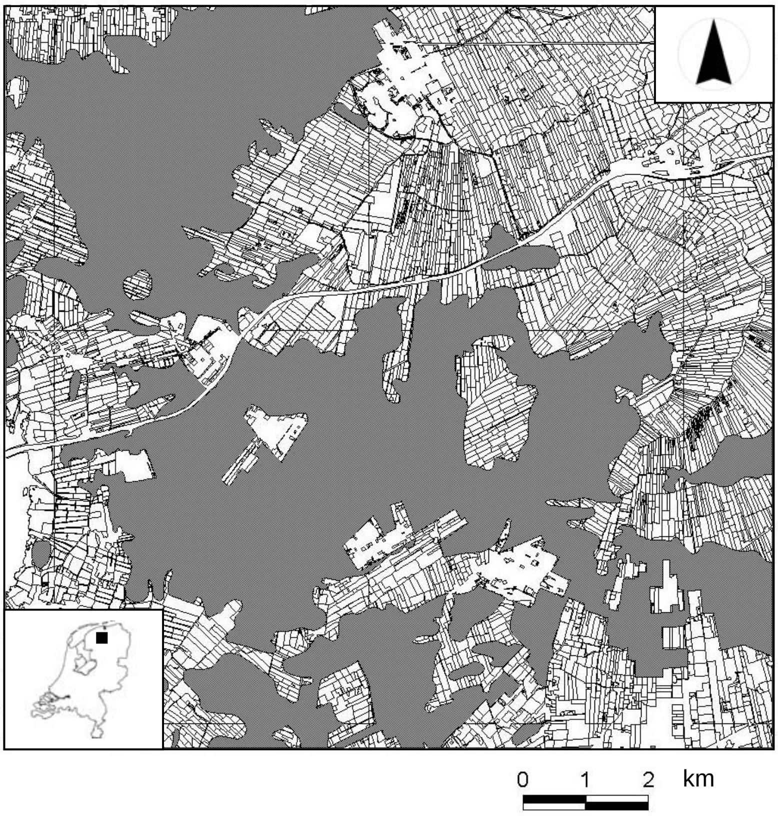

Figure 2.1. Location of the study area in The Netherlands (53:16°N; 6:05°E) and the distribution of the cHn23 soil series.

Most of the soils belonging to the cHn23 soil series in the higher western part of the study area have formed through the build-up of a humose topsoil in a mixed farming system often over centuries (Pape, 1970). Manure was mixed with litter in stables and spread on the fields to improve the fertility of the originally poor sandy soils. The dark topsoil is often quite homogeneous and fragments of pottery and pieces of charcoal are common in these soils. The system was abandoned in the beginning of the twentieth century.

The area has been much wetter in the past but because of drainage and the construction of dykes the area is at present considerably drier. Because of these formerly wet conditions, (high moor) peat developed in the lower eastern part of the area. The peat was dug out for fuel in the 17th and 18th centuries, after which the land was reclaimed for agriculture by mixing the sand from reclamation canals with peat remains. Both types of genesis, old plaggen soils in the western part and peat soils in the eastern part, were described and mapped for the study area (Van der Schans and Vleeshouwer, 1956). Apart from the higher ridges in the landscape, the topsoil often consists of a mixture of old coversand and till sand. Till or subglacial lacustrine clays (Van den Berg and Beets, 1986), starting within a depth of 40 and 120 cm below the surface and with a thickness of at least 20 cm, has been mapped for 36% of the area classified as cHn23.

2.2.2 Field selection

A 1:25 000 scale soil map covered the eastern part of the area (also available in GIS format) and a 1:10 000 scale soil map the western part. Groundwater regimes in the area were mapped into specific classes (Van der Sluijs and De Gruijter, 1985), related to landscape morphology, the occurrence of till and type of vegetation. Groundwater table measurements were used to verify these class estimations. It appeared that 93% of the cHn23 soils fell into two main classes. These classes, Vb and VI, have a mean lowest groundwater table deeper than 120 cm below surface while the mean highest groundwater table lies between 25 to 40 cm or between 40 to 80 cm below surface, respectively.

Twelve farms were selected on the basis of occurrence of the cHn23 soil series and particular groundwater-class Vb or VI. One field on each farm was identified as suitable for sampling using an overlay of the soil map. Specific land use histories for the last 50 years were obtained from the farmer and were grouped into three separate classes: 1) *continuous silage maize* for at least 10 years with a conversion to grassland within the last 5 years, 2) *reseeded grassland,* where the land had been tilled at least once during the last 10 years; and 3) *old grassland*, where no tillage had occurred for the last 50 years (Whitmore *et al.*, 1992).

2.2.3 Data collection

In spring 2000, each field was divided into four quadrants and from each quadrant, samples were taken. A flowchart was constructed to check the criteria for the cHn23 series (Figure 2.2). When the profile characteristics were confirmed, samples were taken from 3–5 points per quadrant subsequently bulked from two layers, 10–20 cm (topsoil) and from the layer starting at 25 cm to the lower boundary of the A-horizon (subsoil). From each field, 2 to 4 subsamples from the same depths were taken for respiration analyses.

Bulk samples were air-dried (40°C), sieved (<2mm) and some basic soil properties were determined: pH-$_{KCl}$ (1M); particle size fractions (Laser Diffraction Technique using a Coulter LS230 apparatus); mineral N extractable with 1M KCl; total soil N; organic N by difference; total soil C.

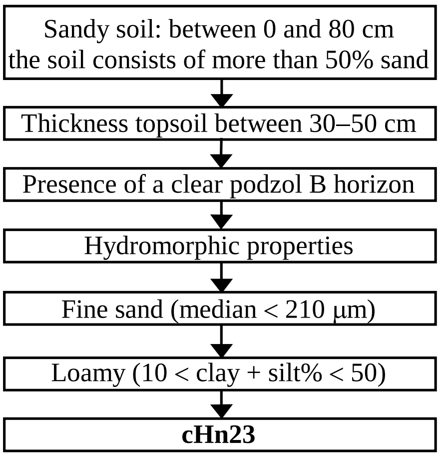

Figure 2.2. Flowchart to determine the selected soil series: cHn23. Hydromorphic properties were assumed from the groundwater table fluctuations.

Respiration was measured on 20 g sub-samples, wetted to a gravimetric water content of 30%, and incubated at a constant temperature of 20°C. Continuous CO_2 production rates (mg hr^{-1}) were measured indirectly by recording the conductivity in a 10 ml 0.1 M KOH solution according to Nordgren (1988). KOH solution was replaced regularly because of saturation. Recordings were first made at one- and later at three-hour intervals over a period of 33 days. Data were corrected for disturbances and day/night cycles and were averaged on a daily basis. Average field CO_2 production rates were then related to the calculated average original organic carbon contents of the samples. To relate these measurements with land use history, statistical analyses were performed with SPSS (SPSS, 1998).

2.3 Results and discussion

2.3.1 Land use characteristics

The selected dairy farms are distributed across the entire research area. Each farm covers on average about 50 ha. The selected fields are on average about 2 ha in size. About half of these fields were used as arable land around 1900. Their land use history for the last 50 years as well as some field characteristics are presented in Table 2.1.

Table 2.1. Topsoil characteristics of the selected fields fulfilling the requirements for the cHn23 soil series.

Field	L.U.H.[a]	Age[b]	Long Term Cultivation History	G.W.C.[c]	pH (KCl)	Texture (% particles <)	
		Years				2 µm	50 µm
5	Maize	3 (14)	Plaggen cover	Vb	5.3	5.3	22.8
47	Maize	3 (12)	Plaggen cover	Vb/VI	5.3	3.8	18.8
12	New	2	Plaggen cover	VI	5.0	6.0	31.7
30	New	1	Reclaimed peat	VI	4.9	5.9	27.2
38	New	10	Plaggen cover	VI	4.8	4.2	24.0
41	New	4	Plaggen cover	Vb/VI	4.8	3.9	17.9
50	New	10	Plaggen cover	VI	4.6	5.3	30.5
10	Old	50	Reclaimed peat	Vb	4.9	6.1	28.1
22	Old	50	Plaggen cover	VI	4.9	5.8	27.3
37	Old	50	Reclaimed peat	Vb	4.8	4.4	20.7
40	Old	50	Plaggen cover	Vb	4.7	4.1	19.9
56	Old	50	Plaggen cover	Vb	4.1	5.2	26.8

[a] L.U.H. = Land Use History. Continuous maize cropping is indicated by *Maize*, reseeded grasslands are indicated by *New* and old grasslands are indicated by *Old*.
[b] Grassland age. For *Maize*, the length of the cropping period is given in brackets. For reseeded grassland, the number of years since tillage and reseeding is given. For *Old*, only the minimum age is given.
[c] G.W.C. = Ground Water Class. Classes Vb and VI have an average lowest groundwater table deeper than 120 cm below surface and an average highest groundwater table between 25 and 40 cm and between 40 and 80 cm below surface respectively. Data is derived from the 1:50 000-scale soil map.

The fields that had been used for maize cultivation had all been converted to grassland within the last 3 years. Before the maize cropping period, these were all grassland. Field 12 also had maize for two years at the end of the 1980s and field 41 also had one year maize in 1996 but both are considered to be reseeded grassland. Values for pH-$_{\mathrm{KCl}}$ were generally below 5.0. Fields that fulfill the requirements for continuous silage maize had significantly higher pH-$_{\mathrm{KCl}}$ values caused by recent liming.

2.3.2 Soil organic carbon and nitrogen as a function of land use history

Organic carbon and nitrogen contents of the top- and subsoils are presented in Table 2.2. Variation in soil organic carbon ranged from 2.4% – 4.5% and fell within the range of the soil database (assuming a carbon content of 50%) (De Vries, 1999).

Table 2.2. Average organic carbon and nitrogen contents and mineralization rates at two depths for three different types of land use history. Standard deviations are given in brackets.

L.U.H.[a]	10–20 cm.				25–50 cm.			
	C-org	N-org	C/N	C mineralized	C-org	N-org	C/N	C mineralized
	%			% d^{-1} (×100)	%			% d^{-1} (×100)
Maize	2.41a	0.25a	9.64	4.10a	2.35a	0.25a	9.40	2.51a
	(0.00)	(0.03)		(0.36)	(0.14)	(0.06)		(0.47)
New	3.51b	0.28a	12.54	2.87b	2.84a	0.24a	11.83	1.61b
	(0.41)	(0.04)		(0.55)	(0.80)	(0.05)		(0.39)
Old	4.03b	0.34b	11.85	2.49b	3.22a	0.24a	13.42	0.88c
	(0.43)	(0.03)		(0.51)	(0.76)	(0.04)		(0.13)

[a] L.U.H. = Land Use History. Continuous maize cropping is indicated by *Maize*, renewed grasslands are indicated by *New* and old grasslands are indicated by *Old*. Values within one column, followed by different letters are significant according to L.S.D. (0.05).

Fields that were used for continuous silage maize cropping have considerably lower organic carbon contents compared to reseeded grassland and old grassland for both top- and subsoil, though not statistically significant for the subsoil. This is in line with the observations of Haynes and Tregurtha (1999) who relate this to the higher organic matter inputs under grassland and an increased rate of decomposition due to cultivation under maize. Only for the topsoil are these lower soil organic carbon contents associated with lower organic nitrogen contents, with a significant difference between the old grasslands and the other two types of land use history. The C/N ratios in Table 2.2 suggest an enrichment of nitrogen relative to carbon under arable land.

Carbon contents of the topsoil were related in a regression equation ($P = 0.05$):

$$\text{C organic (\%)} = 3.51 - 1.10 \times \mathit{Maize} + 0.52 \times \mathit{Old} \tag{2.1}$$

$$R^2_{\text{adj}} = 0.67$$

Where *Maize* has the value 1 for continuous maize cropping and 0 otherwise and *Old* has the value 1 for old grasslands and 0 otherwise. Soil organic carbon contents for the cHn23 are related to land use history. Because of the heterogeneity in soil formation in the landscape, other site-specific information was added by incorporating groundwater class information (GWC) from the available 1:50 000 soil survey map. This yielded the following:

$$\text{C organic (\%)} = 3.40 - 1.54 \times \mathit{Maize} + 0.19 \times \mathit{Old} + 0.55 \times \mathit{GWC} \qquad (2.2)$$

$$R^2_{\text{adj}} = 0.75$$

Where *GWC* has the value 1 for class Vb and 0 for VI. Thus, more of the variation in soil organic carbon could be explained when the *GWC* was included in the model. This indicates that originally poorly drained sites in the landscape are likely to have higher carbon contents in their man-made topsoils and carbon losses are likely to be higher in these sites when pasture is converted into maize since they are in general currently well drained. The soils in the study area that are located in the wettest and often lowest positions in the landscape are likely to have been covered by peat, of which remains were mixed in the soil profile during the reclamation of the land. Wetter conditions in the landscape are also associated with the presence of till or subglacial clay in the sub-surface with poor aeration and low decomposition rates. Improved drainage in recent decades is likely to have reduced the rate of accumulation of organic matter in grasslands.

The drier soils belonging to the cHn23 series, mostly in the south eastern part of the study area used to be mostly heathlands. These areas were generally used for arable farming before 1900, which might also help to explain the observed relationship in Equation 2.2, though exact information of land use before 1950 was not extensively available for the selected fields. Soil organic carbon and nitrogen contents of the topsoil were found to be positively related to each other (Table 2.2). Thus:

$$\text{N organic (\%)} = 0.28 - 3.87 \times 10^{-02} \times \mathit{Maize} + 6.04 \times 10^{-02} \times \mathit{Old} \qquad (2.3)$$

$$R^2_{\text{adj}} = 0.56$$

Adding *GWC* data only slightly improved the model ($R^2_{\text{adj}} = 0.59$). In general, the clay+silt fractions for the cHn23 soils were not related to organic C and N contents. Hassink (1995) noted that most of the C and N in Dutch grassland soils is in the fraction > 50 μm and are related to the amounts of residue being incorporated into the soil and not to soil characteristics. The variation in soil organic carbon contents for the subsoil appeared to be greater than for the topsoil. No significant models could be constructed relating average subsoil organic carbon contents to land use history or ground water class data. In some cHn23 soils with a plaggen cover, the original A-horizon could still be observed in the subsoil. It is possible that subsoil samples have been taken from this buried horizon, thereby confusing the relationships found. The presence of charcoal, might also confuse the relationships since this carbonized material hardly decomposes in the soil (Jenkinson 1988).

2.3.3 Soil organic carbon dynamics as a function of land use history

Mineralization rates, derived from the soil respiration study, were expressed as the percentage of C mineralized per day (Fig. 2.3). This figure shows that relative carbon mineralization rates were lower for the grasslands compared to the former silage maize fields. Within the grassland soils, the variation in mineralization of the reseeded grasslands appears to be greater compared to the old grasslands.

A significant relationship for the topsoil was found between the percentage of C that mineralized per day at the end of the incubation period (Table 2.2) with land use history:

$$\%\text{C mineralized day}^{-1}\ (\times 100) = 2.87 + 1.24 \times \mathit{Maize} - 0.38 \times \mathit{Old} \qquad (2.4)$$

$$R^2_{\text{adj}} = 0.53$$

where again *Maize* has the value 1 for continuous maize cropping and 0 otherwise and *Old* has the value 1 for old grasslands and 0 otherwise. Adding groundwater class information did not improve this relationship. The significance only slightly improved when texture data ($\% < 50\ \mu\text{m}$) was incorporated: $R^2_{\text{adj}} = 0.55$.

There was a statistically significant negative relationship with %C in the soil:

$$\%\text{C mineralized day}^{-1}\ (\times 100) = 7.49 - 0.83 \times \%\text{C} - 6.61 \times 10^{-2} \times \% < 50\ \mu\text{m} \qquad (2.5)$$

$$R^2_{\text{adj}} = 0.76$$

The negative relationship with the soil organic C content alone explained only 59% of the observed variation.

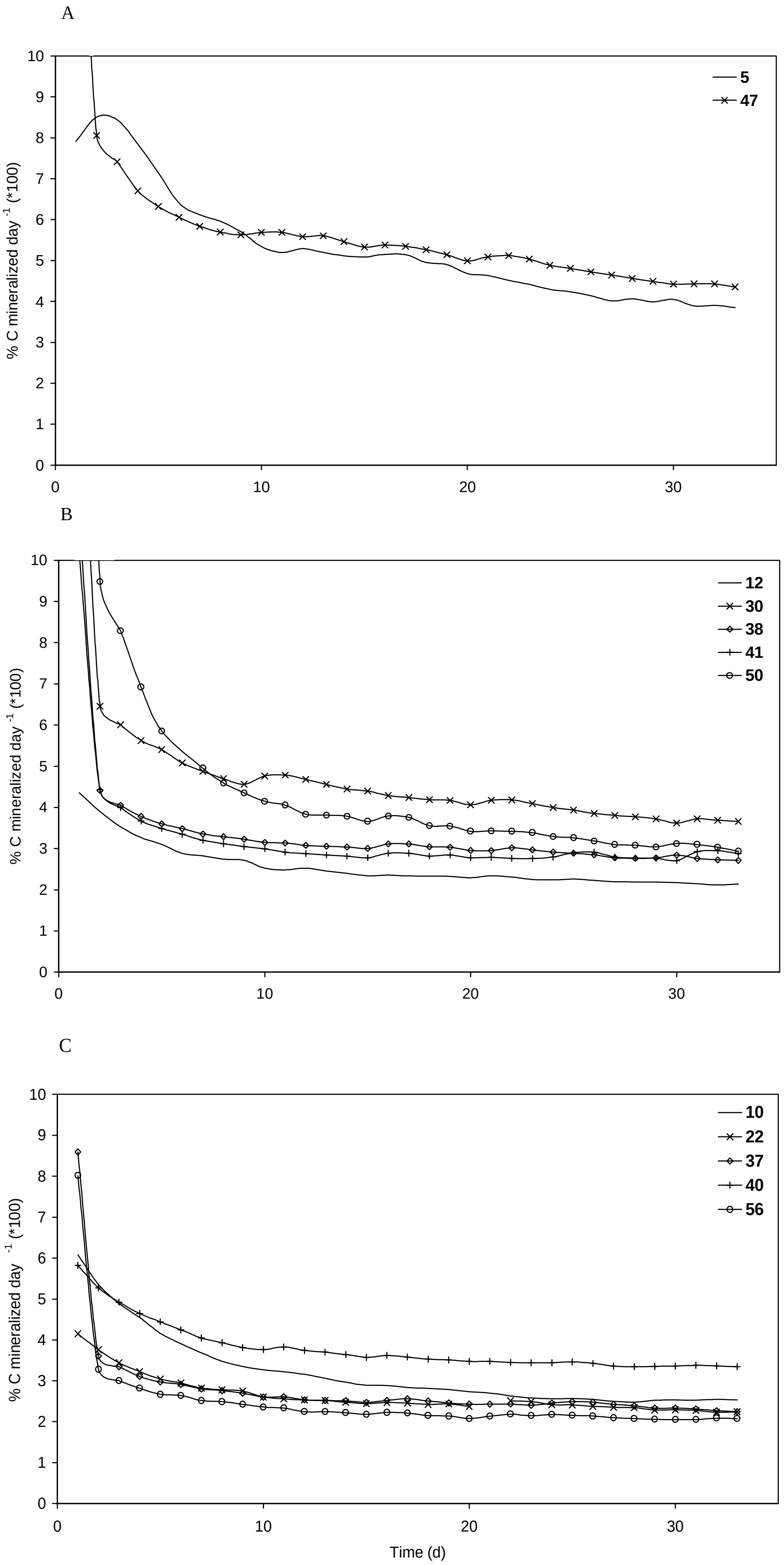

Figure 2.3. Average % of soil organic C mineralized per day (× 100) during laboratory incubation of the topsoil of three types of land use history: continuous maize cropping (A), reseeded grassland (B) and old grassland (C). Numbers relate to field numbers.

For the subsoil, C mineralization rates could also be related to land use history and soil organic carbon content:

$$\%\text{C mineralized day}^{-1}\ (\times 100) = 1.61 + 0.90 \times \textit{Maize} - 0.74 \times \textit{Old} \qquad (2.6)$$
$$R^2_{\text{adj}} = 0.78$$

and

$$\%\text{C mineralized day}^{-1}\ (\times 100) = 3.17 - 0.57 \times \%\text{C} \qquad (2.7)$$
$$R^2_{\text{adj}} = 0.28$$

Higher relative mineralization rates for the subsoil are again found to be negatively related to soil organic carbon content. Incorporating groundwater class data or the % particles < 50 μm did not significantly (P = 0.05) improve the models for the subsoil. The statistical models suggest that the soil organic matter in old grasslands is of a different nature and more stable in terms of mineralization compared to the soil organic carbon in former silage maize fields and reseeded grasslands. Texture data in the form of the percentage of particles < 50 μm improved the relationships found for the topsoil. The percentage of C that mineralized per day was found to be negatively related to the clay+silt fraction. This characteristic is related to landscape position. Soils in the higher southwestern part of the study area (fields 5, 40, 41 and 47) have lower clay+silt fractions compared with fields in other parts of the study area. This can be attributed to the recent geological processes that shaped the area. In contrast to the northwestern and south eastern part of the study area, in the south western part there has been little mixing of till material with the coversand.

2.3.4 Implications for regional nitrate leaching studies

Regional nitrate leaching studies that make use of digital soil survey data can profit from this additional information in soil databases. These studies can roughly be divided into two types. The first type aims to provide information on leaching risks for a specific region which is often done by means of rating soils and creating vulnerability (or leaching risk) maps e.g. (Khakural and Robert, 1993; Navulur and Engel, 1998; Olsen and Kristensen, 1998). The second type aims to provide scenario analyses which are important for evaluation of policy measures or extrapolated management practices (e.g. Hack-ten Broeke *et al.*, 1999; Jansen *et* *al.*, 1999; Groeneveld *et al.*, 2001; Lord and Anthony, 2000). Here, often 'what if?' questions play an important role.

In the first group of 'screening' studies, it may be useful to explore whether conclusions still hold if the whole possible range in soil properties is taken into account. If relationships between these properties and land use history have been established as for the cHn23 soil series, more precise assessments may be performed through the use of older land use databases, land use maps and possibly interviews. This may then lead to more spatially explicit leaching risk maps, especially when specific landscape characteristics are included. Scenario-studies may profit from this research in much the same way. They may also use information on soil organic carbon dynamics to estimate relative organic matter pool sizes in their models. If their goal is to asses environmental impacts of land use change over a period of decades or more, refined soil survey data could also be used to take account of changing soil properties.

When model outputs are used in the policy decision making process, it should be realized that the variability in soil properties is not just variability as such, but that it is a consequence of specific decisions on farms which are often regarded as meaningful in the context of the whole soil-plant-animal system. Moreover, the mapped landscape units and the spatial dynamic relationships that exist among them often structure these agricultural activities. Strategies to control nitrogen leaching must therefore be developed within the context of the whole-farm (Cuttle and Scholefield, 1995) and landscape system.

2.4 Conclusions

Variability of soil organic nitrogen, soil organic carbon and its dynamics in the cHn23 soil series could largely be explained by taking account of three types of land use history: old grassland, reseeded grassland and grassland converted from continuous silage maize cropping. These spatially explicit relationships could even be improved by taking account of specific landscape characteristics. Distinguishing between mowing and grazing regimes or specific silage maize cultivation practices might further explain the variability observed. Regional land use studies using soil information from databases that are confronted with spatial and also temporal variation in these soil properties under grassland can profit from this refined soil survey information.

Chapter 3

Simulation of soil water regimes including pedotransfer functions and land use related preferential flow

Abstract

Differences in land use history among taxonomically identical soils often result in different hydraulic properties, derived from either laboratory measurements or pedotransfer functions (PTFs). Additionally, flow mechanisms in sandy soils may also change through differences in water repellency associated with land use history. The soil water regimes for three sandy soils of the same taxonomic unit and under pasture but with differences in land use history were simulated. The land use histories were old grassland (site A), recently reseeded grassland (site B) and previous maize-cultivated land (site C). Degrees of water repellency, as indicated by the Water Drop Penetration Time (WDPT) test, were found to be highest for the topsoil of sites A and B. Initial simulations, using continuous pedotransfer functions to derive the Mualem-Van Genuchten parameters, corresponded poorly with field measurements (TDR). Additional laboratory measurements did not result in a better correspondence. Taking account of preferential flow in sites A and B, using the mobile/immobile concept, improved modelling performance significantly. Model simulations for a limited time period showed that water storage in the top 50 cm was on average 59 mm higher for site C compared with site A and 23 mm higher for site B compared with site A. Downward fluxes at 50 cm depth were especially larger for site A compared with sites B and C.

Based on:

Sonneveld, M.P.W., Backx, M.A.H.M. and Bouma, J., 2003.

Simulation of soil water regimes including pedotransfer functions and land use related preferential flow.

Geoderma, 112(1-2): 97-110.

3.1 Introduction

Soil series can cover a significant range of non diagnostic soil properties. One of the causes for this observed variation is management history (e.g. Pulleman *et al.*, 2000; Sonneveld *et al.*, 2002; Sparling *et al.*, 1992). To examine the consequences of these human-induced effects on soil water regimes, dynamic simulation models can be used as an explorative tool (Wagenet *et al.*, 1991). Input data which is required for these computer simulation models can be derived from laboratory or field measurements but most of these techniques are relatively time consuming and costly (Hack-ten Broeke and Hegmans, 1996; Wagner *et al.*, 2001; Wösten *et al.*, 1995). Input data may also be derived indirectly, using pedotransfer functions (PTFs) which relate different soil characteristics and properties to one another or to land qualities. Essentially, they act as filters (Wagenet *et al.*, 1991) through which basic soil survey information passes to eventually be used in simulation models. Common PTFs are those which relate basic soil properties obtained in soil surveys, such as texture and organic matter, to parameters describing the soil hydraulic functions. Especially continuous PTFs which predict Van Genuchten parameters using characteristics such as bulk density and organic matter content (Wösten, 1997), are often regarded as useful (Van Alphen *et al.*, 2001).

Different hydraulic functions for the same taxonomic unit can thus be obtained when continuous PTFs use data from soils belonging to that unit but with different land use histories. However, models that simulate water regimes in such soils are not only confronted with a range of input parameters, through the continuous PTFs, but also with heterogeneity in the basic flow mechanisms. For example, differences in land use history may imply differences in the degree of water repellency, leading to wetting front instability and preferential flow paths (Ritsema *et al.* 1993). Bond (1964), DeBano (1969) and Dekker and Ritsema (1996b) reported that the degree of water repellency in grassland soils is a function of time and generally increases with the age of the pasture. Evidently, models describing water movement through hydrophobic soils must provide an appropriate description of water repellency and associated preferential flow (DeBano, 1969; Ritsema *et al.*, 1998). Stable flow is possible in homogeneous and isotropic soils but is quite uncommon in field soils. Preferential flow paths may remain stable for only one growing season in the case of crop cultivation while in untilled fields, persistent patterns may develop (Ritsema and Dekker, 2000).

Clearly, land use differences in a given soil not only result in changes of non diagnostic soil properties, such as soil organic matter content and bulk density, leading to differences in input for PTFs, but these differences may also result in heterogeneous flow patterns.

Consequently, the basic assumptions of a hydrological model should also be critically examined when simulating soil moisture regimes in soils with differences in land use history.

The aim of this chapter is to evaluate how differences in land use history influence soil water regimes in one Dutch sandy soil series under pasture. Field measurements and simulation modelling were used to describe these regimes and this study focused on improvement of modelling performance by (1) testing the use of continuous pedotransfer functions and additional laboratory measurements and (2) assessing the potential occurrence of preferential flow paths.

3.2 Materials and methods

3.2.1 Study area and monitoring

The study area is located in the north of the Netherlands (53,16°N; 6,05°E) where more than 80% of agricultural land comprises dairy farms. The soils may have experienced different types of land use history as a consequence of various factors. Three major types of land use history for grasslands were identified as having occurred the last 50 years: old grassland, reseeded grassland and previously arable land, used for continuous cropping with silage maize. GIS analysis of the 1:50 000 scale soil map of the area revealed that about 40% of the area is covered by one soil series, the cHn23 (StiBoKa, 1981), which is classified as a Plagganthreptic Alorthod (Soil Survey Staff, 1998). This sandy soil has a characteristic anthropogenic topsoil of 30–50 cm thick and is generally found in the higher parts of the landscape (Sonneveld *et al.*, 2002). Three sites with soils fulfilling the requirements for the cHn23 soil series were selected corresponding with the three different types of land use history (Sonneveld *et al.*, 2002). Site A has been permanent pasture for the last 50 years without reseeding. Site B was also permanent pasture but has recently been reseeded; the age of the pasture at the time of sampling was 10 years. The age of the pasture at site C at the time of sampling was 3 years, following a 12-year period of growing maize in continuous cultivation. During spring 2000, TDR sensors were installed at each site at 4 different depths (10, 25, 40 (or 45) and 60 (or 65) cm). A piezometer was also installed and regular recordings for both soil moisture content and ground water level depth were made during the summer of 2000. Precipitation data were derived from two local weather stations as there was some distance between the three sites. The distance between a site and a weather station was less than 7.5 km. Other meteorological data were derived from a meteorological station with an approximate distance of about 20 km. Grassland management for the specific fields was also recorded.

3.2.2 Simulation model

For this study, the agrohydrological model SWAP (Van Dam *et al.*, 1997; Kroes *et al.*, 1999) was used. It is designed to simulate transport processes at field scale during an entire growing season. The model uses Richard's equation for soil water movement in the soil matrix. Top boundary conditions of the simulated soil system are defined by the soil surface with or without a crop and atmospheric conditions. A simple grass growth module was used here to simulate daily potential evapotranspiration. In SWAP, measured groundwater level values were used as input to define bottom boundary conditions. The model was initialized using local meteorological and hydrological data for the years 1998 and 1999.

3.2.3 Water repellency

It is generally recognized that water repellency is a function of the type of organic matter in the soil and may be induced through irreversible drying processes of organic matter, leached organic substances, hydrophobic microbial by-products and intermixing of mineral soil particles with particulate organic matter (Dekker, 1998). A simple and common method to assess and classify water repellency is the empirical Water Drop Penetration Time (WDPT) test (Dekker and Ritsema, 1994). This test consists of placing three water drops on the soil surface and recording the time taken for the water to penetrate the sample. Five classes are distinguished (Dekker and Ritsema, 1994): wettable or non-water repellent (< 5 s), slightly (5–60 s), strongly (60–600 s), severely (600–3600 s) and extremely water repellent (> 3600 s). Here, dried (40°C) and sieved samples of top- and subsoil of three fields were used. They were equilibrated for three days at laboratory conditions (20°C, 50% humidity). Three water drops were placed on each sample and the time required to complete infiltration of all drops was recorded. The procedure was repeated three times and the average time to infiltration was used to determine the degree of potential water repellency.

3.2.4 The mobile/immobile concept

Van Dam *et al.* (1990; 1996) suggested to use the mobile/immobile concept to simulate water movement in a water repellent soil. Though criticized for neglecting hysteresis in the water retention function (Ritsema and Dekker, 2000), their approach seems appropriate where preferential flow is likely to play a role but where the water retention function is surrounded with some uncertainty. The latter is the case when PTFs are introduced. Assuming the water content in the immobile parts is negligible, Van Dam *et al.* (1990; 1996) derived the following equations for the entire soil:

$$\theta^*(h) = F\,\{\theta(h)\} \tag{3.1}$$

$$K^*(h) = F\,\{K(h)\} \tag{3.2}$$

Here, $\theta^*(h)$ and $K^*(h)$ are the water retention function and the hydraulic conductivity function for the entire soil respectively while $\theta(h)$ and $K(h)$ are the retention and hydraulic conductivity function which apply to the preferential flow domain. *F* represents the volumetric fraction of the soil occupied by preferential flow paths. Following Van Dam *et al.* (1990); an inverse optimization method will be used here to estimate *F*.

3.2.5 Pedotransfer functions

The Mualem-Van Genuchten equations (Van Genuchten, 1980) were used in this study to describe the relationship between volumetric soil water content, θ, and hydraulic conductivity, *K*, as a function of the pressure head, *h*:

$$\theta(h) = \theta_r + \frac{\theta_s - \theta_r}{(1 + |\alpha h|^n)^{1-1/n}} \tag{3.3}$$

$$K(h) = K_s \frac{\left[(1 + |\alpha h|^n)^{1-1/n} - |\alpha h|^{n-1}\right]^2}{(1 - |\alpha h|^n)^{(1-1/n)(l+2)}} \tag{3.4}$$

Here, *r* and *s* refer to residual and saturated values while α, *n* and *l* are parameters determining the shape of the curve. The continuous PTFs of Wösten (Wösten, 1997) give predictions of the Van Genuchten-Mualem parameters. The presented functions use basic soil properties as texture, organic matter content and bulk density as input data to derive these parameters for sandy and loamy/clayey soils. Wösten (1997) described transformed model parameters to comply with a number of physical boundary conditions:

$$K_s^* = 9.5 - 1.471D^2 - 0.688OM + 0.0369OM^2 - 0.33\ln(CS)$$

$$\alpha^* = 146.9 - 0.0832OM - 0.395topsoil - 102.1D + 22.61D^2 - 70.6D^{-1} - 1.872\mathrm{CS}^{-1} - 0.3931\ln(CS)$$

$$\begin{aligned} n^* = {} & 1092 + 0.0957CS + 1.336M50 - 13229M50^{-1} - 0.001203M50^2 - 234.6\ln(M50) - \\ & 2.67D^{-1} - 0.115OM^{-1} - 0.4129\ln(OM) - 0.0721D \times CS \end{aligned}$$

$$l^* = 0.797 - 0.591OM + 0.0677OM^2 + 0.573topsoil$$

$$\begin{aligned} \theta_s = {} & -13.6 - 0.01533CS + 0.0000836CS^2 - 0.0973CS^{-1} + 0.708D^{-1} - 0.00703M50 + \\ & 225.3M50^{-1} + 2.614\ln(M50) + 0.0084OM^{-1} + 0.02256\ln(OM) + 0.00718D \times CS \end{aligned}$$

Here, K^*, α^*, n^* and l^* are the transformed parameters in the van Mualem-Genuchten equations (Wösten, 1997). The term CS represents the clay+silt%; OM: the percentage organic matter, D: bulk density, M50: median sand particle size and there is also the discrete variable topsoil (one when true). Continuous pedotransfer functions were used for the top 50 cm while similar class pedotransfer functions based on texture (Wösten *et al.*, 1994) were used to characterize the lower layers of the soil profile.

3.2.6 Laboratory measurements

At the selected fields, soil organic carbon of 4 mixed samples for the topsoil (10–20 cm) and 4 mixed samples for the subsoil (25–50 cm) were measured using an elemental analyser (EA 1108). Soil texture from different depths was analyzed using a laser diffractometer (Coulter LS230). Eight soil cores (300 cm^3) were taken from both top and subsoil for soil physical measurements. Saturated hydraulic conductivity was measured using a Mariotte tube and feeder burette to maintain a constant water pressure on saturated soil samples (Booltink *et al.*, 1991). Bulk densities were determined by oven-drying saturated samples overnight (105°C) and determining water loss by weighing before and after drying. Saturated moisture contents were calculated from these data.

3.2.7 Statistical criteria

Modelling performance was assessed using simulated and measured soil moisture contents. Different criteria have been proposed for evaluation modelling performance (Loague and Green, 1991; Hack-ten Broeke and Hegmans, 1996; Van Alphen *et al.*, 2001). For this study, it was thought to be sufficient to asses whether there is a bias in the simulated results, how much scatter there would be around a 1:1 linear relation between simulated and measured values and how large the maximum error is. Hence, we used mean residual error (MRE), root mean squared residual error (RMSE) and maximum error (ME):

$$MRE = (1/n)\sum_{i=1}^{n}(P_i - O_i)$$

$$RMSE = \sqrt{(1/n)\sum_{i=1}^{n}(P_i - O_i)^2}$$

$$ME = Max\left|P_i - O_i\right|_{i=1}^{n}$$

Here, P_i are the simulated values and O_i are the observed values. Ideally, the values of all criteria should be close to zero. All criteria were calculated for each site using all available data.

3.3 Results

3.3.1 Soil characteristics

Differences in chemical and physical characteristics were observed between the three selected sites (Table 3.1). Soil organic matter contents were calculated assuming that it consists of 50% of carbon (Nelson and Sommers, 1982). The data suggest that site A (old grassland) has about 100 tons ha^{-1} more organic matter in the top 50 cm compared with grassland which is recently converted from continuous maize cropping.

The strongly compacted subsoil of site C and observed differences in root density between sites A and C in the subsoil suggested limited rootability for site C. Based on the data from Table 3.1, the Mualem-Van Genuchten parameters could be derived using the pedotransfer functions of Wösten (1997) and assuming for all soils a median sand particle size of 150 μm. Table 3.2 gives these parameters for both top- and subsoil of the three sites.

Table 3.1 General characteristics for the three selected sites.

Depth: 10–20 cm				Depth: 25–50 cm			
Texture		OM %^{a}	D (g cm^{-3})^b	Texture		OM %^{a}	D (g cm^{-3})^b
%<2	%<50			%<2	%<50		
5.2	26.8	8.1 (0.4)	1.30 (0.06)	4.4	23.5	5.8 (0.6)	1.32 (0.07)
5.3	30.5	6.3 (0.4)	1.36 (0.06)	5.0	31.0	5.1 (0.5)	1.44 (0.04)
3.8	18.8	4.8 (0.1)	1.48 (0.06)	3.6	18.9	4.5 (0.2)	1.54 (0.08)

Standard deviations are given in brackets.

[a] Soil organic matter contents were calculated assuming that it consists of 50% of carbon.

[b] Bulk density.

Table 3.2. Mualem-Van Genuchten parameters calculated from Table 3.1 for both soil and subsoil.

Depth: 10–20 cm					Depth: 25–50 cm				
K_{sat} (cm day^{-1})	θ_{sat} (-)	α (1 cm^{-1})	l (-)	n (-)	K_{sat} (cm day^{-1})	θ_{sat} (-)	α (1 cm^{-1})	l (-)	n (-)
16.0	0.43	0.007	0.944	1.37	23.1	0.43	0.014	-0.350	1.41
16.0	0.42	0.007	0.331	1.36	15.8	0.40	0.009	-0.448	1.33
17.4	0.40	0.008	0.093	1.35	14.7	0.38	0.010	-0.482	1.33

Table 3.3. Saturated conductivity (K_{sat}), saturated water content (θ_{sat}) and Water Drop Penetration Time (WDPT) test values for the three selected sites.

Depth: 10–20 cm					Depth: 25–50 cm				
K_{sat} (cm day^{-1})			θ_{sat} (-)	WDPT (s)	K_{sat} (cm day^{-1})			θ_{sat} (-)	WDPT (s)
min	max	mean[a]			min	max	mean[a]		
20	2892	68 (n=6)	0.45 (0.02)	180[b]	18	4005	138 (n=5)	0.42 (0.02)	3[d]
13	3652	21 (n=2)	0.42 (0.02)	9[c]	8	9756	14 (n=5)	0.40 (0.03)	2[d]
16	3925	27 (n=4)	0.37 (0.02)	4[d]	5	4403	15 (n=5)	0.35 (0.02)	3[d]

Standard deviations are given between brackets for θ_{sat} .

[a] After omitting values that exceeded 10 times the calculated value for K_{sat} from Table 3.2.

[b] Strongly water repellent.

[c] Very slightly water repellent.

[d] Wettable.

3.3.2 Hydraulic soil properties and water repellency

Measured hydraulic parameters are given in Table 3.3. Measured θ_{sat} values were found to be considerably lower for site C compared with site A and B for both top- and subsoil. Measured values for site B were closest to those calculated and given in Table 3.2. The data correspond well with observations from others studies where also higher porosities for grassland soils compared with arable soils were reported (Whitehead, 1995).

The measurements on saturated conductivity yielded quite large ranges for K_{sat} as a consequence of macropore flow. From the obtained K_{sat} values, a mean K_{sat} was calculated after omitting the larger values that exceeded ten times the calculated values for K_{sat} from Table 3.2. Such fluxes are associated with very large continuous pores that hardly contribute to flow under field conditions. The resulting means show for the topsoil two to three times larger values for site A compared with sites B and C. For the subsoil, mean K_{sat} values for site A yielded values which were about 10 times the values calculated for sites B and C. WDPT-tests indicated that the topsoil of site A was strongly water repellent while the topsoil for site B was slightly water repellent (Table 3.3). No indications of potential water repellency for samples from the subsoil of these sites were found. For site C, both top- and subsoil were found to be wettable.

3.3.3 Modelling performance

Simulation modelling using standard continuous PTFs and assuming homogenous flow resulted in a poor modelling performance for site A and B for all criteria (Table 3.4).

Modelling performance for these sites could not be improved by replacing the calculated Mualem-Van Genuchten parameters θ_{sat} and K_{sat} with measured values. For K_{sat}, the mean value as given in Table 3.3 was used. Actually, including these measured values resulted in a somewhat worse modelling performance as ME was higher for site A and RMSE was higher for site B. Modelling performance was considerably better for site C using the standard continuous PTFs and assuming homogenous flow. Here, performance could be slightly improved when measured saturated moisture contents and mean K_{sat} values were used in the Mualem-Van Genuchten equations. Because the WDPT test only revealed potential water repellency for sites A and B, preferential flow was included for these sites. Optimized constant fractions of hydraulically accessible soil (*F*) for different depths are given in Table 3.5.

Table 3.4. Model performance considering simulated and measured moisture content values for different model inputs and model flow descriptions at the three sites.

Parameters determination	Flow type	Model Performance		
		ME	MRE	RMSE
- continuous PTFs	homogeneous	0.29	0.15	0.17
- cont. PTFs and additional measurements	homogeneous	0.30	0.16	0.17
- cont. PTFs and additional measurements	preferential	0.12	0.01	0.05
- continuous PTFs	homogeneous	0.25	0.12	0.13
- cont. PTFs and additional measurements	homogeneous	0.25	0.12	0.14
- cont. PTFs and additional measurements	preferential	0.12	0.00	0.04
- continuous PTFs	homogeneous	0.13	0.04	0.06
- cont. PTFs and additional measurements	homogeneous	0.11	0.04	0.06

Table 3.5. Volume fractions of hydraulically accessible soil. Values were derived through inverse optimization.

Location (site)	Approximate depth (cm)			
	0–10	*10–25*	*25–45*	*45–65*
Site A	0.38	0.49	0.48	1.0
Site B	0.44	0.75	0.68	1.0
Site C	1.0	1.0	1.0	1.0

After optimization, site A showed consistently lower values for the mobile fractions of the soil volume for three depth ranges compared with site B. This trend corresponds with the findings of Table 3.3 for the topsoil. Samples for all three sites were found to be wettable for the depth 25–50 cm but model optimization still showed improvement of modelling performance when preferential flow was assumed for this depth. No WDPT test values were available for the top 10 cm to compare with the F values given in Table 3.5. All criteria for modelling performance for sites A and B, (ME, MRE and RMSE) showed better agreements between simulated and calculated soil moisture contents when preferential flow was included. All three combinations for model input and flow type for sites A and B and the two sets of model input for site C (Table 3.4) were used to create Fig. 3.1. Simulated and measured moisture contents are given in this Figure for the year 2000 at depths of about 40 cm (sites A and B) and 45 cm (site C). A good correspondence between measured and simulated (I and II) moisture contents was obtained for site C for both low and high values.

Both simulations for site C, with calculated and with measured values for θ_{sat} and $K_{sat,}$ overlapped. This can also be seen for sites A and B where differences between both simulations (I and II) are very small. Sites A and B showed especially good correspondence between measured and simulated (III, including preferential flow) moisture contents for low values. Higher moisture contents at the end of the observation period could not be simulated satisfactorily for these sites. This is due to the fact that volume fractions for the hydraulically accessible soil were defined as constant for the whole period. Van Dam *et al.* (1996) noticed that the volume fraction F varies in time with higher values for wetter soils. A relationship (e.g. linear) between h and F was not implemented here because of the low number of observations in wetter periods at the beginning and end of the year. In stead, it was chosen to limit the period for comparing simulated soil water regimes for the three sites between day numbers 150 (end of May) and 250 (beginning of September) which covers most of the observation period.

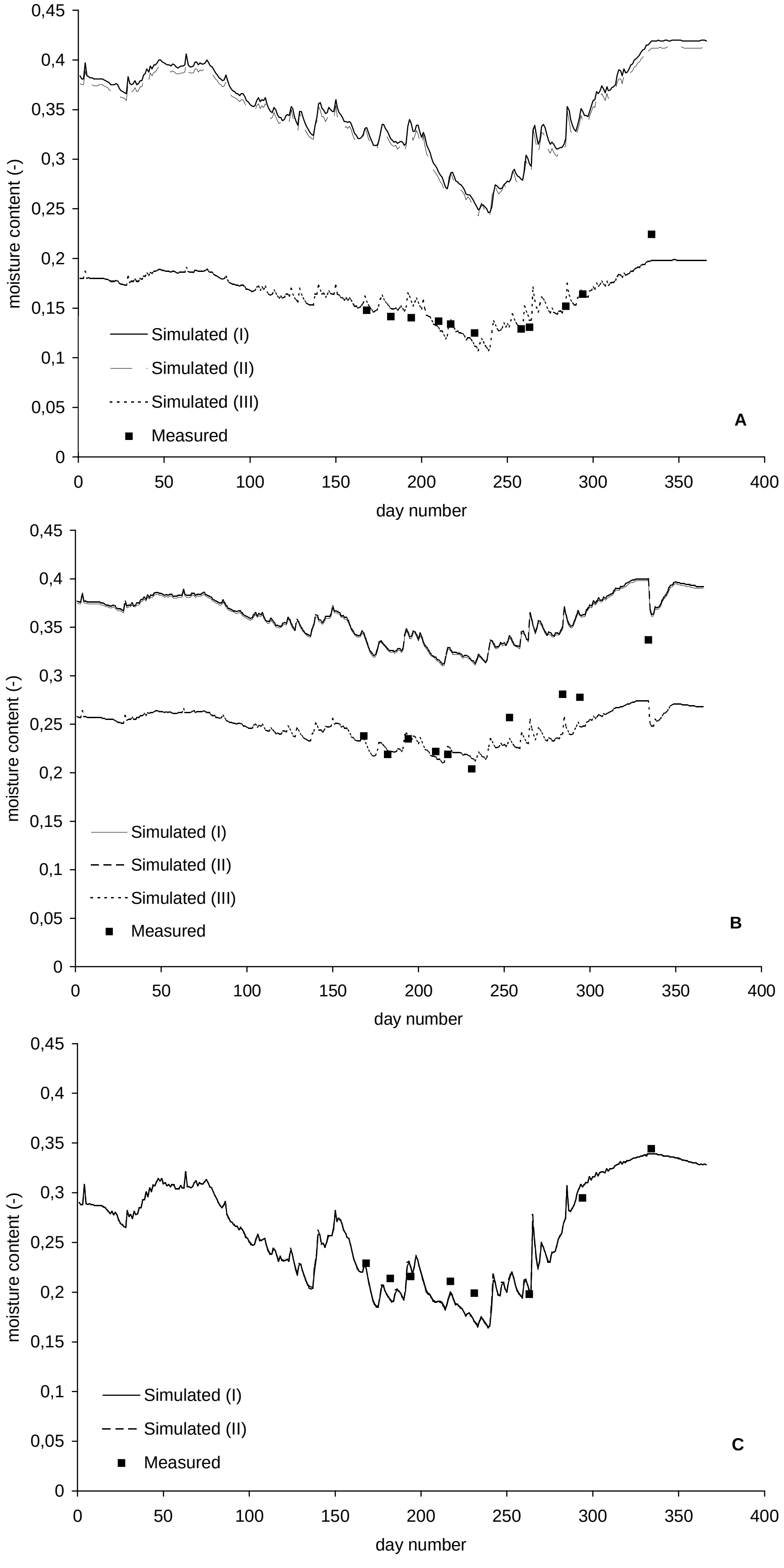

Figure 3.1. Comparison between measured and simulated water contents at 40 (sites A and B) and 45 (site C) cm depth for the year 2000. Simulation using (I) only continuous PTFs, (II) continuous PTFs with additional measurements and (III) continuous PTFs with additional measurements and including preferential flow.

3.3.4 Simulated soil water regimes

Fig. 3.2 shows the simulated water storage for the top 50 cm using standard data for grassland management, groundwater level fluctuations and weather for the year 2000. On average, site A has 59 mm less water stored in the top 50 cm compared with site C and 23 mm less water stored compared with site B. Site B has on average 36 mm less water stored compared with site C. Fig. 3.3 shows the water flux at 50 cm depth for the period between day 150 and day 250. For this period, both maximum upward and maximum downward fluxes were about 30% higher for site A compared with site C. Compared with site B, the maximum downward flux for site A was 7% higher and the maximum upward flux was 19% higher. All the downward peak fluxes for this period, about 15 in total, occurred on the same day except for two. For these days, site A had fluxes which were on average 78% higher compared with site C and 57% higher compared with site B. On day 192, 16.1 mm of rain was recorded for the area (on a total of 209.7 mm for the whole simulation period). On the same day, sites A and B showed in the simulation a peak downward flux of 8.6 and 5.6 mm respectively at 50 cm depth. A simulated peak flux of 5.0 mm for site C followed a day later. Upward fluxes were found to be generally lowest for site C. Especially around day 164 and day 175, site A showed a considerably larger upward flux due to a higher gradient in matrix head.

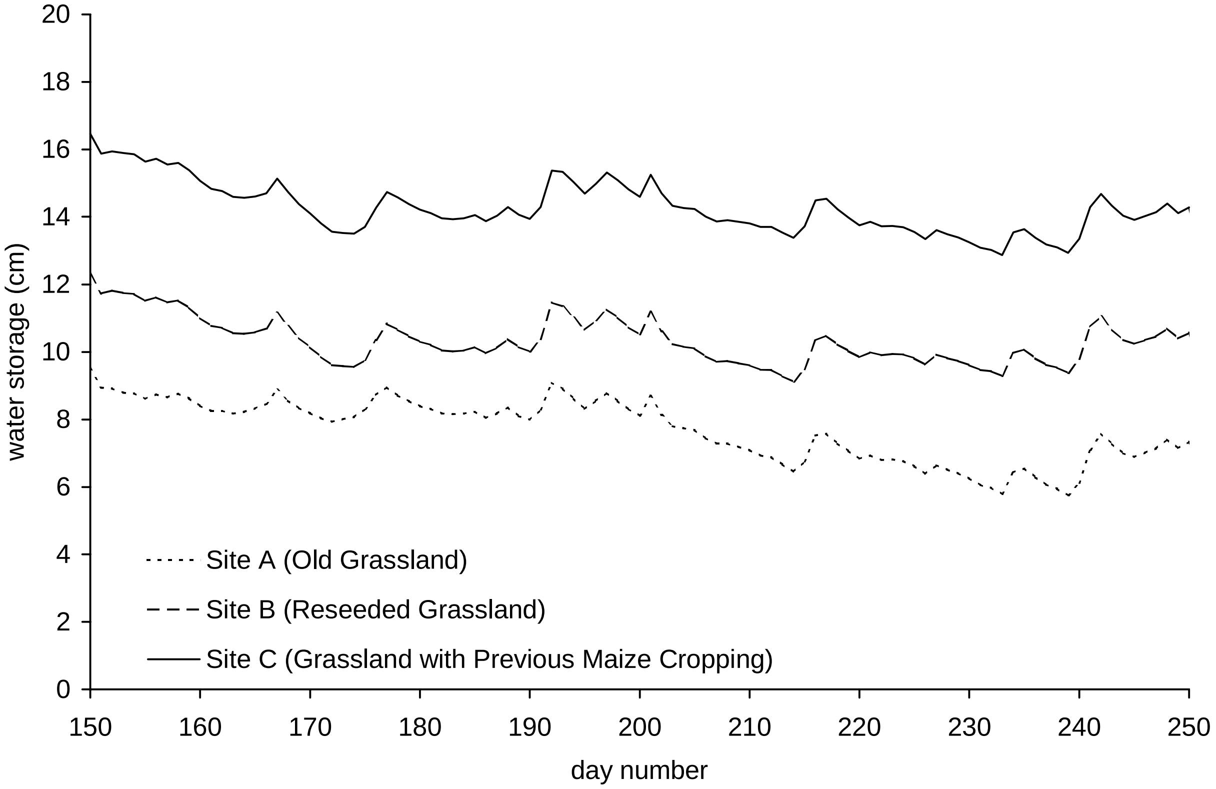

Figure 3.2. Simulated water storage in the top 50 cm for sites A, B and C for the observation period.

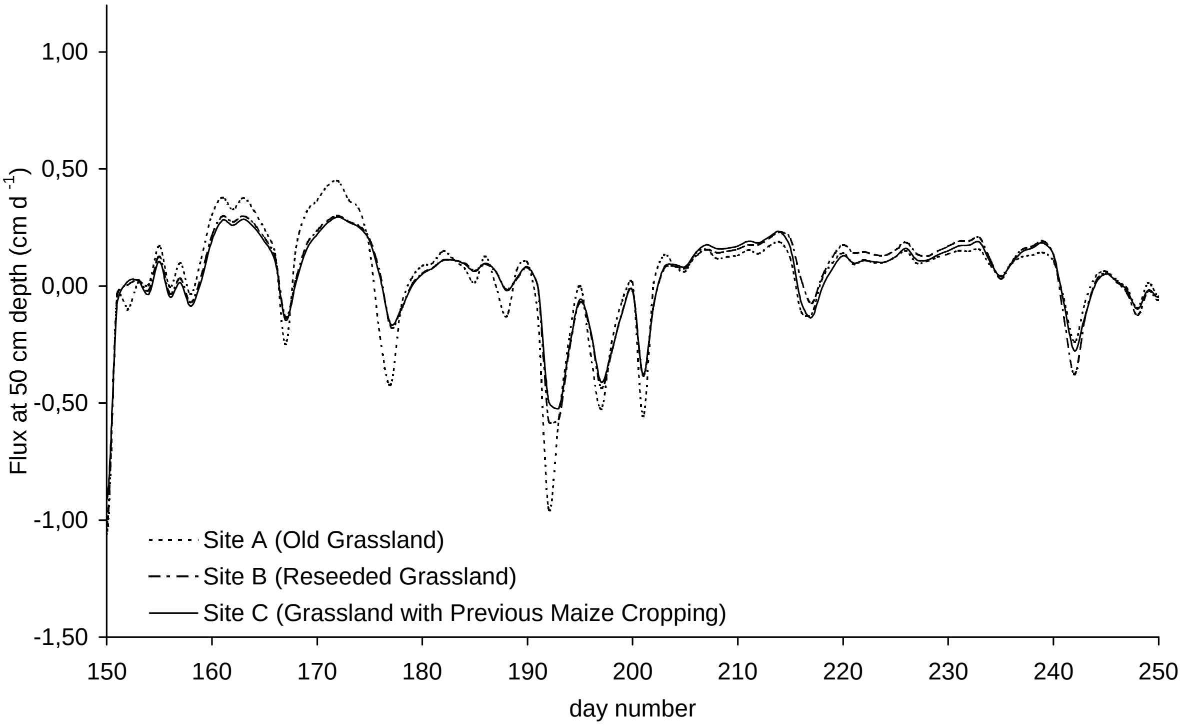

Figure 3.3. Simulated water fluxes for sites A, B and C at 50 cm depth (+ = upward) for the observation period.

3.4 Discussion and conclusions

The way saturated moisture contents were measured in this study ignores air-inclusions that occur when samples are slowly saturated with water and ‘saturated’ water contents are experimentally determined. Application of the crust technique or using larger cores might have resulted in a less wider range of values for K_{sat} than what was observed. Mean values for θ_{sat} and K_{sat} were used as replacement of values derived from the pedotransfer functions but mean K_{sat} values were derived after omitting the very large values. The number of samples to calculate these means (n, Table 3.3) were about the same for all three sites except for the topsoil of site B where only two values remained.

Similar values for potential water repellency as yielded by the WDPT-test have been reported for the topsoil of grasslands on heavy clay soils (Dekker and Ritsema, 1996a). They also show that the degree of water repellency decreases with depth, leaving severe water repellency restricted to the top few centimeters. For their soils, they report mainly wettable samples for the subsoil. Similarly, no water repellent characteristics for the subsoil of sites A and B were found, as indicated by the low WDPT values. It was also found that both topsoil and subsoil of site C (the converted arable land) were wettable.

Including preferential flow for sites A and B (old grassland and reseeded grassland respectively) improved modelling performance considerably. The values used for *F,* the volume fraction of the soil where water movement takes place, follows the trend of potential water repellency for the topsoil as given in Table 3.3. For the subsoil however, the optimized values for *F* are not supported with data on potential water repellency. For the observation period considered, the mobile/immobile concept proves to be a valuable tool to improve modelling performance by incorporating preferential flow. The low number of observations in wetter periods however does not fully justify the use of constant *F* values for longer periods because *F* probably varies in time. The water storage in the top 50 cm as well as the water flux were both considerably influenced by incorporating preferential flow. There are various factors that are likely to influence the effect on water availability for grass growth. Rooting depth for site C is considerably limited due to compaction of the subsoil which will make only water in the topsoil available for root uptake. The compacted subsoil may on the other hand also result in increased water supply due to capillary rise, thereby compensating for the limited rooting depth.

In this study, water retention and hydraulic conductivity from PTFs, based on a number of relatively small samples were assigned to the area of the preferential pathways. The samples taken may however not be fully representative for the preferential flow domains. These areas may show for example less organic matter because of increased leaching compared with the surrounding soil. This might lead to lower porosity values and a decreased saturated hydraulic conductivity but a change in these parameters hardly has any effect on the simulated water regimes once preferential flow domains have been defined. Rapid macro-pore flow may occur more frequently in old grasslands because of the often larger number of soil fauna (e.g. earth worms) compared with recently established pastures. How the distribution of biogenetic macro-pores is related to the occurrence of preferential flow paths (because of water-repellency) is currently not known but these may very well be related.

Days with large amounts of precipitation are likely to lead to significant leaching of solutes in old grassland as compared with recently established swards without preferential flow. Similar effects may also occur during urine excretion from grazing animals.

The finding from this study that soils belonging to one soil series with the same current land use (grassland) may show such variation in soil water regimes suggests that using standard data of the national database for this soil series can lead to erroneous results in crop growth- or nutrient leaching studies that take place at field scale on a daily basis. It was found that the properties used in continuous PTFs to predict the Mualem-Van Genuchten parameters varied widely within one soil series leading to different hydraulic properties. Only when also different flow mechanisms were implemented could measured soil water contents be simulated satisfactorily. The fact that land use history plays a significant role in this variation of soil properties and flow mechanisms implies more emphasis in such studies on the interaction between soil hydrological processes and land use.

Chapter 4

Effects of different combinations of land use history and nitrogen application on nitrate concentration in the groundwater

Abstract

Effects of differences in both land use history and levels of nitrogen (N) application on nitrate concentration in the groundwater were studied for permanent pastures located on a single soil series in the Friesian Woodlands in the north of the Netherlands. The study was carried out for three fields: A, B and C. Field A was an old pasture, field B was a reseeded pasture and field C had been previously used for growing silage maize. The models SWAP and ANIMO were used for long-term simulations of the soil organic matter and soil N dynamics. The soil data from fields A, B and C were combined with different N application levels derived from commercial dairy farms on the same soil series for 2000. Soil organic matter and soil organic N were lower in field C than in fields A and B. In field C also the probability of exceeding the environmental threshold for nitrate in groundwater of 50 mg l^{-1} *was lowest, which was ascribed to net immobilization irrespective of the high levels of N applied. However, this probability increased rapidly when the soil properties were similar to those of the old pasture (field A). Simulated levels of N uptake were higher for field A than for fields B and C at all levels of N applied. On old pasture, reducing N application levels can lower the probability of exceeding the environmental threshold for nitrate by up to 20% whilst hardly affecting N uptake.*

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on nitrate concentration in the groundwater.
Netherlands Journal of Agricultural Science, 51(1/2): 135-146.

4.1 Introduction

Many farmers consider it advantageous to renew ageing swards (Hopkins *et al*., 1995). Renovation has resulted in a substantial portion of the grasslands in the Netherlands being ploughed and reseeded. In the 1990s, this portion ranged from about 4 to 8% of the total grassland area (Verstraten, 1996; CBS, 2000). Grasslands are on average reseeded every 8 or 9 years and in most cases are not older than 15 years at the time of reseeding (Anon., 2000). Although grassland renovation appears common practice, for decades it has been widely debated among scientists (Minderhoud, 1959). It generally is a costly operation with considerable risk of failure, which sometimes even results in a lower dry matter production than was obtained with the original, ageing sward (Hoogerkamp, 1974; Woldring, 1975; Hopkins *et al*., 1995). Recently, environmental concerns have resulted in other arguments against grassland renovation (Verstraten, 1996; Vellinga *et al*., 2000) and even in a complete ban by the Dutch government on ploughing grassland in the period 16 September – 31 January (LNV, 2001a).

Most of the concerns focus on the fact that ploughing grassland will potentially lead to an increase in the leaching of nitrate to the groundwater, which eventually may affect the quality of drinking water. This is caused by mineralization of organic nitrogen (N) after tillage (Whitehead, 1995). Some evidence for this comes from studies that focused on the conversion of pasture into arable land (Garwood and Ryden, 1986; Whitehead *et al*., 1990; Lloyd, 1992). On chalk soils in the UK, Cameron and Wild (1984) identified ploughed grasslands followed by winter wheat as a major potential source of nitrate in drinking water. The study of Whitmore *et al*. (1992) even showed that in some parts of the UK, ploughed grasslands were the main source of nitrate leaching to the groundwater. Substantial amounts of nitrate can also be leached from the soil in ley-arable land systems where short grassland periods are alternated with periods of arable farming (Ryden *et al*., 1984; Francis, 1995; Djurhuus and Olsen, 1997). Although such systems are often favourable for an arable crop like silage maize (*Zea mays*) (Van Dijk *et al*., 1996), the overall effectiveness may be debatable when large amounts of N are lost following ploughing (Catt *et al*., 1992). In general, the amount of N mineralized will increase with (1) increasing time the sward has been remained intact, and (2) increasing fertilization rate, whereas it will also be higher after grazing than with cutting (Whitehead *et al*., 1990). Also the time of ploughing significantly affects nitrate leaching (Whitehead *et al*., 1990; Jarvis, 2000). When ploughing is followed by a long period of arable cropping, e.g. continuous cropping of silage maize, N losses are in general expected to be substantial. In contrast to arable cropping, grassland renovation involves no annual soil tillage and results in a full canopy cover during subsequent winters (Shepherd *et al*., 2001). If grassland is reseeded, the increase in nitrate leaching is expected to be mainly short-lived (Vellinga *et al*., 2000; Jarvis, 2000). This is also confirmed by Shepherd *et al*. (2001) who found that the largest losses through leaching were in the first winter after reseeding; no significant increases in nitrate losses occurred in subsequent years.

Old pastures are expected to have equilibrium levels of organic carbon and organic N in the soil. Estimates of time required for reaching this equilibrium after arable cropping range from 50 to 200 years (Whitehead, 1995). Pastures with organic N levels below equilibrium level will show net immobilization of N. This takes place at rates from 50 to 150 kg N ha^{-1} year^{-1}, but initial high rates will decrease as accumulation of organic N is asymptotic (Ryden, 1984; Scholefield *et al*., 1988; Hassink, 1995). In old pastures nitrogen immobilization is at equilibrium with nitrogen mineralization. In reseeded pastures and pastures converted from arable cropping, both reduced aeration and lower organic matter contents will lead to lower mineralization rates (Scholefield *et al*., 1993).

In the area of the northern Friesian Woodlands, farmers have traditionally opposed ploughing grassland for the conversion to arable cropping or ley-arable land systems (Veenenbos, 1949). Especially old pastures were known to have an ‘old force’ (Edelman, 1949), which withheld some farmers from ploughing their old grassland. Also the fact that in some parts of the fields the less fertile subsoil could be brought to the surface prevented farmers from ploughing and reseeding (Van der Ploeg, 1999). Nonetheless, during the last decades grassland renovation and growing silage maize have increased in this area. An earlier study showed a significant relationship between land use history and organic carbon and organic N contents of the topsoil of a major sandy soil series (cHn23) in this area (Sonneveld *et al*., 2002). Because of increased environmental concerns, some of the reseeded pastures and the pastures established after arable cropping may be used in the future as permanent pasture, i.e., without any cultivation. In terms of nitrate leaching and in comparison with old pastures these ageing swards are likely to respond differently to constant levels of N application over time.

The objective of this study was to describe the effects of both differences in land use history and different levels of N application on long-term nitrate concentrations in the groundwater for cHn23 soils under permanent pasture. This was done using dynamic water and N simulation models. Simulation results will be discussed in the context of the current EU standard of 50 mg l^{-1} for nitrate concentrations in the upper groundwater (European Commission, 1991) and MINAS, the Dutch Mineral Accounting System (Van den Brandt and Smit, 1998).

4.2 Materials and methods

4.2.1 Area description and field selection

The region of the Friesian Woodlands is situated on the northern edge of a till plateau and mostly consists of sandy soils with some peaty and clayey soils in the northern part. Using soil survey data from StiBoKa (1981), it was found that more than 40% of the area belongs to one soil series, cHn23 (Fig. 2.1), which is classified as a Plagganthreptic Alorthod (Soil Survey Staff, 1998). This sandy soil series with its characteristic anthropogenic topsoil is generally regarded as an excellent soil for agricultural purposes. As soils belonging to this soil series are mainly situated on top of the plateau, most of the corresponding hydrological regimes fall into classes Vb and VI, indicating a mean highest groundwater table below 25 cm or 40 cm, respectively, and a mean lowest groundwater table below 120 cm.

From a larger data set (Sonneveld *et al*., 2002), three fields under pasture (A, B and C) were selected at commercial dairy farms that fulfilled the requirements for the cHn23 series but with different land use histories. Field A was an old pasture where no tillage had been applied for the last 50 years. Field B was a reseeded pasture that had been ploughed and reseeded some 10 years ago, with no other crop for the last 50 years. Field C was a recently established pasture following 12 years of continuous cropping with silage maize. In 2000, samples were taken from all fields from both topsoil (10–20 cm) and subsoil (25 cm – lower boundary of A horizon) and analysed for soil organic carbon and soil organic N contents.

4.2.2 Field management

Data available for the year 2000 of several commercial dairy farms located on the cHn23 soil series were used to calculate mineral surpluses using MINAS. This Mineral Accounting System, introduced by the Dutch government to restrict emission of nutrients to the environment, involves registration at farm level of N inputs in fertilizers and feed, and of N outputs in products and manure. Three farms were selected (I, II, and III) that differed in calculated N-surpluses: 278, 250 and 203 kg ha^{-1}, respectively. At each of these farms, two fields were selected where in 2000 only cutting and no grazing had taken place. Field management data from these fields were used as input for the simulation models. Amounts of N applied with organic manure (slurry) were calculated using chemical characteristics of the slurry from each of the three farms. Amounts of N applied to every field for the year 2000 are given in Table 4.1. In this table each field has been given a unique number (N level) that corresponds with a ranking in total N applied.

Table 4.1. Amounts of N applied to the selected fields of each farm for the year 2000 (kg ha^{-1}) and the assigned N management levels.

Farm	N from fertilizer	N from slurry	Total N	N level
I	141.0	100.0	241.0	1
	284.0	275.0	559.0	6
II	159.0	168.0	327.0	2
	159.0	336.0	495.0	5
III	189.0	188.0	377.0	3
	188.0	258.5	446.5	4

4.2.3 Simulation models

To be able to compare the dynamic behaviour of the selected soils, the agro-hydrological model SWAP (Kroes *et al*., 1999; Van Dam *et al*., 1997) and the N model ANIMO (Groenendijk and Kroes, 1999; Kroes and Roelsma, 1998) were used to simulate water and N fluxes in the soil. Model input for calibration purposes was derived by monitoring - during one year - of soil water contents, weather conditions, grassland management and groundwater level fluctuations in fields A, B and C. Soil hydraulic characteristics were derived by means of pedotransfer functions. Since the ANIMO model has been calibrated and validated with annual averages (Hack-Ten Broeke and De Groot, 1998) using the model implies that only annual nitrate concentrations in the groundwater will be compared. Homogeneous flow was assumed for the three sites.

A 30-year (1971–2000) climatic data set was obtained from a nearby weather station, and a long-term record of groundwater-level fluctuations was available from a local piezometer at farm I over the same period (groundwater class VI). Farm-specific data regarding the chemical composition of the slurry were used in the ANIMO model. Distribution of organic matter over different pools in the soil was calculated using data from Römkens *et al*. (1999) assuming a carbon content of 0.58 for organic matter. The ANIMO model was further calibrated assuming that following ploughing and reseeding, a 50-year period of permanent pasture would result in the properties of the old pasture of field A for all levels of N application. In a given year, N from fertilizer or from slurry was applied to the soil on days that corresponded with the days of the actual applications in 2000. The nitrate concentrations were calculated for a depth of 1.7 m below the soil surface as this depth was mostly within the groundwater zone.

Table 4.2. Field and soil characteristics of fields A, B and C. Soil texture is given for the topsoil and total soil organic matter and soil organic nitrogen are given for a 1-meter profile.

Field	Land Use History	Age	Ground-water class	Texture (% particles <)		Total Organic Matter	Total Organic Nitrogen
		(years)		2 μm	50 μm	(tons ha^{-1})	(tons ha^{-1})
A	Old Pasture	50	Vb	5.2	26.8	392.2	16.5
B	Reseeded Pasture	10	VI	5.3	30.5	349.3	14.9
C	Previous Maize Cultivation	3	VI	3.8	18.8	309.7	13.3

4.3 Results and discussion

4.3.1 Field and soil properties

Soil properties and other characteristics of fields A, B and C are given in Table 4.2. Total soil organic matter and soil organic N contents for a 1-m soil profile were calculated on the basis of the analyses and the distribution over the various organic matter pools in the ANIMO model. Soil organic matter content was lower for fields B and C compared with field A (11 and 21%, respectively). Also soil organic N content was lower for fields B and C (10 and 19%, respectively). When field A would be ploughed for arable cultivation, as much as 3.2 t N ha^{-1} could be lost from the soil profile. Because the effects of increased compaction (and hence higher bulk densities) for site C were not taken into account, N losses will probably be somewhat lower in practice.

4.3.2 Model simulations

Mean annual nitrate concentrations at 1.7 m depth were averaged over the different N application levels and plotted as cumulative frequency curves (Fig. 4.1). Nitrate concentrations in the groundwater varied between years and levels of N application from 0 to 107 mg NO_3 l^{-1}. Frequency distributions for fields B and C were almost similar. On average, field A showed the highest probability of exceeding the nitrate standard of 50 mg l^{-1}. This was also true for each N application level separately. Table 4.3 lists the probabilities of exceeding an average annual nitrate concentration of 50 mg l^{-1} at 1.7 m depth for the soil map unit (cHn23-Vb/VI) under grassland with different types of land use history and N application levels. These probabilities were derived by linear interpolation between the closest points.

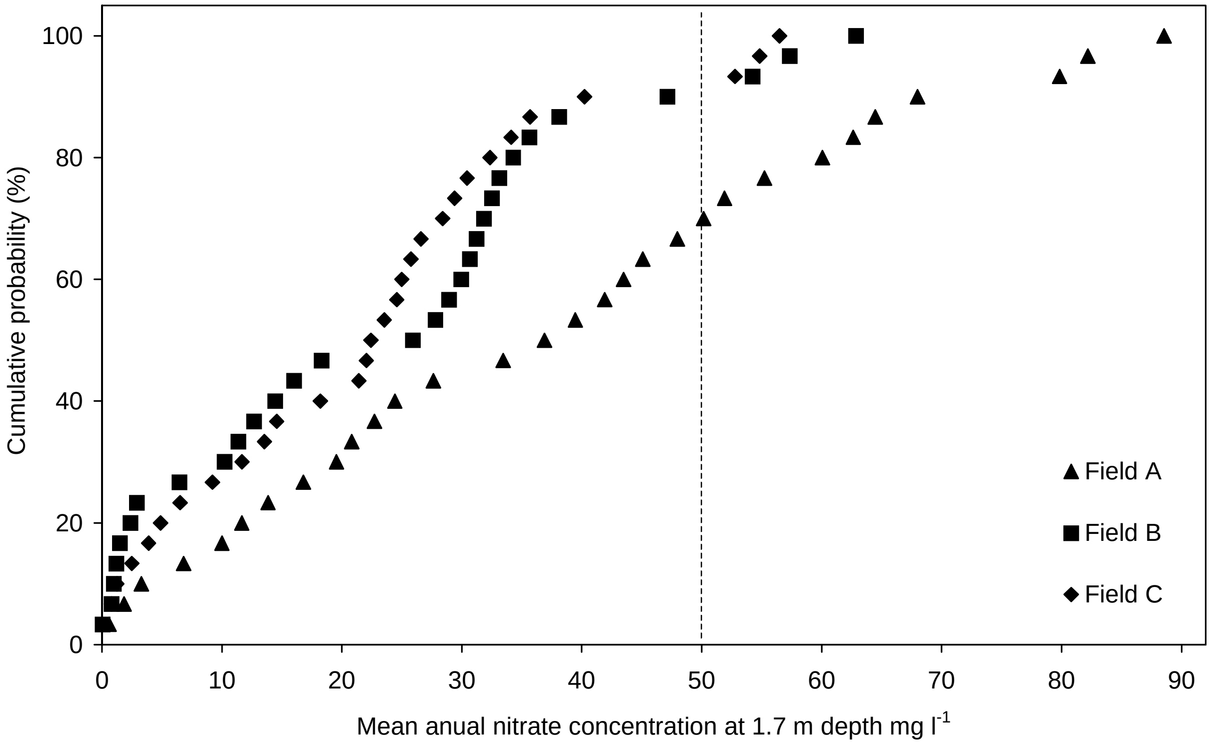

Figure 4.1. Cumulative probability of the mean annual nitrate concentration at 1.7 m depth for one soil map unit (cHn23-Vb/VI) for the three selected fields. Data are averages over the different levels of N applied. Vertical interrupted line indicates the EU standard of 50 mg l^{-1}.

Table 4.3. Probabilities (%) for exceeding an average annual nitrate concentration of 50 mg l^{-1} at 1.7 m depth for one soil map unit (cHn23-Vb/VI) under grassland comparing different types of land use history and nitrogen application levels. Corresponding N levels are given in brackets.

	Total N applied (kg ha^{-1})					
	241 (1)	327 (2)	377 (3)	446 (4)	495 (5)	559 (6)
Field A (Old Pasture)	21.6	18.9	30.2	13.0	19.1	47.1
Field B (Reseeded Pasture)	7.1	7.1	8.7	6.7	7.0	13.2
Field C (Previously cropped to maize)	4.8	0.0	7.8	0.0	7.1	9.9

Probabilities were generally lower for field C than for the other fields. For field C, higher N levels were not always reflected in higher probability values, as could be expected, but showed fluctuating values. For example, probabilities for N-level 3 were always higher than for levels 1, 2, 4 and 5. This can be explained by the different chemical characteristics of the slurry from each farm while at the same time also total amounts of slurry applied varied among N levels. For example, for N-application level 3, only 17% of the total N applied was in organic form whereas all other levels showed values between 20 and 28%. For the highest N application level (level 6), the probabilities for exceeding the 50 mg threshold for a particular field were always higher than for lower application levels. The combination old pasture (field A) with 559 kg N ha^{-1} yielded the highest probability of exceeding the EU nitrate standard (47.1%). Other probabilities for field A were less than half this value, except for N-level 3.

In general the findings are in agreement with Scholefield *et al*. (1988; 1993) who studied the effects of reseeding on nitrate leaching on a clay loam in successive years. In a trial with 400 kg fertilizer N per hectare, Scholefield *et al*. (1988) found that nitrate leaching from an undrained, reseeded sward was a factor 2 lower than from a 40-year old sward. When drainage was included the authors found an even higher reduction in nitrate leaching. Cuttle and Scholefield (1995) reported that nitrate leaching from a grass ley sown after a period of arable cropping was minimal in the second and third year after establishment but increased appreciably in subsequent years in spite of fertilizer inputs remaining unchanged at 200 kg N ha^{-1}. The greater leaching losses from year 4 onwards were assumed to have resulted from an increased supply of N from mineralization of soil organic matter and/or reduced immobilization of fertilizer N. Both processes will also affect the uptake of N by the sward.

For the three fields in this study, also simulated N uptake was given extra consideration. N uptake was always highest for the old pasture (field A) with high levels of N applied (N-levels 4–6). Other N uptakes were calculated per year relative to the ones calculated for field A for these levels. Results were then averaged per field for each level of N application. In Fig. 4.2 the average relative N uptake and the percentage of years in which the EU nitrate standard of 50 mg l^{-1} was not exceeded are plotted for the 6 N levels in each of the 3 fields. N uptake was lowest for the lowest N level in fields B and C (63 and 64%, respectively). N level 6, however, always showed maximum N uptake in all fields. Although leaching to the groundwater at this N level in field C is not much higher than at the other N levels, considerably more leaching is expected when the conditions of the old pasture are met (field A). Moreover, with 182 kg N ha^{-1} less (level 3), a total N uptake close to the maximum can be obtained in field A (96%). For this combination, the probability of exceeding the EU standard is almost 20% lower. If a reduction in N uptake of 10% and a probability of exceeding the EU nitrate standard for groundwater of less than 20% is considered to be acceptable, even 327 kg N ha^{-1} (level 2) can be applied, which is substantially lower than the highest N application level (6). Also the model simulations by Cuttle and Scholefield (1995) showed that any effects of increases in N losses associated with sward age can be avoided provided that fertilizer inputs are lowered to take the increasing supply of N from mineralization of soil organic matter into account.

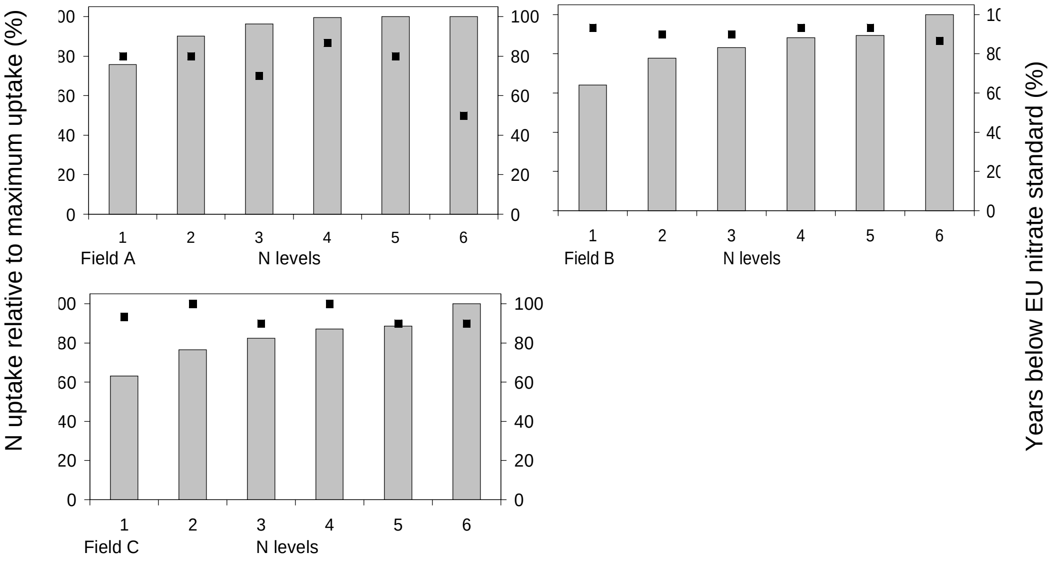

Figure 4.2. Average relative N uptake (bars – left y-axis) and percentage of years in which the EU standard of 50 mg l^{-1} was not exceeded (square points – right y-axis). Data presented for 6 N application levels and three fields with a different land use history. Field A: old pasture; field B: reseeded pasture; field C: previously cropped to maize. Values for N uptake are relative to the N uptake at the highest N level for field A (100%).

4.3.3 MINAS

Field management data were taken from farms with different N surpluses as based on MINAS. Both the highest and the lowest level of N application at field level came from farm I, which had the highest MINAS-N surplus. For this farm, probabilities of exceeding the EU nitrate standard varied between about 5 and 47%, depending on the combination of land use history and level of N applied. Farms II and III showed less variation for the same combinations: between 0 and 19% and 0 and 30%, respectively. However, the MINAS surplus standard, which for 2003 is set by the Dutch government at 180 kg N ha^{-1}, probably does not guarantee that the annual average nitrate concentration in the groundwater is always met. Farm III - with a N surplus of 203 kg ha^{-1}, which is close to the final MINAS standard - still will exceed the

EU nitrate standard of 50 mg l^{-1} in 30% of the years for the simulated combination of old pasture with N level 3. Such probabilities, however, may be regarded as acceptable.

4.4 Final remarks

The results from this study indicate that the cHn23 soils cannot be unambiguously evaluated with respect to the 50 mg l^{-1} nitrate threshold without taking field management and land use history into account. The question as to which probability of exceeding the EU standard is still acceptable from an environmental point of view is not addressed here; the data were only presented to enable others to make such a judgement. Soil properties are not only related to landscape characteristics as defined by soil genetic processes in traditional soil science, but also to land use to the extent that transformations of N can be significantly different within one soil series. Land use studies using soil maps, including so-called ‘representative profiles’, also need to pay attention to land use history.

Our study shows that low N application levels are most suitable for old pastures. Such a combination gives relatively high levels of N uptake with a modest probability of nitrate leaching. If the new sward does not take up all mineralized N and if acceptable levels of N uptake are to be maintained over a long period, reseeded pastures may require higher levels of N input. At farm level, the balance between the benefits of feeding a reduced-N diet of maize silage and the risk of increased leaching after ploughing should also be taken into consideration.

Soils can be considered as results of co-production between nature and man and can reflect social and economic processes in society. This opens up a new perspective for soil science. The search for innovative farmers and contrasting types of land use becomes relevant to discover soils that ‘fit’ in their context of economic and environmental demands. Modern simulation models as used in this study can further characterize their dynamic behaviour. Projects in which both farmers and soil scientists participate to describe and further explore the behaviour of soils, are a useful contribution to the development of sustainable agricultural systems.

Chapter 5

Methodological considerations for nitrogen policies in the Netherlands including a new role for research

Abstract

The Netherlands has attempted to follow EU guidelines in developing national policies to reduce pollution of groundwater by nitrates originating from (over)fertilized agricultural land. The EU has not been satisfied with these policies and this is resulting in legal conflicts. National policies have focused on nitrogen budgeting and on fertilization rates, over-simplifying the crucial role of soils during the leaching of nitrates to groundwater. As an alternative, a dynamic approach using simulation modeling is introduced as is illustrated for a study area in the Netherlands. A number of considerations for future policy directions are suggested, including requirements for research: (i) promotion of research aimed at improving and maintaining nutrient use efficiency at farm level; (ii) promotion of joint learning experiences between farmers and researchers, where farmers' organizations could act as "research consortia"; (iii) emphasis on site and time specific management (precision agriculture) in policy development, and provision of site-specific advice via modern information and communication technologies; (iv) clearer guidelines for groundwater monitoring procedures, including additional monitoring at greater depths and consideration of groundwater quality from an appropriate regional perspective; (v) groundwater monitoring should take place at locations selected according to specific hydro-geological characteristics, rather than being executed at random and (vi) clear goals that are defined within existing and future policies at EU and international level, should allow for regional differentiation in indicators; these being the outcome of negotiations between farmers or their representatives, policy makers and researchers.

Based on:

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5.1 Introduction

Policies on regulating nutrient emissions from agriculture date back to the early nineteen eighties for the Netherlands and early nineties for the European Union. These do not fully match which has resulted in the implementation of legal procedures and risk of high penalties. In the Netherlands, there is currently considerable debate as to how to move on with the existing national policy on manure now that the Advocate General of the European Court has declared that this policy is inadequate. This chapter explores the background of the current debate with special attention to nitrogen (N) and nitrate (NO_3^-) leaching. First, the policies from the EU, the Netherlands and the conflicts between them will be described. Considering the importance of nutrient transformations in the soil, it will be described how land units have been incorporated so far into Dutch policy. An alternative approach is suggested which is illustrated with a case-study from two environmental cooperatives (VEL & VANLA) in the northern Friesian Woodlands in the Netherlands. Finally, six recommendations are made which are believed to be relevant for future policy measures on nitrogen and nitrate leaching, involving a new role for agricultural research.

5.2 European and national manure policies

5.2.1 The European Union

In 1991, the council of environmental ministers of the European Union adopted the council directive concerning the protection of waters against pollution caused by nitrates from agricultural sources, 91/676/EEC (European Commission, 1991). This directive was signed under Dutch presidency and is also known as the Nitrates directive. The directive aims at reducing and preventing water pollution caused by nitrates from agricultural sources. Before the end of 1993, Member States were obliged to identify so-called ‘vulnerable zones’ and establish codes of good agricultural practices. These codes should require that farming is practiced in a way that minimizes the pollution of water by nitrates (Tunney, 1992). To realize the objectives of the Directive, Member States were to establish Action Programs. These should specifically result in nitrate concentrations in groundwater not exceeding 50 mg NO_3^- l^{-1}. Action Programs should include rules for (1) the periods in which the use of certain types of chemical fertilizer are prohibited, (2) the storage capacity for, or the disposal of, livestock manure and (3) limitation of land application of fertilizers. More specifically, account should be taken of local characteristics such as soil conditions (type and slope), climatic conditions and irrigation and balancing N supply with crop N demand, including both chemical fertilizer and manure. Action Programs should also ensure that the amount of livestock manure applied does not exceed the equivalent of 170 kg N ha^{-1}, corresponding with 2 livestock units per ha.

Originally, Member States could allow amounts of manure containing up to 210 kg N ha^{-1} for the first four year Action Program (1996–1999) but the final standard of 170 kg N ha^{-1} has been moved forward and was due to start from mid-December 2002 (De Clercq *et al.*, 2001). Moreover, Member States were allowed to derogate from this standard, but only on the basis of objective criteria relating to crop and soil characteristics. Some examples of criteria have already been provided in the Directive, such as a long growing season, crops with a high N uptake, high net precipitation and soils with a high denitrification capacity.

5.2.2 The Netherlands

By the time the Nitrates directive was adopted, reducing the impact of agriculture on the environment had already been on the Dutch policy agenda for some years. In 1984, the *Temporary Act Restricting Pig and Poultry Husbandry* was introduced to prevent establishment of new pig and poultry farms, and limit expansion of existing farms in specific areas. In 1987, this Act was followed by the *Soil Protection Act* and the *Manure Act*. Legislation based on these acts included application standards for animal manure based on phosphate content, and regulation on timing and methods of application. For the first phase from 1987–1990, farmers were allowed to apply the manure equivalent of 125, 250 and 350 kg phosphate ha^{-1} per year for arable land, grassland and maize land respectively. Originally, the standards were intended to be tightened in the second phase from 1991–1994 with final standards becoming operational in the last year of the third phase (1995–2000). In the early nineties however, it became clear that objectives would not be reached with these policies (CPB, 2000). Increasingly, it was realized that amounts of N also needed to be directly regulated (Schoumans *et al.*, 1998). Taking account of N in animal manure through phosphate application standards proved to be insufficient because of varying composition and, moreover, because a large quantity of N was being applied through chemical fertilizers. In 1995, additional measures were proposed by the Dutch Ministry of Agriculture (LNV, 1995). Now the focus moved from *application* standards to *loss* standards for both nitrogen and phosphate at individual farms. This farm-budget approach, called MINAS (MINerals Accounting System), was meant to comply with the EU Nitrates Directive (Van den Brandt and Smit, 1998). Starting from the 1st of January 1998, intensive dairy farms, later followed by most other dairy farms and arable farms, were obliged to reach given farm surplus standards in 2008/2010. These levy-free surplus standards were set at 20 kg ha^{-1} for phosphate, at 180 kg ha^{-1} for nitrogen on grassland, and at 100 kg ha^{-1} for nitrogen on arable land. For farm-surpluses exceeding these levels, levies should be paid. An overview of the original levy-free standards is given in Table 5.1.

Table 5.1. Original levy-free surplus standards for phosphate and nitrogen (kg ha^{-1}).

	1998	2000	2002	2005	2008/2010
Phosphate	40	35	30	25	20
Nitrogen on grassland	300	275	250	200	180
Nitrogen on arable land and maize land	175	150	125	110	100

5.2.3 Conflicting policies

The Netherlands was one of six Member States that designated the entire country to be a Nitrate Vulnerable Zone. In 1998, however, the European Commission made it clear that policies in the Netherlands were inadequate to comply with the Nitrates directive. Basically, the Commission considered MINAS to be an addition to their policy rather than an alternative. It stated that the surplus standards for the dry sandy soils were not sufficient and that the target should not be 2008/2010, but the last year of the second Action Program, 2003. Finally, the Commission also ruled that the levies for surpluses exceeding the maximum allowable surplus limits were too low.

In September 1999, the Dutch Ministers of Agriculture and Environment presented the letter *Integrated Approach on Manure Problems* (Brinkhorst and Pronk, 1999). In their letter, they formulated new proposals: (1) Shifting the final levy-free surpluses backward from 2008 to 2003, (2) Increasing the levies from EUR 0.68 to EUR 2.27 per excess kg of nitrogen and from EUR 2.27 to EUR 9.08 per excess kg of phosphate without further differentiation in excess surpluses, (3) Introduction of the Manure Transfer Agreement System (MTAS, Ondersteijn *et al.*, 2002) in which farmers with excess manure must have contracts to ensure the proper disposal of this excess. Maximum allowable amounts of animal manure on the land are, in this system, directly related to the standards of the Nitrates Directive. For N on arable land, 170 kg ha^{-1} is allowed while for grassland 250 kg ha^{-1} is allowed. This maximum for grassland is part of a derogation request from the Dutch government to deviate from the standard in the Nitrates directive and is based on a scientific report (Willems *et al.*, 2000). Nitrogen production is calculated from manure production using provided standards for each animal. Such standards are fixed amounts for specific groups of animals for a given year. They are set at 85% of the average expected manure production per animal for 2002 and at 95% for 2003, which implies that for average conditions, more N would be applied per hectare than requested in the derogation. Essential for the control of MTAS is the construction of a parcel registration system that contains geographic and land use characteristics of each parcel in the Netherlands. Meanwhile lower loss standards have been defined for the dry sandy soils (LNV, 2001b). A field is now defined as a dry sandy soil, or leaching-sensitive soil, when two-third of its area has a groundwater class (GWC) VI or higher (see Table 5.2). For the coming years, the most stringent loss standards however only apply to the 'dry leaching-sensitive soils' (GWC > VI) covering about 140 000 ha.

In combination with other regulations, MTAS and MINAS are supposed to comply with the Nitrates Directive. These other regulations, already mentioned in the original Nitrates Directive, include restrictions on the application period, the application technique for applying manure and the construction of storage tanks for organic manure.

The existence of different policy measures in Dutch manure policy has led to bizarre consequences. Farmers that do not meet MTAS requirements are obliged to arrange (expensive) contracts to sell their excess N. Many of them do, however, meet the MINAS requirements and this has led to so-called 'empty' contracts, which have no practical consequences except that they are quite costly. This phenomenon has been attributed to the use of fixed standards in MTAS and the fact that the actual disposal of manure in MINAS is calculated according to the balance for phosphate. Furthermore, the opposite occurs for some other intensive dairy farms. These have to arrange a smaller area for disposal of their manure than what would actually be necessary (RIVM, 2002).

Table 5.2 Definition of groundwater classes (GWC) according to groundwater table fluctuations (expressed in cm below surface).

	I	II	III	IV	V	VI	VII
Average highest groundwater table	0	0	< 40	> 40	< 40	40–80	> 80
Average lowest groundwater table	< 50	50–80	80–120	80–120	> 120	> 120	> 120

5.2.4 The need for change

Nutrient budgeting, as performed with MINAS, facilitates understanding the effects of farm management on nutrient use efficiency, using the nutrient surplus as a performance indicator (Neeteson, 2000). The introduction of MINAS was facilitated by the observation that farmers are quickly able to understand and to work with farm budgets. The approach appears attractive as it focuses on the individual farmer (Van den Brandt and Smit, 1998). Farmers have the flexibility to follow different management procedures to decrease the N-surplus to acceptable levels. Indeed, considerable achievements have been made in the last years. N surpluses at farm level reduced from 400 kg ha^{-1} in 1986 to 225 kg ha^{-1} on average in 1999/2000 (RIVM, 2002). From 1998 onwards, part of this achievement can be attributed to MINAS. However it is also realized that with current MINAS standards for 2003, final objectives for groundwater quality will not be met (RIVM, 2002a). Bouma *et al.* (2002) demonstrated that meeting MINAS requirements does not imply that quality standards for groundwater are met as well. In fact, the relationship between MINAS surpluses and groundwater quality is unclear. In addition, there are severe administrative problems with MINAS: the Dutch Government Accounting Office has reported that there is a backlog of data administration and that the accuracy of data is poor (Anonymous, 2001). Farmers complain about the high administrative burden involved and about the restrictive character of the regulations, leaving little room for creative and innovative management measures focused on improving groundwater quality, the original objective of the nitrate guideline.

Integrated N-approaches have been suggested in other publications (Davies, 2000; Jarvis, 2000; Lord and Anthony, 2000). These approaches vary from farm or catchment scale to even national scale levels. The main objective of most integrated N-studies is to improve the efficiency with which this nutrient is used (Davies, 2000). Efficiency refers to the conversion of input of N into desired output for a defined system. Following these developments, integrated N-policies have also been suggested for the Netherlands which, according to Erisman *et al.* (2001), should at least have the certainty that the pressure of impact is not shifted in time, to areas abroad or towards other (environmental) themes. The concept of nitrogen use efficiency does not, however, directly refer to losses to the environment at lower scale levels. Even though farms may have a high efficiency at farm level, specific fields with a high leaching risk may still receive too much N and exceed the groundwater quality standard.

With respect to N, the correction of existing policy measures to reach quality objectives for ground- and surface water (and air) has recently received more attention than the quality objectives themselves. Soils, which form the essential link between fertilization activities and losses to the environment can, however, be approached differently than has been done so far, as will be discussed in the next section.

5.3 Perspectives on land units

5.3.1 Policy and science: development of MINAS surplus standards

For the last twenty years, government policies on manure regulation were mainly characterized by quantitative aspects that needed to be addressed. National statistics on livestock production reported substantial increases in the numbers of cattle, pigs and chickens from 1950 to 1980, with a corresponding increase in manure production (CPB, 2000). Nutrient surpluses expressed in kg ha^{-1} yr^{-1} were the highest in the world (Schoumans *et al.*, 1998). These figures were so evident in the 1980s that warnings by experts on the environmental impact of excess nutrients could no longer be neglected or ‘neutralized’ by the Ministry of Agriculture (Frouws, 1994). Policies were developed but these showed no differentiation between soils in the first and second phase. There were only different standards related to land use, covering grassland, maize land and arable land.

One of the guiding principles in earliest legislation was ‘balanced fertilization’. The objective was that in 2000 no more phosphate and nitrogen should be applied to the land than a crop could take up (Dekker and Van Leeuwen, 1998). In other words, losses to the environment should equal zero. However, due to an increased awareness that not all losses to the environment are harmful, and that certain agricultural losses are unavoidable, the focus changed from application standards to loss standards (Van Leeuwen *et al.*, 1995). To come up with standards for losses at farm level, a project team started in 1994 with representatives from both the agricultural sector and the government (Dekker and Van Leeuwen, 1998). In this project, use was made of two types of surpluses. The *agricultural surplus* was calculated using most input and output fluxes at farm level, while the *environmental surplus* was calculated using terms for leaching to ground- and surface water, denitrification, NH_3 emission and immobilisation/ mineralization (Van Eck, 1995). Included were objectives as the 50 mg NO_3^- l^{-1} threshold for groundwater. A range of environmentally acceptable N losses resulted from this desktop study using the standards for ground- and surface water. These are given in Table 5.3 (Van Eck, 1995). The MINAS standards, as finally accepted, are a compromise between what is environmentally acceptable and what is attainable for agriculture (Schoumans *et al.*, 1998) with an additional correction for the dry sandy soils, which are particularly vulnerable to nitrate leaching. Still lower allowable losses, as defined by MINAS, were considered unacceptable for socio-economic reasons (Dekker and Van Leeuwen, 1998; RIVM 2002). However, another important problem for the "rational development of standards" was the apparent lack of information from relevant research. Results from agricultural and environmental research on nitrogen and phosphate behavior in the soil could not be combined satisfactorily, and the available models did not seem to match very well (Van Eck, 1995). According to Dekker and Van Leeuwen (1998, p. 142), "the available agricultural research on fertilization had hardly given any attention to environmental impacts of agricultural practices". Soon after the implementation of MINAS, studies showed that objectives would not be met, especially not for the dry sandy soils (Oenema *et al.,* 1998; Schoumans *et al.*, 1998). The lack of differentiation in soil types did not change with the introduction of MINAS. Final levy-free nitrogen surpluses were originally defined only for grassland and arable land respectively while the loss standard for phosphate was not differentiated at all. Apparently, more than half a century of research in soil science (Buurman and Sevink, 1995) had not resulted in any substantial differentiation of soils within these agricultural policies except for the 'dry sandy soils' (LNV, 2001b).

The approach taken for defining levy-free surpluses was supposed to follow a rational chain of (1) definition of environmental quality standards, (2) definition of environmentally acceptable losses and (3) implementation of restrictions/consequences for agricultural practices (Dekker and Van Leeuwen, 1998). Between the first and second step, land units are implicitly introduced. The land units (combinations of major soil types and groundwater classes) were grouped into six categories initially using classical subdivisions that were made for recommendations on N fertilization. Moreover, land qualities such as moisture supply capacity, soil organic matter content and nitrogen supply capacity were supposed to be difficult to influence by the farmer and were therefore not considered (Van Eck 1995 p. 37.).

Table 5.3. Ranges of environmentally acceptable losses (kg N ha^{-1}) for grassland and arable land as a function of soil type and drainage status.

	Sand (wet GWC[a] < VI)	Sand (medium GWC = VI)	Sand (dry GWC > VI)	Peat, Clay
Grassland	50–250	50–115	70–130	80–170
Arable land	10–115	10–50	25–65	35–200

[a] GWC = groundwater class

5.3.2 Land units as contextual phenomena

Soils, once classified and mapped, are often regarded as static (Bouma, 1994). However, considerable ranges in properties exist within basic categories recorded in databases, partly as a consequence of land use. Recent publications have related the variability of these properties to farm management and land use history. For any one map unit or soil series present in the database, variations resulting from different land use or management practices can be observed in the field. Properties such as bulk density, soil organic matter content and organic N have been found to vary substantially within one unit. Droogers and Bouma (1997) tried to capture this by introducing the concept of genoforms (the genetically defined soil series) and phenoforms, (expressions within one soil series resulting from different types of management). Recently, Sonneveld *et al.* (2002) were also able to refine soil survey information by using land use history to better assess the environmental impact of fertilization strategies on grassland. This dynamic approach of looking at the soil resource is not only much more realistic than the static approach, but it also represents a very different way of research. For a given area, basic soil processes operate on a geologic time scale and, to a certain extent, limit land use in its economic potential and environmental constraints, but not to an extent that eliminates all human creativity. Looking at land units as phenomena which act as windows of opportunity (e.g. Bouma, 1994), provides a challenge for both agriculture and research. Potential solutions to environmental problems, then, should not solely originate from sources outside agriculture, using e.g. desktop studies, but should also come from within farms themselves.

At farm level, aims are to integrate livestock, fields, land-use and technology in such a way that farm objectives are achieved. In practice, considerable heterogeneity in these objectives exists, which can be related to different farming styles (Van der Ploeg, 1991). Associations that exist between farms in rural areas or, more specifically, in regional environmental cooperatives, can also have a specific farming style in common. This acts as a complex set of experiences, rules and interpretations governing a specific way of farming (Van der Ploeg, 1991). Social interactions that exist among farmers influence efficiency at farm level, which might not have been achieved without this structure. Cross-farm visits, discussions and mutual dedication to formulated objectives can result in much synergy.

5.4 VEL & VANLA case study

5.4.1 Environmental cooperatives

Two environmental cooperatives, VEL (Vereniging Eastermars Lânsdouwe) and VANLA (Vereniging Agrarisch Natuur- en Landschapsbeheer Achtkarspelen) have been working actively since 1992 towards achieving a clean environment, maintaining an attractive landscape and to further consolidate natural elements. Located in the Friesian Woodlands of the province of Friesland, they cover almost 60 dairy farmers in an estimated total area of about 15 000 ha (Fig. 5.1). Farms are on average 49 ha in size, of which most is used as grassland and a small percentage (5%) is used for the cultivation of silage maize. About 16% of the area consists of clay, 9% consists of peaty soils and almost 60% consists of sandy soils, most of which are strong loamy. Of these sandy soils, about 39% are classified as ‘dry sandy soils’ (GWC VI and higher). The area covers the northern part of a till plateau and the majority of the sandy soils (60%) have glacial till within 1.2 m depth. This till varies in thickness to a maximum of several meters below surface, covering well-sorted pro-glacial sands.

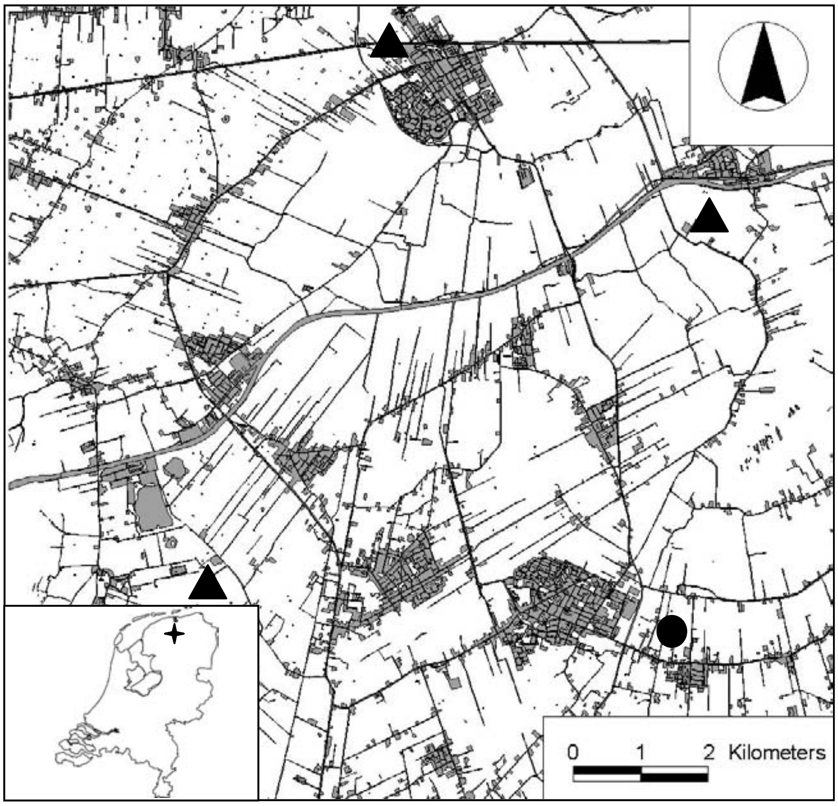

Figure 5.1. Location of the VEL & VANLA area in the Netherlands. Monitoring locations from the NMN are indicated by (▲) whereas the monitoring location from the PMN is indicated by (●).

5.4.2 Environmental performance

In 1998, the project VEL & VANLA was initiated, which specifically aimed at reducing nutrient losses to the environment and also involved researchers from Wageningen University. Over the years, the VEL & VANLA farmers have shown considerable achievements. MINAS-N surpluses decreased from 261 kg ha^{-1} in 1997/1998 to 163 kg ha^{-1} in 2001/2002, a great deal of which could be attributed to a decrease in the use of fertilizer-N. The percentage of farmers that met with the final MINAS standards of 2003 increased from 11% to well over 60% for the same period. To illustrate their environmental performance, in the year 1999/2000 33% of all dairy farmers in the project reached surpluses below the levy-free surplus standards of 2003 while the achievement was only 10% for dairy farms at national level (RIVM, 2002). More importantly, the efficiency of N use at farm level increased from 21% to 33%.

5.4.3 Nitrate leaching risk within one 'homogeneous' unit

Combinations of soil types, hydrological units and land use are often grouped into so-called ‘homogeneous’ units which are supposed to have uniform responses in nitrate leaching given a certain input of N from fertilizer and animal manure. For the VEL & VANLA area, about 80% of the land use consists of grassland and 60% is sandy soil of which half belongs to one soil series with an anthropogenic topsoil, classified as cHn23. More than half the area of the cHn23 (65%) has a GWC of VI. The combination cHn23/GWC VI/Grassland is therefore one of the most common combinations in the VEL & VANLA area. Because of differences in land use history, however, a wide range of soil properties such as soil organic matter and soil organic nitrogen can be found within this combination. As a consequence, nitrate leaching will not be uniform with respect to N management. To evaluate this, model simulations using the models SWAP (Kroes *et al.*, 1999) and ANIMO (Kroes and Roelsma, 1998) were performed using this combination of cHn23/VI/grassland and using, in addition, data from fields with differences in land use history. One field was considered to be ‘old pasture’, one field was considered to be ‘reseeded pasture’ and one field was recently converted from a long period of ‘silage maize cultivation’. Only grassland was simulated using the fertilization data from one field in the VEL & VANLA area in the year 2000. Both N from fertilizer (159 kg) and N from slurry (168) was taken into account making a total of 327 kg N ha^{-1}. Because yearly differences in weather have a major impact on N transformations, simulations were made for a 30-year period using real weather data. The applied N-levels were assumed to be the same for each year. No direct deposition of N through manure and urine during grazing was taken into account in this simulation, nor were fertilization activities related to weather conditions in any given year as they were set at fixed dates. Further details are given by Sonneveld and Bouma (2003). Fig. 5.2 shows the results of a 30-year model simulation as cumulative probability graphs for the groundwater at 1.7 m depth. Between different years, there is considerable variation in the mean annual nitrate concentration, which ranges from 0 to 80 mg l^{-1}. The probability of exceeding an average annual concentration of 50 mg NO_3^- l^{-1} ranges between 0 and 20% for the same soil unit under grassland. Previous arable land never exceeds this standard because of immobilization of N in the soil, but old grassland is at higher risk. Fig. 5.2 indicates that this combination of soil, GWC and land use type cannot be unambiguously evaluated with respect to nitrate leaching.

The amount of N applied through organic manure does not exceed the standard of 170 kg ha^{-1} and in most years, the average annual concentration of nitrate in the groundwater at 1.7 m depth does not exceed the threshold of 50 mg NO_3^- l^{-1}. Nor are higher levels of N applied through organic manure expected to lead to large concentrations in the groundwater, especially not at greater depths (see next section). Because of relatively low numbers of livestock at each farm, no manure contracts for MTAS have been signed for 2002, when an application standard of 300 kg N ha^{-1} for grassland was operational. When, however, the strictest standards for MTAS are introduced (170 kg N ha^{-1}) and full account is taken of N production for each animal, this situation will radically change. All farmers will then exceed the standards and further action will need to be taken. It is estimated that, on average, a farmer needs to make a contract for an extra 20 ha of land to meet the EU standard of 170 kg ha^{-1} (Verhoeven and Dijkstra, unpublished data).

5.4.4 Nitrate monitoring

Concentrations of nitrate in the groundwater are monitored in the study area as part of three different operational monitoring networks. The Soil Quality Monitoring Network (SMN) of the province of Friesland was initiated in 1996 and approximately 10 locations are situated in the VEL & VANLA area, mostly distributed throughout the southwestern part. Concentrations of nitrate in the upper meter of the groundwater were measured in 1996, 1999 and 2000, and are described in separate reports (CSO, 1997; CSO, 2001). Values were specifically reported for grassland on sandy soils with an anthropogenic topsoil for the whole province. Median values for the separate years were 56.5 (n=72), 61 (n=35) and 83.5 (n=32) mg l^{-1} respectively, indicating that more than 50% of the samples in all three years exceeded the standard of 50 mg l^{-1}. Concentrations of nitrate in the groundwater are also measured at three locations that are part of the Netherlands National Groundwater quality Monitoring

Network (NMN) (Reijnders *et al.*, 1998). These locations are indicated in Fig. 5.1. The two locations in the western half of the map have been described as grassland on organic-matter rich sandy soils, whereas the location in the eastern half of the map has been described as marine clay. Measurements have been performed annually from 1980 onwards, often in late summer or early autumn, and samples were taken from 8–10, 12–14 and 23–25 m depth. All samples taken showed nitrate concentrations below 20 mg l^{-1}, which is well below the EU standard, and most samples were even below 1 mg l^{-1} (98%). Measurements at the two sandy locations in the early eighties revealed decreases in total organic carbon content at greater depth. These decreases were on average 29% between 8–10 and 12–14 m and 55% between 8–10 and 23–25 m. Denitrification is likely to be one of the processes causing this decrease.

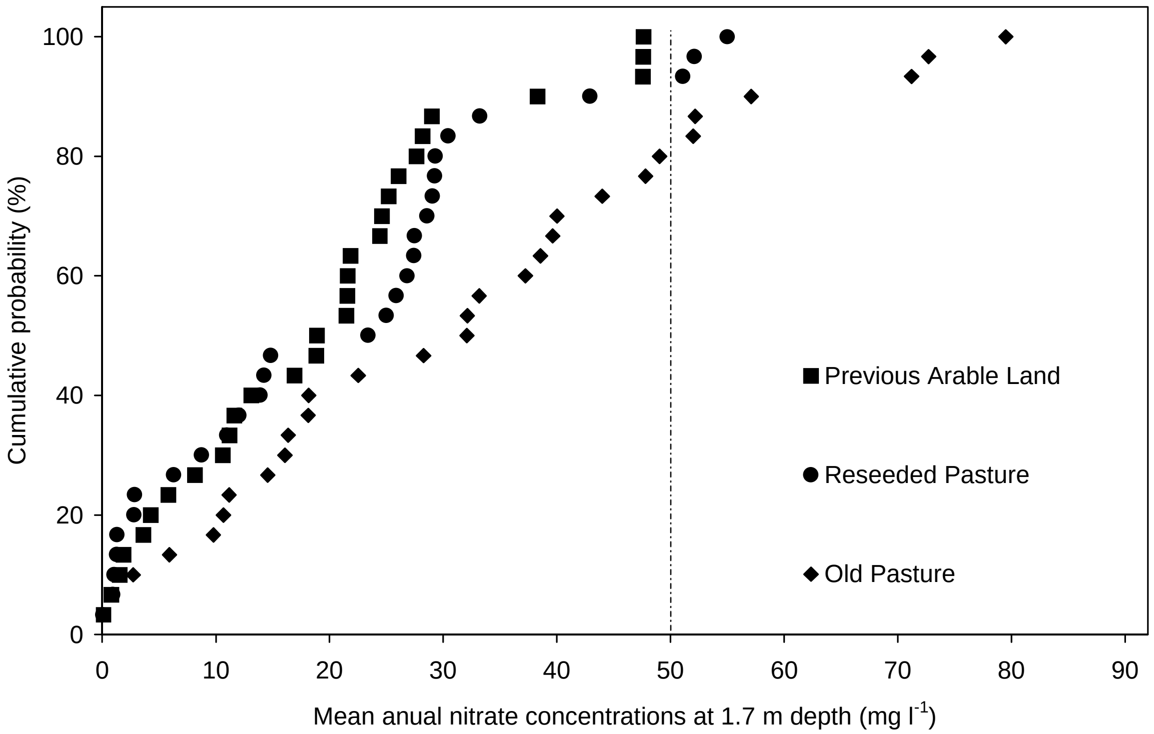

Figure 5.2. Cumulative probability curves of the mean annual nitrate concentration at 1.7 m depth for one soil map unit (cHn23-VI) under grassland. The EU standard of 50 mg l^{-1} is indicated by the vertical dotted line.

Finally, information is also collected within the Provincial Groundwater quality Monitoring Network (PMN) involving one location in the southeastern part of the study area (Fig. 5.1). At this location, measurements have been taken annually from 1994 onwards at depths similar to the NMN. Nitrate concentrations in the 8–10 m layer are on average twice as high as the EU standard of 50 mg l^{-1}, but are always below 2 mg l^{-1} in the deepest layer (23–25 m). Again, total organic carbon content of the samples shows a decrease of 37%, on average, from the uppermost layer to the deepest layer, suggesting denitrification. No additional information on land use or soil type is provided for this location, but this section of the study area generally has a much thinner topsoil with lower organic matter content compared with the rest of the study area. This would probably lead to less denitrification in the upper layer of the groundwater.

Thus, significant differences in nitrate concentrations have been observed at different groundwater depths, between the SMN location and the NMN and PMN locations, respectively. All locations from the NMN and the PMN show very low nitrate concentrations (< 1 mg l^{-1}) in the deepest section of 23–25 m. Drinking water is extracted in the area at depths exceeding 60 meters below surface. Clearly, if denitrification is a significant process in the region, the drinking water quality standard of 50 mg NO_3^- l^{-1} at 1 m depth represents a very high safety margin.

5.4.5 Heterogeneity within a mapped hydrological regime

The actual fluctuation of the groundwater level within a given field can vary considerably in the VEL & VANLA area, even within one GWC. In Fig. 5.3, the fluctuation of the groundwater level in the year 2000 is indicated for 4 pasture-fields, belonging to different dairy farms but with the same sandy soil and with GWC VI (StiBoKa, 1981). During summer, groundwater levels may show differences of up to 1 meter but differences become smaller in the wet season (autumn and early spring). The location of a particular field is of importance in understanding such differences. Fields 1 and 4 are located on the till plateau (in the western part of the area) and can experience rather wet conditions after rainfall events in autumn and spring because of the slowly permeable subsoil. In summer however, the groundwater level drops below the till layer. Fields 2 and 3 are located in the lower central part of the area on the edges of the till plateau. These locations are likely to receive groundwater from the surrounding area resulting in wetter conditions throughout the summer period. The depth of the anthropogenic topsoil, which is rich in organic matter, is around 50 cm for these soils. For early autumn, this indicates larger potential losses of N through denitrification in fields 2 and 3 compared with fields 1 and 4. Interpretations of nitrate monitoring data should take account of the effects of site location. Random placement of observation piezometers can lead to great variability, much of which can be explained by considering geological and landscape conditions. Considering such conditions when selecting observation locations is therefore strongly recommended.

5.4.6 Land units in context

Fig. 5.4 is presented as a scheme to conceptualize the above. Combinations of soils, hydrology and land use are described in this figure as spatial contextual phenomena. The agricultural context, in the top-part of the Figure, is largely influenced by human decisions and specific management. The location of the field with respect to farm buildings, the strategy of the farmer or existing landscape management programs that are shared among farmers influence land use (grassland or arable land) and land management (fertilization, harvesting, etc). The environmental context, the lower part of the Figure, is largely influenced by the structure of the environment and natural processes that exist within. The terminology of a certain combination, as e.g. “grassland on dry sandy soil”, will surely capture some of its characteristics but specific properties and dynamics are *also* relevant to asses the environmental impacts of agricultural practices. Specific land use history, subsurface properties and hydrological regime can, to a great extent, determine whether or not environmental thresholds are exceeded.

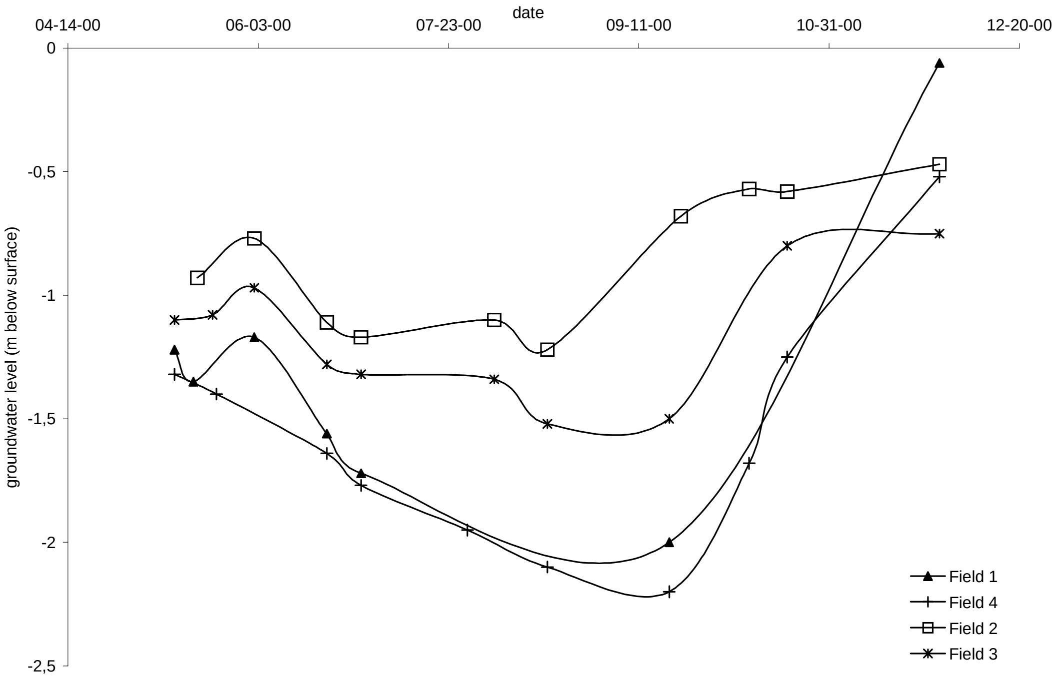

Figure 5.3. Groundwater level fluctuation for 4 fields with GWC VI in the VEL & VANLA area in 2000.

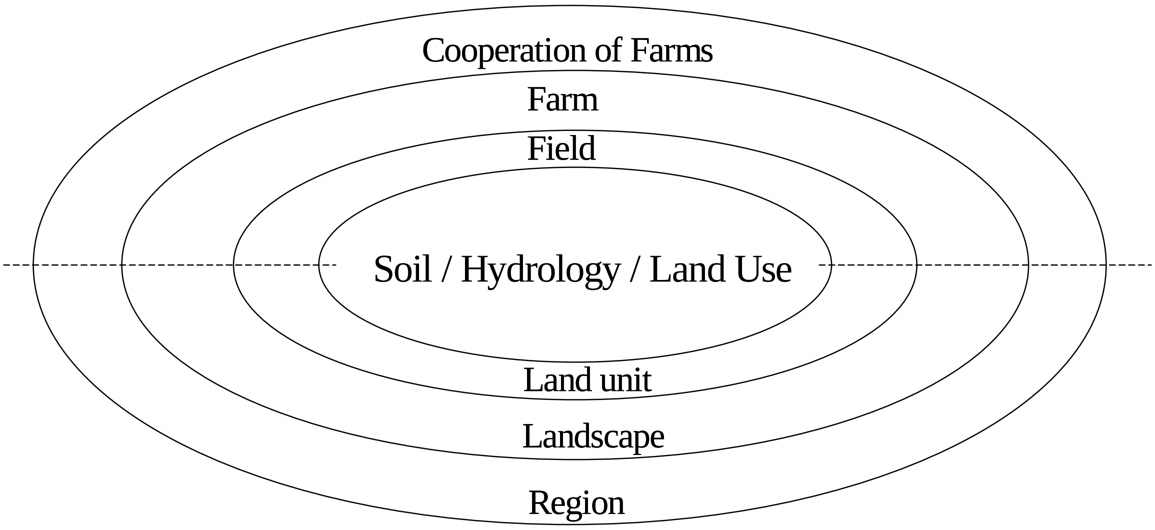

Figure 5.4. Schematic representation of a standard combination of soil type, hydrology class and land use type as existing within the context of specific agricultural activities and natural processes.

5.5 Basic considerations for new policy measures

New approaches are to be formulated by close consultation and negotiation between policy makers, stakeholders, NGO’s and researchers. This chapter suggests a number of considerations, on which this negotiation process could conceivably be based, including requirements for research:

i) Improving and maintaining high nutrient use efficiency, especially at farm level should be a key aspect of both agricultural policy and agricultural research. Opportunities should be encouraged for farmers and researchers to design innovative management procedures that lead to desired environmental performances. Some farmers already have a high nutrient use efficiency at farm level. Their management practices should be studied, documented and communicated widely by using, for example, nutrient budgeting tools at farm level. A systems approach, which considers the entire farming system and all interrelated N fluxes in soil, air and water, is needed. Modern information and communication technology (ICT) provides the opportunity to distribute such knowledge widely. Particular attention should be given to the study of soil processes in farms where nitrate leaching occurs at acceptable levels. Similar land units can follow different land use trajectories, which implies that standard evaluation procedures with respect to nitrate leaching using homogeneous units may not be appropriate anymore. Again, ICT gives the opportunity to facilitate interactive research between farmers and researchers.

ii) Joint learning experiences between farmers and researchers, as suggested in point (i) are likely to be more successful when occurring in the context of regional farmers' organizations such as environmental cooperatives. For governmental agencies to deal individually with some 90 000 farmers is extremely difficult, both from the viewpoint of enforcement of regulations and of research cooperation. Conceivably, such co-operatives could also become responsible for enforcing regulations and could be funded to guide and guard research procedures by forming 'research consortia'.

iii) Rules and regulations that focus on emission pathways but also on issues as animal welfare should be mutually consistent. The apparent incompatibility of the current MINAS and MTAS rules, which constitute the two pillars of Dutch manure policy, is unacceptable. A farm nutrient budgeting approach (such as MINAS) provides a clear management instrument for farmers, but the relationship between groundwater quality and the EU guideline of 170 kg N ha^{-1} derived from organic fertilizer is undefined. Application standards currently used in MTAS have a direct relationship with this EU guideline, although these are currently implemented at farm level and not at field level. As stated above in the VEL & VANLA case, introduction at this time of the 170 kg N ha^{-1} limit within MTAS would have severe consequences: many farmers would need to buy extra land or sign manure contracts. A standard of 170 kg N ha^{-1} from manure was defined but the Nitrates directive gives examples of objective scientific criteria to justify higher amounts of N application from organic manure. Also, the use of chemical fertilizers is mentioned in the directive. Basically, the Nitrates directive aims at meeting the 50 mg l^{-1} standard for groundwater by taking into account specific soil types and soil conditions, weather conditions, and supply of nitrogen through mineralization. In other words, the Nitrates Directive aims at *Site and Time Specific Management (precision agriculture).* Research should play a much more active role here, by focusing on the objective to balance supply and demand of nutrients to plants to the effect that excess nutrients will not be lost through leaching or ambient emissions. Using data from the parcel registration program, and soil and crop specific information in combination with actual weather data and dynamic models with various levels of complexity, an online service could be constructed providing fertilization recommendations for both chemical fertilizer and organic manure at field level (e.g. Bouma et al, 2002).

iv) Water quality is the central objective of the EU nitrate guideline, which specifies a threshold value of 50 mg l^{-1} nitrate for groundwater, to be measured in the ‘upper’ part of the aquifer, and thresholds for the quality of surface water. The threshold for groundwater has been interpreted in the Netherlands in terms of water quality of the upper meter, but the Directive does not specify any depth or monitoring frequency. At greater depths, concentrations often rapidly decrease because of denitrification, as has been shown for the VEL & VANLA area. Concentrations in sandy soils can decrease by 35% over a range of 1.5 m whereas concentrations in loamy soils have been reported to decrease by 60% over a range of 3.5 m in the upper groundwater (Kronvang *et al.*, 2002). The Nitrates Directive allows measurement depths from 0–5 meters, as practiced in Denmark. There is no reason why this should not also be done in the Netherlands, which would put a new perspective on the entire problem. The objective here is not to shift the pressure of impact of agriculture to other pathways but to realize that in some areas natural processes such as denitrification do not justify a rigid short term implementation of very strict measures. A reasonable time-path such as that followed by the VEL & VANLA farmers to reduce their N losses appears to be much more appropriate (see point i).

v) Water quality measurements vary considerably in space and time, partly as a result of soil heterogeneity or of geologic conditions. Monitoring the concentration of nitrate in groundwater at every agricultural field throughout the year is not feasible. High spatial and temporal variability, especially under grazed pastures, is therefore likely to result in low accuracy or high cost due to intensive monitoring. In addition, nitrate concentration in the groundwater reacts rather slowly to changes in nutrient management at farm level. This is even more pertinent when considering nitrate concentrations at pumping stations for drinking water, because of the considerable lag times and the fact that field fluxes from different farms converge to such locations. Measurements should therefore be made in a given representative area, such as a well-defined watershed, at locations that have been selected after studying hydro-geological conditions, as is also practiced in Denmark. This will ensure representative and consistent measurements of water quality and can be beneficial for both understanding environmental processes as well as evaluating the effectiveness of existing regulations.

vi) A different approach to nitrogen policies, and environmental policies in general, becomes more urgent because of the new farm policies to be implemented by the European Union: price support will gradually be replaced by income support for farmers to be conditioned upon the quality of their management in terms of environmental and product quality and animal well-being. Clear goals, formulated at EU level and beyond, should allow for different indicators as means to reach these goals. Differentiation at regional level is able to take account for natural regional variation and provides room for negotiation with farmers or their representatives. Both in the Netherlands, in other EU countries as well as beyond EU boundaries, this approach can stimulate the formation of relationships between farmers, researchers and policy-makers that build on trust and joint efforts to reach overall objectives.

Chapter 6

Dynamics of land degradation in communal grazing areas; a case study for KwaZulu-Natal, South Africa

Abstract

For a case study area in the Okhombe catchment in the province of KwaZulu-Natal, South Africa, a multi-scale analysis of land degradation dynamics was performed. At sub-catchment level, the dynamics of erosional features were investigated by means of aerial photographs. At site level, the dynamics of soil properties were investigated by means of a fence-line contrast study. Both external and intrinsic factors related to erosional features were studied.

At sub-catchment level, an increase in the number of erosional features of mainly strongly active and weakly active gullies was observed from 1975 to 2000 but this followed a substantial inactivation of erosional features from 1962 to 1975. This is most likely due to a wet spell in combination with a low population density in the area; measured by the number of dwelling units. It was also found that increases in erosional activity in 1962 were related to abandoned cultivated fields.

At site level, a significant decrease in soil C/N ratio was observed for the fenced site within 3 years. Total carbon, saturated hydraulic conductivity and bulk density were not significantly different inside the fenced area compared with outside. Textural breaks and transitions between permeable and less permeable layers appear to be the driving factors for subsurface erosion features instead of chemical characteristics. In the study area, it was furthermore found that local people have the capacity to participate in experiments that can lead to improved understanding of the system and implementation of sustainable natural resource management.

based on: Sonneveld, M.P.W., Everson, T.M. and Veldkamp, A.
Dynamics of land degradation in communal grazing areas;
a case study for KwaZulu-Natal, South Africa.
Land degradation and development (submitted)

6.1 Introduction

Soil erosion is regarded as one of the biggest environmental problems in South Africa (Laker, 1999). Reports on the extent and intensity of soil erosion in this country date back to the first decades of the twentieth century (e.g. Jacks and Whyte, 1939), indicating that it has long been recognised as a serious issue and a key component of land degradation. Past official perspectives especially perceived communal grazing areas as being heavily degraded beyond recovery with overstocking and overgrazing as the main driving factors (Harrison and Shackleton, 1999). As a consequence, past and present practices to prevent further land degradation have included destocking, rotational grazing schemes to rest areas and restrictions on the use of key resources (Scoones, 1992). To reorganise land use practices and centralise formerly scattered residences into nucleated settlements, rehabilitation or 'betterment' programmes were designed in rural black areas in the middle of the twentieth century. Stock reduction was enforced to curtail overgrazing, common land was allocated for rotational grazing, arable land was consolidated and soil conservation measures were implemented (Cooper, 1991; Dollar and Goudy, 1999). In a way, the former homelands were subject to 'social engineering' on a grand scale (Auerbach, 1999). Local people were hardly involved in the development of management systems of their natural resources (Behnke and Scoones, 1993). Livestock farming in black rural areas had been, on the whole, dismissed as being traditional and backward (Adey *et al.*, 2002; Cooper, 1991).

Local people often do recognize the problem of soil erosion but have different perceptions on this issue compared with what is expressed in official views or held by researchers (Cartier van Dissel and De Graaff, 1998). They can be well aware of the dynamics of soil erosion (Critchley, 1998), they are sometimes able to identify key factors that are important in soil erosion processes (Scoones, 1992) and they can be effectively involved in the development and evaluation of alternative land use practices (Auerbach, 1999).

Though communal grazing areas are currently also seen predominantly as being strongly degraded (Meadows, 2003), recent insights in the dynamics of these systems have indicated that soil erosion processes involve more complex interactions between land use, climate and soil properties than what has been assumed in historic interventions. Studies of the dynamics of soil erosion phenomena by means of sequential aerial photographs in combination with analyses of land use, settlement patterns and climate variables have indicated that alternating stages of increased and decreased land degradation occur (Kakembo and Rowntree, 2003; Marker, 1988; Meadows, 2003; Watson, 1996). The fence-line contrast study of Harrison and

Shackleton (1999) in communal lands, including sites throughout South Africa with grazing and without grazing, showed for Sourveld a remarkable resilience in vegetation, especially in species composition and in basal cover. This may be attributed to its rapid maturity and consequent unpalatabilility during the dry winter season. These stages of recovery may for example be induced after drought periods with cattle losses (Harrison and Shackleton, 1999). Next to the findings that stages of land degradation are actually dynamic instead of irreversible and continuously deteriorating, these studies therefore also indicate that other external factors apart from overstocking and overgrazing play a role in land degradation dynamics.

Other studies have highlighted that intrinsic factors play a role as well. Botha (1996) revealed strong relationships between gullies and specific bedrock types, which indirectly influence slope profiles and colluvial layers. Rienks *et al.* (2000) studied a gully in KwaZulu-Natal and found that erodibility was strongly correlated with the values for exchangeable sodium percentage (ESP) and sodium adsorption ratio (SAR). Others (e.g. Laker, 1999) have indicated the susceptibility to erosion of duplex soils with a sharp textural break and a permeable horizon overlying a less permeable horizon. These properties may also lead to soil piping through subsurface erosion (Bocco, 1991).

Few studies have so far combined assessments of soil erosion dynamics and their drivers at different scale levels such as (sub-) catchment level and site level. Such approaches are however considered to be relevant if future steps are being taken towards controlled grazing management in communal areas. In degraded communal grazing areas, there is therefore a need for multi-scale analyses that identify the dynamics of land degradation processes as well as the dominant factors that contribute to them. Moreover, there is a need to involve community members in interventions within these degraded areas.

For a case study area in KwaZulu-Natal in South Africa, a multi-scale approach will be presented. First of all it will be investigated whether there has been a consistent increase in the occurrence and intensity of soil erosion in past decades. Secondly, it will be investigated how a recent exclusion of grazing and burning from a selected area in communal grazing land affects some basic soil properties. Thirdly, both intrinsic properties as well as external factors, which are most likely to contribute to erosion processes, will be identified. More specifically, the role of land use and land use change in soil erosion will be evaluated. Finally, aspects that need to be considered in the evaluation of interventions in communal areas will be discussed.

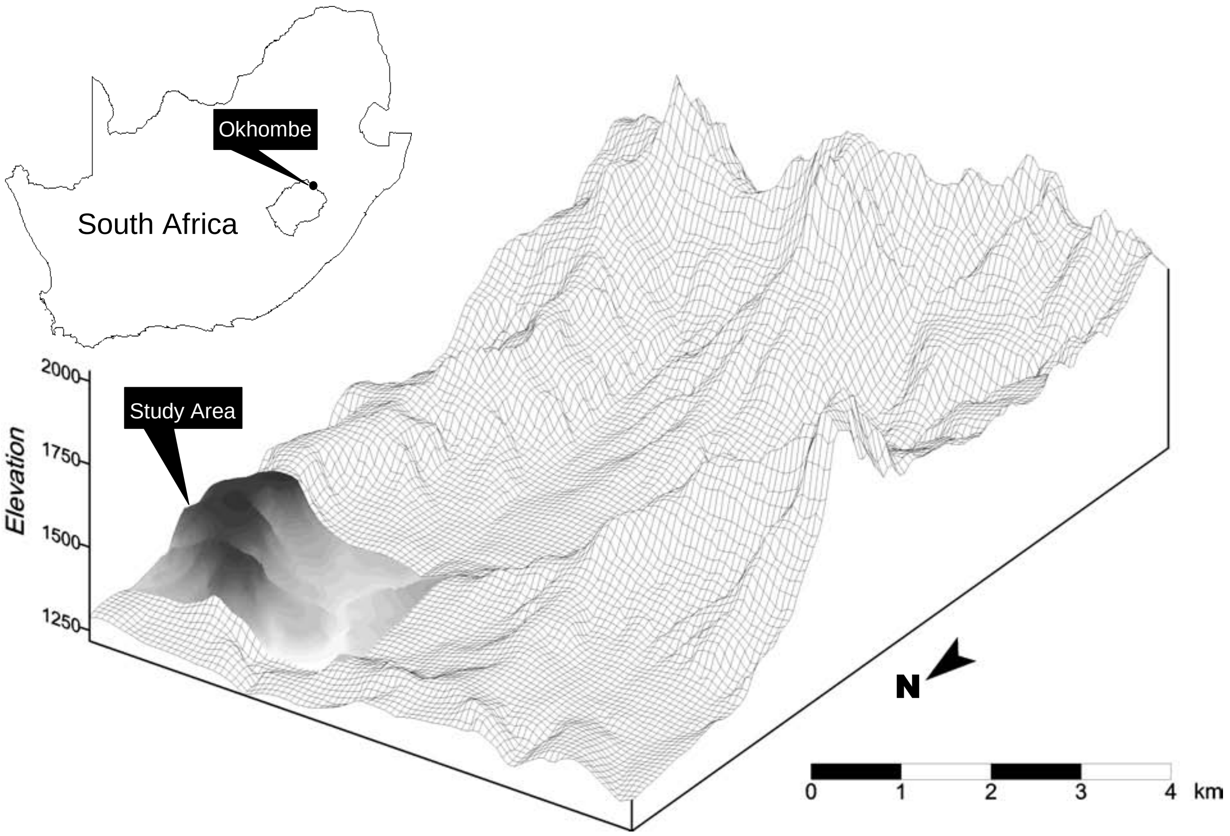

Figure 6.1. Location of the Okhombe catchment in South Africa and the study area within the catchment.

6.2 Study area

6.2.1 Description

The study area covers part of the catchment of Okhombe (28°42' S; 29°05' E), located in the Upper Thukela[2] catchment area in the province of KwaZulu-Natal, South Africa (Fig. 6.1).

Its main geology consists of rock types of Triassic and Permian age, belonging to the Beaufort Group (Verster, 1998). The plateau consists of sandstone of the Tarkastad formation whereas the lower area consists of shales and sandstone of the Adelaide formation. In the lowest part, dolerite of Jurassic age is present. The valley is located between 1000 and 1500 m altitude and receives some 800 – 1000 mm of rain per annum. Most of the precipitation falls in the months October to March (Schulze, 1997).

The Okhombe ward consists of six subwards of which one, Ngubhela, is covered by the study area indicated in Fig. 6.1. In general, the Okhombe community, consisting of about 4000

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[2] Thukela is the Zulu name for Tugela

inhabitants, relies heavily on the surrounding natural resources for their daily living. Approximately 4000 head of cattle and 2000 small stock, mainly goats, occupy the area. Grazing on the hillslopes takes place in summer (September – May) while in winter, cattle are allowed to graze the remains of crops, mostly maize stalks, in the bottom of the valley since grass has then become unpalatable. In most of the area, burning of the vegetation is a common practice, which takes place in and around August, although out of season burning during the growing season is also common.

Based on slope and parent material, five different landscape units can be distinguished: 0) River Valley, 1) Footslope, 2) Midslope, 3) Upperslope and 4) Plateau. These are located between elevations of 1250–1300 m, 1300–1330 m, 1330–1360 m, 1360–1425 m and 1425–1500 m respectively (Fig. 6.2). Data for soils located within landscape units 1, 2 and 3 are given in Table 6.1 with classifications according to the Bionomial Classification System (Soil Classification Working Group, 1991). Soil profile 1, located in an arable field on a gently sloping hill (2%) and convex-convex curvature is classified as a Hutton Form or as a Rhodic Ferralsol and is located on dolerite parent material. Soil profile 2 was located on a side-wall of an actively eroding gully with soil pipes and cavities existing in close vicinity. The surface was slightly sloping (5%) with straight-convex curvature and high grass cover (> 80%). The soil was classified as a Sepane Form or as a Haplic Lixisol. Soil profile 3 was located on a moderately steep hillslope section (15%) with convex-convex curvature and high grass cover (> 80 %). This soil developed in unconsolidated colluvial deposits and is classified as a Magwa Form or as a Dystric Cambisol.

Table 6.1. Soil properties for soil profiles in the Footslope (1), Midslope (2) and Upperslope (3) sections. Soils are classified as Hutton Form, Sepane Form and Magwa Form respectively. NIR = near infra red; n.d. = not determined.

		Depth (cm)	Colour	NIR clay (%)	NIR C (%)	Exchangeable bases (c. mol charge/kg soil)				CEC (c. mol charge/ kg soil)	ESP-CEC	Soil pH	Soil Salinity (Saturation Extract)						
						Sod.	Calc.	Mag.	Pot.			KCL	% Sat.	EC (mS/m)	Na (me/L)	Ca (me/L)	Mg (me/L)	K (me/L)	SAR
Profile 1	Ap	0-20	5YR 4/4	40	1.6	0.36	1.54	0.58	0.79	3.65	9.86	4.00	*n.d.*	*n.d.*	*n.d.*	*n.d.*	*n.d.*	*n.d.*	*n.d.*
	AB	20-40	2.5YR 3/6	43	1.4	0.35	1.23	0.46	0.77	4.20	8.33	4.11	*n.d.*	*n.d.*	*n.d.*	*n.d.*	*n.d.*	*n.d.*	*n.d.*
	B	40-120	2.5YR 4/6	56	1.0	0.34	3.24	0.82	0.74	3.38	10.05	4.31	*n.d.*	*n.d.*	*n.d.*	*n.d.*	*n.d.*	*n.d.*	*n.d.*
Profile 2	A	0-20	10YR 3/2	29	1.0	0.33	3.67	1.61	0.99	5.57	5.92	4.11	41.7	9.9	0.10	0.27	0.19	0.17	0.21
	B	20-45	10YR 3/3	38	1.2	0.33	4.60	1.65	0.86	6.89	4.79	3.98	44.6	7.6	0.17	0.38	0.25	0.15	0.30
	BC	45-70	10YR 4/4	39	<0.5	0.33	2.89	1.17	0.81	6.35	5.20	3.93	40.0	5.1	0.07	0.01	0.09	0.15	0.31
	C	70-120	10YR 5/6	36	<0.5	0.35	3.03	1.89	0.84	7.40	4.73	3.90	46.1	3.6	0.09	0.02	0.06	0.10	0.45
Profile 3	A	0-20	10YR 3/2	30	2.5	0.33	4.46	1.69	1.20	7.85	4.20	4.29	46.7	11.3	0.05	0.13	0.23	0.36	0.12
	AB	20-40	10YR 3/3	40	<0.5	0.33	2.25	1.10	0.84	5.75	5.74	4.00	38.7	6.5	0.05	0.03	0.04	0.09	0.27
	B	40-100	7.5YR5/6	28	<0.5	0.34	1.59	1.00	0.78	4.84	7.02	3.99	45.2	4.4	0.03	0.01	0.15	0.22	0.11
	C	100-120	10YR 4/6	24	<0.5	0.37	1.21	0.87	0.78	3.29	11.25	4.01	39.5	3.8	0.02	0.00	0.05	0.09	0.13

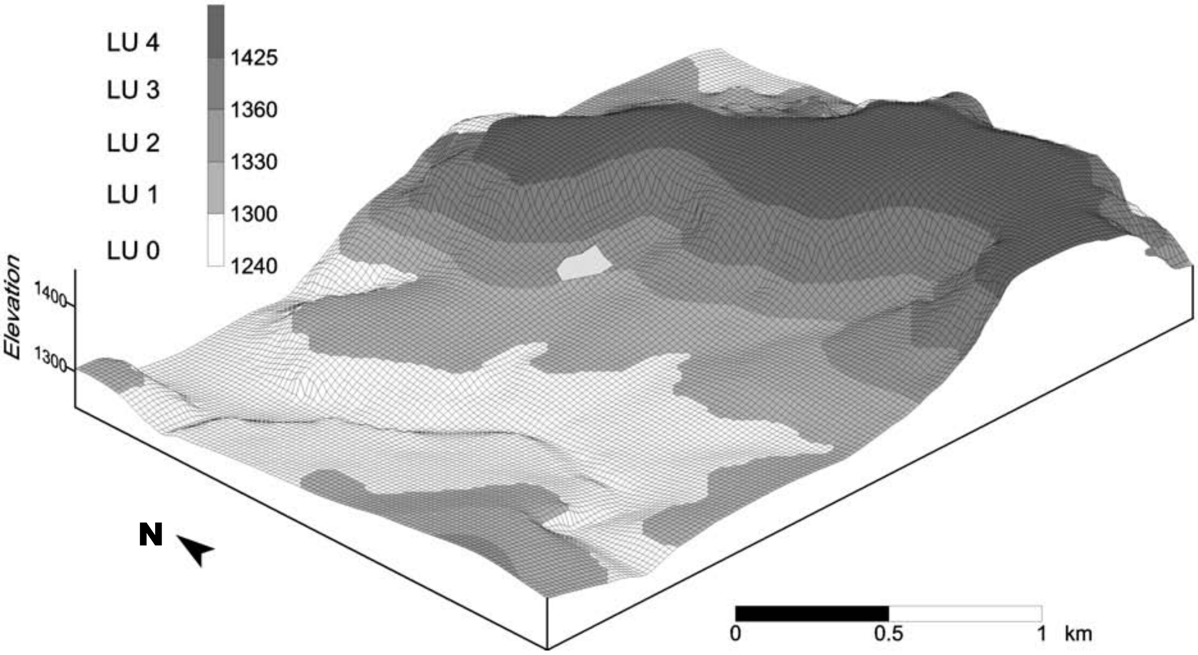

Figure 6.2. Distribution of distinguished landscape units (LU) in the study area and location of the fenced site (hatched).

6.2.2 History

The Okhombe catchment was re-planned for agricultural production in the early 1960s (Von Maltitz and Evans, 1998). Mountain slopes and plateaux were designated as communal grazing land, the people were forcibly removed to one of six closer settlements (sub-wards) at the foot of the slopes and lower areas adjacent to rivers were designated for cropping. Grazing camps, surrounded by fences, were designed for the communal grazing areas to accommodate different types of cattle in different parts of the camp. The grazing land was divided between sub-wards and into two sections. The slopes of the hills, close to the homesteads were planned for dairy cows and cattle belonging to woman-headed homesteads to save the women the long walks needed to retrieve their cattle. The hilltops were reserved for the main livestock herds.

Currently, these grazing camps are no longer functional. Most fences have been destroyed or have collapsed because of ageing and lack of maintenance. Sometimes, fences were also removed by local residents of the Okhombe community in order to have access to other pieces of land. As a consequence, there is no control of movement of cattle and grazing patterns. Furthermore, the lack of security and theft have led to the situation that most cattle are kept near the homestead and are daily moved up and down the slopes. Crop destruction from cattle is another issue that has become more important in recent years because of the poor fencing, making crop growing a more risky activity. Currently, there is also no restriction on how many cattle are kept by an individual and hence no control on stocking density.

In 1994, a catchment management project was initiated in Ngubhela. The project was initiated to address the Okhombe communities' concerns that soil erosion was increasing in their catchment area resulting in reduced water supply. There were also increasing reports of livestock that had died because of falling into the gullies. One of the initiatives undertaken was the establishment in 1999 of a pilot project to rehabilitate a severely degraded hillslope. A community capacity building programme was initiated in which community members were trained in soil amelioration techniques such as laying contours using an A-frame, planting Vetiver grass along the contours to promote infiltration, building swales to slow down run-off and planting over 400 trees to bind the soil. Because of damage to the trees by cattle, the tree establishment was less successful than anticipated. In response to this, local community members decided to erect a fence around this site to protect the trees, prohibiting grazing as well as burning.

6.3 Materials and methods

6.3.1 Aerial photograph analysis at sub-catchment level

Aerial photographs were available for the Okhombe catchment for the years 1945, 1962, 1975, 1986 and 2000. Enlargements were made for the research area around Ngubhela producing photographs with a scale of 1:3 000 for 1945 and 1:5 000 for the following years. Soil erosion features can be classified using different criteria (Dardis *et al.*, 1988). In this study, mainly longitudinal erosional features with confined flow were mapped for each year. Sheet erosion features with unconfined flow were mapped but only when they preceded confined flow paths in following years. Additionally, their intensity was also recorded, using vegetation coverage. Two main channel networks exist in the area with erosional features upslope and these were recorded as separate elements. The distinct soil erosion features were classified and mapped according to Table 6.2. Soil pipe systems, once identified, were not attributed to any other class.

Table 6.2. Soil erosion classes and relation to the existing classes of Dardis et al. (1988).

Erosion Class	Description	Existing classification
A	Strongly active gully, steep and bare sidewalls with no or little vegetation	7,8
V	Weakly active gully, steep but vegetated sidewalls	7,8
B	Strongly active linear channel (rill) with bare surface	3
L	Weakly active linear channel (rill) with vegetated surface	3
P	Linear closed-open channel. Overland and subsurface flow; vegetated soil (soil pipe)	6
S	Sheet, undulating surface; bare soil	1

Besides the identification of soil erosion features for the different years, attention was also paid to the dynamics of individual features. It was noted whether features could be identified on all previous photographs, when they were observed for the first time, when they could not be observed anymore and whether they showed fluctuations in existence and non-existence. To illustrate the methodology, Fig. 6.3 shows for three different time-steps how these dynamics were identified and recorded.

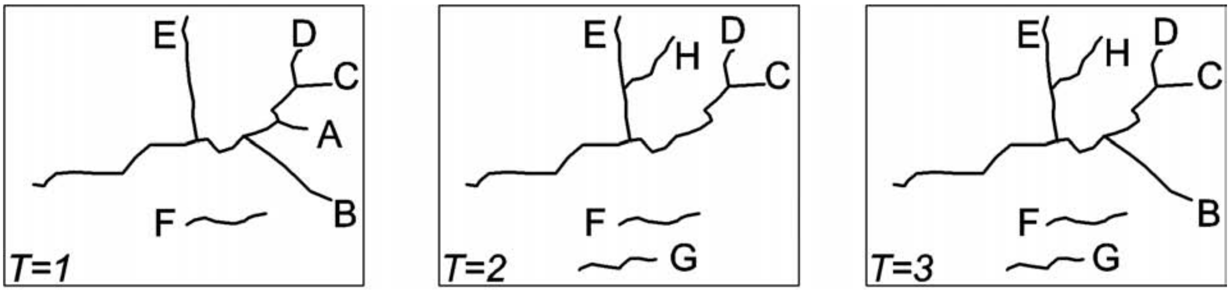

Figure 6.3. Example of soil erosion dynamics at three time-steps.

At T=1, 6 features are observed (I). Features G and H are not observed (II). At T=2, two features, G and H are observed for the first time (III). Two other features, A and B are not observed anymore (IV). Four other features, C, D, E and F are observed as they were also at T=1 (I). At T=3, features G and H are observed but they could not be continuously observed throughout all time-steps (VIII). Feature B is observed again after previous observation and subsequent no-observation (VII). Feature A is not observed for the second time (V). Features C, D, E and F are observed as they were also at T=1 and T=2 (I).

Land use surrounding the erosional form was also mapped using land use classes. A distinction was made between grazing land, cultivated land, residential area (including paths) and areas with woody vegetation such as trees and bushes. If vegetated areas could not be identified as cultivated land, they were designated as grazing land (Kakembo and Rowntree, 2003). Special attention was paid to features that were crossed by the fence erected in the 1960s.

The detail of the enlarged aerial photographs was also sufficient to distinguish between man-made structures. Dwelling units were therefore also recorded for the different years and for the different landscape units as an indication of settlement dynamics (Marker, 1988). Daily rainfall data were obtained from a weather station near Winterton from 1973 onwards and annual trends were derived from this data.

6.3.2 Fence-line contrast study at site level

The fenced area on the hillslope section in Ngubhela provided the opportunity to perform a fence-like contrast study, which was executed in March 2002. Two transects were defined for the fenced area which was located on a south-west facing slope (Fig. 6.2). Transect X, parallel to the slope contour, started in the right hand top corner and extended 104 m along the northern fence. At a distance from about 40 to 60 m, this transect also crossed a small gully. Transect Y, perpendicular to the slope contour, also started in the right hand top corner and extended 72 m along the eastern fence. Along transect Y, the slope varied from 15% (0–18 m), 25% (18–46 m) and 5% (46–72 m). Along transect X, the slope varied from 0% (0–36 m), 15% (36–47 m), 25% (47–64 m), 15% (64–81 m) and 0% (81–104 m). Both transects are graphically depicted in Fig. 6.4.

At regular intervals (approx. 10 m) soil samples were also taken from inside and outside the fence at 4 m distance for total carbon and total nitrogen. These were taken from two depths, 0–5 cm and 5–10 cm. Samples were air dried, sieved (2 mm) and milled and analysed using an Autoanalyser. For the same intervals, core samples (300 cm^3) were taken from the topsoil from both inside and outside the fence for saturated conductivity measurements and bulk density determination. Paired data were statistically analysed using a one-sample t-test ($P < 0.05$). Additional samples were taken for particle size analysis by the pipette method. Finally, a rough estimate of topsoil thickness was also made on the grazed side of the fence.

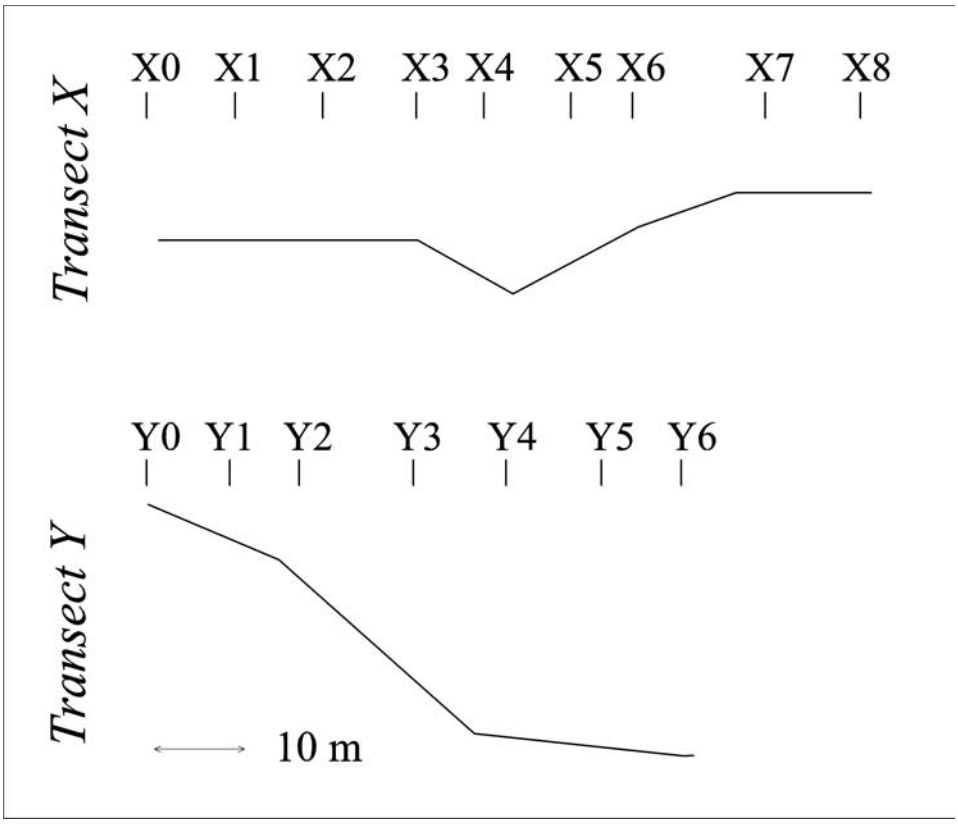

Figure 6.4. Cross sections and sample locations of the transects along the contour line (X) and perpendicular to the slope contour (Y).

6.4 Results and discussion

6.4.1 Soil erosion dynamics at sub-catchment level

For the study area, the aerial photographs revealed a total of 52 erosion features that could be identified on one or more of the aerial photographs, excluding connecting elements and channel network outlets that drained on the Khombe river. All features are indicated in Fig. 6.5. Nine elements belonged to one channel network and 14 to the other one. The other features mainly occurred as separate features in the landscape.

Of all elements, 4 were located within the River Valley, 12 were located on the Footslope, 20 were located on the Midslope, 16 were located on the Upperslope unit and none were observed on the Plateau. These figures are approximate indications as many of them cross the boundaries between the different landscape units. Data on the dynamics of the mapped features are given in Table 6.3. Table 6.3 indicates whether the phenomena could be observed in a particular year, without distinguishing between erosion classes or land use. The code numbers included refer to the example in Fig. 6.3. On the aerial photograph of the year 2000, 43 active features were observed against 42 in 1945. Of these 42, 28 were continuously observed on all aerial photographs. The 1962 aerial photograph shows the existence of fences, erected as part of the betterment plan. It also showed the highest number of erosion features that could be observed, 50 in total. In 1975, 15 of them (30%) were inactivated, i.e. could not be identified as erosion features. This year showed the lowest number of active erosion phenomena. By 1986, 6 of these inactivated phenomena were re-activated again while the rest (60%) could still not be observed. Additionally, 5 other features were also inactivated. Of the 9 phenomena that remained inactivated in 1986, 6 could still not be identified in the year 2000. In other words, they appear to have been permanently inactivated on a time-scale of decades. Furthermore, 5 features were reactivated leading to a number of erosion features that is the second highest for all years.

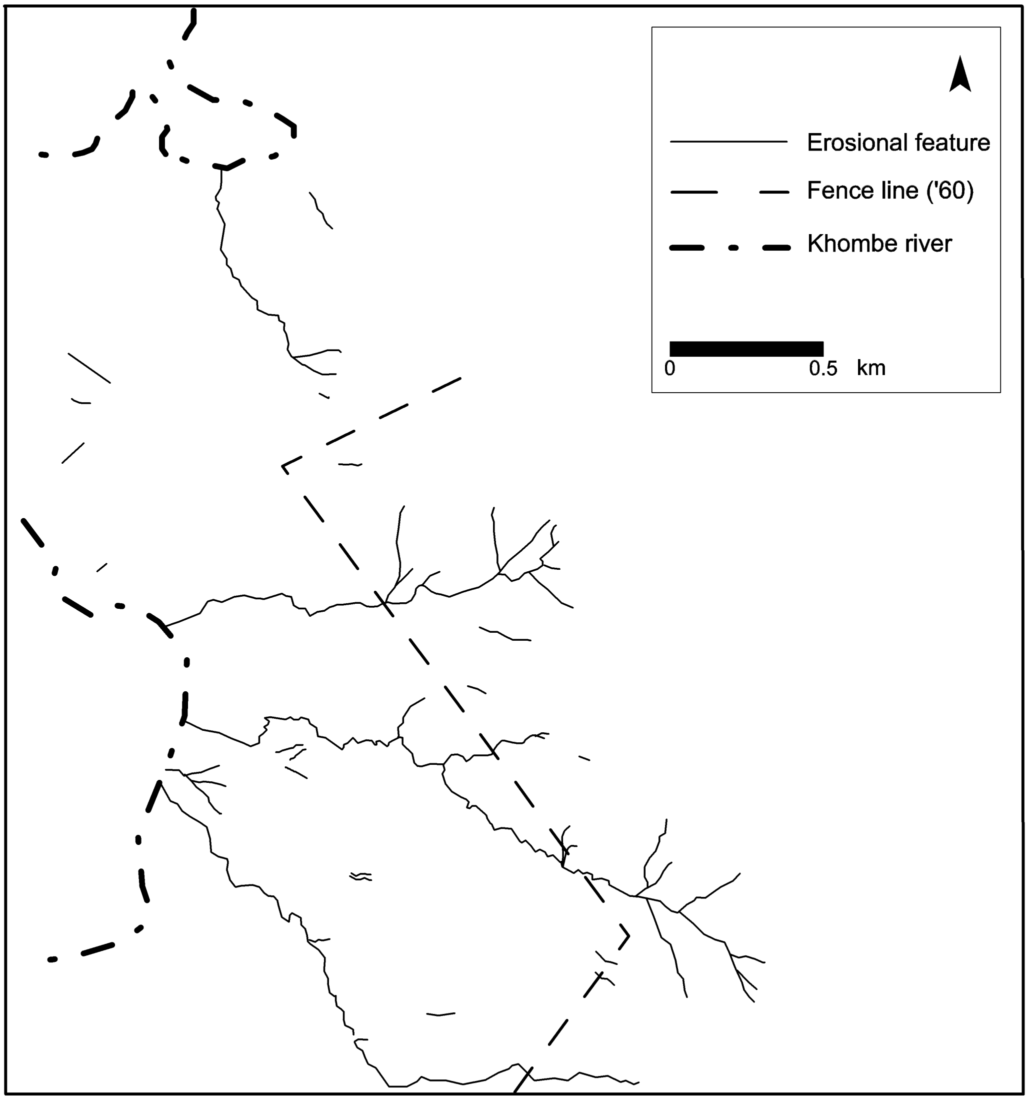

Figure 6.5. Erosional features identified in the study area, location of the Khombe river and location of the fence line.

Table 6.3. Dynamics of individual erosional features for the years 1945, 1962, 1975 and 2000[a]

	Code[b]	1945	1962	1975	1986	2000
Observed in specific year and all previous years	I	42	42	33	28	28
Not observed in specific year or previous years	II	10	2	1	0	0
First time observed (activated)	III	0	8	1	1	0
First time after earlier recording not observed	IV	0	0	15	5	0
Second time after earlier recording not observed	V	0	0	0	9	3
Third time after earlier recording not observed	VI	0	0	0	0	6
Observed after previous observation and subsequent no-observation (re-activated)	VII	0	0	0	6	5
Observed with previous intervals of no-observation	VIII	0	0	2	3	10
Total active features		*42*	*50*	*36*	*38*	*43*

[a] The 1945 aerial photograph is regarded here as the starting point.

[b] This refers to a common element of distinguished dynamics and Fig. 6.3.

Table 6.4. Recorded erosional features for five different years.

Erosion Class	1945	1962	1975	1986	2000
A	2	4	4	7	9
V	13	18	14	17	19
B	3	8	2	1	2
L	18	15	14	9	9
P	4	5	2	3	4
S	2	0	0	1	0

The soil erosion features were mapped according to the classification in Table 6.1 and results are given in Table 6.4. The number of strongly active gullies (A) has been steadily rising from 2 in 1945 to 9 in 2000. The number of weakly active gullies (V) has fluctuated somewhat and shows a pattern that resembles the number of active features in a certain year though these are, of course, related. The lowest number of vegetated gullies was observed in 1975 (14), with a gradual increase from there onward to 2000 (19). Weakly active linear channels or rills (L) decreased significantly from 1945 onwards to 2000. This was due to the fact that some became permanently inactivated whereas others developed into gully systems with steep sidewalls. At both locations where sheet erosion was observed in 1945, unvegetated rills (B) were observed in the 1962 photograph. One of them was again classified with sheet erosion in 1986 but the 2000 aerial photograph showed again an unvegetated channel feature at that location. Most of the pipe systems identified, with a maximum of 5 in 1962 could not be identified anymore in the 1975 aerial photograph. They appear to have a high resilience to become re-activated again. The one that was not active anymore in 2000 could not be identified in 1945 either. Though pipe-systems are difficult to study using aerial photographs, the data indicated that these features are not continuously active phenomena.

Land use data for the mapped features are given in Table 6.5. In all years, most erosion features were located in grazing land (60, 74, 86, 79 and 88% in 1945, 1962, 1975, 1986 and 2000 respectively). The increase after 1945 is mainly the result of the Betterment Plan for the area, which aimed at centralising cultivation practices on the lower slopes and in the valley bottom whereas grazing was planned to take place on the higher slopes. For the same reason, the number of erosion features in cultivated and residential areas decreased strongly between 1945 and 1975 with a small increase in cultivated areas thereafter. Some features were clearly surrounded by trees and bushes in 1975 and 1986 but these could not be observed in 2000.

Table 6.5. Types of land use associated with active erosional features.

Land Use	1945	1962	1975	1986	2000
Woody Vegetation	0	0	2	3	0
Grazing Land	25	37	31	30	38
Cultivated Land	12	12	3	5	5
Residential	5	1	0	0	0

Table 6.6. Number of dwellings in different parts of the landscape.

Landscape Unit	1945	1962	1975	1986	2000
0	0	0	0	2	41
1	10	28	21	91	240
2	58	25	0	53	78
3	13	3	0	0	0
4	0	0	0	0	0

Population dynamics, as indicated by the number of dwellings counted, are given in Table 6.6 per landscape unit. In general, the number of dwellings strongly increased in the last 55 years, from 81 dwellings to 359 in 2000, though this increase was not linear. Very few dwellings were counted in 1975 (21) and also 1962 (56), during the implementation of the Betterment Plan. Settlement distribution also changed drastically. In 1945, 72% of all dwellings were located on the Midslope areas but this figure decreased strongly from 45% in 1962 to 22% in 2000. In 1975 no dwellings were recorded for this area. In 2000, most dwellings (67%) were located on the Footslope section of the area. In 1962, it could be observed that 18 dwellings were still located on the eastern side of the newly created fence. In 1975, no dwellings were being used anymore on the eastern side of the fence but many remnants could clearly be identified.

6.4.2 Soil property dynamics at site level

Within the fenced area, vegetation cover and species diversity had changed remarkably within 3 years. In most cases a mulch layer of dead grasses, about 2 cm in thickness, covered the soil surface yielding a topsoil with a cooler temperature during daytime and a higher moisture status. Samples were taken from 16 locations in total, all having sandy clay loam texture (Fig. 6.4). Soil data inside and outside the fenced area are given in Table 6.7. Both transect X and transect Y showed varying depths. For transect X, this was mainly related to the gully that was crossed between locations X4 and X5. Minimum and maximum values for the percentage of clay were recorded at these same locations, and are mainly the result of particle-size selective processes of removal and deposition. For transect Y, a consistent decrease in clay% was recorded from Y0 to Y5 although the contrary was expected.

Soil organic carbon was not significantly higher or lower within the fenced area. Total carbon to nitrogen (C/N) ratios for the layer of 0–5 cm were significantly lower for samples from inside the fence compared with samples from outside the fence. Differences were not significant for the 5–10 cm layer. The top 5 cm for transect X did not consistently show lower C/N ratios inside. The existence of trees and more bushy vegetation along the contour as well as the existence of charcoal remnants may have confused the pattern that was more clearly observed for the transect Y.

At first instance, the reverse would be expected. Outside the fence, grazing takes place and increased inputs of N through excrements would result in lower C/N ratios. However, livestock excrements are collected by the community and used for fuel so this contribution could be less than expected. The higher returns of grass materials inside the fence with a high C/N ratio would also result in higher values inside the fenced area. The fact that, for the topsoil of transect Y, the largest differences were observed at the steepest slope suggests that selective removal of N takes place on the slope, most likely in mineral form such as nitrate or ammonium. N is more selectively removed outside the fence (more sheetwash) than inside the fence. Beneath the site, three local swales had been created along the slope contour of about 5 m in length and 1 in width. Though unfortunately no monitoring record was kept for these soil conservation structures, the sediment traps on average contained some 7 cm from the degraded slope, indicating that sheet wash is indeed an active phenomenon on the hillslope. Hence, soil fertility is directly affected by this intervention. Again, the topsoil of transect X, along the contour did not show this pattern and variation in C/N ratios seems to be natural variation. However, in the layer of 5–10 cm, again consistent lower C/N ratios were observed inside the fence and differences were also larger with increasing slope.

Average soil bulk densities are lower for both transect X and transect Y inside the fence (1352 and 1398 kg m^{-3} respectively) compared with outside the fence (1410 and 1416 kg m^{-3} respectively), but differences are small and not significant. Harrison and Shackleton (1999) also did not find any significant change in bulk density, not even for sites that showed long periods of alternative land use. Saturated conductivity (K_{sat}) was on average 40 mm hr^{-1} higher inside the fence compared with outside the fence but the difference was not significant. Locations Y2, Y3, Y4 and Y6 all showed lower values for soil bulk density and largest differences in saturated conductivity were found for locations Y2, Y3 and Y6. For transect X, no pattern could be observed.

Table 6.7. Soil properties at locations along transect X and transect Y for both inside (I) and outside (O) the fenced area.

Transect	Distance	Topsoil	Clay %	C%				C/N				Bulk Density		K_{sat}	
				0 - 5 cm		5 - 10 cm		0 - 5 cm		5 - 10 cm		(kg m^{-3})		(mm hr^{-1})	
				I	O	I	O	I	O	I	O	I	O	I	O
X0	0	25	29	2.39	3.88	2.34	3.09	11.95	11.76	14.63	11.88	1403	1461	165	168
X1	12.5	40	30	2.80	2.51	2.18	1.98	11.67	12.55	12.82	11.00	1417	1413	16	22
X2	23.5	30	29	2.47	2.74	2.05	2.46	11.76	11.42	12.81	10.70	1474	1436	69	154
X3	34.0	15	28	1.81	2.73	1.62	2.27	11.31	11.38	11.57	10.81	1167	1342	122	31
X4	44.1	10	32	2.09	3.01	1.65	2.32	11.00	12.54	10.31	11.05	1345	1468	152	24
X5	55.8	10	23	2.20	5.03	1.08	2.30	12.94	12.90	9.82	13.53	1377	1434	55	89
X6	67.4	25	26	3.21	3.02	2.49	2.06	12.35	12.08	11.32	12.88	1340	1418	123	82
X7	82.5	15	26	3.67	2.43	2.51	1.93	13.11	12.15	11.95	13.79	1276	1451	261	128
X8	98.5	20	28	2.22	3.44	1.92	2.51	13.06	11.86	11.29	12.55	1439	1359	32	77
Y0	0	20	29	2.61	3.11	2.10	2.24	12.43	15.55	11.67	13.18	1298	1267	153	104
Y1	11.8	25	28	2.27	1.77	1.95	1.65	11.95	17.70	11.47	12.69	1378	1413	148	30
Y2	20.0	30	27	1.61	1.29	1.48	1.21	10.73	18.43	11.38	11.00	1360	1501	86	12
Y3	37.1	35	27	2.19	2.88	1.89	1.93	10.95	14.40	11.81	12.87	1440	1483	57	59
Y4	49.2	26	35	2.12	2.13	1.81	2.09	11.16	14.20	11.31	12.29	1538	1357	57	84
Y5	61.5	25	25	2.51	2.81	1.76	1.82	11.95	12.22	11.73	13.00	1376	1477	244	46
Y6	71.5	*n.d.*	*n.d.*	2.79	2.67	1.93	2.07	11.63	12.14	11.35	12.94	*n.d.*	*n.d.*	*n.d.*	*n.d.*

Landscape unit 3, the Upper slope area, contains many of the erosional features. Its morphology, however, is not uniform throughout the research area. In the northern part it consists of a combination of sandstone and mainly mudstone and shales, leading to concave slope profiles. In the more southern parts, the slopes occasionally consist of predominantly sandstone leading to more straight and even convex slope profiles. Cattle and goats, which are brought into the grazing area on a daily basis, often move to the vegetation-rich plateau area (landscape unit 4) using these slopes. The top-fence, erected in the early sixties and still existing in 1975, divided such a straight slope between a section that could still be used by cattle, and a section that could not be reached by cattle. The fenced area could be reached by cattle from the Plateau but is likely to have experienced a much lower intensity of grazing and trampling compared with the other side of the fence. Only a slight improvement in vegetation cover could be observed for this area in 1975 compared with the accessible part of the slope. Extremely shallow soils that are located here apparently make it difficult for vegetation to recover. This finding, in combination with observations from the recently established fenced area that was sampled, leads to the notion that resilience of vegetation in communal grazing land is a strongly localised phenomenon.

Under the grazing system which prevails in Okhombe there have been major changes in species composition. The most distinctive feature of the rangeland at Okhombe is the loss of highly palatable grass species such as Themeda triandra. These species are preferred by livestock and have been eliminated with overgrazing. The dominance of poor grazing grasses such as Eragrostis plana and Sporobolus africanus is an indicator of overgrazing. However, the grass cover formed by these species is generally more dense, making it more resistant to physical degradation. This is supported by the high basal cover values recorded in Ngubhela (17.7–21.2%). This is mainly due to the dense nature of the Eragrostis curvula tufts and to the stoloniferous nature of the invader grass Paspalum dilatatum.

The veld condition of other study sites in Ngubhela ranged from poor (15.9%) to reasonable (64.8%), with an average of 46.3%. The degraded sites were characterized by a large percentage of Increaser IIb species which are indicative of long term overgrazing. The veld condition of these sites could be improved with good management practices such as resting. However, some of the sites were severely degraded with a high percentage of Increaser III species such as Aristida junciformis. Aristida is a hard, wiry-leaved, unpalatable grass which has virtually no grazing value. If selective overgrazing continues Aristida will be virtually impossible to eradicate with normal veld management practices.

6.4.3 Intrinsic landscape properties as contributing factors to soil erosion

Many signs of surface erosion are visible within the entire Okhombe catchment. In some cases on the upper slope sections, the combination of sloping sandstone surfaces, on which overland flow accumulates in volume and kinetic energy, overlying highly erodible shale and mudstone layers make these areas very susceptible to removal of soil material. The advance of gully features in colluvial deposits is driven by a wide variety of factors, among which is soil piping.

Subsurface erosion phenomena are common in the study area and are mainly found in landscape units 2 and 3. Values for Electrical Conductivity (EC) were around 10 and 11 mS m^{-1} for profiles 2 and 3 respectively and decreased consistently with depth to below 4 mS m^{-1}. This is considerably lower than what has been reported by Rienks *et al.* (2000), especially compared with the samples from their study that showed a high level of erodibility (1370–1830 mS m^{-1}). Their samples also showed much higher values for ESP-CEC (>20) and SAR (>15). Similar to their findings, it was found that ESP and CEC were strongly correlated for all samples (Table 6.1), especially when an exponential curve was fitted ($r^2 = 0.97$). No relationships were found between clay% and ESP or CEC.

Piping does occur in materials with low ESP and SAR values. Beckedahl (1996) found, for example, that these phenomena occur in soils with ESP ranging from 0–19 and SAR from 0.3–21.9. In this study area, it is not likely that the chemical properties of these soils drive the initiation and development of soil pipe systems. It was noted that the ESP value of the deepest layer of profile 3 is relatively high (11.3). This may lead to a somewhat higher susceptibility for removal of particles compared with the layers above though the difference will probably be small (Rienks *et al.*, 2000).

The soil pipes and tunnels around soil profile location 2, close to a gully, were more intensively studied. The soil had developed in colluvial material and has a low percentage of organic carbon (Table 6.1). The texture was coarser compared with soil 1. A strong textural break was observed at around 20 cm depth with an increase in clay of 10%. Soil structure changed from granular (topsoil) to prismatic and even columnar from 70 cm depth. Some clay cutans were also visually observed at this depth but this was not expressed in a higher clay% compared with the layer directly above it. When this layer was exposed, its consistency was very hard. It was observed that around the nearby gully, most of the soil had been eroded to this layer, forming a very hard floor with hardly any vegetation. ESP values increased slightly with depth. At the surface, a continuous soil crack of about 1 cm in width was observed at some 30 cm distance from the gully head.

One particular tunnel showed a diameter of 0.7 m when it entered the gully through a sidewall. At about 6 m upslope, away from the gully, the tunnel roof had collapsed but at 10 m distance, the roof was still intact exhibiting a cavity sufficiently large for a man to sit in. Substantial amounts of fresh sediment were observed inside. Further upslope, the form revealed itself as a longitudinal depression in the landscape which could also be traced back on aerial photographs, even on the one from 1945. Within the gully, the tunnel had developed just above the B-horizon, i.e. just above the textural break recorded at location 2. Other parts of the side-wall showed seepage of water at the same depth. Small soil pipes were also observed at greater depth of approximately 1.20 m. Strong seepage was furthermore observed at 1.60 m which could be related to a layer of shales just below. Strong breaks in permeability, related to the texture of the material and in combination with water transport on sloping conditions seem, therefore, to be the most important mechanism behind the initiation and development of soil pipes and tunnel systems, at least on these landscape units. These phenomena may trigger other erosion phenomena such as gully expansion through headward retreat. At locations where several pipe systems converge, even slumping may develop when most of the subsoil has been eroded away. This has also been observed.

6.4.4 External contributing factors to soil erosion

Annual rainfall data from 1974 to 2000 are given in Fig. 6.6. Although the data unfortunately do not cover the entire time-span of the aerial photographs and lack data for 1994, some useful information can be obtained. First of all, the data show a cyclic pattern in annual rainfall with a recurrence time of approximately ten years. Secondly, 1975 and 1986 have comparable values for precipitation (993 and 948 mm respectively) but the year 2000 was much drier with a cumulative rainfall of 716 mm. The relatively wet year 1975 showed a strong inactivation of erosion features. It is interesting that both Marker (1988) and Watson (1996) also had 1975 aerial photographs and rainfall data available. For the year 1975, they also found a decrease in erosional activity for their study areas in 1975 compared with previous years. The study of Marker (1988) revealed that rills could be re-activated in later years, similar to what has been found in this study. Climate therefore appeared to have a substantial impact on soil erosion through an increase in vegetation and a smoothing of erosional scars.

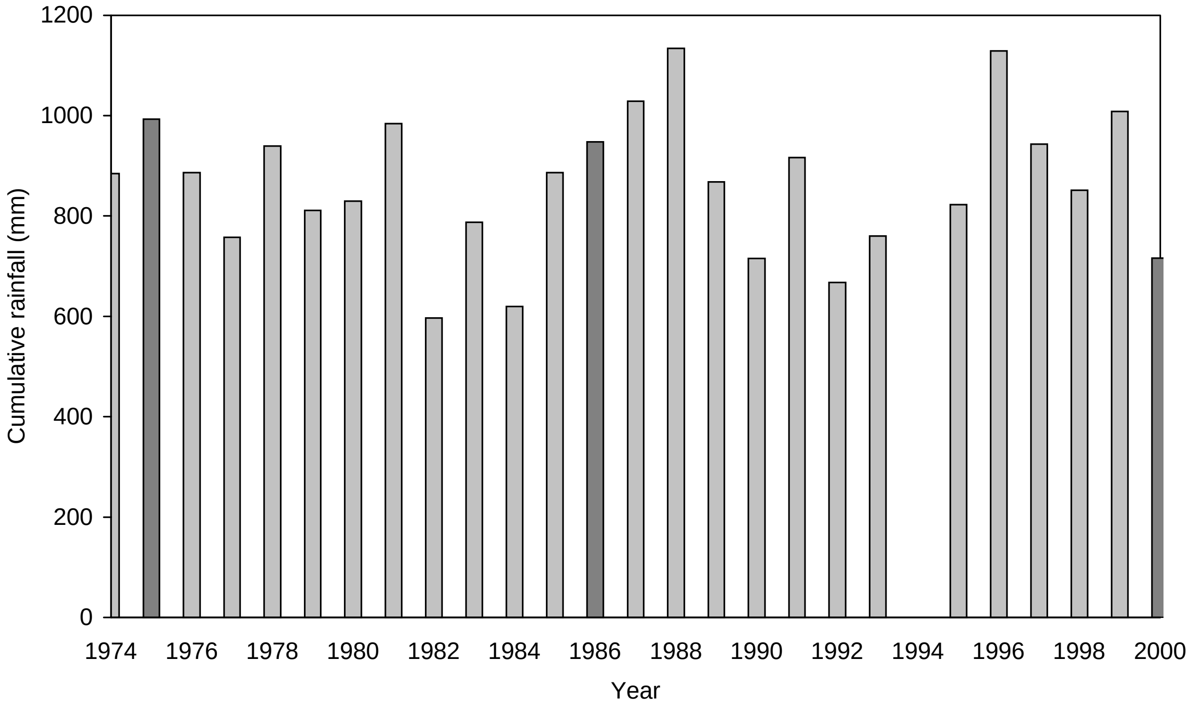

Figure 6.6. Annual rainfall for the research area as obtained from a nearby weather station. The dark-coloured bars indicate data for years that match the aerial photographs.

It should be noted that 1975 was also the year with the lowest number of people in the area but it is not exactly clear why. Local inhabitants have reported late plantings and low yields in the early 1970s because of late rain. Perhaps this has stimulated the emigration of some of the residents. In 1986, many more people lived in the area as the number of dwellings increased by almost 600% and it is likely that cattle numbers also increased, but probably less drastically. Compared with 1975, this year and the previous year had about the same amount of rain but the erosion pattern was somewhat different. The number of soil erosion phenomena remained more or less the same (slightly higher) but the number of both active and weakly active gullies increased. The year 2000 appeared to have the components for more intensive soil erosion; high population pressure (increase of 146% compared with 1986) and a dry year. Indeed, strong erosional activity was observed on the August aerial photographs and the number of gullies again increased. However, the year before (1999) showed a relatively high level of rainfall that may have stimulated vegetation growth resulting in a lower susceptibility for surface erosion. In the following year (2000), erosional activity may therefore have been less intense than what could have potentially occurred when a long-term dry period had preceded this year. In 2000, none of the mapped features were associated with trees or covered by bushes. Most likely, the woody material that was observed in 1975 and 1986 has been removed by local residents for fuel purposes.

It is interesting to investigate the reasons for the incidence of soil erosion in 1962. For this year, population and probably also grazing pressure decreased in the studied area compared with 1945. Climatic data from Marker (1988) and Watson (1996) indicated that conditions were not very dry for this year or previous years. Yet, soil erosion took place at a high rate with many gullies and unvegetated rills. The erosional features that were activated i.e. for the first time observed in 1962 were therefore more closely investigated on aerial photographs from 1945 and 1962. As an example, Fig. 6.7 is included which indicates how these features were identified for two different years. It appeared that out of the 8 activated features in 1962 (all rills), 6 were actually located in abandoned cultivated fields and not actually cultivated fields. Similarly, Kakembo and Rowntree (2003) also noted a strong spatial correlation between abandoned cultivated land and soil erosion, which, in their case, implied gullies. Marker (1988) also found that soil erosion, in her case sheet erosion, can be correlated with abandoned cultivated lands. This can be attributed to the fact that grass seedling establishment recorded on disturbed sites has been found to be extremely low (< 0.7 seedlings m^{-2} after four years following disturbance). When cultivation of fields is no longer carried out, annual sedges and *Oxalis obliquifolia* are among the first plants to establish. These species contribute little to basal cover and therefore make such locations especially vulnerable for soil erosion by overland flow.

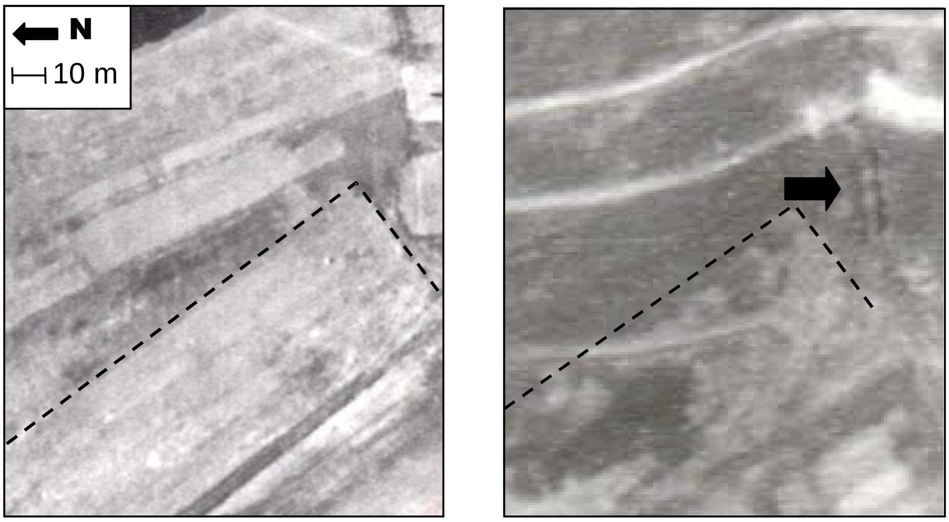

Figure 6.7. Comparison of a selected area between 1945 (left) and 1962 (right). Arrow indicates an erosional feature on an abandoned cultivated field. As a reference, the dashed line indicates the boundary of one particular field.

Excluding the main channel networks, four erosional features were identified that were dissected by the lower fence erected in the early 1960s, all located in landscape unit 2, the Midslope section. Two of these could not be identified on the 1945 photograph but were both activated by 1962, being located on abandoned cultivated fields. Three features were also identified as active on the photograph from 2000 but only on the eastern side of the fence, where grazing takes place. These were classified as one soil pipe system (P) and two gullies that had developed from rills. In this part of the landscape, grazing and trampling by livestock, possibly preferentially along the fence, have contributed to a continuous susceptibility for soil erosion processes whereas the exclusion of both activities has led to stabilisation.

6.4.5 Environmental aspects of interventions

Communal areas can be studied at different levels ranging from site, sub-catchment to catchment level. At each of these levels different soil system properties can be defined as indicators of land degradation. At site level, organic matter content, bulk density and infiltration capacity can be taken as indicators. At sub-catchment level, the distribution, type and vegetation cover of erosional features can be taken as indicators. At catchment level, the response of the drainage network as well as the sediment content of river streams can be taken as variables. In this study, the first two levels were taken into account and at both levels it was found that degradation is not an irreversible, continuous process.

Removal of soil material, through sheet wash, subsurface flow or headward retreat of gullies has been identified on each available aerial photograph and these processes have been taking place for at least 60 years. The observations of many existing buried paleosols suggest long-term dynamics of the biophysical landscape system involving stable and active stages for thousands of years. This falls out of the scope of this study and has more extensively been described by Botha (1996). The observations, however, indicate that erosion and sedimentation processes are part of the natural system. Moreover, removed materials from upslope areas have often been re-deposited in the lower parts of the valley, yielding soils that are suitable for growing crops such as maize. After crop harvesting, the maize stalks provide a relatively good quality of fodder in wintertime, acting as a ‘key resource’ (Scoones, 1992). In this study, more attention was paid to the grazing land on the hill slopes. However, it is clear that indications of increasing land degradation at sub-catchment level cannot simply be scaled-up to diagnose a decreased carrying capacity at catchment level. When grazing management schemes are implemented, such ‘key resource’ areas must also be taken into account. Also from site to sub-catchment level, soil property dynamics cannot simply be scaled-up. In general, a multi-scale assessment of land degradation dynamics provides clues as to which factors are important and how alternatives of land use affect soil system processes at various levels (Veldkamp *et al.*, 2000).

The most common soil conservation measure to decrease the rate of soil erosion through overland flow, the most visible form of soil erosion, is to increase the rate of infiltration. This may be achieved either by increasing or changing the vegetation or by means of slowing the velocity of overland flow through stone lines or swales. When large subsurface tunnels have already developed, which can be identified mainly by occasional roof collapse, increased infiltration will most likely further stimulate expansion of subsurface erosion or head ward retreat of down-slope gullies.

Management of cattle movement is most strongly needed along gully sidewalls and in the vicinity of gully heads. As gullies develop in (slightly) sloping areas, cattle movement near gully heads will likely result in soil cracks leading to infiltration of overland flow in these cracks, seepage along the side walls of the gully (on less permeable layers) and eventually collapse of the topsoil (soil topple) into the gully. It must be realized though that fencing on slopes stimulates the selective movement of cattle and goats along these fences and is therefore likely to promote the formation of cattle tracks with reduced vegetation cover and more susceptibility to soil erosion. Specific grazing control through fencing may therefore be considered for some areas but it should be evaluated whether the problem of topsoil removal is not simply being re-allocated somewhere else.

6.4.6 Social aspects of interventions

In this study, attention is paid to alternative land use trajectories by means of interventions at two different scale levels: at (sub) catchment level and at site level. At (sub) catchment level, intervention took place in the early 1960s. Nucleated settlements were created against people’s will and some people were forcibly removed from the upper slopes to one of the sub-wards. Within 10 years, the established fences were already fallen apart leading to increased cattle theft and increased crop destruction by cattle. Within the community as a whole, there has apparently been a lack of ownership, financial resources and accessibility to governmental bodies in order to maintain the established land use system. It is not clear how the landscape would have responded to changes in population-density, livestock and climate if no land use re-organisation had taken place. It is clear however that especially abandoned cultivated fields on the hillslopes are particularly vulnerable for soil erosion, which corresponds with the findings of others (Kakembo and Rowntree, 2003; Watson, 1996). The 2000 aerial photograph shows that of the rills developed in abandoned cultivated fields in 1962, 50% had developed into weakly and strongly active gullies. It also showed that erosional features that were dissected by the established fence (Fig. 6.5) were only active in the grazing area, not in the cultivated area on the west side of the fence. This indicates that certain erosional features will indeed be stabilized when they are not subjected to grazing and trampling. Altogether, the number of erosional features in the study area in 2000 was not substantially different from the number of features in 1945 although the number of both weakly and strongly active gullies had increased. Within the sub-catchment as a whole, responses to interventions thus vary in space but also in time.

The fenced site in Ngubhela showed a rapid recovery of vegetation, which strengthened community ownership of this site. The ‘visual resilience’ has stimulated discussions in other subwards for setting up similar sites. Thus, the outcome was ‘novel’ for the area and has had an impact beyond the boundaries of the plot. The site that was investigated is important as it shows the impacts of local interventions on soil and vegetation properties. Within an active communal grazing system at (sub) catchment level, the challenge is now to develop grazing management systems that fulfill the needs of community members but at the same time allow sufficient recovery of both soil properties and vegetation characteristics.

This initiative has contributed to the request of community members for a project for the whole ward. This led to the launch of the Okhombe Landcare project in 1999, which aims to address land degradation through sustainable farming practices, grazing management and soil conservation measures. This project, in which many community members participate, has further strengthened self-governance of the natural resources and may further strengthen the experimental capacity of land users to develop more sustainable land use practices in the future.

Diverse landscapes, such as in Okhombe, will result in strongly varying localised responses of soil and vegetation characteristics to land use and climate dynamics. All responses are difficult to unravel and scientific literature is more and more stressing the complex interactions that occur (Tapson, 1993). There is a need to localise research as well. The local people of Okhombe have been very well able to point out important changes in their environment in the last 25 years but the availability of aerial photographs from previous years has made it possible to give a longer term perspective on land degradation dynamics. Projects in which researchers and local community members work together to identify, monitor and evaluate land degradation processes, provide the opportunity to execute research in combination with the active involvement of local people and a potentially more successful adoption of alternative management practices. Increasing awareness (e.g. Reynolds and Stafford Smith, 2002) on both the temporal and spatial dimensions of soil erosion is believed to be vital in assessing land degradation and taking action in improved current and future natural resource management. In the case of Okhombe, the strongly increased population density in combination with increased erosional activity and the destruction of past implemented grazing schemes especially raise new questions that only can be addressed by mutual efforts and joint learning.

6.5 Conclusions

At sub-catchment level, communal land in the Okhombe catchment does not exhibit a status of continuous and irreversible degradation. An increase in the number of erosional features of mainly strongly active and weakly active gullies was observed from 1975 to 2000 but this followed a substantial inactivation of erosional features from 1962 to 1975.

At site level, vegetation cover and species diversity in communal grazing land may strongly increase after excluding grazing but the response varies throughout the landscape. A strong decrease in soil C/N ratio was observed for the top-soil of a fenced sloping site with sandy clay loam soils within 3 years. The decrease is most likely the result of a decreased removal of mineral N from the topsoil through soil wash processes. For the same site, total carbon, saturated hydraulic conductivity and bulk density were not significantly different inside the fenced area compared with outside. It is concluded that through fencing, fertility of the topsoil may locally be preserved.

Intrinsic landscape properties are important factors to understand soil erosion dynamics. Soil pipes are natural phenomena that are common in the area, especially within the communal grasslands. These may develop into substantial subsurface tunnels, which can also be identified on large-scale aerial photographs because of roof collapse and longitudinal depressions in the landscape. These phenomena appear to be easily inactivated as was observed in the 1975 aerial photograph. The chemical properties of the investigated soils show low values for ESP and SAR, making it improbable that chemical characteristics are driving factors behind these soil piping phenomena. Their initiation and development is mainly the result of textural breaks and transitions between permeable and less permeable layers.

External factors are also important to understand soil erosion dynamics at sub catchment level. The decrease in soil erosion features from 1962–1975 is most likely due to a wet spell in combination with a low population density in the area; measured by the number of dwelling units. The high level of erosional activity observed in 2000 is most likely the result of a dry year in combination with high population density and probably related increased grazing activities. The implementation of the Betterment Plan for the area in the early 1960s has resulted in a considerable change in settlement distribution. In accordance with findings from other studies, it was found that erosional activity increased on abandoned cultivated fields.

Intervention projects in communal lands that are based on up-scaled results of dynamics at lower levels may overlook important environmental aspects. This regards the relationship between soil property dynamics and soil location at sub-catchment level as well as the soil erosion dynamics at sub-catchment level and the existence of key grazing areas at catchment level. Furthermore, important local processes such as sub-surface erosion may be enhanced by standard soil conservation solutions that aim for increased infiltration in the soil.

In South Africa, past intervention projects in communal lands that failed to enlist local people in the development and implementation of land use re-allocation have not been successful. Furthermore, past scientific evidence for interventions that result in deep social disruptions proved to be weak and not sufficient to justify such drastic interventions. In the study area, it was found that local people are very well able to identify key processes, and that with training, they have the capacity to participate in experiments that can lead to improved understanding of the system and implementation of sustainable natural resource management. Given the increasing awareness in the scientific community that the interactions between land use, climate and soil- and landscape processes are complex and often strongly localised, new research constructions in which both researchers and local people participate, need to be developed.

Chapter 7

Synthesis

7.1 Impressions of Interactions

In both case-study areas, the Friesian Woodlands in The Netherlands and the Okhombe catchment in South Africa, environmental issues play an important role in the development of local agricultural systems. In the Dutch case-study, nitrogen surplus levels and nitrate leaching within dairy farming systems have been on the political agenda for some time now. In the South African case study, soil erosion and, more specifically, land degradation in communal areas, are important elements in the debate on sustainability. Soil systems with their soil and landscape processes are in both cases fundamental components of local land-based agriculture.

This thesis gives some impressions of the interactions that have taken place between farmers and their land. It also gives some information on the impressions that land users themselves have of these interactions. But mainly, this thesis describes the impressions of the interactions on the land itself. Indeed, these impressions are real mirrors of land use. The land reflects agricultural activities in various static and dynamic properties and, in turn, influences future land use on both the short and the long term. In the introduction (chapter 1), it was proposed to use co-production as a framework for studying soil systems. Given the material presented in the previous chapters, five elements can be derived that show the potential of this framework for providing a new path towards sustainability.

7.2 Co-production as a new framework for research

7.2.1 Environmental relevance

Soil series serve as a vehicle to transfer soil information and research knowledge from one area to another (SSSA, 1987). In contrast to the real soils (polypedons) they refer to, soil series are conceptual groups (Arnold, 1983). As such, they encompass a whole set of soils that have been used under different land use practices and thus can act as carriers of land use history. A single soil series can therefore be regarded as providing a characteristic “window of opportunities” (Bouma, 1994), covering multiple phenoforms. In the past, impacts of different types of management on soil properties were usually not included within soil taxonomic systems, often because of a lack of sufficient knowledge. As a consequence, soil databases are either filled with average values, or with a range of values for non-diagnostic properties or with a complete omission of some properties because of the difficulties in expressing them quantitatively.

Now, for example by using regression functions, properties such as organic carbon content or organic nitrogen content can be related to land use history. For a Dutch sandy soil series, soil organic N, soil organic C and its dynamics could be related to land use history (chapter 2). When established for any given soil series and stored into databases, such relationships can significantly improve the soil survey input into dynamic models working at a regional scale level. At site or field level, probabilities of exceeding the environmental threshold of 50 mg nitrate l^{-1} in the groundwater have been shown to vary considerably within one soil series as a consequence of varying N application levels, but also with land use history (chapters 4 and 5). This indicates that, with respect to environmental quality, soil series cannot be evaluated unambiguously without taking account of land use history.

Environmental themes as land degradation and, more specifically, soil erosion are important in many countries all over the world. This also holds for communal grazing land in South Africa. For the study area in Ngubhela, it was found that C/N ratios within a non-grazed fenced site were significantly lower compared with locations where cattle could graze freely (chapter 6). Most likely, this is caused by less removal of N on the slope compared with conventionally grazed areas. Grazing control or rotational grazing schemes where resting areas are incorporated will imply less removal of nutrients, particularly N, through runoff. Especially when some kind of grazing control is implemented on a larger scale, important contributions can be made to improve and maintain the quality of the environment.

New environmental issues have emerged in recent decades of which especially global climate change is one of the most important ones. Climate is an important factor to understand soil erosion dynamics in South Africa’s communal grazing land on a (sub-)catchment scale. Forecasts for climate change have predicted substantial increases in precipitation (Meadows, 2003). Although wet years may contribute to a decrease in erosional features (chapter 6), current high levels of grazing pressure and population density are different from the past, and increases in erosional activity are not unlikely. Moreover, when rainfall becomes relatively more concentrated in intensive storms, excessive runoff, leading to increased frequency and magnitude of flooding, will occur. This is especially the case where upland wetlands are eroded leading to a decrease in storage and gradual release of water from the soil to the rivers (Laker, 1999). Within the non-grazed fenced site in the study area in Okhombe, almost all sample locations showed a mulch layer of several cm in thickness. This layer is important as it reduces the impact of rainfall on the topsoil. Moreover, higher levels of infiltration are also likely to occur although the higher saturated hydraulic conductivities were not found to be significant.

In general, a stronger buffering capacity of the system may be attained in the future through improved grazing control, including resting times for the land.

Within the debate on global climate change, there is also an increased interest in storing carbon into (agro-) ecosystems (Wolf and Janssen, 1991). In this respect the approach of farmers in the VEL & VANLA area also appears to be relevant because old pastures will have high levels of carbon stored in the soil (chapter 2).

7.2.2 Social relevance

The changes in soil and landscape properties that have been brought about can affect the user in his or her awareness of the ability to build (or to degrade). This will often happen through characteristics that are not included within standard research enquiries though these are related to defined land and soil quality parameters. For example, one farmer in the Friesian Woodlands expressed that because one of his fields felt like “concrete” after ploughing instead of resilient, he decided to avoid grassland renovation through ploughing as much as possible. This indicates that there is a ‘looking backwards’ element within farming which affects future activities. Specific tasks are continuously observed, interpreted, evaluated and adjusted (Van der Ploeg, 1987). Dairy farmers that have not followed the common trend to frequently renovate their pastures can now in fact be considered as ‘modern’ farmers in terms of grassland management. They are able to use less fertilizer without affecting production too much. Moreover, in the past, less nitrogen will have moved from these fields to the groundwater compared with frequently renovated pastures.

The strong visual response of the vegetation in the study area in Ngubhela (chapter 6) stimulated further discussions within the community on natural resource management. The site was for example visited for discussions during meetings of the Okhombe Partners, the members of the Landcare project.

Within both projects where the research was conducted, common shared visions have proved to be essential components. In Okhombe, a competition among community members for a Landcare slogan in Zulu lead to “Thandizwe”, meaning love for the land (Salomon and Zuma, 2003). Additionally, some people mentioned that “if one would respect the soil, the soil would take care of him”. In the VEL & VANLA area, the phrase “Natuurlijk in Balans” was adopted as a title for a number of magazines that stressed the importance for farming to be in balance with soils, plants, animals but also society. Moreover, farmers in this area earlier expressed a strong feeling of injustice because regulations did not fit in with their small-scale landscape. They felt that they were punished for the fact that they maintained the landscape created by their forefathers (Renting and Van der Ploeg, 2001). In both cases, it is clear that farmers have other goals than only economic and material ones that define their relationship with the land.

The actual units of land form in fact particularly relevant elements that bind the people together. In a recent evaluation meeting of the VEL & VANLA mineral project, held in spring 2003, it turned out that a grown solidarity among the farmers was the most important result of the mineral project. The approach that was followed created much coherence among the farmers. In the recent evaluation of the Okhombe Landcare project (Sisitka, 2002), local actors also reported that bringing people together, knowing each other and good communication were important results of the project. Additionally, project members in both evaluations reported an increased capacity to do things themselves, to take control of the agricultural development in their areas. This can further strengthen collective action towards adopting alternative land use strategies that aim for a higher level of ecological sustainability.

From an environmental point of view, process-based land units such as catchments can be more relevant than land units based on administrative boundaries. In many cases where farmers share connected resources, collective action will be needed to address concerns for a declining environmental quality at these levels (chapters 5 and 6). It must be realized however that the concept of a watershed or catchment exists for (soil-) geographers, but it may be very abstract for most people who regard themselves as living near a particular town or administrative area rather than to a particular catchment (Auerbach, 1999). The need for a sustainable management of these resources (basically commonly owned) will then also imply building a common awareness.

7.2.3 Scientific relevance

The framework of co-production opens a whole new range of research questions that presents exciting challenges for (soil) science. In the past, new features that followed from the interaction between land and land use were sometimes considered to be a ‘nuisance’ because they implied adaptations of scientific theories and/or computer simulation models (Bouma, 1989). Here, these features move to the center of research. In chapter 3, it is indicated how water repellency is related to land use history and what the potential effects are on the soil water regime. Moreover, this phenomenon is likely to interact with fertilization and nutrient movement in the soil. To extend this further, Droogers (1997) reported differences in soil structure within one soil series following land use history and this further influenced hydrological processes in the soil (Droogers and Bouma, 1996). These findings first of all indicate that common assumptions on the actual behaviour of soils in the field are challenged. Secondly, the fact that there is a land use component in these phenomena makes the challenges even bigger: what other expressions of the soil can be found, or created, that are not known today? In practice, it can be argued, a wide variety of ‘field experiments’ is already there, waiting to be studied (Pulleman *et al.*, 2000).

Soils that are the outcome of a specific land use trajectory may very well fit into the agricultural system but at the same time require specific conditions in terms of management options. For example, Pulleman *et al.* (2003) showed that long term organic farming leads to higher organic matter contents that can result in better soil structure but only with specific management. Such soils with a higher organic matter content run a higher risk of being compacted by tillage, vehicular traffic or grazing under wet conditions. In other words, soils, through their land use history, canalise management practices and the use of specific technology. Earlier, farmers from within the VEL & VANLA area have raised similar points (Eshuis *et al.*, 2001). In the absence of suitable practices or technologies, soils will obtain different properties compared with what could have been obtained. The question then becomes: which expressions require which kinds of field management so that they can lead to sustainable agro-ecosystems?

In Okhombe, field research indicated that subsurface erosion through soil piping is a common element in the landscape (chapter 6). In general, research has not paid much attention to this type of soil erosion as most studies are related to splash-, sheet-, rill- or gully erosion. Only recently, the study of Beckedahl (1996) specifically addressed sub-surface phenomena in KwaZulu-Natal. Questions emerge when attention is paid to the interactions between land use and these phenomena. What happens, for example, when the infiltration capacity of the topsoil increases following land use change? Sofar, hardly any attention has been paid to this. In the study described in chapter 6, increases in soil erosion have especially been found on abandoned cultivated fields. Others have reported similar findings. Some suggestions are given on possible mechanisms that explain this but further research is needed.

Concepts of uniform responses of soil systems have shown to be in need of refinement. Simultaneously, fixed intrinsic properties that are co-influencing soil and landscape processes need to be identified and investigated. Some soil and landscape properties can be regarded as fixed boundary conditions. Duplex soils in sloping areas with aggressive rainfall such as Okhombe simply result in a higher risk of soil erosion compared with deeply draining soils. In

The Netherlands, sandy soils result in a higher risk of nitrate leaching compared with clay or peat soils. Altogether, this suggests that there is a need for *realistic* descriptions of soils, both static and dynamic, that can be used to answer questions raised regarding agriculture and environment.

The questions that are raised require different types of knowledge to be available, ranging from qualitative to quantitative and from empirical to mechanistic (Bouma and Hoosbeek, 1996). Especially where detailed quantitative information is required, modern research technology provides some relevant tools. This includes well-tested simulation models, operational Geographic Information Systems and so on (see also chapter 5). These tools are especially useful because some questions involve a whole set of complex interactions. This holds for example for questions that address the future of land use in communal areas under climate change (e.g. Meadows, 2003) or questions that address the dynamics of water quality in complex catchments following land use change (Lord and Anthony, 2000). Moreover, these 'what if' type of questions can be much more realistically answered when this is done so within a farmer's or community context.

Simultaneously, there is also a need for multi-scale descriptions of soil systems. In some cases, impacts of land use transcend the boundaries of fields, farms or, as in the case of Okhombe, sub-catchments. An assessment of these off-site effects forms an important element in studying land use systems. Moreover, results from field- or site experiments cannot always be simply scaled up to higher spatial levels and different methods of analysis may be required at various scale levels. With a change in a definition of the boundaries of the soil system, the properties of the system also change. At soil, or polypedon, level, other (macro-) properties are recorded in soil surveys compared with higher levels (chapter 6). Each of these properties can be influenced by land use activities and hence become part of co-production. Co-production is thus in itself not limited to a spatial scale level.

7.2.4 Inherent reflection on the impact of soil science

Within a scientific discipline, a dominant view can prescribe the trajectories along which agriculture should develop and other possible routes are then possibly ignored or even suppressed. Such a dominant view can be described as a *regime*, being basically a shared set of cognitive, social and technical rules (Roep *et al.*, 2003). From the beginning of the second half of the 20th century, the agricultural scientific regimes in The Netherlands became so strongly integrated with the technological and institutional regimes that they could govern the development of an entire sector in society i.e. the modernization of Dutch agriculture (Van der Ploeg, 1999). Dominant scientific regimes will influence research procedures and research objectives. A dominant scientific regime can also become ‘materialized’, i.e. become reflected in land properties. During the modernization period in The Netherlands, landscapes in different regions often became rationalized to improve the efficiency of farming. This affected the geometry of drainage patterns, landscape diversity and nature areas. At field level, also a kind of materialization took place through intermediate technology and land use. In the past, for example, several researchers reported that maize silage could be fed without problems as the only forage to dairy cows (Hijink, 1979) and this no doubt further stimulated the cultivation of this crop. As becomes clear from chapter 2, this practice strongly affects soil organic carbon and soil organic nitrogen levels but also soil structure (Alblas, 1990).

Another example is that researchers sometimes regarded the large amounts of organic matter in old pastures as “locked-up capital, bearing no current interest” which could only be used when a conversion to arable land would take place (Davies, cited by Hoogerkamp, 1984). The large-scale conversion of pasture to arable land that indeed took place on the basis of such recommendations led to enormous environmental problems because of water contamination. In fact, Whitmore *et al.* (1992) has indicated that in some areas in the UK, the entire nitrate problem can be directly related back to this practice (see also chapter 4). Clearly, the recommendations of converting old pastures into arable lands considered only the beneficial effects of mineralised N on the subsequent crops, not on the potential for leaching of nitrate to the groundwater.

Interventions into local areas can also take place by means of regulations. In the recent allocation of ‘dry’ sandy soils (see also chapter 5), a substantial amount of protest letters was delivered to the government. Surprisingly relatively many were also coming from the VEL & VANLA area (Van Kekum *et al.*, 2002). There were, in other words, substantial discrepancies between farmers’ perception of their land and the description of the corresponding soils on maps. Local actors have clearly expressed great concerns about the procedure. Other publications have reported comparable findings in other areas (e.g. Bloemendaal, 1995). This does not only show that there is some level of distrust between farmers and the Dutch Ministry of Agriculture but also between farmers and the scientific community.

Looking at the Okhombe catchment, similar observations can be made. This area was re-planned for agricultural production in the early 1960s. Mountain slopes and plateaux were designated as communal grazing land, the people were forcibly removed to one of six closer settlements (sub-wards) at the foot of the slopes and lower areas adjacent to rivers were designated for cropping. Though this land use scheme is still largely in place today, there are environmental problems with the communal grazing lands (see chapter 6) and, in general, the capacity of the local people to manage their own resources effectively seems to have been greatly affected. In the 1990s, expensive soil conservation structures (gabions) were implemented in the area by government agencies to stop the further development of the existing gullies. These have not been maintained however, leading to their disintegration and currently to an accelerated removal of sediment stored behind these structures. Moreover, important factors such as cattle movement along gully boundaries and the contribution of sub-surface erosion to the growth of these erosional features have not been addressed in these interventions.

How is research related to these interventions? One common element is that when scientific knowledge enters the local domain as such without any re-contextualisation, disruptions occur, affecting the environment, the actors or both. When scientific output is directly introduced in the locality in the form of prescriptions, regulations, technologies or land use plans, many detrimental side effects can occur. Both in the VEL & VANLA area (part of the Friesian Woodlands) as well as the Okhombe catchment (part of the Tugela Basin) initial soil survey reports were remarkably optimistic on their influence to change the local agricultural systems through planning (see chapter 1 or Veenenbos, 1949 and Van der Eyk *et al.*, 1969). Actually, hardly any distinction seemed to be made between the surveys, land evaluation and technical implementation (planning). It is now clear that a distinction does exist between scientific knowledge on soils, how these are valued and the actual interventions that take place. In general, scientific knowledge obtains a technological appearance when the de-contextualised objects (land units) form the direct basis of regulations or prescriptions for land management.

Within the framework of co-production, it could be argued, there is an inherent reflection on the impact of research. This reflection relates to direct regulations, prescriptions or technologies that are being developed on the basis of products from soil science. The fact that defined taxonomic classes involve a range of different expressions in the field with corresponding relevant processes call for field, farm or region-specific approaches. Correspondingly, this also holds for soil map units, especially those given on small-scale soil maps.

There is, in other words, a need for re-contextualising scientific (soil) knowledge. This re-contextualisation involves taking account of existing economic, social and environmental objectives.

7.2.5 Revaluation of the role of local actors

There is an urgent need for a different research approach to deal with the management of land. In the classic approach, experiments are only carried out on experimental fields or using desktop simulation modelling with subsequent transfer to farmers. This transfer-of-technology model does not suit the current challenges for agriculture anymore. The traditional views on research are now being strongly contested and new modes of doing research have been suggested such as joint learning environments (e.g. Bouma, 2001; Dent *et al.*, 1996; Pretty, 1998; Scoones and Thompson, 1994).

Research within the framework of co-production involves different groups of actors as e.g. scientists from different disciplines, farmers and engineers. Joint learning implies that local and scientific knowledge are valued on an equal basis. Recently, it was noted that learning from experience appears to be a more important factor in understanding the possibilities of nutrient management on a Dutch dairy farm than what is assumed in 'information dissemination' pyramids that follow a top-down approach (Ondersteijn *et al.*, 2002). New roles for scientists therefore have to be taken up because traditional science in laboratories and on research stations has to go together with learning from and with farmers (Pretty and Chambers, 1994). For example, the databases mentioned in paragraph 7.2.1 can also hold the 'stories' of farmers; i.e. their experience and knowledge. These databases then reflect the effects of management on soil properties and this broadens the basis for communication with farmers. Thus, the variability in soil properties that exists within soil system databases is linked to land use history, both with technical and sociological aspects. In other words, the story of land is linked to the story of people.

This is more than just tapping the relevant information that farmers hold to formulate research questions or develop technologies. It must be realized that innovations in agriculture are not only being developed in institutions and universities. The experimental capacity that exists among farmers needs to be recognized and supported by scientists. Farmers and researchers have unique experiences, which can be combined productively to address present and future challenges for agriculture.

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Summary

During recent decades, considerable efforts have been made in the discipline of soil science to develop soil classification systems, soil maps and simulation models. Moreover, land evaluation procedures were developed as means to contribute to agriculture. In its early stages, just around and after the Second World War, it was known that farmers can and do adapt soils to their need, resulting o.a. in man-made soils. Additionally, soil classification made extensive use of local (soil) knowledge and land evaluation was basically systematized farmers experience. However, it has proved to be difficult to include local actors systematically within a general research approach for soil science. It appears that, in time, the broad relationships between farmers and their land have been lost.

Now that agriculture is confronted with environmental problems in many areas all over the world, pleas for other research approaches have been made. In this thesis, it is proposed to use co-production as a new framework for research. In general, co-production has been regarded as the on-going interaction between and mutual transformations of farmers and living nature. The main objective of this thesis is to describe soil systems in two case study areas along the lines of co-production.

The research has been executed in two case study areas. One is located in the northern Friesian Woodlands in Friesland, The Netherlands and the other one is located in the Okhombe catchment which is part of the Upper Tugela catchment in KwaZulu-Natal, South Africa. Dairy farming is the dominant type of land use in the Dutch case study area. This agricultural sector has received an increasing pressure from society to reduce nutrient losses to the environment. Especially nitrogen (N) losses to atmosphere, ground- and surface water has received considerable attention. Therefore, there is a strong need to develop more sustainable dairy farming systems. The research was executed within the context of a running mineral project called the VEL & VANLA project. In the South African case-study area, communal grazing is the most dominant type of land use. Signs of land degradation are common and increasing emphasis is put on sustainable natural resource management. Research was executed within the context of a running Landcare project.

For a major sandy soil series in the Dutch case study, variability of soil organic nitrogen, soil organic carbon and its dynamics could largely be explained by taking account of three types of land use history: old grassland, reseeded grassland and grassland converted from

continuous maize cropping. This implies that soil survey information can be refined by taking account of land use history.

For the same soil series, varying degrees of potential water repellency were observed, related to land use history. In the presence of water repellency, field soils will show a level of preferential flow. A hydrological simulation model was used to evaluate the effects of preferential flow on water storage and water fluxes. Taking account of preferential flow improved modeling performance significantly. Model simulations for a limited time period showed that water storage in the top 50 cm was lower for old grassland compared with reseeded grassland or previous arable land. Downward fluxes were especially larger for old grassland.

In another model simulation study, involving both a hydrological model and a nutrient model, attention was paid to nitrate leaching. Combinations of the mentioned types of land use history and nitrogen application were simulated for a long-term period and probabilities for exceeding the environmental threshold for groundwater were calculated. It was found that this soil series cannot be unambiguously evaluated with respect to nitrate leaching.

Both EU and national policies aim to reduce the pollution of the groundwater caused by nitrate leaching. These do not match which has resulted in legal procedures. Attention was paid to the development of policies at both levels and more specifically to the perspective on land units within Dutch policies. A number of considerations for future policy directions are being given, including: (i) promotion of research aimed at improving and maintaining nutrient use efficiency at farm level; (ii) promotion of joint learning experiences between farmers and researchers, where farmers' organizations could act as 'research consortia'; (iii) emphasis on *site and time specific management* (precision agriculture) in policy development, and provision of site-specific advice via modern information and communication technologies; (iv) clearer guidelines for groundwater monitoring procedures, including additional monitoring at greater depths and consideration of groundwater quality from an appropriate regional perspective; (v) groundwater monitoring at locations selected according to specific hydro-geological characteristics and (vi) allowance for regional differentiation in indicators; these being the outcome of negotiations between farmers or their representatives, policy makers and researchers.

For the case study area in the Okhombe catchment, a multi-scale analysis of land degradation dynamics was performed. At sub-catchment level, the dynamics of erosional features were investigated by means of aerial photographs. At site level, the dynamics of soil properties were investigated by means of a fence-line contrast study. Both external and intrinsic factors related to erosional features were studied.

At sub-catchment level, an increase in the number of erosional features of mainly strongly active and weakly active gullies was observed from 1975 to 2000 but this followed a substantial inactivation of erosional features from 1962 to 1975. It was also found that increases in erosional activity in 1962 were related to abandoned cultivated fields. At site level, a significant decrease in soil C/N ratio was observed for the fenced site within 3 years. Textural breaks and transitions between permeable and less permeable layers appear to be the driving factors for subsurface erosion features instead of chemical characteristics. In the study area, it was furthermore found that local people have the capacity to participate in experiments that can lead to improved understanding of the system and implementation of sustainable natural resource management.

A synthesis was made of the research that had been done in both The Netherlands and South Africa. Five elements could be derived that indicate the potential of co-production as a framework for research. These are a) the relevance from an environmental point of view, b) the relevance from a social point of view, c) the relevance from a scientific point of view, d) the inherent reflection on the impact of soil science and finally e) the revaluation of the role of local actors.

Samenvatting

Gedurende de afgelopen decennia zijn er binnen de bodemkunde aanzienlijke ontwikkelingen geweest in de richting van de bodemclassificatie, de bodemkartering en de modellering. In het bijzonder werden er landevaluatie procedures ontwikkeld om een bijdrage te leveren aan de landbouw. In de begintijd, rond en net na de Tweede Wereldoorlog, werd binnen de bodemkunde nadrukkelijk onderkend dat er een wisselwerking bestond tussen boeren en hun land. Dit uitte zich onder andere in antropogene gronden. De bodemclassificatie maakte daarnaast uitgebreid gebruik van aanwezige lokale kennis over de bodem terwijl de landevaluatie in principe bestond uit gesystematiseerde boerenkennis. Het is echter moeilijk gebleken om de rol van boeren systematisch binnen de bodemkunde te integreren. In de loop van de tijd lijken de brede relaties die bestaan tussen landgebruikers en het land binnen de bodemkunde uit het oog verloren.

Nu milieukundige randvoorwaarden een belangrijke rol zijn gaan spelen binnen de landbouw is de behoefte uitgesproken voor andere onderzoeksbenaderingen. In dit proefschrift stel ik voor om co-productie te gebruiken als een nieuw raamwerk voor onderzoek. In het algemeen wordt co-productie voorgesteld als de voortgaande interactie tussen en wederzijdse transformatie van boeren en levende natuur. De hoofddoelstelling van dit proefschrift is om bodemsystemen te beschrijven volgens de noties van co-productie.

Het onderzoek is uitgevoerd in twee case studie gebieden. Eén is gelegen in de noordelijke Friese Wouden in Friesland. Melkveehouderij is het belangrijkste landgebruikstype in dit studiegebied. Deze agrarische sector ondervindt een toenemende druk vanuit de maatschappij om de uitstoot van nutriënten naar de omgeving te beperken. Dit geldt in het bijzonder voor stikstofverliezen naar de atmosfeer en naar het grond- en oppervlakte water. Er is daarom een sterke behoefte aan meer duurzame melkveehouderijsystemen. Het onderzoek is uitgevoerd binnen de context van een lopend mineralen project, het VEL & VANLA project. Het andere studiegebied is gelegen in het Okhombe stroomgebied wat deel uitmaakt van het grotere Tugela stroomgebied in KwaZulu-Natal in Zuid-Afrika. Hier vormt gemeenschappelijke beweiding het belangrijkste landgebruikstype. Landdegradatie is een veel voorkomend verschijnsel en er is sprake van een toenemend accent op duurzaam beheer van natuurlijke hulpbronnen. Het onderzoek in dit gebied is uitgevoerd binnen de context van een lopend *Landcare* project.

Voor een veelvoorkomende bodemeenheid in het Nederlandse studiegebied kon de variabiliteit in organische stikstof, organische koolstof en koolstof dynamiek voor een belangrijk deel verklaard worden door rekening te houden met drie typen landgebruikshistorie: oud grasland, opnieuw ingezaaid grasland and grasland met een voorgaande periode van snijmaïs in continuteelt. Dit impliceert dat informatie binnen de bodemkartering verfijnd kan worden door rekening te houden met de geschiedenis van het landgebruik.

Voor hetzelfde bodemtype werden verschillen in de mate van waterafstotendheid geobserveerd, gerelateerd aan landgebruikshistorie. In de aanwezigheid van waterafstotendheid kan er binnen gronden een zekere mate van preferente stroming optreden. Om de effecten van preferente stroming op vochtgehalte en water fluxen te evalueren is een hydrologisch simulatiemodel gebruikt. Het in acht nemen van preferente stroming verbeterde het model aanzienlijk. Model simulaties voor een beperkte tijdsperiode lieten zien dat de waterberging in de bovenste 50 cm lager is in oud grasland vergeleken met vernieuwd grasland of voormalig bouwland. Neerwaartse waterfluxen waren in het bijzonder groter voor oud grasland.

In een andere model studie, waarbij zowel een hydrologisch als ook een nutriënten model gebruikt zijn, stond nitraatuitspoeling centraal. Voor een lange termijn simulatie is gebruik gemaakt van combinaties van de genoemde types van landgebruikshistorie en verschillende stikstof toedoeningsniveaus. In het bijzonder zijn de kansen berekend voor het overschrijden van de gegeven milieukundige kwaliteitsnorm voor grondwater.

Zowel Europese als nationale regelgeving is erop gericht om de verontreiniging van het grondwater als gevolg van nitraatuitspoeling te reduceren. Deze komen niet geheel overeen wat geresulteerd heeft in juridische procedures. Er is aandacht besteed aan de ontwikkeling van regelgeving op beide niveaus en meer specifiek aan de beschouwing van land eenheden binnen het Nederlandse mestbeleid. Een aantal overwegingen voor toekomstig beleid worden gegeven: 1) stimuleren van onderzoek gericht op het verbeteren en instandhouden van de nutriënten efficiëntie op bedrijfsniveau; 2) stimuleren van gezamenlijke leerervaringen tussen boeren en onderzoekers waarbij boeren organisaties zouden kunnen optreden als onderzoeks consortia; 3) nadruk op plaats- en tijd specififiek management (precisie landbouw) in beleidsontwikkeling en voorzien van plaats-specifiek advies met behulp van moderne informatie en communicatie technologie; 4) duidelijke richtlijnen voor grondwater monitoring procedures, inclusief aanvullende monitoring op grotere diepten en beschouwing van grondwater kwaliteit vanuit regionaal perspectief; 5) grondwater monitoring op locaties die geselecteerd zijn met in achtneming van specifieke hydro-geologische omstandigheden en 6) toestaan van regionale differentiatie in indicatoren welke de uitkomst kunnen zijn van onderhandeling tussen boeren of hun vertegenwoordigers, beleidsmakers en onderzoekers.

Voor het studiegebied in het Okhombe stroomgebied is een meervoudige schaal analyse uitgevoerd van de dynamiek van landdegradatie. Op sub-stroomgebieds niveau is de dynamiek van erosiepatronen bestudeerd met behulp van luchtfoto’s. Op lokaal niveau is de verandering in bodemeigenschappen onderzocht door middel van een omrasterings studie. Zowel externe als intrinsieke factoren gerelateerd aan erosie patronen zijn hierin meegenomen.

Tussen 1975 en 2000 vond een toename plaats in erosiepatronen van hoofdzakelijk sterke en matig sterk ontwikkelde geulen op sub-stroomgebied niveau. Dit volgde op een aanzienlijke inactivering van erosie patronen tussen 1962 en 1975. De toename in erosiepatronen in 1962 kon gerelateerd worden aan het verschijnsel van verlaten akkers. Op lokaal niveau werd binnen 3 jaar een significante afname in bodem C/N verhouding waargenomen binnen de omheining. Overgangen in textuur en doorlatendheid van de bodem blijken de belangrijkste sturende factoren te zijn voor erosie onder het bodemoppervlak. Daarnaast is gevonden dat lokale mensen de capaciteit bezitten om deel te nemen aan experimenten die kunnen leiden tot een toegenomen begrip van het natuurlijke systeem en tot duurzaam beheer van natuurlijke hulpbronnen.

Een synthese is uiteindelijk gemaakt op basis van het onderzoek wat is uitgevoerd in zowel Nederland als Zuid-Afrika. Vijf componenten konden afgeleid worden die tezamen de potentie aangeven van co-productie als raamwerk voor onderzoek. Deze zijn a) de relevantie vanuit milieukundig perspectief, b) de relevantie vanuit sociaal perspectief, c) de relevantie vanuit wetenschappelijk perspectief, d) de inherente reflectie op de invloed van de bodemkunde en als laatste e) de herwaardering van de rol van lokale actoren.

Curriculum Vitae

Matthijs Pieter Wiggert (Marthijn) Sonneveld werd op 16 december 1975 geboren te Bleiswijk. Na het behalen van het VWO-diploma aan het Revius College the Rotterdam begon hij in 1994 met de studie Bodem, Water en Atmosfeer aan Wageningen Universiteit. Een eerste afstudeeronderzoek bij het Laboratorium voor Bodemkunde en Geologie richtte zich op het modelleren van landschapsprocessen. Een volgend afstudeeronderzoek werd verricht bij het Centrum voor Geoinformatie in Wageningen en betrof het monitoren van landgebruiks-veranderingen in Nederland. In 1999 sloot hij zijn studie cum laude af met als specialisatie ruimtelijke bodemkunde. In hetzelfde jaar trad hij in dienst als assistant in opleiding (AIO) bij het Laboratorium voor Bodemkunde en Geologie (Wageningen Universiteit). Sinds maart 2003 is hij hier eveneens in dienst als toegevoegd onderzoeker.