Land use and groundwater quality

How technical instrumentation and scientific knowledge can support groundwater planning

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How technical instrumentation and scientific knowledge can support groundwater planning

Landgebruik en grondwaterkwaliteit

Hoe technische instrumentatie en natuurwetenschappelijke kennis kunnen bijdragen aan grondwaterplanning

(met een samenvatting in het Nederlands)

PROEFSCHRIFT

Ter verkrijging van de graad van doctor aan de Universiteit Utrecht
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ingevolge het besluit van het college voor promoties
in het openbaar te verdedigen op

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Cors van den Brink

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Contents

Voorwoord

Werkend op het grensvlak van het ontwikkelen van kennis en het toepassen daarvan, raakte ik geïntrigeerd door de rol van technisch-inhoudelijke kennis in besluitvorming rond het beheer van grondwater. Deze ambitie werd versterkt door de werkgroep grondwater van de Technische commissie bodembescherming waar ik tussen 1998 en 2003 lid van was en die zich vanuit een technisch-inhoudelijke invalshoek boog over vragen op het gebied van onder meer grondwaterbeheer. Ik was één van de weinige werkgroepleden die niet was gepromoveerd. Professor van den Akker, destijds voorzitter van de werkgroep, vroeg mij dan ook bij gelegenheid wel eens of het niet tijd werd ook eens een boekje te schrijven. Om de kans van slagen te vergroten, bestond de eerste uitdaging uit het creëren van een project dat in tijd en diepgang een voldoende basis voor een promotie-project bood. Dit leidde tot het project 'Ruimtelijke ontwikkelingen en grondwaterbeheer' dat gesubsidieerd is door de Stichting Kennisontwikkeling en –overdracht Bodem (SKB) en samen met TNO/Deltares is uitgevoerd. Dit project heeft voor de rode draad gezorgd die nodig was voor mijn promotie-onderzoek. Daarnaast heb ik van Iwaco/Royal Haskoning de ruimte gekregen die nodig was om er een aantal samenhangende aspecten uit te lichten en in de vorm van wetenschappelijke artikelen uit te werken en te bundelen in dit boekje. Dit proefschrift is daarmee niet alleen de proeve van bekwaamheid die elk proefschrift is, maar ook een illustratie hoe universiteiten, kennisinstituten en adviesbureaus elkaar kunnen ontmoeten en versterken.

Wat begon met een inspirerend spanningsveld tussen 'Goede praktijk heeft theorie nodig' en 'Je hebt pas wat aan kennis waneer je er wat mee doet', ontwikkelde zich in de loop der jaren tot een persoonlijke evaluatie van ruim 15 jaar grondwaterbeheer. Een periode waarin we aanvankelijk – begin jaren '90 – met groot enthousiasme en inhoudelijke betrokkenheid instrumenten in elkaar knutselden waarmee de grondwaterkwaliteit beschreven kon worden. Instrumenten en uitkomsten die gezien vanuit de optiek van de ontwikkelaars voor zich spraken en dus hun toepassing wel zouden vinden. Na deze eerste euforie realiseerden we ons dat er meer nodig is dan een 'handig instrument' om een verandering teweeg te brengen in het beheer van grondwater. Een diagnose van een grondwaterconflict, zelfs één die gedeeld werd door de inhoudelijke mensen die verantwoordelijk zijn voor het grondwaterbeheer, bleek onvoldoende voor het oplossen van zo'n conflict. Vanuit de gedachte dat we het dus anders moesten aanpakken, werd een voorstel voor SKB geschreven. De kern van het SKB-project bestond uit afstemmen van 'inhoud en proces' om niet alleen diagnostisch maar zeker ook oplossingsgericht aan de slag te gaan. De aanpak die we ontwikkelden voor een concrete praktijksituatie sloot niet alleen goed aan bij ontwikkelingen om actoren te betrekken bij modelleringen, maar plaatste ook de eerdere pogingen op dit punt in perspectief. Terugkijkend lijkt het er toch sterk op dat projecten effectiever zijn in het daadwerkelijk veranderen van het grondwaterbeheer – en leuker - wanneer sprake is van wederzijds leren in plaats van projecten waarbij 'de probleemeigenaar' wordt verteld 'hoe de wereld in elkaar zit'.

Ik heb het als buitengewoon inspirerend ervaren om gedurende een langere periode intensief en doelgericht samen te werken met zeer competente mensen. En hoewel het project doelgericht is georganiseerd en uitgevoerd, was de weg zeker zo belangrijk. Of zoals de Zuid-Afrikaners zeggen: 'Soek jou geluk onderweg, nie by die bestemming nie, want dan is die reis voltooi'. Dit betekent overigens niet dat het geluk evenredig toeneemt met de lengte van de weg. Ik vond het uitdagend om het project te bedenken, heb veel geleerd en genoten van de uitvoering en sluit het nu met een voldaan gevoel af.

Ik bedank Peter de Ruiter, Jasper Griffioen en Willem Jan Zaadnoordijk voor jullie betrokkenheid. Dit was de basis voor het succesvol afronden van het promotieproject. Ik heb jullie leren kennen als scherpe denkers die pas genoegen nemen met een tekst wanneer alle beleidsmatige wolligheid en lucht eruit geklopt is, er staat wat er moet staan – en de tekst liefst korter geworden is dan een eerdere versie.

Ik wil de coauteurs Giuseppe Frapporti, Bas van der Grift en Saskia Burgers bedanken voor hun bijdrage. Met name de inspanning van Giuseppe en Saskia in het toegankelijk maken van statistiek voor mij als een niet-statisticus loog er niet om.

I would like to thank Vince Tidwell. You were a complete unknown to me when Mary P. Anderson suggested me to contact you about an issue paper for the 'Ground Water' journal. Your reply to my initial e-mail took away all my doubts of writing about a topic that had been refused by two journals already (including Ground Water). It was really a pleasure writing the issue paper together. I really enjoyed exchanging our ideas and experiences and combine them into one paper.

Ik wil Martha Buitenkamp bedanken voor de vele gesprekken waarin we nu eens aspecten ten aanzien van het grondwaterbeheer analyseren, dan weer ideeën uitwisselen of plannen maken.

Ik wil Theo Kleinendorst en Kees van Immerzeel bedanken voor het programmeren van respectievelijk FLUNIT en MDSAT, de rekenharten van de methodieken die we hebben gebruikt in onder andere de projecten in Halasztelek en Holten. Ik wil Han Komdeur en Adriana de Bruin bedanken voor de ondersteuning bij het maken van de tekeningen en het combineren van alle teksten tot een boekje.

Ik wil de Quintessensen, Ton le Coultre, Joke van Olst, Marijn Nijboer en Jorien Rippen, bedanken voor het gezamenlijk muziek maken. Samen muziek maken is niet alleen heel erg leuk, maar vereist ook een absolute concentratie waardoor gedachten over te schrijven boekjes effectief onderbroken worden.

Ik wil Jan Erik van Deijnen, Reinout Schoonbeek, Jan Smittenberg, Marien de Bakker en Ger Blijham bedanken voor het jarenlang aanhoren van de stand van zaken, zonder daarbij de indruk te wekken dat er twijfel zou kunnen bestaan aan de uitkomst.

Ik wil Evert Holleman bedanken voor zijn steun en vertrouwen bij het – in het perspectief van een adviesbureau – 'eindelijk' afronden van mijn promotie-project. Egon Coolen, bedankt voor je steun bij het over de eindstreep gaan, maar meer nog dan dat, bedankt voor je vriendschap.

Huub, Marieke en Nanke, de opgewektheid en nieuwsgierigheid waarmee jullie de wereld instappen, zijn voor mij een grote bron van vreugde en inspiratie. Mareen ben ik dankbaar voor haar liefde, en geduld in met name de laatste maanden waarin het geheel is afgerond. Hoewel ik het promotie-project zoveel mogelijk als project heb beschouwd, was het – met het einde van het project in zicht – niet altijd makkelijk mijn aandacht goed te verdelen tussen werk en gezin. Jouw begrip was daarbij een grote steun. Tenslotte dank ik mijn ouders voor het nest waarin ik ben opgegroeid en waar betrokkenheid bij elkaar en de wereld om ons heen vanzelfsprekend was en is. Lieve Mam, ik ben dankbaar dat jij het afronden van het boekje hebt meegemaakt en in mijn gedachten is Pap er ook bij.

Abstract

The primary aim of applied science is to develop knowledge to solve societal problems. Environmental sciences for example, focus on the human impacts on natural resources in order to prevent overexploitation or deterioration of these resources. It is, however, difficult to implement scientific knowledge in management and policy processes in an adequate way. In my PhD study I have focused on the question of how to improve the use of scientific knowledge in the current societal issue of sustainable groundwater management. I especially focus on the relationship between land use and groundwater quality and how understanding this relationship can be the starting point of groundwater planning.
Land use in the Netherlands is very intense and different types of land use occur within relatively short distances of each other. Land use effects on groundwater quality may result in different kinds of 'conflicts'. For example, conflicts between agricultural land use and water quality at drinking water abstractions. Also conflicts between different land uses can occur, for example agriculture and urban land use, needing low groundwater levels versus natural ecosystems requiring high groundwater levels. The challenge of groundwater planning is, therefore, to take the best course of action in a context of different stakeholders, land use and groundwater functions, policy fields and institutions. For my study, I have developed and modified groundwater models in such a way that they analyze and quantify the impacts of the various land uses on groundwater quality and that the model results can indeed support groundwater planning processes. This latter aspect was studied by application of the models in actual cases of groundwater management.

Firstly, a model was constructed that links a groundwater flow model with a nitrate-leaching model into a Geographical Information System (GIS). This model analyzes the effects of agricultural land use on groundwater quality abstracted at a drinking water abstraction site in Halásztelek, Hungary. The aim of the model was to understand the impact of the fertilizer use on the nitrate concentrations in abstracted groundwater. The model output provided quantitative insight in transport processes in groundwater and the cause-effect chain in groundwater quality. The model did, however, not result in a groundwater management scenario that could be implemented by policy makers.

Next, a comparable model was constructed together with stakeholders with the aim to improve the applicability of the model. This modeling was made part of a groundwater planning process in Holten, Overijssel, the Netherlands. The stakeholders represented particular interests in this conflict: agriculture, municipality, recreation, province, nature and drinking water supply. The model was expanded to include also urban land use and nature. In this way the interests and role of each stakeholder was addressed to enable the use of the model as Negotiation Support System (NSS). Because of the small scale heterogeneity in urban areas, the model adopted the 'inverse approach' to analyze the impact of urban land use. This approach quantifies the impact of land use from spatial observations on groundwater quality data, instead of calculating the groundwater quality from land use practices with deterministic relations. During the course of model development and application, the NSS was used to identify and analyse alternative land use scenarios together with the stakeholders. This resulted in the general accepted 'adapted land use scenario'. This scenario resulted in a plan for sustainable groundwater management, embedded in a regional development plan. This adapted scenario led to a transformation toward sustainable groundwater management, in that the land use safeguards long term drinking water supply. A sensitivity analysis showed that the NSS could sufficiently distinguish between various land uses to get the contrasts between scenarios and that science-based decisions could be made with the NSS and the information that was available.

With this PhD study, I have showed that technical, or scientific, knowledge can help to solve a societal issue like that of sustainable groundwater planning, but also that it is required that (1) the technical information is acceptable to stakeholders and (2) that this acceptability can be greatly improved by interaction and cooperation with the stake stakeholders in the set-up of the model, the choice of scenarios and the analysis and interpretation of the results towards spatial planning alternatives. I have learned that such cooperative modeling does not only improve the 'quality' of the planning processes, but also vice versa, that it improves the quality of the technical process.

Introduction

Groundwater is an important resource for drinking water and agriculture, and is intrinsically valuable for ecosystems. Groundwater provides two-thirds of the necessary drinking and industrial water in the Netherlands. Groundwater bodies are dynamic systems with water coming in at infiltration areas and water leaving the system in seepage areas or by abstraction. Groundwater flows very slow leading to residence times of groundwater in the subsoil of decades to hundreds or even thousands of years. Therefore, it may take the same or even longer periods before adverse effects of human activities on groundwater quality are noticeable when groundwater leaves the subsurface through abstraction. For the same reason, remedial measures may take equally long to be effective. As result of the very long recovery times and the sometimes large spatial scales of the impact of human activities, groundwater is considered to be extremely vulnerable to human influences (Dobson et al., 1997).

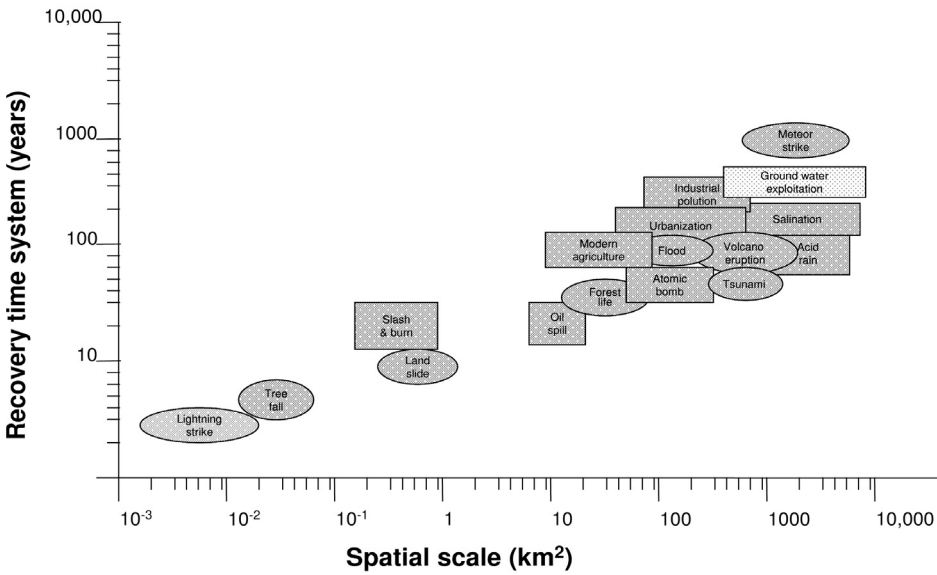

Figure 1. Relation between the spatial scale of natural and anthropogenic disasters and their approximate expected time to recovery. Natural disasters are depicted in ellipses and anthropogenic disasters are represented by rectangles (Dobson et al., 1997). The figure shows a deterioration of groundwater can have a very large spatial dimension and that recovery may take hundreds of years.

Understanding the effects of human activities on groundwater quantity and quality is of utmost importance for the management of groundwater. For sustainable groundwater management to be effective, it should consider the interests of various groundwater 'users' such as farmers, water supply companies, recreation entrepreneurs and nature conserving organizations. At the same time, various policy fields having different objectives, means and priorities are relevant for groundwater management. Relevant policy fields are environmental management aiming to prevent or limit inputs of pollutants into groundwater, water management aiming at adjusting the water system to human functions and vice versa, management of the natural heritage aiming at a specific groundwater flux and quality and spatial planning balancing a wide range of land use functions with a wide range of criteria. The term groundwater 'planning' is used instead of 'management' when effects of land use are under discussion. Land use may have a strong effect on groundwater quality, while land use planning has a strong relation with environmental policy and has become embedded in the social and economic processes (Driessen and Glasbergen, 2002).

Land use in the Netherlands is very intense and different types of land use occur within relatively short distances of each other. Land use effects on groundwater quality may, therefore, result in conflicts between various land

uses and between land use and groundwater functions such as drinking water abstractions (e.g. Secunda et al., 1998; Thirumalaivasan et al., 2003; Lake et al., 2003). For example, nitrate leaching from farmlands can cause nitrate levels to reach concentrations above the drinking water limit of 50 mg NO_3 per liter at nearby drinking water abstractions (see, e.g., Van Drecht, 1993; Hansen et al., 1991; Steenvoorden et al., 1997). Groundwater abstraction may cause lower crop yields by increasing the depth of the groundwater table, which leads to soil-moisture deficiency. In turn, water management optimizing agricultural land use may cause a decline of the seepage flux in groundwater-dependent natural ecosystems. A last example is when groundwater-dependent natural ecosystems become too enriched with nutrients from nearby agricultural fields (e.g., Runhaar et al., 1996).

Because of this complex relationship between land use and groundwater quality, various technical and scientific models have been developed to support groundwater planning. For example, transport models have been developed to assess the impact of various types of land use on groundwater quality at abstraction sites (e.g. Refsgaard et al., 2005). Also, there are models available that generate quantitative information on the relationship between land use and groundwater quality at a regional scale. These models have been developed to provide information for a specific policy field, such as agriculture, water management, environmental management and drinking water protection. A restriction of these models is that they do not address the combined influence of various types of land use on all relevant policy fields and by this the possible conflicts among the land uses. The analysis of such conflicts is especially important when the aim of the models is to support groundwater planning in the context of multiple land use.

Aim

In this thesis I have developed and modified land use and groundwater models that analyse the effects of spatially organised different kinds of land uses on groundwater quality. An important aim of the integrated model was that they can adequately support stakeholder driven groundwater planning. The first step was to construct a technical instrument that generates scientifically sound information on the effects of the various land uses on groundwater quality. The next was to investigate how such scientific information can be made suitable for groundwater planning. This was done by further development and application of the model in a stakeholder driven groundwater planning process in Holten, the Netherlands. The central question of the thesis is:

– How can we construct a model that provides scientifically sound information on the impact of various land uses on the groundwater quality that is acceptable to stakeholders and can be used in groundwater planning?

The role of such a model in groundwater planning is illustrated by its name: Negotiation Support System (NSS).

Approach

The cooperative modelling approach I have chosen is schematically illustrated in Figure 2. Two domains are distinguished, viz., the domain of scientific knowledge and the domain of the planning process. The connection of scientific knowledge to the planning process has three components: 1. the connection at the land use level defines how the technical description of land use effects on groundwater quality is related to specific stakeholders, 2. the connection at the NSS level defines how the technical information is acceptable to and useful for stakeholders and 3. the connection at the results level defines the identification and analysis of the alternative land use scenarios.

Set-up of the thesis

In **chapter 1** a model is presented that analyses the impact of agricultural land use on groundwater quality. This model is constructed by linking a groundwater flow model with a deterministic leaching model into a Geographical Information System (GIS). The model is used to quantify the effects of agricultural management on the nutrient concentration in the groundwater at a drinking water abstraction site in Halásztelek in Hungary. The model includes spatial information regarding land use, groundwater tables, and soil type as well as temporal variables such as manure and fertilizer application and atmospheric deposition.

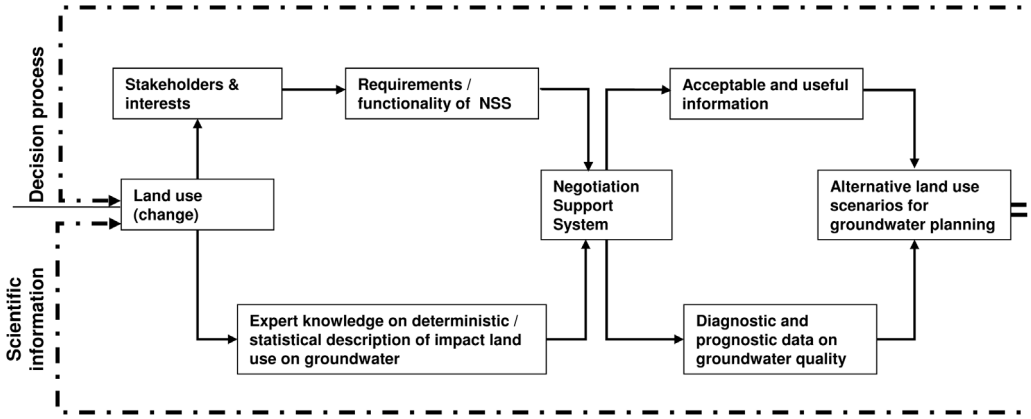

Figure 2. The NSS connecting scientific information on the impact of land use on groundwater quality to the decision process of groundwater planning.

In **chapter 2** a so-called 'inverse approach' is presented with which the effects of urban land use on groundwater quality can be established. The inverse approach derives the impact of land use from spatial observations on groundwater quality instead of calculating the groundwater quality with deterministic relations from land use practices. **Chapter 3** describes the development and application of the Negotiation Support System (NSS), including different kinds of land use: urban land use and recreation, nature and the agricultural land uses grassland, arable land and maize land. The cooperative modelling approach is described in terms of how the most important stakeholders are engaged in the modelling process. The results of the NSS visualise the effects of land use on groundwater quality in the case of groundwater planning in Holten, Overijssel, the Netherlands. The need for groundwater planning in Holten originated from the concern of the local water company and the provincial authority about high nitrate concentrations in abstracted and shallow groundwater. The stakeholders included the water company, provincial authority, the municipality, agriculture and recreation entrepreneurs. The application of the NSS on the Holten case resulted in a preferred 'adapted land use' scenario, that was accepted by the stakeholders, and which was implemented in a regional spatial plan. In **chapter 4** the results of the NSS are submitted to a stochastic uncertainty and sensitivity analysis. This analysis shows that the NSS is sufficiently sensitive to adequately distinguish between various land use types. In addition, the sensitivity analysis shows that the NSS is considered relatively robust, as the uncertainty of input parameters is not affected by various fertilizer and manure inputs of the scenarios. **Chapter 5** discusses how cooperative modeling projects can best be designed. Three valuable elements are distinguished, i.e., public participation, modeling, and their fusion. Public participation aims at structuring the involvement of stakeholders in the modeling process. Modeling aims at providing scientific information to support the decision process. Fusion aims at providing alternatives for land use planning and selecting a preferred scenario. In **chapter 6** I discuss the results of foregoing chapters from the perspective of 'adaptive governance'. I used this concept to judge the role and value of scientific knowledge in groundwater planning in relation to how the modeling facilitates a common learning process of scientists and stakeholders.

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Chapter 1

Evaluation of groundwater contamination from non-point sources: a case study

Cors van den Brink and Willem Jan Zaadnoordijk

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Impression of the study area Hálasztelek, 1992 (Photo: Willem Jan Zaadnoordijk).

Evaluation of groundwater contamination from non-point sources: a case study

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Abstract

In many countries a substantial part of the drinking water supply comes from subsurface water resources. During the last several decades an increasing extent of diffuse pollution has endangered these water resources. Knowledge of the changes in groundwater quality is necessary in order to know if, and to what extent, groundwater quality is endangered. Changes in groundwater quality can be predicted by means of models describing both the groundwater flow and the transport of contaminants. A Geographic Information System is an efficient tool to handle the storage and manipulation of the large amount of data needed for the description of diffuse pollution. A methodology to predict changes in groundwater quality, which is called FLUNIT, has been built around the programs FLUZO, NITRON, ARC/INFO, dBASE and WELCON and the groundwater flow package TRIWACO. The method runs on a PC (with 386 processor). The system has been applied to a well field at Halásztelek on Csepel Sziget (near Budapest, Hungary). The purpose was not so much the prediction of the groundwater quality, but much more the evaluation of groundwater protection strategies based on risk analysis and effectivity of possible measures. The system has been designed in such a way that the important information can be used with the available resolution and that new and more detailed data can be added efficiently. Change of the nitrogen concentrations in the abstracted water can be predicted. The resolution is sufficient for the evaluation of alternative groundwater protection strategies. Moreover, detailed insight into changes in groundwater quality can be obtained quickly.

Introduction

Subsurface water resources are used for the drinking water supply in many countries. Subsurface water is generally of good and constant quality so that simple treatment (such as aeration followed by rapid sand filtration or disinfection) is enough to turn it into drinking water. However, increasing diffuse pollution has endangered the groundwater resources during the last several decades. In the case of nitrate, the pollution mostly results from agricultural application of manure and fertilizer and is not easily stopped or prevented. In order to know if, and to what extent groundwater utilization is endangered, knowledge of changes in groundwater quality is needed. Accordingly, the impact of different protection policies on groundwater quality should be quantified.

Predicting the groundwater quality by means of statistical extrapolation of current trends is not satisfactory, and does not allow quantification of proposed remedial measures. Knowledge of concentration determining processes, such as mineralisation, denitrification and recharge, allows calculation of concentration changes with deterministic programs. The parameters needed as input parameters for these programs may differ in time and space. Thus a large amount of data is needed for the prediction of concentration changes due to diffuse pollution. Efficient data storage and manipulation are needed. This can be done by means of a Geographic Information System (GIS) that uses maps for representation. It conveys several types of information about the area represented on a map. Spatial information is represented by points, lines and areas.
In this paper a system is presented to obtain insight in the effects of remedial measures on the quality of the abstracted groundwater. This system is called FLUNIT, after the programs FLUZO and NITRON, that are the core of the method. Differences in soil type, land use, manure and fertilizer application can be accounted for. The application of the system is illustrated by a groundwater protection study.

Tools

Several programs have been applied successfully to model various aspects of groundwater flow and contaminant transport. The programs NITRON[1] and FLUZO[2] are used for the unsaturated zone while TRIWACO[3] and WELCON4 are used for the saturated groundwater flow. The programs had been used together in local studies and in cases of point source pollution. It became necessary to automate the data transfer between the programs when studies were extended to regional problems with diffuse pollution. The data storage and manipulation could be efficiently carried out by the GIS PC-ARC/INFO[5] which contains the database dBASE. Therefore, a system has been set up consisting of the programs that have been mentioned above tied together by means of data exchange via ARC/INFO. The programs will be briefly described.

NITRON

NITRON is a program that simulates the leaching of nitrogen or a few other solutes from the unsaturated zone to the groundwater table. Nitrogen in the soil primarily originates from inorganic and organic fertilizers, organic matter, wet and dry deposition and irrigation water. It is mainly removed by crops, leaching, denitrification and volatilization of ammonia during manure application. A suitable model for regional use should be based on a clear and quantitative description of the following processes:

1. leaching and accumulation of nitrogen;
2. denitrification related to (partial) anaerobiosis.

These processes take place in two zones, the root zone and the sub soil, which are two subzones of the unsaturated zone. The processes and transport of solutes in the unsaturated zone can be described one-dimensionally. The one-dimensional program NITRON has been developed by the Dutch Province of Gelderland and IWACO (Broers, 1988; IWACO, 1990; IWACO, 1993).

NITRON is a deterministic model, carrying out dynamic simulations. The nitrogen-cycle is described by four transformation reactions, see Figure 1. These first-order transformation reactions connect four different types of nitrogen: slowly decaying organic nitrogen, quickly decaying organic nitrogen, ammonium and nitrate. Furthermore volatilization and linear adsorption of ammonium are taken into account. Values for the groundwater table, manure and fertilizer application, plant uptake and atmospheric deposition may be specified for each input time step of 10 days. The fluxes in the unsaturated zone and the soil moisture content are calculated by the program FLUZO for each time step. The transformation coefficients, soil type and soil use are considered constant in time but may vary in space.

The program NITRON has been calibrated and verified on local scale (Broers, 1988; Broers, 1990) and regional scale (van Ommen et al., 1992; Van den Brink et al., 1993a) and has been proven to be a useful tool for the prediction of the nitrate concentration in the shallow groundwater (i.e. the groundwater just below the groundwater table).

FLUZO

The program FLUZO (FLow of water in the Unsaturated ZOne) simulates unsaturated groundwater. The program is based on a reservoir concept (root zone, sub soil and saturated zone) and uses relations developed by van Genuchten and co-workers (van Genuchten, 1980; van Genuchten and Nielsen, 1985). It has been developed by IWACO (Smidt and Emke, 1991). The set up of FLUZO is geared to a short calculation time so that it is possible to calculate the unsaturated flow at each node of a regional finite element model for each time step in transient calculations. The results of the program compare well with those of other programs for flow in the unsaturated zone for a test area in the Netherlands (Smidt and Emke, 1991).

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1 NITRON, NITRate flux to shallOw grouNdwater, (c) Prov. Gelderland and IWACO B.V.
2 FLUZO, FLow of water in the Unsaturated ZOne, (c) IWACO B.V.
3 TRIWACO, finite element package for simulation of groundwater flow, (c) IWACO B.V.
4 WELCON, concentration of abstracted groundwater in abstraction wells, (c) IWACO B.V.
5 ARC/INFO is a registered trademark of Environment Systems Institute, Inc. Redlands, CA, USA.
6 dBASE is a registered trademark of Ashton-Tate Corporation.

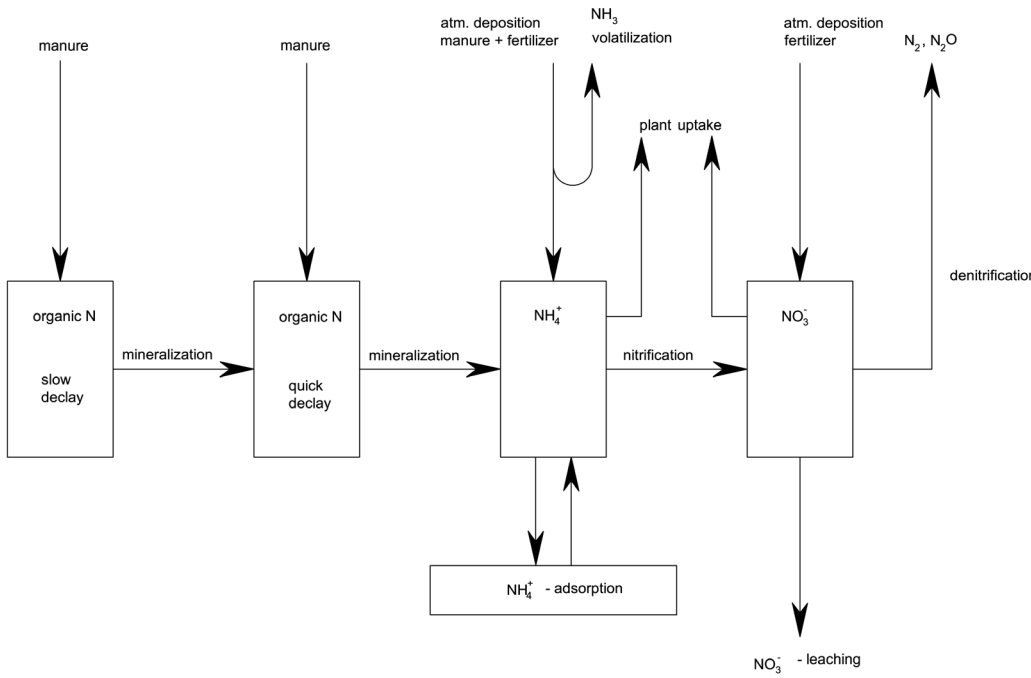

Figure 1. Overview of the nitrogen-cycle as schematized in the program NITRON.

TRIWACO

TRIWACO is a program package for numerical simulation of quasi three-dimensional groundwater flow based on the finite element technique (IWACO, 1992b). It has been developed by IWACO, based on programs of the Dutch National Institute of Public Health and Environmental Protection RIVM. TRIWACO is capable of handling a vast variety of steady state and transient groundwater flow problems in multi-aquifer systems. It contains many options for the calculation of the recharge at the top of the groundwater system. These options can be used together with FLUZO, the module for the unsaturated zone, to describe this flux even more accurately.
The grid generation program is called TESNET. It generates a grid of triangular elements. TESNET is very flexible and allows an easy inclusion of wells and rivers.
The program TRACE also is a part of TRIWACO. It determines path lines and travel times of water particles. It is possible to trace the path lines both upstream and downstream.

WELCON

The program WELCON calculates trends in the quality of the water abstracted at a well field, based on distributions of recharge fluxes, travel times and concentrations of pollutants within the total recharge area. The pollution load or leaching of a specific pollutant must be known as a function of time. Subsequently, the abstracted water quality is calculated as a function of time by integrating the contributions of the individual path lines within the recharge area (equation 1). The impact of retardation and first order transformation on abstracted water quality is incorporated:

$$C_t = \sum_{i=1}^{n} A_i \; c^*_{i,t} \; \frac{q_i}{Q} \qquad (1)$$

in which:

$$c^*_{i,t} = c_{i,(t-R\,t_i)} \, e^{-k\,R\,t_i} \qquad (2)$$

where:
C_t = abstracted water quality concerning the parameter to be forecasted at time t (mg L^{-1})
n = number of distinguished path lines (-)
A_i = representative recharge area of a path line (m^2)
q_i = recharge for a path line (m d^{-1})
$c_{i,t}$ = shallow groundwater quality for a path line at time t (mg L^{-1})
$c^*_{i,t}$ = groundwater quality for a path line at abstraction well at time t (mg L^{-1})
t = time (yr)
t_i = travel time of a path line (yr)
Q = abstraction rate of well field (m^3 d^{-1})
k = decay rate (yr^{-1})
R = retardation factor (-)

In the program the following assumptions are made:
– the recharge and travel times are constant in time;
– the impact of dispersion on concentrations in abstracted groundwater is negligible.

Abstraction rates generally vary in time so groundwater flow can hardly be considered fully steady. However, the transient effects have a minor influence on the water quality when the impact of diffuse pollution is considered, whereby small changes in boundaries of the recharge area does not influence the pollution load much.

The assumption that dispersion is negligible is justified in the case of diffuse pollution, since the spreading in concentrations of the parameter to be forecasted as a result of spreading in travel time is much larger than the spreading caused by dispersion and diffusion.

So, without describing the behaviour of solutes in detail, a quick insight in changes in the quality of abstracted groundwater due to changes in shallow groundwater can be obtained (IWACO, 1992a).

Methodology

To predict and evaluate groundwater contamination from non-point sources, several processes have to be assessed. These processes determine the changes in water quality between the infiltration at the soil surface and the abstraction at the wells. One of the most important contaminants from non-point sources, nitrate, has been chosen as the first contaminant to use in the methodology.
The recharge to the groundwater has to be quantified. Knowledge of the water flow in the unsaturated zone enables calculation of the solute transport to the shallow groundwater. Parameters determining the leaching of nitrate from the unsaturated zone (such as soil type, manure and fertilizer application, land use and thickness of the unsaturated zone) vary in time and space.
Quality changes during the flow in the saturated zone can be represented by retardation and decay.
The recharge area of the abstraction site is determined by a TRIWACO model of the groundwater flow. The leaching of nitrate in the recharge area is determined by FLUZO and NITRON. WELCON is used to determine the resulting concentrations in the abstracted water.
A large amount of data has to be transferred from one program to another. This is done by means of a Geographic Information System (GIS). The GIS used is PC-ARC/INFO version 3.4D, in which the database dBASE is implemented. This GIS allows efficient data storage, transfer and manipulation for the proposed methodology. The geographic distribution of input parameters and calculated nitrate concentrations are stored in ARC/INFO maps, so that regional differences can be taken into account. The data manipulation is carried out by dBASE.

The method consists of two main steps:
1. determination of the leaching of nitrate from the unsaturated zone to the saturated groundwater on the basis of the nitrogen load at the soil surface. The leaching of nitrate will be quantified as a function of physical, chemical and microbial processes by deterministic programs;
2. determination of the quality of the abstracted groundwater on the basis of the travel times and distribution

of travel times of the groundwater within the recharge area in combination with the determined leaching of nitrate.
In this way, the effects of remedial measures on groundwater quality can be quantified and compared.

Leaching of nitrate
The fluxes of water in the unsaturated zone are simulated by means of the program FLUZO. The result is used by the program NITRON that calculates the leaching of nitrate to the shallow groundwater (leaching of other solutes like sulphate, potassium and pesticides can also be calculated). The input data of these programs may vary in space and time. Land use and soil type vary in space only, while the values of manure and fertilizer application and the thickness of the unsaturated zone depend both on space and time.
FLUZO and NITRON perform one-dimensional vertical simulations. Multiple calculations are needed to account for the variation in space. The geographical distribution of each input parameter is stored in ARC/INFO as a map. These maps consist of areas with a constant value of the parameter. ARC/INFO combines all maps and overlays the areas, thus creating a map with the so-called 'homogeneous areas'. This is illustrated in Figure 2. So the parameters do not vary in space within a homogeneous area.

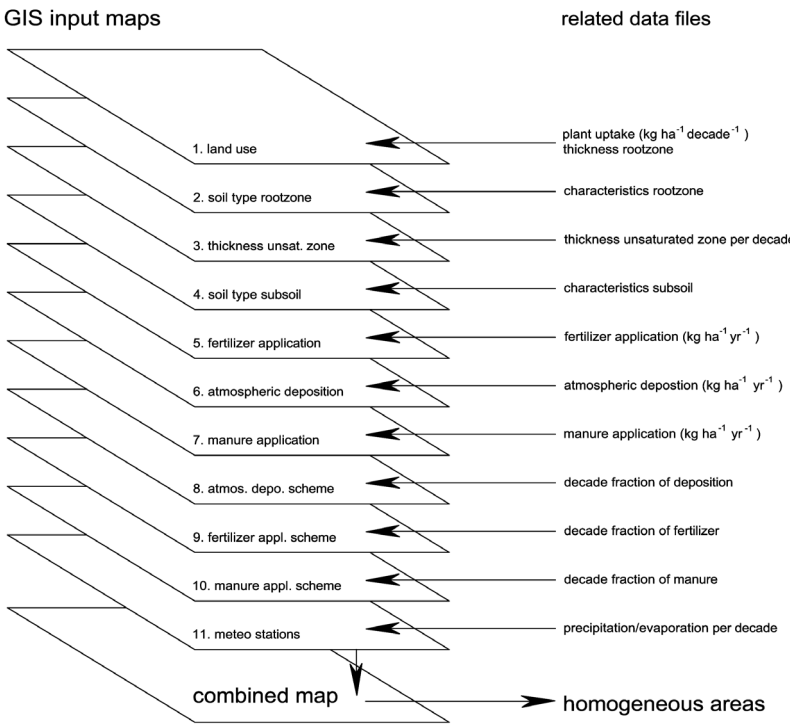

Figure 2. Relation between GIS input maps and related dBASE data files for the creation of the map representative areas.

The values are not stored directly in the maps. Each area in a map contains a label. These labels refer to dBASE-files which contain the actual data of the specific input parameter. In this way it is possible to use just one map for a parameter that changes in time (like manure or fertilizer application and plant uptake or precipitation and evaporation). In such cases for each label the actual value of the input parameter has to be given for each time step.
The leaching of nitrate is calculated for each homogeneous area. First, the FLUZO input files are generated by ARC/INFO and the fluxes of water in the unsaturated zone are calculated. This FLUZO-calculation produces the hydrological input data (for every time step) for NITRON. The remaining NITRON-input data are generated by ARC/INFO. The resulting time-series of the leaching of nitrate out of the unsaturated zone are stored in ARC/INFO. The leaching rates are averaged per year. As a result, two-dimensional maps of the leaching rates are

obtained for each year of the simulation. The data exchange has been automated and the resulting system is called FLUNIT (FLUzo-NITron linkage).

Quality of abstracted groundwater

Within the recharge area water that has leached out of the unsaturated zone moves towards the abstraction site. Some quality changes may take place during this saturated groundwater flow. However, because of the negative charge of the nitrate ion, retardation is assumed not to take place. Furthermore, denitrification is neglected in case the water in the aquifer contains oxygen. So for aerobic water no quality changes are considered in the saturated groundwater.

The homogeneous areas for the leaching of nitrate have not been selected with respect to recharge and travel time. Therefore, the recharge area is divided into so-called "representative areas". Each representative area is situated inside a homogeneous area of the nitrate leaching and is small enough, so that it can be represented by a single value for travel time and a single value for the recharge. The travel time is determined by tracing a path line from the center of the representative area. The program TESNET (part of TRIWACO) is used to generate a grid of these starting points of the path lines.

The path lines are traced using the program TRACE, also part of TRIWACO. So not only the leaching of nitrate out of the unsaturated zone is known, but also the recharge and travel time are known for each representative area. Finally, the changes in the quality of the abstracted groundwater are calculated with the program WELCON (Van den Brink et al., 1993b; IWACO, 1992a). WELCON adds the contribution of each path line.

Results

The system FLUNIT has been applied in a groundwater protection study in Hungary (IWACO, 1992c), that focused on the Halásztelek well field of the Budapest Water Works. Halásztelek is a village on Csepel Sziget, an island between two branches of the Danube just south of Budapest (see Figure 3), where leaching of nitrate plays an important role in the pollution of groundwater originating from the upland area (László and Homonnay, 1986; Alföldi, 1988)

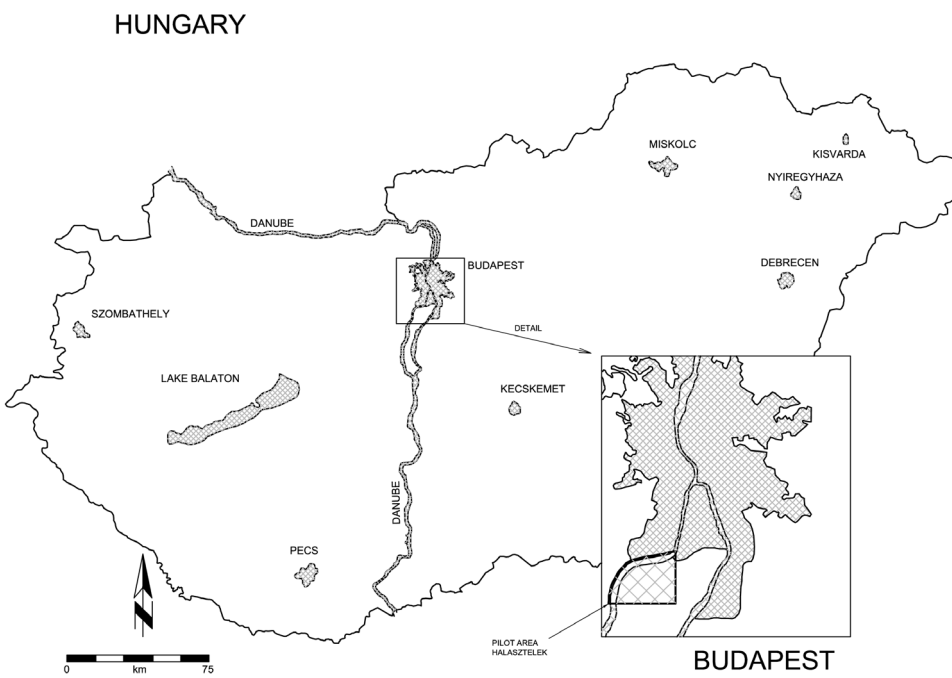

Figure 3. Model area.

Application

The well field at Halásztelek consists of 19 radial collector wells located along the Danube river. In the present situation wells number 3, 4, 5, 6, 14 and 15 (counting from north to south) are in operation. About 90% to 95% of the abstracted water comes from the river. The remaining 5% to 10% of the abstracted water originates from the upland (IWACO, 1992c). The nitrate concentration in the river averaged 13 mg NO_3 L^{1-} in the time period 1965-1985, while the maximum value was 23 mg NO_3 L^{-1} (László and Homonnay, 1986). This is well below the permissible concentrations for drinking water. However, the nitrate concentrations in the upland water are much higher and at several wells nitrate concentrations in the abstracted water exceeded the permissible limits.
A model has been set up that calculates the expected groundwater quality. The aim of this model is not so much the prediction of the groundwater quality, but much more the evaluation of groundwater protection strategies based on risk analysis and effectivity of possible measures.

The nitrate concentrations in the abstracted water depend strongly on the leaching of nitrate out of the unsaturated zone. The system FLUNIT has been used to predict nitrate concentrations in the abstracted groundwater, extrapolating the current situation and predicting future changes in case remedial measures are taken.
The calculations have been carried out in three steps. Firstly a groundwater flow model has been set up to determine the recharge area and the travel times. Secondly the leaching of nitrate out of the unsaturated zone has been calculated. Thirdly the quality of the abstracted groundwater has been predicted.
The land use, soil type and thickness of the unsaturated zone of the model area are shown in the Figure 4. Vegetable gardens characterize the villages. The land outside the villages is mainly used for maize, cereals and fruit trees. The most common soil type is loamy sand. The thickness of the unsaturated zone is greater than 2.5 meter everywhere. These conditions facilitate the leaching of nitrate, so the model area can be considered vulnerable to the leaching of nitrate.
The input of nitrogen can be divided into atmospheric deposition, manure application and fertilizer application. Estimates of the ranges are given in Table 1.

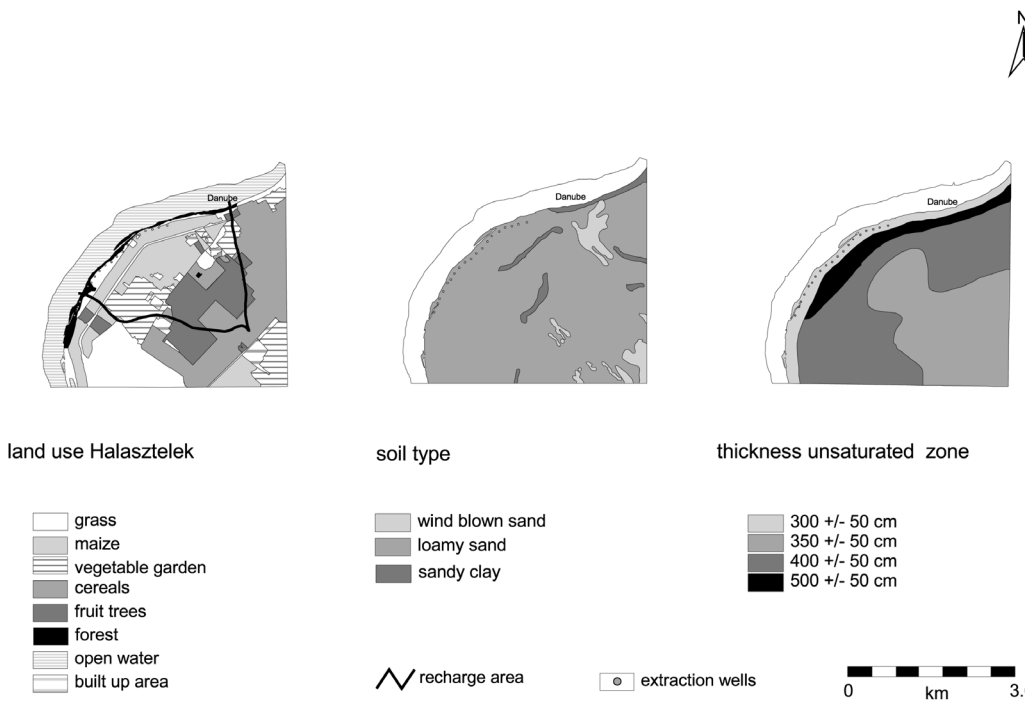

Figure 4. Land use, soil type and depth of piezometric surface below land surface in model area.

The values increase from the lower bound to the upper bound in the period 1955-1990 and are assumed to be constant thereafter (ENVIMARK, 1992). The atmospheric deposition is constant in the model area. The manure and fertilizer application depend on the type of crop. The manure application is assumed to take place in the spring while the application of fertilizer is divided into an application of 60% in the spring and 40% in the autumn. Predictions have been made for two scenarios that run until the year 2010, so insight in the efficiency of remedial measures on the quality of the abstracted groundwater can be obtained. The current application of fertilizer and manure is continued in scenario 1. The total nitrogen application is reduced to a maximum of 125% of the potential plant uptake in scenario 2. Details with respect to the nitrogen input in the scenarios are given in Table 1.

Table 1. Ranges of nitrogen input (kg ha^{-1} yr^{-1}) in period 1955-1990 (ENVIMARK, 1992; László, 1992) and the extrapolation of this input for the scenarios 1 and 2 till 2010.

Crop	Atmospheric deposition	Manure application	Fertilizer application	Total input as % of the potential plant uptake	
				Scenario 1 1955-2010	Scenario 2 1992-2010
Grassland	10-20	-	-	3-7	7
Maize	10-20	50-75	130-300	95-198	125
Cereals	10-20	10	130-300	79-174	125
Vegetable garden	10-20	50-75	130-300	100-208	125
Fruit trees	10-20	25-50	70-150	55-116	116

Long term averages (1941-1989) of the recharge and evaporation were used to determine the groundwater recharge for the model (ENVIMARK, 1992). It ranges from 55-75 mm yr^{-1}, with the exception of vegetable gardens, for which a recharge of 365 mm yr^{-1} was used, because of irrigation. The recharge occurs in the months November, December, January and February.
The resulting actual nitrate concentration in the abstracted groundwater depends on the nitrate concentration in the Danube and the upland portion of the abstracted groundwater. As mentioned before, the average and maximum nitrate concentration in the Danube are 13 and 23 mg NO_3 L^{-1}, respectively. Furthermore, the upland fraction of the abstracted water is assumed to range from 7.5% in the average situation to 15% as a worst case. Two combinations are distinguished:
I. combination representing the most likely situation in which the nitrate concentration in the Danube is 13 mg NO_3 L^{-1}, while the upland fraction of the groundwater is 7.5%;
II. combination resulting in upper bounds of the nitrate concentrations, in which the nitrate concentration in the Danube is 23 mg NO_3 L^{-1}, while the upland fraction of the groundwater is 15%.

Shallow groundwater quality: current situation
The quality of the shallow groundwater (i.e. the groundwater just below the groundwater table) is fully determined by the quality of the water that has leached out of the unsaturated zone. So the output of (FLUZO and) NITRON can be compared to the nitrate concentrations in the shallow groundwater.
The calculated values are yearly averages. For scenario 1 (continuation of current manure and fertilizer application) FLUNIT predicts very low nitrate concentrations of the shallow groundwater under grassland and forest in 1990 (see Figure 5a). In the shallow groundwater under cereals, fruit trees, maize and vegetable gardens the calculated nitrate concentrations range from 33 to 266 mg NO_3 L^{-1} (Table 2). A direct comparison between the calculated and measured concentrations is not possible, because no concentration measurements of the shallow groundwater are available. The nitrate concentrations have been measured at greater depths in the same aquifer (ENVIMARK, 1992). These concentrations are also influenced by other factors such as the saturated groundwater flow. This means that the measured nitrate concentrations are not as directly related to the set of input parameters as the calculated values.

The measurements have been added as an illustration for the order of magnitude or the nitrate concentrations occurring in the aquifer. No data are available that can be related to the calculated quality of the shallow groundwater more closely.

Table 2. Calculated nitrate concentrations in shallow groundwater and nitrate concentrations (mg NO_3 L^{-1}) measured deeper in the aquifer in the period 1984 - 1991 (ENVIMARK, 1992).

Crop	Calculated concentration shallow groundwater (1990)	Measured concentration deep groundwater (1984-1991)
Grassland	<1-2	<1-26*
Maize	125-266	12-170
Cereals	69-168	<1-204
Vegetable garden	191-214	12-292
Fruit trees	33-81	134-302

*minimum - maximum values of the measurements in the period 1984-1991.

Shallow groundwater quality: extrapolation into future

Figure 5b shows the nitrate concentration of the shallow groundwater when the current nitrogen application is continued till 2010 (scenario 1). The increase after the year 1990 is large. The current nitrogen concentrations in the shallow groundwater are not yet in equilibrium with the current nitrogen application at the soil surface. The high nitrate concentrations corresponding with the current nitrogen application have not yet reached the water table because of the low recharge rates.

Scenario 2 shows the improvement of the abstracted groundwater when nitrogen application in the upland is limited, starting in the year 1992, to a maximum of 125% of the potential plant uptake. The resulting concentration of the shallow groundwater in 2010 is shown in Figure 5c. The increase in the nitrate concentration is still large although not as large as in scenario 1. Toward the end of the simulation the nitrate concentrations reach an equilibrium with the (reduced) nitrogen applications.

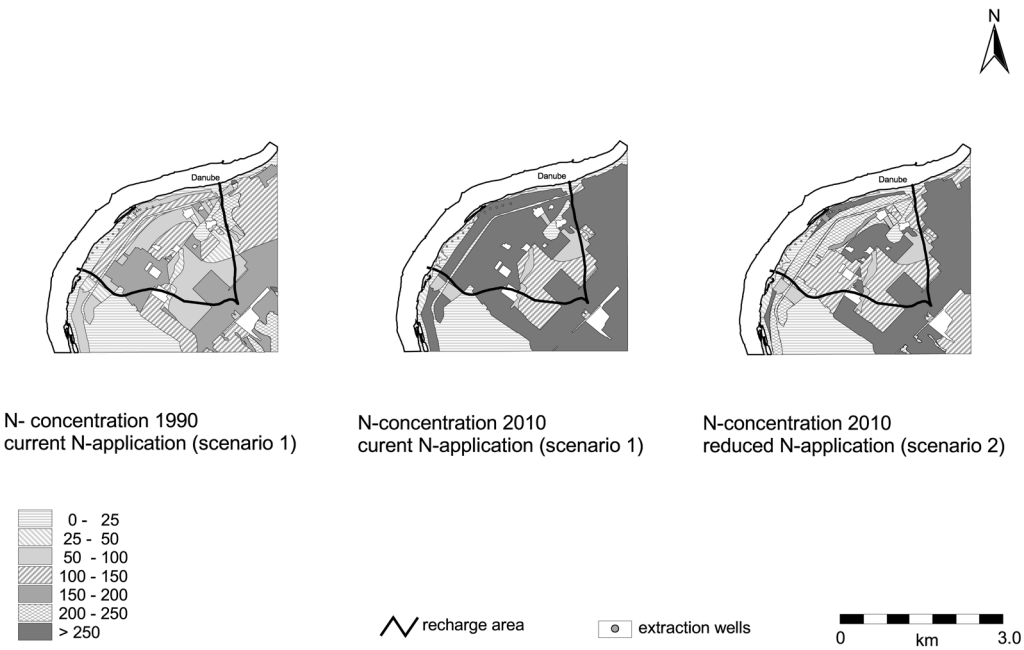

Figure 5. Calculated nitrate concentration in shallow groundwater (mg NO_3 L^{-1}) for 1990 and 2010 when no protecting measures are taken (scenario 1) and when the total nitrogen application is limited to 125% of the potential plant uptake (scenario 2).

Quality of abstracted groundwater

Shallow groundwater within the recharge area, which is indicated by a heavy line in the Figures 5a, b and c, eventually reaches the wells of the Halásztelek water works. WELCON calculates when this water will arrive at a well and also what the resulting concentrations will be in the abstracted water. Nitrate is assumed to behave like a conservative solute because of aerobic conditions in the aquifer. Figure 6 shows for the scenarios 1 and 2 the quality of the upland water that reaches the wells.

The abstracted water is a mixture of upland water and water from the Danube. Two values of the nitrate concentration in the river water have been used in the calculation of the resulting concentrations in the abstracted water, viz. 13 and 23 mg NO_3 L^{-1}. The former is the average value over the past 20 years and the latter is the maximum value observed during the same period. Figure 7 shows that the nitrate concentrations in the abstracted water will rise. The concentrations will continue to rise after 2010 if no measures are taken to reduce the total nitrogen application (scenario 1). The models predict concentrations to stabilize at about 30 mg NO_3 L^{-1} in 2010 if the nitrogen application in the upland is limited to 125% of the plant uptake (scenario 2).

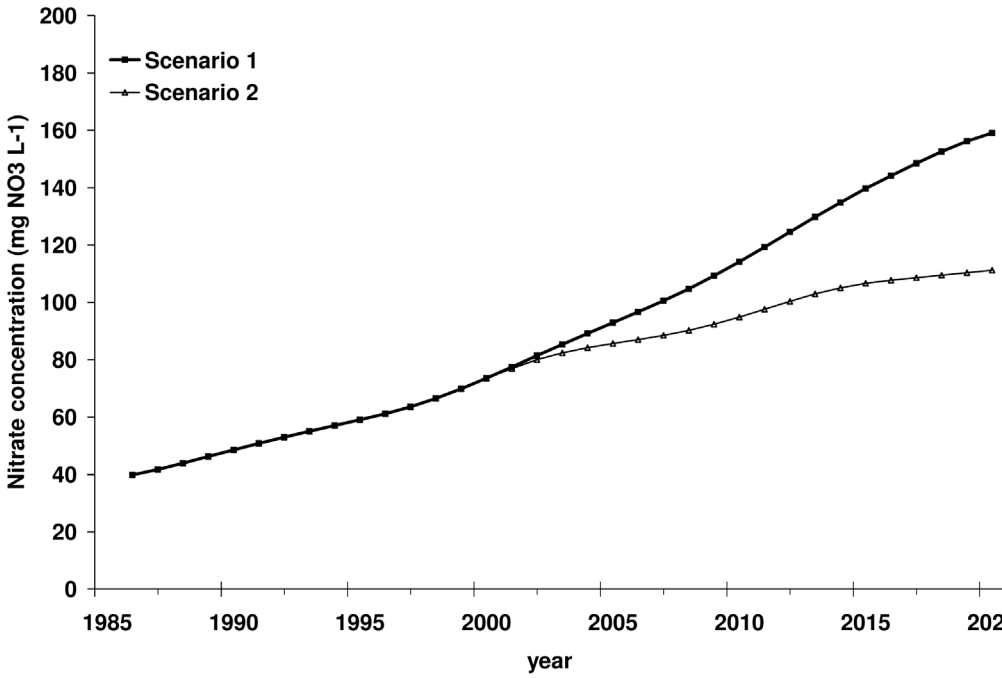

Figure 6. Calculated nitrate concentration in upland portion of the abstracted groundwater (mg NO_3 L^{-1}) when the current application amount is continued (scenario 1) and when the total application is reduced to 125% of the potential plant uptake starting in 1992 (scenario 2).

Discussion

Discussion of the methodology

The calculated nitrate concentrations of the shallow groundwater show rather good agreement with the measured values that have been measured at a greater depth (see Table 2). The influence of horizontal transport is not very large, because otherwise the measured nitrate concentrations in the lower groundwater underneath grassland would show a much higher nitrate concentration. It is expected that the influence will increase in the future due to the fact that the groundwater velocities have increased since the start of groundwater abstraction in 1986.

The calculations are averages both in time and space. This means that not all of the observed fluctuation can be reproduced.

This does not pose much restriction on the simulation of a macro parameter such as nitrate. Foster et al. (1991) have found that transport by preferential flow is not very important for macro parameters, although it can be the dominating process for the transport of micro parameters, such as pesticides.

The system is designed in such a way that the important data can be used with the available resolution. The distribution can easily be refined when better data has become available. It can be seen from this case study that the use of several relatively simple deterministic programs to simulate the transport of nitrates through soils and groundwater can be of great value for the planning and evaluation of remedial measures.

Discussion of the results

The current situation has been extrapolated to the year 2010. Figure 5b shows the concentrations in the shallow groundwater if the current application of manure and fertilizer is continued (scenario 1). The levels are approximately 80% higher than the current levels. Thus, continuing the current situation till 2010 will result in increasing nitrate concentrations. There is no equilibrium between total input of nitrogen and the resulting concentration in the shallow groundwater in the current situation.

Scenario 2 is the reduction of the nitrogen application from the current values to a maximum of 125% of the potential plant uptake. Even in that case an increase of the current nitrate concentrations is predicted.

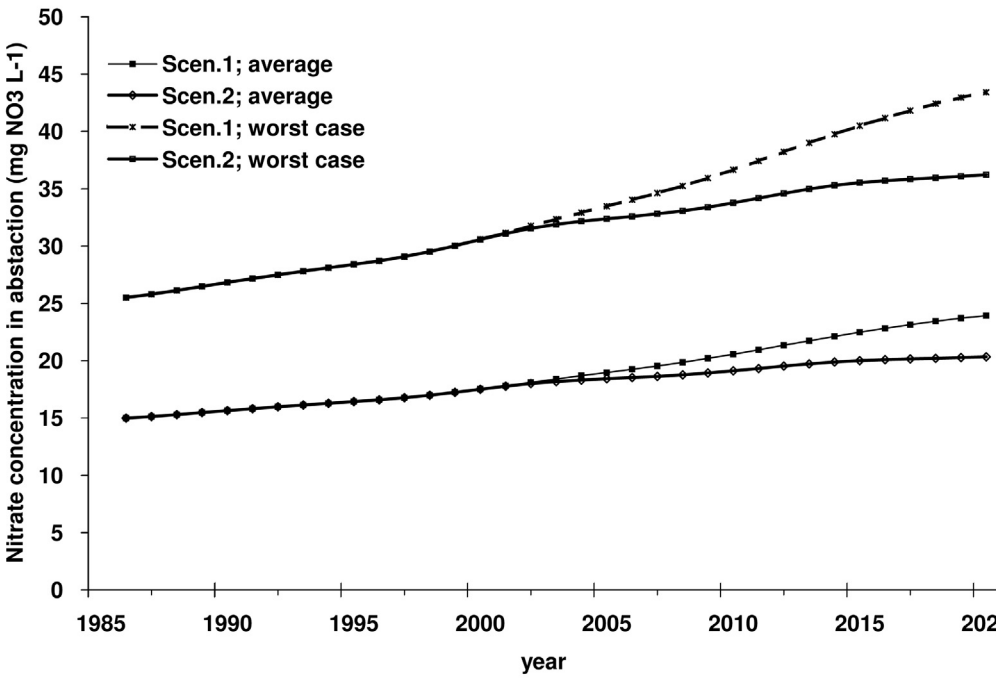

Figure 7. Extrapolation of the current situation: calculated nitrate concentration in the abstracted water (mg NO_3 L^{-1}) with the combinations:
Scenario 1 for nitrogen application; nitrate concentration Danube 13 mg NO_3 L^{-1}; upland fraction 7.5%;
Scenario 1 for nitrogen application; nitrate concentration Danube 23 mg NO_3 L^{-1}; upland fraction 15%;
Scenario 2 for nitrogen application; nitrate concentration Danube 13 mg NO_3 L^{-1}; upland fraction 7.5%;
Scenario 2 for nitrogen application; nitrate concentration Danube 23 mg NO_3 L^{-1}; upland fraction 15%.

It must be stressed, that the actual plant uptake in most cases is less than the potential plant uptake, even in cases when the total nitrogen application exceeds the potential plant uptake. Some of the nitrogen leaches out of the root zone before it can be taken up by plants. So, a nitrogen shortage in the root zone may coincide with the leaching of nitrate out of the unsaturated zone (van Ommen et al., 1992). This means that even a reduction

of the total nitrogen application to 100% of the potential plant uptake would not prevent leaching of nitrate to the groundwater. Furthermore, some of the applied nitrogen will be denitrified and some will be immobilized in organic matter. This means, that the nitrate leaching is not proportional to the nitrogen application.

It can be seen from the Figures 6 and 7 that reducing the total nitrogen application to a maximum of 125% of the potential plant uptake causes a significantly lower increase of the nitrate concentration than would be the case if the current nitrogen application is continued.

The nitrate concentrations in the abstracted water will continue to rise after 2010 if no measures are taken. If the current nitrogen application is continued, it is very likely that during a part of the year the nitrate concentrations will exceed the value of 40 or even 50 mg NO_3 L^{-1}. The impact of reducing the nitrogen application stabilizes the predicted concentrations at about 35 mg NO_3 L^{-1}. The impact of the measure is not seen in the abstracted water until the year 2002. This time lag is caused by the small velocity of the unsaturated groundwater and the relatively long travel times in the saturated zone.
The Hungarian standard for drinking water allows a maximum nitrate concentration of 40 mg NO_3 L^{-1}. International standards permit slightly higher values. The World Health Organisation has set a maximum permissible level of 50 mg NO_3 L^{-1} in 1992. It has also published standards in 1984, in which a maximum of 10 mg N L^{-1} (=45 mg NO_3 L^{-1}) was mentioned. The European Community allows a maximum of 50 mg NO_3 L^{-1} in drinking water. These values are well above the predicted 35 mg NO_3 L^{-1} in 2010 if nitrogen application is limited to 125% of the plant uptake. However, the nitrate concentration in the abstracted water will rise to approximately 50 mg NO_3 L^{-1} if the current level of nitrogen application is continued. It is very likely, then, that the standard for drinking water will be exceeded at least during part of the year.

Conclusions

The system FLUNIT is a useful instrument in the planning and evaluation of groundwater policies. By using several relatively simple deterministic programs together with GIS, the data can be handled in the same resolution as it becomes available. Two-dimensional time dependent distributions of the quality of the water near the groundwater table can be calculated. Consequently, good predictions of the quality of abstracted water can be made with a set of travel times from path lines covering the recharge area of the abstraction site.
Furthermore, it offers fast and detailed insight into changes in groundwater quality due to remedial measures such as reducing the nitrogen application or changing the (yearly) application period of manure and fertilizer. This means, that the efficiency of different remedial measures can be compared, so FLUNIT can be a powerful tool for the evaluation of groundwater protection strategies.

The presented system FLUNIT has been used to predict the quality of the groundwater near the water table and the abstracted water at Halásztelek on Csepel Sziget (near Budapest, Hungary). The nitrate concentrations in the unsaturated groundwater are not yet in equilibrium with the nitrogen application. Therefore, the concentrations in the shallow groundwater will continue to rise.
If the current nitrogen application is continued, the concentration in the abstracted water will continue to rise for several decades. The level will be about 50 mg NO_3 L^{-1} in the year 2010, on the edge of international standards. The concentrations will stabilize at about 35 mg NO_3 L^{-1} if the nitrogen application will be limited to 125% of the potential plant uptake, which is well below the standards for drinking water.

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Chapter 2

Statistical analysis of anthropogenic versus geochemical-controlled differences in groundwater composition in the Netherlands

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Monitoring wells of the Dutch National Groundwater Quality Network (Photo: RIVM and H.P. Broers).

Statistical analysis of anthropogenic versus geochemical – controlled differences in groundwater composition in the Netherlands

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Abstract

Quantitative insight into the impact of land use on groundwater composition is vital for the sustainable management of aquifers, especially those situated in urban areas. The objective of the present study was to determine the effects of land use on groundwater composition throughout the Netherlands, relative to the influence of groundwater age, groundwater origin, and geochemical processes. Using a nationwide dataset on shallow groundwater from the Dutch National Groundwater-Monitoring Network, land use effects were quantified statistically for all major ionic substances. Analysis of variance was applied to the entire dataset, to a priori groups based on groundwater age, and to non-a priori statistical clusters determined by fuzzy c-means clustering. The results indicated that the effect of land use on groundwater composition in the Netherlands strongly depends on geochemical processes and groundwater age. Land use effects were most pronounced in younger, geochemically less-evolved groundwater: for example, urban areas showed an increase in Ca, Na, Cl, and HCO_3 by a factor 2-4, an increase in K and B by a factor 4-10, and a decrease in Al by a factor 10 compared to natural areas. Similarly, agricultural areas showed an increase in Ca, NO_3, and Fe by a factor 2-4, an increase in K by a factor 4-10, and a decrease in Al by a factor 2. Compared to urban areas, agricultural areas contained less Na, Cl, and Al and more NO_3. Our results demonstrate that the importance of the processes controlling groundwater composition in the Netherlands is scale-dependent, with geochemical processes being more dominant on a national scale and land use on regional and local scales.

Introduction

Land use affects the composition of (infiltrating) groundwater either directly by adding or removing components or indirectly by changing soil physicochemical conditions. The impacts of agricultural and natural land use can be predicted with forward (deterministic) models if the required data on the application of substances and the physicochemical conditions are available (e.g., Tiktak, 1999; Van Drecht, 1993; van den Brink and Zaadnoordijk, 1995). The prediction of the impact of urban activities, necessary to manage and sustain the quality of urban aquifers, however, is hampered by the greater spatial heterogeneity of these activities.
Several studies have quantified urban and agricultural land use effects on groundwater composition. Trojan et al. (2003), for example, showed significant effects of land use on groundwater composition in the uppermost meter of the Anoka sand plain aquifer in Minnesota (USA), with the highest concentrations of trace inorganic chemicals and volatile organic compounds being found in sewered urban and commercial areas, respectively. Nitrate concentrations, herbicides, and metabolites were highest under (irrigated) agricultural and non-sewered urban areas. In addition to the significant differences in groundwater composition beneath existing land use areas, these researchers observed significant changes in groundwater composition following changes in land use. Based on their observations, Trojan et al. (2003) argued that land use is the dominant factor affecting the shallow groundwater composition of the Anoka aquifer. Lerner et al. (1999) described the loadings of non-agricultural nitrogen on shallow (5-7 meters below surface [mbs]) groundwater composition under Nottingham (UK). They found that nitrate concentrations were equally high in the groundwater under Nottingham and the surrounding agricultural area and that urban sources of nitrogen in the groundwater included leaking sewers (13%), leaking water mains (36%), and non-isolated landfills, public gardens, and industrial chemical spills (together 50%). Other authors (Cronin et al., 2003; Fukada et al., 2004), however, found that the nitrate

contamination of shallow Nottingham groundwater (7 mbs) primarily resulted from sewage and atmospheric deposition. With regard to deep groundwater abstracted by industrial wells and the public water supply in the same city, Barrett et al. (1999) were unable to find suitable markers for precipitation and mains leakage. This confirmed the need for a multi component approach rather than the use of individual marker species to obtain insight into the impact of sewer leakage on groundwater quality. Even though Barrett et al. (1999) demonstrated that the impact of sewer leakage on the quality of deep groundwater under Nottingham was generally not high, other researchers (e.g., Foster et al., 1998; Foppen, 2002) considered domestic and industrial effluents in urbanized areas to be a source of the widespread contamination of shallow groundwater.
The aforementioned studies generally focused on one locality and used marker species to find the sources of contamination. In that context, land use is the only factor explicitly considered important for groundwater composition. A different approach was undertaken by Absil (1997) in a large-scale study of the Netherlands. He compared shallow groundwater composition below urban areas with those of adjacent agricultural areas and accounted for their geographical influences on groundwater quality by assuming an a priori spatial relationship between groundwater composition in urban wells and their agricultural neighbours. Nonetheless, he did not explicitly account for other factors such as geochemistry, groundwater age, and groundwater source (seawater, surface water, and rainwater), factors which have been shown to affect Dutch groundwater composition (Stuyfzand, 1993; Frapporti et al., 1993; Broers, 2002). The aim of the Dutch National Groundwater-Monitoring Network is understanding of groundwater composition and the timely detection of trends (RIVM, 2000). This requires knowledge of all of the factors influencing groundwater chemistry.
The main objective of the present study, therefore, was to determine the impact of land use on groundwater composition across the Netherlands relative to the influence of groundwater age, groundwater origin, and geochemical processes. The hypothesis was that the relative impact of various land use functions on all major groundwater solutes and general groundwater quality parameters (EC, pH, temperature) can be analyzed as a function of a priori-assigned or non-a priori-established dominant factors. Using a nationwide dataset on shallow groundwater from the Dutch National Groundwater-Monitoring Network, the relative impact of land use on groundwater composition was established by statistically analyzing land use effects on (1) the undivided entire dataset, (2) 'fuzzy' data clusters which were determined without any a priori knowledge of the controlling processes, and (3) data subsets (groups) based on groundwater age. The first analysis focused solely on land use effects and served as a reference. The second analysis regarded the effect of land use to be dependent on a dominant factor established by the cluster procedure itself. Finally, the third analysis considered the effect of land use to be dependent on the more dominant factor, groundwater age. In studies by the Dutch National Groundwater-Monitoring Network, this dominant factor turned out to be geochemistry.

Materials and methods

Hydrogeology of the Netherlands

Geologically, the Netherlands can be divided into a Holocene and a Pleistocene part. The low-lying western part of the country is composed of shallow Holocene marine and peri-marine deposits as well as fluvial deposits from the Rhine and Meuse rivers. To the north-northwest, the aquifer system is covered by semi-impervious coastal deposits from the Holocene epoch. The eastern and southern Pleistocene part of the Netherlands comprises older fluvial deposits and glacial or peri-glacial deposits near the surface. The groundwater basin is wedge-shaped, with a thickness of about 50 m in the east-southeast to about 250 m in the west-northwest (e.g., Frapporti, 1994; Huisman et al., 1998; Broers, 2002).

Groundwater hydrology in the Netherlands is controlled by the presence and lithology of unconsolidated sediments, deposited in a still subsiding sedimentary basin (Huisman et al., 1998). The principal sources of recharge of the natural groundwater system are precipitation and riverbank infiltration. In the eastern and southern (i.e., Pleistocene) part of the country, groundwater is regionally discharged through a natural drainage system of rivers and brooks. This natural drainage system has been extended artificially and leads to many local, shallow hydrogeological systems superimposed on the regional systems, sometimes with flow directions opposite the regional flow (Dufour, 2000). Surface runoff in this part is 0 – 30 mm yr^{-1}, while the recharge of these aquifers is 70 – 350 mm yr^{-1} (Huisman et al., 1998). The Pleistocene deposits form the freshwater aquifers of the Netherlands. In some parts of the country, however, the Pleistocene aquifers are covered by confining

Holocene fluvial deposits. These aquifers are recharged with freshwater coming from rain or river water. In other areas, coastal dunes serve as a natural area of recharge or as an artificial area of recharge for the production of drinking water through the infiltration of pre-purified Rhine and Lake IJssel water. In the northern and western (Holocene) part of the Netherlands, groundwater is regionally discharged into polders through a system of drains and ditches in a confining layer of peat and marine clay. Surface runoff in this part is 20 – 100 mm yr^{-1}, while the recharge of these aquifers is 30 – 100 mm yr^{-1} (Huisman et al., 1998).

Figure 1 summarizes some of the relevant information on the geology and hydrology of the Netherlands and shows the position of the monitoring wells of the Dutch National and Provincial Groundwater-Monitoring Networks

Dataset

The Dutch National Groundwater Quality Monitoring Network (LMG) was established in 1979 by the Dutch National Institute for Public Health and the Environment (RIVM) and reached its present size in 1992. Its major goal is to assess and quantify human impact on groundwater quality over space and time (Van Duijvenbooden and Van Waegeningh, 1987). The monitoring wells are evenly distributed throughout the country and are located in areas where the influence of point-source pollution is thought to be minimal (Van Duijvenbooden et al., 1985; Van Duijvenbooden and Van Waegeningh, 1987). Each well has two 2" (5.08 cm) screens that are two meters in length. The screens are located at approximate depths of 9 and 24 mbs (the exact depths depend on local hydraulic properties). All of the wells were newly drilled using a cable tool-drilling system (Van Duijvenbooden et al., 1985). In addition to the national network, there are also regional monitoring networks (Provincial Groundwater Quality Monitoring Networks; PMGs). The first one was established in 1989. The objective of these regional networks is to yield more detailed information for use in regional water management and soil protection strategies (Broers, 2002). The present study used data from both the LMG and the PMGs.

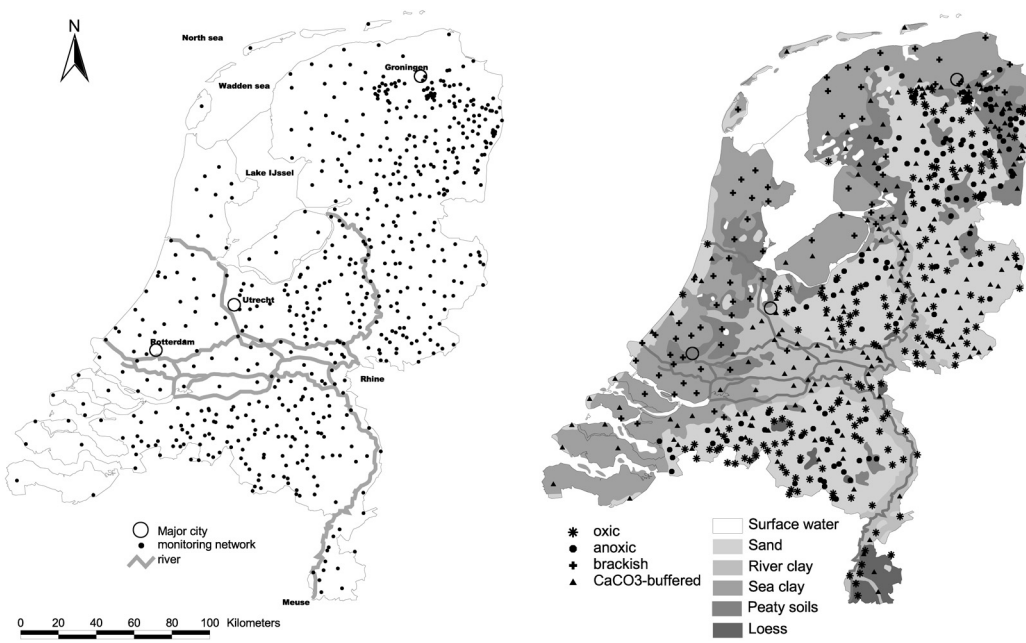

Figure 1. Schematized maps of (a) the monitoring network and main water courses and (b) soil types (after Huisman et al., 1998) and the distribution of samples over FCM clusters.

LMG and PMG groundwater data are obtained in a uniform, controlled way throughout the country. Sampling and analysis of the national (LMG) wells is performed by RIVM researchers; the provincial wells are sampled and analyzed by contractors. Standard operating procedures (SOPs) and quality assurance and control (QA-QC) procedures are used by all parties to ensure quality and comparability of the national and provincial data. The SOPs, developed by the RIVM, are updated annually/continually and comprise the sampling and conservation of specimens. Special care is taken during both the SOP and QA-QC procedures to minimize the introduction of bias or trends. The SOPs have limited validity – only for the length of the sampling period – to ensure that the operating procedures are up to date and that RIVM is able to keep track of the sampling activities and the conservation and handling of the specimens. With regard to these parameters and to ensure sound chemical results, the methods used to analyze the specimens were accredited by the Dutch accreditation council (RVA, 2006). Finally, a procedure was developed to evaluate the probability of the results of the chemical analyses. Outliers were separated and evaluated individually with regard to possible changes in either the wells or the labeling of the samples (Reijnders et al., 2004).
The 9- and 24-mbs screens are sampled annually using a submersible Grundfoss pump. Three well volumes are pumped before sampling. Electrical conductivity (EC), pH, temperature, alkalinity, and dissolved oxygen are measured in the field. Concentrations of Na, K, Mg, Ca, Fe, Mn, Al, NH_4, HCO_3, Cl, NO_3, SO_4, and total PO_4 are determined in the laboratory. The present study used LMG and PMG data from the 1997-1998 period. It focused on data from the 9-mbs screens to establish the impact of land use on shallow groundwater quality with minimum bias and a relatively short period between infiltration and observation. Assuming an infiltration rate of 1 m yr^{-1} (Huisman et al., 1998), it takes the infiltrating groundwater approximately 9 years to reach the 9-mbs screens. During this period, the horizontal groundwater movement at this depth is approximately 9 – 90 m and thus within the size of the land use plots in which the wells are centered. The dataset included data on natural (103 wells), agricultural (452 wells), and urban (60 wells) areas (615 wells total). Note that, due to atmospheric deposition, the groundwater composition under natural areas was not entirely free of human impact.
Using the Freeze and Cherry (1979) charge balance definition, we tested the individual chemical analyses for charge balance as follows:

$$\% \text{ Charge Balance Error} = \frac{\sum zm_c - \sum zm_a}{\sum zm_c + \sum zm_a} \; .100\ \% \qquad (1)$$

where z is the absolute value of the ionic valence, m_c is the molality of cationic species, and m_a is the molality of the anionic species. Less than 5% of the samples analyzed showed a >10% discrepancy in ion balance. This is an acceptable charge balance error for the statistical treatment of a large hydrochemical dataset (Frapporti, 1994; Güler and Thyne, 2004).

Data Analysis
Many techniques are available that can divide a heterogeneous dataset into homogeneous groups. Although traditional deterministic methods classify data on the basis of general hydrogeochemical knowledge, they are often not optimal for a specific dataset. The Discriminant Function Analysis (DFA), for example, is suitable if homogeneity is expected with a known classificatory parameter. However, while DFA may confirm the influence of known parameters, it will not identify an unknown or unexpected parameter. Because of this, we explored the impact of land use on groundwater quality by analyzing the variance of the entire data set, of non-a priori clusters determined by fuzzy c-means clustering, and of a priori groups based on groundwater age.

Fuzzy c-means clustering (FCM clustering) – Cluster analysis attempts to reorganize the dataset into homogeneous groups, making no use of any a priori knowledge. Unknown parameters, which may cause inhomogeneity of the data, may thus be identified. The use of c-means clustering is warranted if a partition in a predetermined number of groups is appropriate. A major disadvantage of both DFA and c-means clustering is their rigidness, i.e., they assign a sample unambiguously to a cluster or group. Any compositional overlap between groups, a common phenomenon in hydrogeochemical datasets, therefore, is not adequately dealt with. Clustering may also be adversely affected by outliers.

Fuzzy c-means clustering can deal better with an overlap between clusters and with outliers (Frapporti et al., 1993). The technique allows some vagueness – fuzziness – in the description of the cluster model. Similarity between a sample and a cluster is expressed as a membership function, varying from 0 (completely different from the cluster) to 1 (exactly the same as the cluster, i.e., the cluster center). A sample is allocated to the cluster it has the highest membership with if that membership is at least twice as high as the sample's next highest membership. If it cannot be assigned to a cluster according to this rule, the sample is considered an outlier and excluded from further statistical analysis (Frapporti, 1994). This was the case for two of the 615 samples in the present study. The cluster centers were calculated iteratively for all groundwater properties simultaneously. This appears to be an appropriate classification technique for (hydro)geochemical datasets in general since chemical and physical properties of groundwater systems vary continuously rather than abruptly (Frapporti, 1994, Güler and Thyne, 2004). The fuzzy c-means algorithm we used was described by Vriend et al. (1988).

Functionals for determining the appropriate number of clusters in a dataset are often inconclusive (Frapporti, 1994). We chose the number of clusters heuristically on the basis of the regional distribution, the hydrogeochemical interpretability, and the unimodality of the distribution of the parameters per cluster. Four clusters were considered optimal to achieve an adequate division of the variation within the dataset as well as to maintain an adequate number of samples within each cluster in order to perform meaningful analyses of variance. The consistency of the clustering was supported by similar results for the same dataset from another year (Frapporti et al. 1993).

A priori grouping based on groundwater age – Data were also grouped by groundwater age since this is a key factor in determining the concentration of dissolved contaminants in the subsurface (Broers, 2002). We were able to establish groundwater age (pre-1950 or post-1950) in 439 (71%) of the 615 wells using tritium concentrations. Groundwater was classified as 'young' (post-1950) if the measured concentrations exceeded the detection limit of 2 tritium units (TU; 1 TU = 1 ^{3}H atom per 10^{18} atoms of H) (Broers, 2002). The use of tritium as an environmental tracer for 'young' groundwater was based on the large-scale introduction of tritium to the atmosphere by aboveground nuclear experiments in the period 1950-1965 (Fritz and Fontes, 1980; Robertson and Cherry, 1989). Tritium is a valid tracer for groundwater age in the Netherlands since the regional groundwater systems resemble piston-flow systems, where mixing is negligible (Robertson and Cherry, 1989; Maas 1994). The present natural tritium concentration of rainwater in the Netherlands is approximately 5 TU (Meinardi, 2003).
Analysis of variance (ANOVA) – Univariate analyses were performed in SPSS version 12 (SPSS, 2003). One-way ANOVAs were applied for each chemical parameter. The ANOVAs tested the hypothesis that the mean of each chemical or physical parameter did not change with regard to the various land uses (natural, agricultural, and urban areas). The homogeneity of the parameter variances was tested using Levene's test. A least squares difference test (LSD test) was used to perform pair-wise comparisons between the group means. If the variance was not homogeneous, Dunnett's T3 test was used to determine the differences between the parameter means. Dunnett's T3 test is a pair-wise comparison test based on the Studentized maximum modulus.

Results

FCM clustering

Table 1 shows the properties (centers) of the four clusters. Based on their characteristics, the clusters were separated into oxic, anoxic, carbonate-buffered, and brackish water types. The groundwater of the first three clusters originated from rainwater, including riverbank infiltration, whereas the source of water in the fourth cluster was Holocene marine paleowater (Post, 2004). In strong agreement with Frapporti et al. (1993), geochemical processes and groundwater origin appear to be the strongest causes of the variation in groundwater composition.
The oxic cluster was characterized by a rather low pH (5.5) and bicarbonate concentration (63.4 mg L^{-1}) and a high concentration of aluminum (1620 µg L^{-1}). The mean concentrations of chloride (38.6 mg L^{-1}), potassium (21.1 mg L^{-1}), sodium (24.4 mg L^{-1}), nitrate (25.6 mg L^{-1}), and sulfate (78.5 mg L^{-1}) were relatively high compared to rainwater, even when evapotranspiration was taken into account. This indicates additional ion sources such as road salts and fertilizer. Ninety-seven percent of the groundwater in the oxic cluster had >2 TU tritium and thus infiltrated after 1950. In other words, this cluster consisted almost entirely of young, recently infiltrated groundwater.

Table 1. Groundwater properties (centers) of the four FCM clusters (in mg L^{-1} [Al and B in µg L^{-1}], pH in units, EC in mS m^{-1}, T in ^{o}C, tritium age in % of groundwater infiltrated after 1950).

Cluster	Oxic (n=173)				Anoxic (n=131)				$CaCO_3$-buffered (n=229)				Brackish (n=80)			
Parameter	Cluster center	Min.	Max.	St.Dev.	Cluster center	Min.	Max.	St.Dev.	Cluster center	Min.	Max.	St.Dev.	Cluster center	Min.	Max.	St.Dev.
Calcium	58.1	3.1	228	47.3	11.8	0.3	90.0	10.9	95.3	16	260	46.7	211	25.7	1471	209
Magnesium	12.5	1.9	47.1	8.3	4.2	0.4	27.0	3.8	10.7	1.9	73.5	7.6	124	10.6	1284	219
Potassium	21.1	0.4	211	25.5	3.4	0.4	36.0	4.6	6.3	0.3	100	11.5	45.3	2.0	304	58.6
Sodium	24.4	3.1	120	18.0	25.1	3.3	161	29.4	35.2	5.1	268	32.9	970	28.4	9461	1815
Chloride	38.6	3.0	200	30.0	35.8	1.2	282	40.2	54.4	4.8	398	49.3	1725	29.6	17085	3304
Bicarbonate	63.4	3.0	514	112	39.3	3.0	301	44.9	281	18.0	756	136	808	109	3198	498
Sulfate	78.5	10.0	480	56.9	40.3	1.5	360	41.4	62.2	1.5	318	59.1	116	1.5	2121	313
Nitrate	25.6	0.1	136	26.1	0.6	0.1	12.1	1.6	0.8	0.1	50	4.1	0.2	0.1	1.5	0.2
Aluminum	1620	2.0	17151	3107	502	2.0	10000	1412	28.8	2	900	85.3	53.3	2.0	920	121
Manganese	0.5	0.0	11.5	1.2	0.2	0.0	2.9	0.4	0.7	0.0	37.0	2.5	1.2	0.1	11.2	1.5
Iron	0.4	0.0	33.0	2.6	9.9	0.0	109	12.5	11.7	0.0	77.9	13.4	14.5	0.0	120	17.5
Boron	53.9	5.4	418	59.2	16.1	3.0	181	24.6	61.0	3	869	99.9	719	15.1	7597	1219
Temperature	11.2	8.7	17.0	1.7	10.8	8.5	16.0	1.3	11.0	8.8	16.4	1.2	11.0	9.8	13.6	0.8
Electrical conductivity	59.4	10.2	189	30.8	27.3	7.0	103	16.6	70.7	22.2	157	26.3	575	82.6	4670	891
pH	5.5	3.9	7.8	1.0	5.8	4.2	8.6	0.7	6.9	5.5	8.2	0.43	6.9	5.8	7.8	0.3
% with TU > 2	97				79				73				36			

The anoxic cluster showed an absence of nitrate and a presence of iron (9.9 mg L^{-1}). The pH (5.8) was still rather low, but closer to neutral than the oxic cluster (5.5). The chloride (35.8 mg L^{-1}) and sodium (25.1 mg L^{-1}) concentrations were comparable to those of the oxic cluster, while the concentrations of potassium (3.4 mg L^{-1}) and sulfate (40.3 mg L^{-1}) were lower. Compared to the oxic cluster, groundwater of the anoxic cluster appeared to be influenced more by subsurface reactions, especially redox processes, resulting in the disappearance of nitrate and the appearance of iron. This was confirmed by the lower proportion of samples containing >2 TU tritium, i.e., that infiltrated after 1950 (79% in the anoxic cluster versus 97% in the oxic cluster). The interaction of water with the aquifer sediments resulted in lower redox levels. The lower electrical conductivity and potassium and sulfate concentrations suggest less anthropogenic influence in the past and buffering in the subsurface.

The carbonate-buffered had increased concentrations of calcium (95.3 mg L^{-1}) and bicarbonate (281 mg L^{-1}) and a neutral pH value (6.9). In addition, concentrations of iron (11.7 mg L^{-1}) and manganese (0.7 mg L^{-1}) were relatively high and concentrations of oxygen and nitrate low due to redox processes. The chloride (54.4 mg L^{-1}), sodium (35.2 mg L^{-1}), and sulfate (62.2 mg L^{-1}) concentrations were comparable to those of the anoxic cluster. Compared to the oxic and anoxic clusters, groundwater of the carbonate-buffered cluster was older: 73% of the samples contained >2 TU tritium, i.e., infiltrated after 1950 compared to 79% in the anoxic cluster and 97% in the oxic cluster. This cluster, therefore, was influenced even more by subsurface geochemical reactions, with carbonate dissolution as an important additional process. As a result, the electrical conductivity was higher than in the anoxic and oxic clusters.

Finally, the brackish cluster had high mean concentrations of chloride (1725 mg L^{-1}), sodium (970 mg L^{-1}), potassium (45.3 mg L^{-1}), and magnesium (124 mg L^{-1}). The sulfate-to-chloride ratio was far below the seawater ratio, indicating the subsurface reduction of sulfate. The composition can be explained by the mixing of seawater with river surface water or rainwater. Cation exchange, carbonate chemistry, and sulfate reduction all determine the groundwater composition of this cluster (Beekman, 1991). This cluster also had the lowest percentage of young groundwater: only 36% of the samples contained >2 TU tritium, i.e., infiltrated after 1950.

The distribution of the clusters throughout the Netherlands is shown in Figure 1. The oxic and anoxic freshwater clusters are predominantly located in the sandy Pleistocene areas in the south, east, and center of the country, while the carbonate-buffered freshwater cluster occurs in both the Pleistocene river clay and sandy areas. The brackish cluster occurs predominantly in the western and northern parts of the Netherlands, where marine transgressions occurred during the Holocene Epoch (Post, 2004). In the middle of the Holocene Epoch, the shoreline was several tens of kilometers more inland than it is today. The confining Holocene layer in this area consists of peat, marine clay, and marine sand.

Impact of land use

The impact of land use on groundwater quality was explored by analyzing the variance of the entire dataset, of data grouped by groundwater age, and of the four FCM clusters described above.

In the ANOVA of the entire data set (Table 2), urban land use showed significantly higher concentrations of Ca, K, and HCO_3 and higher temperature (T) and pH values than that seen in natural areas. It also showed reduced aluminum levels. Agricultural land use resulted in significantly higher concentrations of Ca, K, HCO_3, NO_3, and Fe and a higher T value than that seen in natural areas. A comparison of urban and agricultural land use showed that concentrations of NO_3, Al, and Fe were significantly higher in the groundwater below agricultural areas, while T and pH were higher in urban areas.

Table 2. Impact of land use: ANOVA of the entire dataset and of groundwater age groups (in mg L^{-1} [Al and B in µg L^{-1}], pH in units, EC in mS m^{-1}, T in ^{o}C).

	Entire data set (n=615)			Young groundwater - TU>2 (n=330)			Old groundwater - TU<2 (n=109)		
Land use type n=	Natural 103	Urban 60	Agricultural 452	Natural 75	Urban 44	Agricultural 211	Natural 5	Urban 7	Agricultural 97
Parameter	Mean	Factor	Factor	Mean	Factor	Factor	Mean	Factor	Factor
Calcium	38.6	B 2.4	B 2.3	35.2	B 2.7	B 2.3	101	1.0	1.5
Magnesium	9.0	2.0	3.2	7.1	B 1.9	B 2.0	46.9	1.0	1.7
Potassium	4.0	B 4.6	B 4.3	3.4	B 5.1	B 4.7	12.2	1.7	1.9
Sodium	41.9	2.7	4.4	31.0	B 2.1	1.3	309	1.5	2.1
Chloride	67.1	2.4	4.8	46.6	2.0	1.3	553	1.3	2.1
Bicarbonate	101	B 3.2	B 2.5	87.5	C 3.4	B 2.2	425	1.3	1.2
Sulfate	64.5	1.1	1.1	48.6	1.1	B 1.7	8.2	5.7	9.0
Nitrate	3.1	1.3	B 3.0	2.9	1.6	B 4.6	0.6	4.8	1.7
Aluminum	1192	B 0.1	0.4	1357	B 0.1	0.5	12.7	3.1	6.7
Manganese	0.3	2.0	2.3	0.3	2.3	2.7	0.5	0.8	1.6
Iron	4.0	1.6	B 2.5	3.8	AB 1.3	B 2.1	10.2	1.0	1.3
Boron	66.3	2.7	3.4	76.3	1.7	0.9	44.9	9.3	10.6
Temperature	10.5	C 1.2	B 1.0	10.2	C 1.2	B 1.1	9.9	C 1.2	B 1.1
Electrical conductivity	47.9	2.4	3.0	41.9	B 2.1	B 1.8	181	1.6	2.1
pH	6.1	B 1.1	1.0	6.1	B 1.1	1.0	6.9	1.0	1.0

The mean values of the natural areas are considered 'A' (unmarked in the table). The mean values of the other areas are also A (and thus unmarked) if they do not differ from that of the natural area. The mean value of an area is marked B if it differs significantly from that of the natural area ($p<0.05$). If the mean values of two areas are marked B, they differ significantly from the natural area, but not from each other. If the mean values of the two areas also differ from each other, one is designated B and the other C. The mean value of an area is marked AB if it does not differ significantly from either A or B; i.e., it lies between the significantly different mean values of the other two areas ($p<0.5$).
Factor: Ratio of deviation of the urban and agricultural mean values compared to the mean values in natural areas; EC: electrical conductivity; T: temperature; TU: tritium units.

Similar ANOVA tests on separate age groups of groundwater (pre-1950 and post-1950) indicated that land use effects were strongest for young, recently infiltrated groundwater (Table 2). In this age group, urban land use resulted in significantly higher concentrations of Ca, Mg, K, Na, and HCO_3 and higher T, EC, and pH values compared to natural areas. Similarly, agricultural land use showed significantly higher concentrations of Ca, Mg, K, HCO_3, SO_4, NO_3, and Fe and increased T and EC values compared to natural areas. In addition, both land use types had significantly reduced aluminum levels compared to natural areas. When comparing urban and agricultural land use, SO_4, NO_3, Al, and Fe concentrations were higher in the groundwater under agricultural areas, while Na and HCO_3 concentrations and T and pH values were higher below urban areas. With respect to old groundwater (TU<2), only the temperature differed between the various land uses. It was lowest under natural areas and highest under urban areas. The absence of differences in chemical parameters suggests that this increased temperature was caused by conduction rather than geohydrological or geochemical processes.

Separate ANOVA tests on the four geochemical clusters (Table 3) showed that the effects of land use not only varied with groundwater age, but also depended on geochemical processes. The most pronounced effects of land use were observed in the oxic cluster, while the least effects were seen in the brackish cluster.

In the oxic cluster (Table 3), both urban and agricultural land use showed significantly higher concentrations of Ca, K, Na, and Cl and higher T and EC values compared to natural areas. They also had reduced aluminum levels. The concentration of B was also increased, but was not statistically significant. The temperature below urban areas was significantly higher than below agricultural areas. Compared to natural and urban areas, agricultural land use resulted in significantly higher Mg and NO_3 groundwater concentrations. The higher NO_3 concentration is likely related to the agricultural use of nutrients. The higher Mg concentration might be due to cation exchange reactions in which Mg is forced into the soil solution by elevated Ca and K; however, this would be expected for both urban and agricultural areas (Beekman, 1991). This higher concentration may also be caused by the Mg-calcite in the chalk used in agricultural liming. The Mg-calcite content of chalk ranges from 4% to 19% (Rinsema, 1985).

The significant effects of urban land use on groundwater properties in the anoxic cluster were limited to temperature, which was increased compared to natural areas. There were indications of other effects (indicated by 'AB' in Table 3), but they were not significant due to the large confidence intervals caused by the low number (3) of urban samples in the cluster. The groundwater under the agricultural areas showed significantly higher concentrations of K, Na, Cl, HCO_3, and Fe and higher T and EC values compared to natural areas. When comparing urban and agricultural land use, temperature was significantly higher under urban areas.

The carbonate-buffered cluster showed significantly increased concentrations of Na and HCO_3 and higher T, EC, and pH values in the groundwater under urban areas compared to natural areas (Table 3). Chloride was also higher, but not significantly. Groundwater under agricultural areas had significantly higher HCO_3 and Fe concentrations and a higher pH value than in natural areas. The sodium concentration and T and EC values were significantly higher in urban areas than in agricultural areas. The lower pH in urban and agricultural areas compared to natural areas was, however, surprising. One would expect a lower pH in natural areas or no difference at all since $CaCO_3$ buffering reduces potential differences in land use impact on pH. One possible explanation for the lower pH in agricultural areas is a higher CO_2 pressure due to the higher turnover rate of organic matter compared to natural areas. Alternatively, acid production associated with the nitrification of organic nitrogen from added nutrients (agricultural areas) or leaking sewage systems (urban areas) in the unsaturated zone may have led to a lower pH of the groundwater below these land use types.

Table 3. Impact of land use: ANOVA of the four FCM clusters (in mg L^{-1} [Al and B in µg L^{-1}], pH in units, EC in mS m^{-1}, T in ^{o}C).

Cluster	Oxic (n=173)			Anoxic (n=131)			CaCO3-buffered (n=229)			Brackish (n=80)		
Land use type	Natural	Urban	Agricultural	Natural	Urban	Agricultural	Natural	Urban	Agricultural	Natural	Urban	Agricultural
n=	31	16	126	42	3	86	27	28	174	3	13	64
Parameter	Mean	Factor	Factor	Mean	Factor	Factor	Mean	Factor	Factor	Mean	Factor	Factor
Calcium	27.0	B 2.5	B 2.4	12.3	1.4	0.9	83.2	1.2	1.2	126	1.0	1.8
Magnesium	7.4	1.3	B 1.9	3.5	0.9	1.3	9.3	1.2	1.2	99.6	0.5	1.4
Potassium	4.4	B 4.1	B 5.8	1.9	AB 2.3	B 2.2	2.8	3.6	2.3	41.1	1.0	1.1
Sodium	17.2	B 2.3	B 1.4	9.4	AB 5.5	B 3.4	37.6	B 1.5	0.8	792	0.4	1.4
Chloride	24.9	B 2.2	B 1.6	13.8	AB 5.5	B 3.3	67.3	1.1	0.7	1247	0.4	1.6
Bicarbonate	37.8	3.7	1.6	21.2	AB 1.4	B 2.3	206	B 1.5	B 1.4	868	0.7	1.0
Sulfate	70.0	0.9	1.2	38.1	0.9	1.1	57.4	1.3	1.1	6.6	7.2	20.5
Nitrate	7.9	1.5	B 4.0	1.1	0.3	0.4	1.0	1.7	0.7	0.3	1.0	0.3
Aluminum	2995	B 0.1	B 0.5	685	0.8	0.6	39.4	0.3	0.8	16.0	2.9	3.5
Manganese	0.5	0.6	1.0	0.3	0.3	0.7	0.4	1.5	2.3	0.3	3.3	4.3
Iron	0.2	1.0	2.5	4.9	AB 1.0	B 2.6	6.8	AB 1.3	B 1.9	5.7	1.6	2.8
Boron	28.8	4.1	1.7	8.1	2.7	2.8	36.8	2.8	1.6	1353	0.3	0.6
Temperature	10.5	C 1.2	B 1.1	10.4	C 1.2	B 1.1	10.6	B 1.1	1.0	AB 10.8	B 1.1	A 1.0
Electrical conductivity	35.9	B 1.7	B 1.8	18.2	AB 2.2	B 1.7	66.7	B 1.3	1.0	419	0.6	1.5
pH	5.3	1.1	1.0	5.9	0.9	1.0	7.3	B 0.9	B 0.9	7.4	B 0.9	B 0.9

Unless otherwise indicated, the mean values of the natural areas are considered 'A' (unmarked in the table). The mean values of the other areas are also A (and thus unmarked) if they do not differ from that of the natural area. The mean value of an area is marked B if it differs significantly from that of the natural area ($p<0.05$). If the mean values of two areas are marked B, they differ significantly from the natural area, but not from each other. If the mean values of the two areas also differ from each other, one is designated B and the other C. The mean value of an area is marked AB if it does not differ significantly from either A or B; i.e., it lies between the significantly different mean values of the other two areas ($p<0.5$).
Factor: Ratio of deviation of the urban and agricultural mean values compared to the mean values in natural areas; EC: electrical conductivity; T: temperature; TU: tritium units.

Few land use-related differences were expected in the brackish cluster because most of the water in this cluster was old and therefore not affected by current land use. Moreover, a substantial number of the wells in the brackish cluster had an upward groundwater flow. Nevertheless, land use effects were observed for temperature and pH (Table 3). The temperature under urban areas was significantly higher than under agricultural areas. This was most likely due to conduction. No temperature differences were found between these land uses and natural areas, probably because of the small sample size of the latter (denoted by 'AB' in Table 3). Groundwater pH was significantly lower in urban and agricultural areas than in natural areas. This trend is similar to the one observed in the carbonate-buffered cluster. Again, the result is surprising since groundwater age in the brackish cluster was the highest and one would have expected that geochemical processes would have masked the land use effect even more.

In summary, the results of the univariate analysis showed that, except for aluminum, various major ions in groundwater below agricultural and urban areas were increased compared to natural areas. The strongest impacts of land use were observed in the data subsets that were least influenced by geochemical processes, i.e., the oxic cluster and the post-1950 age group (TU>2) (Tables 2 and 3). These impacts are illustrated by the ratios (factors) by which groundwater properties in urban and agricultural areas differed from natural areas. In the oxic cluster, for example, groundwater under urban areas showed an increase in Mg, SO_4, NO_3, Mn, Fe, T, EC, and pH by a factor 1-2, an increase in Ca, Na, Cl, and HCO_3 by a factor 2-4, an increase in K and B by a factor 4-10, and a decrease in Al by a factor 10 compared to natural areas. Groundwater under agricultural areas in the oxic cluster showed an increase in Mg, Na, Cl, HCO_3, SO_4, Mn, B, T, EC, and pH by a factor 1-2, an increase in Ca, NO_3, and Fe by a factor 2-4, an increase in K by a factor 4-10, and a decrease in Al by a factor 2 compared to natural areas. In recently infiltrated groundwater (post-1950, TU>2), the effects of urban and agricultural land use were very similar to the trends seen in the oxic cluster.

Discussion

Processes controlling groundwater composition

The results of the non-a priori clustering procedure showed that subsurface geochemical processes and groundwater origin (rainwater versus seawater), not land use, are the two major factors that characterize the shallow groundwater composition on a national scale (Table 1). It is certainly possible that land use would have surfaced as an additional source of variation in the clustering procedure if we had allowed more than four clusters (Frapporti et al., 1993). More clusters, however, would have increased the problem of low numbers of replicate observations within the separate clusters and thus decreased the statistical power of the analysis (e.g., anoxic-urban and brackish-nature, Table 3).

When analyzing the four geochemical clusters separately, land use emerged as a factor that influenced groundwater composition at both regional and local scales (Figure 1, Table 3). The impact of land use was strongest in the oxic cluster, suggesting that anthropogenic contamination is best recognized in recently infiltrated groundwater. This was confirmed by the separate analysis of the a priori groundwater age groups: it showed that the impact of land use was strongest in post-1950 groundwater (TU>2) (Table 2). The progressive geochemical processes in older groundwater may increasingly mask or erase potential land use effects on groundwater composition.

As previously mentioned, the land use effects were strongest in recently infiltrated groundwater (TU>2) and the oxic cluster. These groundwater types are typical of the sandy Pleistocene infiltration areas in the eastern, southern, and central parts of the Netherlands (Frapporti et al., 1993). The precipitation excess, i.e., precipitation minus evapotranspiration, which recharges the groundwater, infiltrates to deeper aquifers within these areas rather than being quickly drained by the surface-water system. Anoxic and carbonate-buffered groundwater can be found in infiltration areas containing more reactive sediments or in parts of groundwater systems with upward seepage or horizontal flow. As a result, the relationship between current land use and groundwater composition was less clear in these clusters than in the oxic cluster and TU>2 group. This is illustrated by the lower number of significantly different parameters (Table 3). A weak relationship between current land use and groundwater composition was expected for the brackish groundwater type as most of the groundwater in this cluster infiltrated before anthropogenic influence became apparent and a substantial part of the excess precipitation today is discharged shallowly by drains and removed from the area via the surface-water system of ditches and channels.

A comparison of the results of the a priori-assigned age groups and the non-a priori-established geochemical clusters suggested that the clustering procedure resulted in a statistically more sensitive classification. The oxic cluster showed very similar effects with similar significance levels with regard to land use compared to the post-1950 age group, although the former had only half the number of observations (173) as the latter (330). Besides these statistical reasons, the non-a priori method is preferred over the a priori grouping because it is unbiased and lets the data 'speak for themselves'. In our case, the non-a priori method helped pinpoint the dominant influence of geochemical processes on groundwater quality on a national scale. The timely detection of trends in groundwater-monitoring networks depends on sensitive and unbiased data-analysis methods.

Impact of land use functions

The present results showed that agricultural and urban activities add solutes to the groundwater (illustrated by the increased EC). Although the effects of urban land use were not always significantly different from those of agricultural land use, a pattern was found. Concentrations of solutes that can be related to sewers and road salting were elevated under urban areas. These substances include potassium (sewage constituent), bicarbonate (oxidation product of organic matter from sewage), boron (detergent), and sodium and chloride (both in road salt) (see, e.g., Barrett et al., 1999). The lower aluminum concentration in groundwater under urban areas compared to natural areas could be due to the fact that a substantial part of (acid) rainwater in urban areas is discharged by the sewage system, is pH-neutralized, and does not infiltrate locally. Concentrations of solutes that were higher under agricultural areas are related to the use of manure and fertilizer salts: potassium (constituent of manure and fertilizer salts), nitrate (oxidation product of mineral nitrogen), calcium and magnesium (both due to the production of acid by nitrification and liming of sandy acid agricultural soils), and iron (subsurface product of pyrite oxidation by leached nitrate).

In addition to the effects on solute concentrations, agricultural and urban land use also caused a significant rise in groundwater temperature, with the highest temperatures being measured under urban areas. Temperature was the only groundwater property that showed significant land use effects across all geochemical clusters (Table 3) and groundwater age groups (Table 2). In the pre-1950 age group (TU<2) and the brackish cluster, the increase in temperature was most likely due to heat conduction rather than the advective transport of groundwater as there were no underlying differences in groundwater chemical composition.

While the impact of urban and agricultural land use on groundwater composition was most significant in the oxic cluster, it was also clearly present in the anoxic and carbonate-buffered clusters. The number of significant differences between agricultural and natural areas in the oxic, anoxic, and carbonate-buffered clusters was 9, 7, and 6, respectively (Table 3). This corresponds well with the proportion of young groundwater (TU>2) in these clusters (97%, 79%, and 73%, respectively). There are two possible explanations for the weaker effects of land use in the anoxic and carbonate water types: (1) the anthropogenic impact before 1950 was lower and these clusters contained more observations of pre-1950 groundwater (TU<2); and (2) subsurface geochemical processes, which transform oxic water type into anoxic and subsequently carbonate-buffered water types, may have masked or erased land use effects.

Conclusions

Quantitative insight into the impact of land use on groundwater composition is vital for a sustainable management of aquifers. Our results demonstrate that the importance of processes controlling groundwater composition is scale-dependent, with geochemical processes being more dominant on a national scale and land use on regional and local scales. Such knowledge is valuable for spatial planning and evaluation purposes: e.g., to allocate land use functions and define requirements for reducing the impact of land use at specific aquifers. It can also be useful to predict the effect of changes in land use or land use intensity, e.g., reducing agricultural N applications.

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Chapter 3

Using a groundwater quality negotiation support system to change land use management near a drinking-water abstraction in the Netherlands

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Impression of the study area Holten, January 2001 (Photo: Cors van den Brink).

Using a groundwater quality negotiation support system to change land use management near a drinking-water abstraction in the Netherlands

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Abstract

A negotiation support system (NSS) was developed to solve groundwater conflicts that arose during land use management. It was set up in cooperation with the stakeholders involved to provide information on the impact of land use, e.g., agriculture, nature (forested areas), recreation, and urban areas, on the quality of both infiltrating and abstracted groundwater. This NSS combined simulation programs that calculate (1) the concentrations of nitrate in shallow groundwater for each land use area and (2) the transport of nitrate in the groundwater-saturated zone. The user interface of the NSS enabled scenario analyses. The NSS was validated at a drinking-water abstraction near Holten (the Netherlands) using a spatial planning process aimed at sustainable land use and groundwater-resource management. Two land use scenarios were considered: a base scenario reflecting the autonomous development and an adapted land use scenario. The calculated results for shallow groundwater provided an explicit spatial overview of the impact of historical land use and N application on the quality of abstracted groundwater as well as insight into the impact of changes in land use and N application. Visualization of the conflicting interests of agriculture and the drinking-water abstraction helped all stakeholders accept the necessary changes in land use identified by the adapted land use scenario of the NSS. These changes were included in the preferred land use management option in the regional planning process, which has since been formalized. The NSS provided system insight, scoping analyses, and education, in addition to generating quantitative information on the impact of land use functions on groundwater quality.

Introduction

Different types of land use, e.g., water abstraction, agriculture, recreation, and urban activities, can have strong, often negative, impacts on the quality and quantity of groundwater and as a result influence each other. For example, nitrate leaching from farmlands can cause nitrate levels to reach harmful concentrations (above the drinking-water limit of 50 mg NO_3 L^{-1}) at nearby drinking-water abstractions (see, e.g., Van Drecht, 1993; Hansen et al., 1991; Steenvoorden et al., 1997). Groundwater abstraction may cause lower crop yields by increasing the depth of the groundwater table, which leads to soil-moisture deficiency. A third example is when groundwater-dependent natural ecosystems become too enriched with nutrients from nearby agricultural fields (e.g., Runhaar et al., 1996).

The impacts of one type of land use on another via groundwater are often characterized by long-term delays and relatively long distances between the cause and the effect. Groundwater heads, for example, may be influenced by abstraction or polders at distances of tens of kilometers and time frames of days to months (e.g., Gehrels et al., 1994). Groundwater quality may be influenced by the quality of recharge in the groundwater system and by groundwater flow characteristics such as the residence times between infiltration and exfiltration points. Even on a small local scale, groundwater residence times may exceed tens of years (e.g., Zaadnoordijk et al., 2004). In addition, groundwater quality may be changed by geohydrochemical reactions in the subsoil, such as denitrification (Tesoriero et al., 2000). These long-distance and long-term effects can hamper the analysis of the impact of current land use on groundwater quantity and quality.

Several studies have been conducted on the relationship between land use and groundwater quality (e.g., Secunda et al., 1998, Thirumalaivasan et al., 2003 and Lake et al., 2003). Their emphasis, however, has shifted from

describing the vulnerability of the groundwater system to describing the impact of land use since understanding the impact of land use can be used to generate guidelines for sustainable groundwater management (Collin and Melloul, 2001; Kersenbaum et al., 2003). Quality control of groundwater is important because it is a major source of drinking water throughout the world (see, e.g., Foster et al., 1998). Moreover, effective groundwater resource management requires that the socioeconomic context is taken into account (Melloul and Collin, 2001). Burke et al. (1999) argued more generally that social, institutional, and political factors are the primary obstacles to the sustainable management of the world's groundwater resources. Groundwater resource management, therefore, should become part of a common socioeconomic interest and integrated into local spatial planning.

Land use in the Netherlands is intense and, due to the high population density, different types of land use occur within relatively short distances of each other. As a result, groundwater conflicts between the various land uses frequently occur. During the last decades, Dutch environmental policy-making has increasingly moved from the national to the regional and local authorities. At the same time, environmental policy has become embedded in the social and economic processes of parties other than governmental organizations, such as businesses, nongovernmental organizations, and citizens (Driessen and Glasbergen, 2002). As a consequence, many spatial planning processes in the Netherlands are now organized as bottom-up processes in which the stakeholders negotiate with each other in a network structure instead of top-down processes with the government as the dominant party (e.g., De Roo, 1999; Driessen and Glasbergen, 2000; Niekerk, 2000). In such bottom-up processes, it is important that the stakeholders are provided with the knowledge and information they need to actively take part in the process.

One tool that provides such information is a negotiation support system (NSS). Computer-aided NSSs were first used in negotiation processes during the second half of the 1990s (e.g., Hill and Jones, 1996). Thiessen et al. (1998), for example, described the use of an NSS to solve water-resource conflicts. They described the algorithms used for analyzing preferences and for generating alternative feasible agreements. The spectrum of NSSs include (Starke and Rangaswamy, 1999) (1) expert systems that use accumulated knowledge to aid stakeholders preparing for negotiations, (2) a system that combines technologies for individual decision support (DSS) and group decision support (GDSS) and facilitates the actual negotiation process in multiagent settings, and (3) "autonomous agents" that are programmed to negotiate on behalf of their human principals. The NSS presented in this paper provides information about the impact of multiple land use (e.g., agriculture, nature, and recreational and urban activities) on the quality of both infiltrating and abstracted groundwater. Moreover, it aids negotiators in overcoming their cognitive limitations and in identifying their (and others') real interests rather than focusing on negotiating positions (see also Fisher and Ury, 1983).

The purpose of the present study was to develop an NSS to solve groundwater conflicts in land use management. The NSS was designed not to provide a scientific evaluation of the groundwater nitrate issue but to identify the best course of action to resolve the stakeholder-accepted conflict between land use management and groundwater quality. It calculated the impact of land use on groundwater quality in terms of nitrate concentrations because nitrate levels illustrate the conflict between agricultural land use and drinking-water abstraction. Thus, our NSS can be considered to be a groundwater application of a planning support system (PSS), as defined by Geertman and Stillwell (2003). A PSS generates information that can be communicated to and among the stakeholders and used in the planning process. This is different from a decision support system (DSS), which primarily helps decision-makers find the best solution (c.f. Loucks, 1995). The set-up of all three of these support systems requires considerable interaction between the developers and the users.
The applicability of our NSS in changing land use management was validated in a pilot study of a spatial planning process near the village of Holten in the eastern part of the Netherlands. Conflicts in this region exist between the land uses drinking-water abstraction, agriculture, a natural ecosystem preserve, and recreation. The stakeholders participating in the planning process were the water company, the provincial government, farmers, recreational businessmen, and the municipality. All of these participants could directly use the information from the NSS in the bottom-up spatial planning process as the technical basis for management decision-making (c.f. Simonovic, 2002).

The NSS setup, the scenario definition, and the analysis of the results were carried out in close cooperation with the stakeholders. NSS results were used in all phases of the process to enable validation in terms of (1) the ability of the NSS to simulate nitrogen dynamics in groundwater and (2) the impact on the outcome of the spatial planning process. For the first validation, the modeled nitrate concentrations were tested using observed concentrations in abstracted groundwater. For the second validation, we investigated whether or not the stakeholders supported a change in land use based upon the modeling results.

Materials and methods

NSS Setup

Requirements and structure of the NSS

The Dutch authorities involved in conflicts regarding drinking-water quality and land use, e.g., water boards, provinces, and water companies, participated in workshops on the use for and setup of the NSS. At this stage, the participants emphasized the need for visualizations like spatial maps and time graphs so that they could compare the different land use scenarios. The main result was a spatially explicit map of the quality of the groundwater at the groundwater table (henceforth referred to as shallow groundwater), because it shows the impact of land use and is linked to the quality of the abstracted groundwater.

The land use types accounted for in our NSS were agriculture, nature (mostly forest), urbanized areas, and permanent recreation. The drinking water abstraction is not considered a land use function for it uses the groundwater rather than the land surface. Three types of agriculture were distinguished because the use of nutrients varies between them: grassland, maize, and other arable crops.

The NSS used in this study was a newly developed tool made up of three components: the Modular Dynamic Spatial Analysis Tool (MD-SAT); the Integrated Transport Model (ITM; Van der Grift et al., 2002); and TriShell (http://www.triwaco.com). It performed the calculations in two steps (Van den Brink and Zaadnoordijk, 1995). The first step calculated the quality of shallow groundwater as a function of land use and other important influences like atmospheric deposition and soil characteristics. The second step calculated the quality of the abstracted groundwater as a function of the current and historical quality of shallow groundwater, groundwater residence time, and the reactivity of the subsoil. The first step was calculated using MD-SAT and the second using ITM. These NSS modules are discussed in the next section.

Components of the NSS

<u>MD-SAT: concentrations in shallow groundwater for each land use function</u>

MD-SAT calculated the quality of shallow groundwater based on the current and historical load, land use type, and soil characteristics. MD-SAT combined various existing simulation programs, each specific for the calculation of the quality of the shallow groundwater for a particular aqueous component and a particular land use type. These programs were all one-dimensional, vertical transient descriptions. MD-SAT can run the programs in parallel for many locations, incorporating the specific land use and conditions of each location. Within MD-SAT, leaching from agricultural lands was calculated using the SPREAD program (Beekman, 1998) on the basis of the soil type, depth of groundwater table, crop (grassland, maize, other arable land), atmospheric N deposition, and manure and fertilizer application. SPREAD considered the following processes: (1) background N leaching in relation to soil type and crop type; (2) plant uptake as a function of the crop, soil type, and N application; (3) leaching as a result of the available mineral nitrogen and the nitrogen uptake by crops; (4) leaching as a result of the application of fertilizer and manure outside the growing season; and (5) denitrification in the zone between the root zone and the shallow groundwater as a function of the groundwater table depth (Steenvoorden et al., 1997). Technical details of SPREAD are listed in Appendix 1.

Leaching from forests was calculated according to a combined empirical and mass-balance approach using SPREAD and the model based on Boxman (2002). This model assumes that background N leaching is zero as long as the soil matrix is not yet saturated with N (before the 1980s) and is described as a mass balance output balance thereafter (Gundersen et al., 1998; Boxman, 2002). Denitrification in the zone between the root zone and

the shallow groundwater as a function of the depth of the groundwater table was also calculated using SPREAD. Due to the roughness of the forest surface, the atmospheric deposition in the forest was increased by a factor of 1.25 when compared to the deposition in the open field (Tietema, 2000). Leaching from natural grasslands was modeled like agricultural grasslands, but did not include manure or fertilizer. Leaching from urban and recreational areas was assumed equal to the load, so the quality of the shallow groundwater was calculated using an input-output model in which accumulation in the soil was neglected. The inputs were (Grimmelikhuyse et al., 1998): the annual atmospheric N deposition and constant values for the application of fertilizers in parks, leakage from the sewage systems, and infiltrating sewage. The recreational activities in the study area consisted of the semipermanent use of summer homes without sewers.

ITM: transport of compounds in the subsoil

ITM simulates three-dimensional reactive groundwater transport and is essentially a combination of the simulation programs MT3DMS (Zheng and Wang, 1999) and PHREEQC (Parkhurst and Appelo, 1999) in a GIS shell. In this study, ITM calculated the quality of the abstracted water. One of the input variables was the temporally and spatially variable concentration of the groundwater recharge. It was taken from the quality of shallow groundwater input, as provided by MD-SAT.

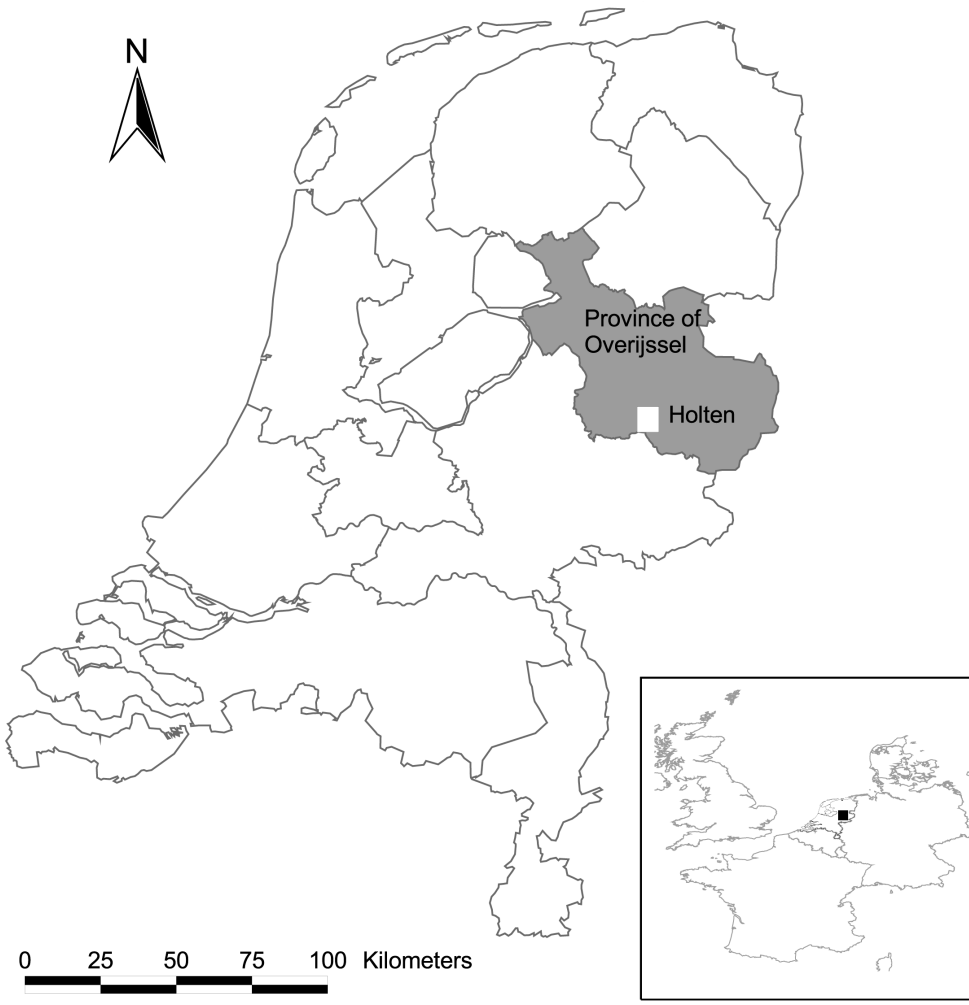

Figure 1. Study area of the Holten case in the province of Overijssel, the Netherlands.

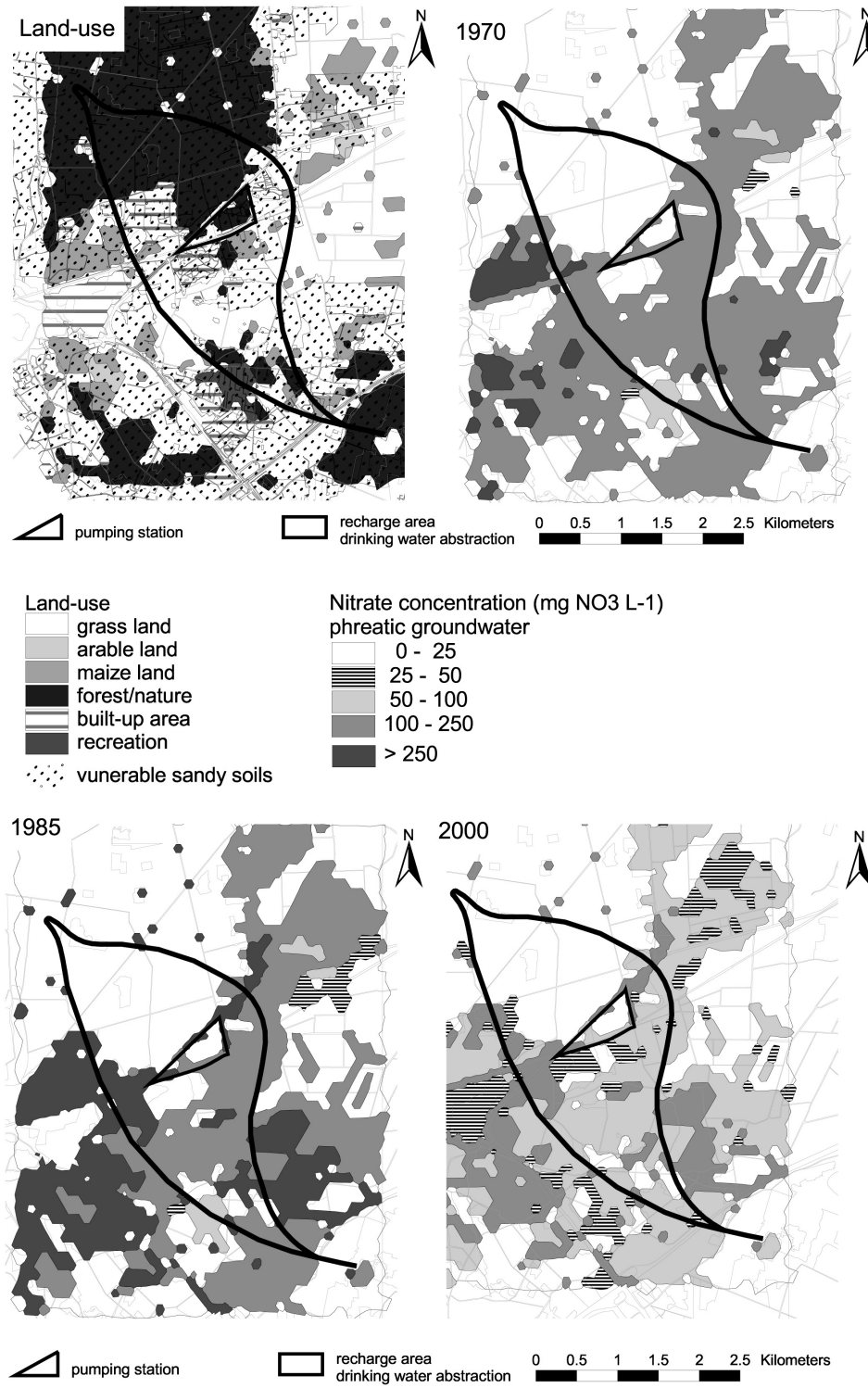

Figure 2. Land use and calculated nitrate concentrations (mg NO_3 L^{-1}) in shallow groundwater in 1970, 1985, and 2000.

User interface: TriShell

TriShell organizes modeling studies in so-called 'projects', which can contain various models and aspects. In the present study, TriShell served as the user interface for the NSS and carried out the necessary data manipulation, scenario management, and visualization of the output. Data manipulation consisted of the transfer of data between the models and the conversion of input data to model input using the Time Series Allocation version of the spatial allocation available in the TriShell. Scenario management by TriShell allows the user to implement scenarios, shows the modified input, and allows a comparison of the results of various scenarios. The visualization includes maps and time lines with export options to, e.g., ArcView.

Application and validation of the NSS

The Holten area

Geography and hydrogeology

The village of Holten is located in the eastern part of the Netherlands, where thick Pleistocene phreatic aquifers are found (Figure 1). The northern and southernmost parts of the area consist of ice-pushed ridges covered with deciduous forest and pines. Nature and recreational functions dominate in this area. The central-southern valley is mainly used for intensive agriculture. The altitude of the ice-pushed ridges is several tens of meters above the valley floor. The subsoil contains one aquifer consisting of three distinctly different geological layers upon a geohydrological basis at approximately 90 meters below the soil surface. Fine nonreactive, fluvioglacial, Late Pleistocene deposits with a thickness of approximately 15–20 meters form the upper aquifer layer in the central-southern area. The second layer (25–30 meters) consists of coarse nonreactive fluvial Middle Pleistocene deposits, which are also found continuously at the surface of the ice-pushed ridges. The third layer is present throughout the region and has a thickness of approximately 40 meters. It is built up of reactive fine marine Early Pleistocene sands. Groundwater abstraction at the base of the ice-pushed ridges started in 1960 with 10 wells and most abstraction screens occur in the second, most permeable layer. Five new wells were installed in the third layer in 1985 to avoid well clogging. Clogging can occur when shallow oxic and deep Fe-anoxic groundwater mix. The permitted abstraction volume is 2.5 10^6 m^3 yr^{-1}, resulting in a recharge area of 5.5 km^2 (Figure 2). The actual abstracted volume is approximately 2.2 10^6 m^3 yr^{-1}. The groundwater table is mostly several meters below the valley surface and several tens of meters below ridge surface. The study area was restricted to the recharge area of the groundwater abstraction. According to Dutch law, a groundwater protection zone of 4.8 km^2 is defined around an abstraction location.

Groundwater quality

The conflict between agricultural land use and drinking-water abstraction can be studied using three categories of water-quality data: data from abstraction wells; soil moisture below agricultural land; and observation wells throughout the entire area. The average nitrate concentration in the abstracted water has steadily increased over the years, from about 5 mg NO_3 L^{-1} in 1960 to approximately 25 mg NO_3 L^{-1} in 1998 (Figure 3). Other parameters have increased as well: chloride and sulfate, for example, by factors of 3.0 and 2.9, respectively (Water company Vitens Overijssel, internal documents). In 1993, the average nitrate concentrations in the soil moisture 1.5 m below the surface of fields of grassland and maize were 228 and 325 mg NO_3 L^{-1}, respectively (Van Beek et al., 1994). Grassland and maize are the main types of agricultural land use in the region. These concentrations were lower than the average 1990 measurements of 252 and 700 mg NO_3 L^{-1}, respectively. The nitrate concentration of groundwater in the observation wells varied from 140 to 170 mg NO_3 L^{-1} below agricultural land and from 8 to 70 mg NO_3 L^{-1} below forested (natural) areas.

Planning process and stakeholders

The aim of the spatial planning process was to change current agricultural practice in such a way that it fit within the environmental legislation. To do this, the water company (the main stakeholder) and the provincial authorities (the legislative body responsible for groundwater quality) initiated a spatial planning process to reduce the impact of agricultural land use on the quality of the abstracted drinking water. Other stakeholders in the spatial planning process were representatives of agricultural, recreational, and urban land use representing particular local socioeconomic interests. Less than five farms were entirely dependent economically on

agricultural activities within the capture zone of the groundwater abstraction site. The other farmers had other major sources of income. Threats for the agricultural stakeholders included an extension of the groundwater protection zone and a strict enforcement of the EU Nitrate Directive (EC, 1991). The EU Nitrate Directive is European Union legislation that restricts the application of manure to agricultural land in order to limit the leaching of nitrate into the groundwater. Both measures would strongly limit the current agricultural practice. The farmers were informed of the benefits of the planning process and offered support in developing and implementing groundwater-friendly agricultural practices. The stakeholders related to recreational land use were in favor of extensive agriculture since it would make the area more attractive to national tourists. The main interest of the local authorities was related to the enhancement of the quality of the landscape with tourism being the main source of regional income. Nature in the study area was not sensitive to the impacts of the other land uses on groundwater quality since the forested area is located in the high, regional infiltration area. Moreover, no changes were expected regarding the current nature areas.

In addition to this local planning process focusing on groundwater protection, the province is formulating a regional plan in response to Dutch legislation on the reconstruction of areas engaged in intensive livestock farming (Reconstruction Law). The purpose of this legislation is to reach a new balance between various functions in rural areas. The legislation stipulates the differences between areas where agriculture can develop, areas where various functions are intertwined, and areas where agricultural activities are more extensive (Driessen and De Gier, 2004).

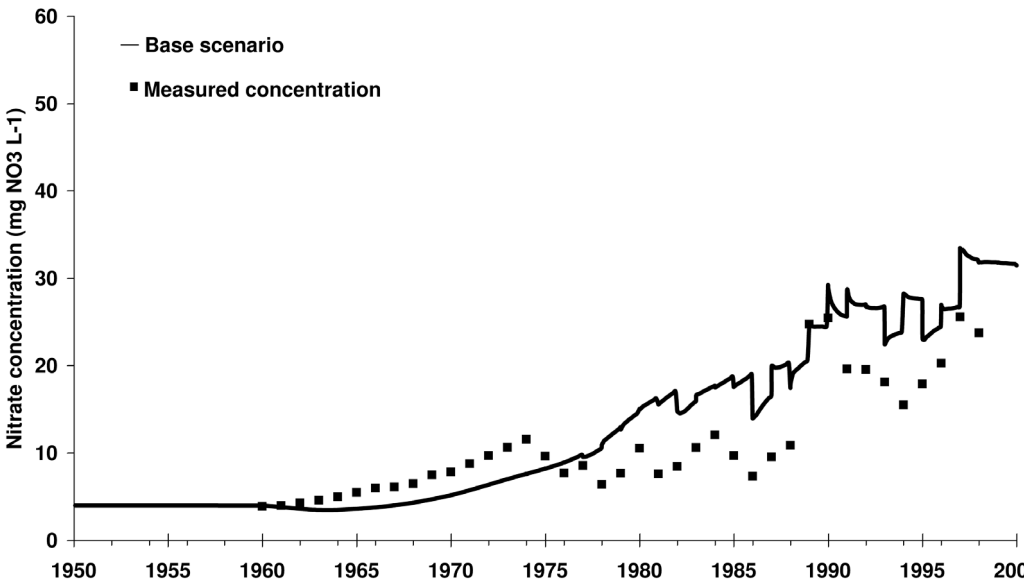

Figure 3. Comparison between calculated and measured nitrate concentrations (mg NO_3 L^{-1}) during the period 1960-1998.

Procedures (input – output – validation)

The NSS requires spatial and temporal input data regarding land use, soil and groundwater characteristics, and N application. The geohydrological input data were taken from an existing finite-element geohydrological model (Iwaco, 1993) and transformed into a finite-difference MODFLOW model (Harbaugh and McDonald, 1996) from which ITM imported the groundwater velocities. This flow model was set up to calculate the recharge area and the groundwater residence times from the groundwater table to the abstraction wells as a function of the recharge of the individual land uses, the elevation of the soil surface, geological information such as the thickness of the aquifer units and the transmissivity of each unit, yearly averaged data taken from a nearby meteorological station, and yearly averaged abstraction volumes of individual abstraction wells. The permeability of the sediments was calibrated using available head measurements. An overview of the input data is given in Table 1.

Input data – unsaturated zone

Data on current land use were obtained from a digital topographic map (1:25,000) based on 2000 LANDSAT satellite images (LGN4, 2003). Using historical topographic maps from 1882, 1922, 1954, 1976, and 1998, it appeared that the use of land today is the same as that in 1950. Soil type and the depth of the groundwater table were obtained from digital maps (1:50,000; Stiboka, 1983). A digital map (1:50,000) with 'vulnerable sandy soils' was also used. This map indicates areas in which an additional reduction in manure application has been implemented since 2001 (Figure 2). The historical and current load of nitrogen in the study area was calculated using information on manure application taken from national institutes, regional practice, and general literature (Table 1).

Table 1. Overview of the source and quality of the input data for the NSS.

Data need	Input data	Quality	Reference
Area for planning process			
Recharge area	Groundwater flow model	+	Iwaco (1993)
Shallow groundwater			
Agriculture	Soil use	++	LANDSAT LGN4 (2003)
	Soil type	++	Stiboka (1983)
	Groundwater table	+	Stiboka (1983)
	Vulnerable sandy soils	++	Province of Overijssel
	Atmospheric deposition	+	National Institute for Public Health and the Environment. Data from the National Rain Water Quality Network over the period 1978 – 1999; Bleeker and Erisman, 1996)
	Manure production	++	CBS, agricultural statistics
	Manure application	+	National policy & regional practice
	Manure composition	++	Rinsema (1985)
	Fertilizer	+	Regional data
	N uptake	+	Beekman (1998)
	Future N load	+	National & European policy + regional practice
Forested area (nature)	Atmospheric deposition	+	National Rain Water Quality Network
	Age & type of forest	+	Topographic data (1882, 1922, 1954, 1976, 1998)
	N uptake	+	De Vries (1994)
	Saturation of soil matrix	o	Boxman (2002)
	Roughness factor	o	Tietema (2000)
Urban & recreational areas	Atmospheric deposition	+	National Institute for Public Health and the Environment. Data from the National Rain Water Quality Network over the period 1978 – 1999
	Function-related N load	o	Grimmelikhuyse et al., 1998
	Road salting	o	Grimmelikhuyse et al., 1998
Abstracted groundwater			
Subsoil	Denitrification coefficient	++	Field work
	Abstraction rate	++	Water company

++: reliable data with adequate spatial and/or temporal resolution
+: reliable data with a lesser spatial and/or temporal resolution adapted to a specific situation
o: data extrapolated/estimated from other study areas
-: poor-quality data

The parameter values reflecting nutrient application for the three agricultural land uses (grassland, maize, and arable land) were assessed at a regional level in close co-operation with the stakeholders and agricultural experts. Manure application before 1990 was set equal to manure production. From 1990 on, N application was set equal to agricultural need reflecting the new national and regional regulations. Historical, current, and future fertilizer applications were estimated from regional information provided by regional agricultural organizations and the drinking-water company. Atmospheric deposition was assumed to decrease from 30 kg N ha^{-1} yr^{-1} in 2000 to 25 kg N ha^{-1} yr^{-1} after 2005 (Bleeker and Erisman, 1996). N input for the various land use types is shown in Table 2.

Table 2. Calculated N input* (kg N ha^{-1} yr^{-1}) and N leaching^ (mg NO_3 L^{-1}) for the various land uses.

Year	1950		1970		1985		1990		2000		2005#	
	Input	Leaching	Input	Leaching	Input	Leaching	Input	Leaching	Input	Leaching	Input	Leaching
Grassland	172	20	562	201	751	322	571	215	450	125	405	114#
Arable land	235	95	390	157	377	152	386	146	332	125	285	135#
Maize land°			513	291	549	323	476	260	202	72	155	78#
Nature	5	2	13	4	25	7	26	8	46	13	31	10
Urban	40	16	45	18	53	21	53	21	66	26	36	23
Recreation	200	50	205	52	213	55	213	55	226	61	196	55

* N input for agricultural land-use types equals the manure and fertilizer application and atmospheric deposition; N input for nature equals the atmospheric deposition times the roughness factor (1.25); N input for urban activities equals the atmospheric deposition and leakage from sewers; N input for summer homes without sewers (estimated to be 120 kg N ha^{-1} yr^{-1} in the summer and 40 kg N ha^{-1} yr^{-1} in the winter) is added to the urban impact.

^ N leaching is the N input minus the N uptake by crops and the denitrification in the root zone divided by the groundwater recharge of the specific land use. The denitrification in dry sandy soils - with mean groundwater tables >140 cm even in spring – is neglected (see Table 4 Appendix 1).

\# The 'vulnerable sandy soils' have been allocated by the provincial authorities since 2001. With this legislation, an additional reduction in the application of manure was implemented in 'vulnerable sandy soils', resulting in nitrate concentrations below those of grassland, arable land, and maize land (92, 114, and 64 mg NO_3 L^{-1}, respectively) (Figure 2).

° Maize is not distinguished as a separate crop type before 1970, because it was not planted as such.

Input data – saturated zone

Two new borings to approximately 50 meters below the surface were available for the geochemical characterization of the subsurface together with borehole descriptions of about 30 existing ones. The redox reactivity was determined experimentally for 18 sediment samples of the two new borings using continuous incubation experiments (c.f. Hartog et al., 2002). This characterization revealed no redox reactivity (i.e., no denitrification) in the fluvioglacial and fluvial deposits of the first two (sub)oxic layers of the aquifer. The oxidation of pyrite was observed in the marine deposits of the anoxic third layer. The denitrification rate constant was, therefore, set to zero for the first two (sub)oxic layers of the aquifer, assuming that the denitrification rate in nitrate-containing layers of the aquifer was negligible. The denitrification rate constant in the third anoxic layer was deduced from the geochemical characterization of the subsurface. The value was set to 1.0 day^{-1}, reflecting the anoxic conditions.

Until 1985, water was abstracted from wells in the first and second aquifer layers. To prevent well clogging caused by the mixing of shallow oxic and deeper Fe-anoxic groundwater, five deep wells were installed in 1985. The amount of groundwater abstracted per well was derived from the operational hours averaged per year. Groundwater abstraction for the period since 1998 was set equal to that in the year 1998 for the individual abstraction wells.

Validation

The applicability of the NSS in changing land use management to solve groundwater conflicts was validated in the spatial planning process. Two approaches were used for the validation, namely a technical and a social approach.

The technical approach tested the ability of the NSS to simulate nitrogen dynamics in groundwater. This validation was carried out using groundwater quality data on the abstracted groundwater and data on the pumping hours. Data on the quality of groundwater abstracted during the period 1960-1998 were available to validate the calculation results of the NSS. Pumping hours of the individual wells were available from 1960 to 1998. The planning process took place from 2000 until 2004. Analyses of abstracted groundwater were available per individual well from 1960 to 1998, with the temporal frequency increasing from twice a year in 1960 to four times a year in 1976. The quality of the abstracted groundwater was tested by calculating the individual mass flux per well (average yearly concentration times yearly pumping hours times specific hourly pumping capacity) and dividing it by the total yearly water flux of all operational wells. Incidental analyses of 24 monitoring wells near and at the abstraction location were also available. The monitoring screens of these wells were at depths

Table 3. N balance as a function of land use for the base and adapted land-use scenarios.

	Base scenario			Adapted land-use scenario		
Land use	Input 2005 (kg N ha^{-1} yr^{-1})	Area (ha)	N-input base (kg N ha^{-1} yr^{-1})	Input 2005	Area (ha)	N-input adapted (kg N ha^{-1} yr^{-1})
Grassland	405	88.4	35802	405	66.3	26852
Grassland vulnerable sandy soils	365	143.8	52487	365	82.9	30259
Arable land	285	0.0	0	285	0.0	0
Arable land vulnerable sandy soils	245	0.0	0	245	0.0	0
Maize land	155	5.5	853	155	5.5	853
Maize land vulnerable sandy soils	115	22.1	2542	115	11.1	1277
Nature	31	221.1	6854	31	315.1	9768
Urban	36	66.3	2387	36	66.3	2387
Recreation[3]	196	5.5	1078	196	5.5	1078
Total		552.7	102003		552.7	72474 (-29%)

[3] The land-use type 'recreation' refers to the semi permanent use of the summer homes without sewers.

ranging from 3.5 to 100 meters below the surface. Of these, three wells monitored urban and recreational areas, four monitored forest (nature areas), and seven monitored agricultural areas. The 10 remaining wells could not be attributed to a single land use type. Finally, historical data on groundwater quality were available for three wells sampled in 1913. These data served as the reference for groundwater composition in 1950. Data collected since 1998 were not included because they were not used in the planning process.

The social approach tested the impact of the NSS on the outcome of the spatial planning process. This validation was carried out qualitatively by studying (1) the level of acceptance of the information generated by the NSS and (2) the extent to which this information moved the stakeholders to change their views.

Scenarios

Two land use scenarios were considered in this study: the 'base' and the 'adapted land use' scenarios. The base scenario calculated autonomous development, assuming no changes in the land use pattern. It did, however,

include the reduction of the N application in agricultural areas implemented from 1990 on (Table 2). The provincial policy regarding an additional reduction of manure application in so-called 'vulnerable sandy soils' (Figure 2) was reflected by the reduction in N application on agricultural land since 2005 (Table 2). Half of the grassland and most of the maize land was located on these 'vulnerable sandy soils' (Figure 2). The adapted land use scenario was identical to the base scenario until 2005. The adapted land use scenario was defined by the water company and the province and based on (1) the calculated quality of the shallow groundwater in 2005 obtained from the base scenario and (2) the residence times of the groundwater calculated using a calibrated geohydrological model (Figure 4). This adapted land use scenario changed agriculturally used grassland into natural grassland without any manure or fertilizer application from 2005 onwards, thus establishing an ecological connection zone between the northern and central natural parts of the study area. The western parts of the study area retained their agricultural function (Figure 4). The nitrogen input of the adapted land use scenario was 29% lower than that of the base scenario (Table 3).

The calculations of the scenarios were carried out for the period 1950–2100. This period allowed an evaluation of the scenarios in which the concentrations of the abstracted groundwater reflected the current function-related N application to the soil surface as this period covers the residence times of groundwater. The annual load per land use type was set constant after 2005 since the aim of the study was on the evaluation of the impact of the adapted land use scenario. For the sake of transparency, the calculations were based on publicly available information and no calibration was carried out regarding the N application. Although changing the input parameters might improve the calculation results of the historical concentrations, it will not provide a unique calibration solution and will therefore not improve the prediction of future concentrations (Thorsen et al., 2001). Because the relevant processes are considered by the NSS, differences between the calculated and the measured concentrations provide additional information on local conditions and how to handle this in the planning process.

Results

The results of the NSS are given as (1) the calculations of the effects of land use on the nitrate concentration in groundwater and (2) the outcome of the application of the NSS in the planning process. The effects of land use on groundwater were first calculated for the quality of the shallow groundwater and subsequently for the quality of the abstracted groundwater. The calculated quality of the abstracted groundwater was then compared with measured nitrate concentrations. The results of the scenarios are presented as time series.

Calculation of the groundwater quality

Effects of land use on the quality of shallow groundwater
Current land use together with the calculated nitrate concentrations in shallow groundwater according to the base scenario are shown in Figure 2 for the years 1970, 1985, and 2005 and in Figure 4 for the year 2005. An overview of the calculated concentrations for the various land uses of dry sandy soils during different years is presented in Table 2. The calculated concentrations below grassland and maize land most explicitly reflect the increasing intensity of agricultural practice during the period 1950–1985 (see also Table 2). Concentrations of approximately 320 mg NO_3 L^{-1} were found in the shallow groundwater. The impact of agriculture on groundwater quality decreased in the period 1985–2005 (see Table 2 and Figures 2 and 4).

When national and regional policies were implemented, a strong decrease was seen in the calculated nitrate concentration of the shallow groundwater. The implementation of additional policies for vulnerable sandy areas caused an even further decrease in the concentrations. The general policies resulted in a concentration of up to 114 mg NO_3 L^{-1} below grassland in 2005, i.e., a 65% reduction compared to the 1985 concentration. In the vulnerable sandy areas, there was an additional 20% decrease in the calculated concentration (to 92 mg NO_3 L^{-1}) in 2005. The general regulations resulted in a calculated concentration of up to 78 mg NO_3 L^{-1} under maize land in 2005, i.e., a 76% reduction compared to the 1985 concentration; in 2005, the vulnerable sandy areas showed an additional 18% decrease in the calculated concentration (to 64 mg NO_3 L^{-1}). The calculated concentrations below arable land showed an increase in the first stage of the period (1950–1970) and then remained rather

constant in the period 1970–1985. Below arable land, the general regulations resulted in a 12% decrease in 2005 compared to the 1985 concentration; in 2005, the vulnerable sandy areas showed an additional 16% decrease in the calculated concentration (to 114 mg NO_3 L^{-1}).

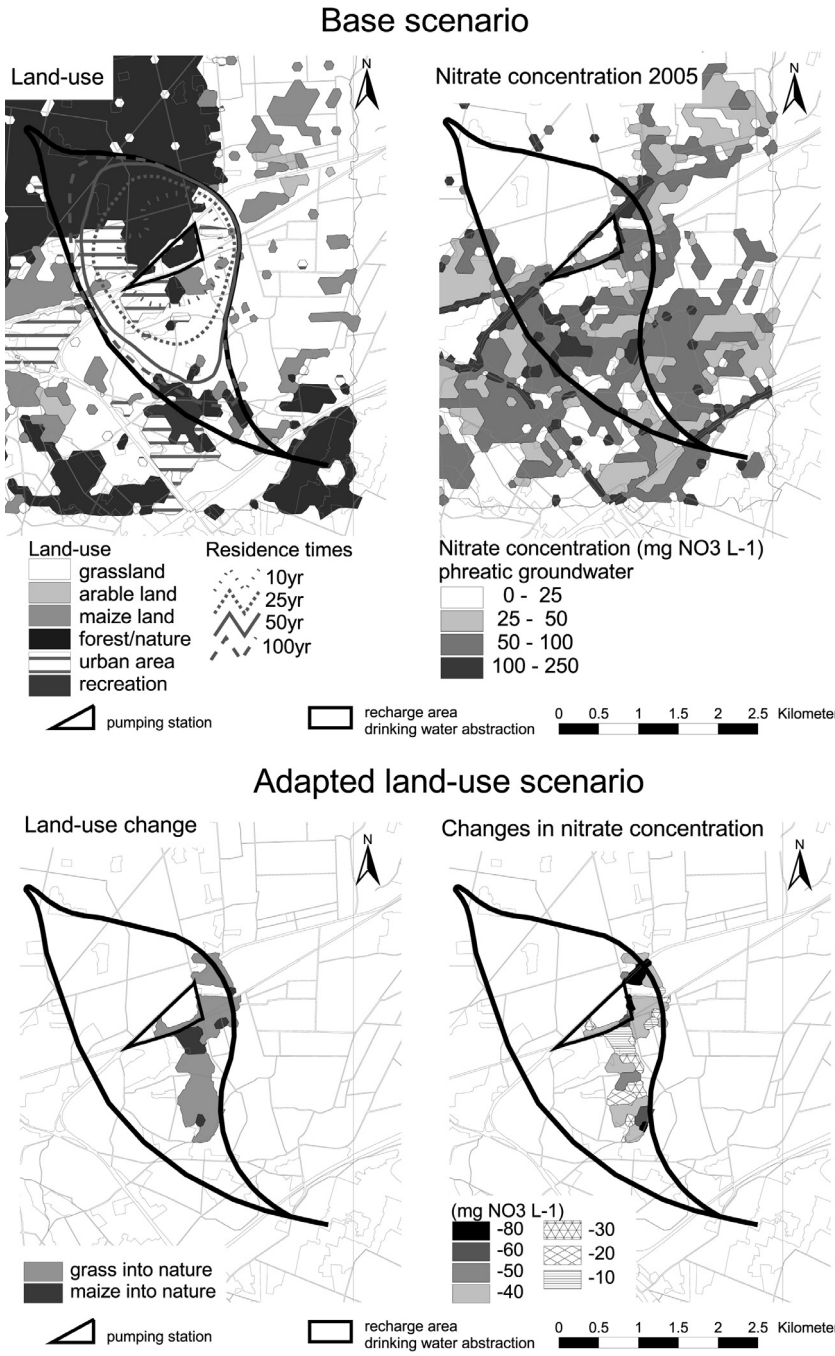

Figure 4. Land use, residence times, and calculated nitrate concentration (mg NO_3 L^{-1}) in shallow groundwater in the base scenario (2005) and the changes in land use and calculated nitrate concentration in the adapted land use scenario.

The calculated nitrate concentrations in the shallow groundwater underneath the forested/natural areas increased between 1970 and 1990. Since 1990, the soil matrix has become saturated with nitrate (Boxman, 2002) and the calculated nitrate concentrations reflect the atmospheric deposition corrected for the N uptake by the trees and denitrification in the zone between the root zone and the groundwater table. The calculated concentrations in 2005 were 10 mg NO_3 L^{-1} (Table 2). The calculated concentration of the shallow groundwater remained rather constant below areas with urban and recreational activities. Those concentrations were approximately 20 and 55 mg NO_3 L^{-1}, respectively. It is noteworthy that in 2005 the concentrations in the shallow groundwater below agricultural land still exceeded the limit of the EU Nitrate Directive in the – prevailing – sandy areas with deep groundwater tables.

Effects of land use on the quality of abstracted groundwater

Figure 3 shows a comparison between the calculated average nitrate concentrations and the measured average concentrations in the abstracted groundwater of the base scenario. Starting from the initial measured concentration of 4 mg NO_3 L^{-1} in 1960, the concentration gradually increased to 24 mg NO_3 L^{-1} in 1998 due to the increased input of nitrogen. Because of groundwater mixing from various land uses and the different ages of the abstraction wells, the spatially and temporally heterogeneous impacts of agricultural land use on groundwater quality became strongly smoothed. The temporal fluctuations, however, became sharper over time as a result of the increasing changes in the pumping rates of the individual wells and the strong heterogeneous increase in nitrate leaching out of the unsaturated zones at particular land use areas. Until 1976 (i.e., from 1960–1976), the calculated nitrate concentrations were lower (average difference: 24%) than the measured ones (Figure 3). After 1976 (1976-1998), they were higher (difference of 59%).

Scenario calculations

Historical and autonomous developments (base scenario) resulted in a continuous rise in the calculated concentrations up to the year 2000 (Figure 3). As a result of the reduction in N application in the early 1990s, the increasing trend of the nitrate concentrations seen in the period 1960–2000 is expected to level off to a rather constant concentration of approximately 32 mg NO_3 L^{-1} in the period 2000–2015. A maximum nitrate concentration of 33 mg NO_3 L^{-1} was calculated for the period 2004 to 2012. Between 2015 and 2050, the calculated concentration is expected to decrease to 24 mg NO_3 L^{-1} (Figure 5). The calculated concentration of 20 mg NO_3 L^{-1} in the abstracted groundwater in 2100 reflects an equilibrium between the land use pattern, the leaching of nitrate, and the transport from the groundwater table to the abstraction.

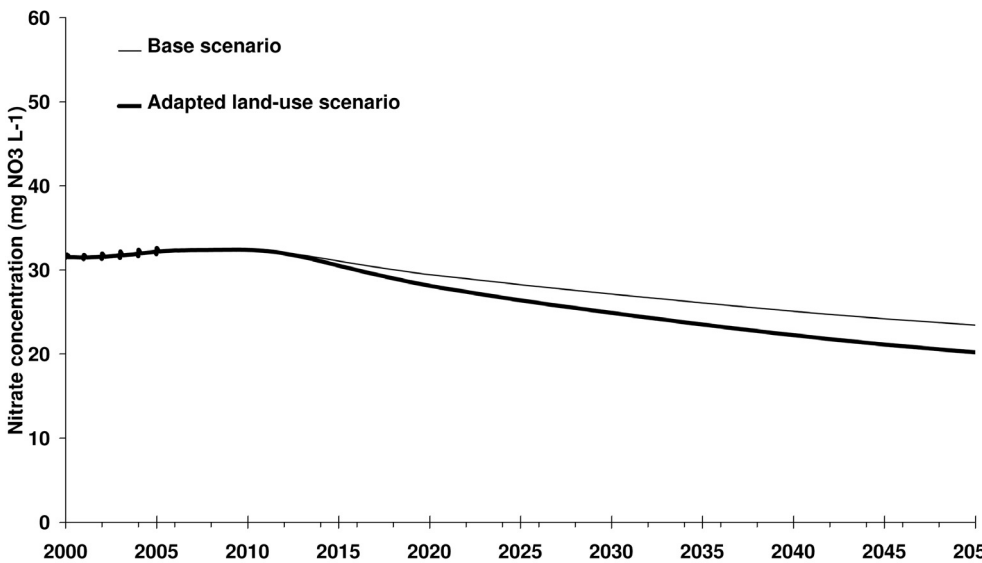

Figure 5. Calculated nitrate concentrations (mg NO_3 L^{-1}) in abstracted groundwater in the base and adapted land-use scenarios during the period 2000-2050.

We found that grassland was the main source of nitrate. This was due not only to the extent of the grassland area, which was 42% of the recharge area (Figure 2), but also to the high levels of N applied to the grasslands (Table 2). The contribution of grasslands to the total nitrate load in the abstraction wells ranged from 60% in the period 1960–1970 to 89% in 2010. Five percent 5% of the recharge area consisted of arable land that was converted to maize land after 1970. The contribution from maize land appeared only after 1980 because 50% of the maize land in the recharge area had residence times of approximately 10 years and 50% had even longer residence times. Even though the contribution of groundwater infiltrating from maize land increased after 1970, the impact of maize land on the quality of abstracted groundwater decreased from 9% in 1980 to an expected 5% in 2100 due to the substantial reduction in the amount of N applied. The total agricultural contribution to the nitrate load at the abstraction site ranged from 84% in 1960 to 97% in 2000, decreasing thereafter to 93%. The relative contribution of nature, covering 40% of the recharge area, was 10%. The contribution of the other land uses, mainly urban land use, was less than 2%.

In the adapted land use scenario, 15% of the grassland and 2% of the maize land in the recharge area was changed into nature (grassland with no added manure or fertilizer) in 2005 (Figure 4). For the period up to 2015, the nitrate concentrations in the base and the adapted land use scenarios were the same. However, the adapted land use scenario resulted in calculated nitrate concentrations of 20 mg NO_3 L^{-1} in 2050 (Figure 5) and 17 mg NO_3 L^{-1} in 2100 in the abstracted water. This is a reduction of 17% and 18%, respectively, compared to the base scenario.

Outcome of the application of the NSS in the planning process

The output generated by the NSS was organized in the form of maps and time series, as requested by the stakeholders. The maps provided an explicit spatial overview of the quality of the shallow groundwater for a selected year. The time series provided insight into the quality of shallow groundwater for a selected location or point in the study area and into the quality of the abstracted groundwater. This information quantified the conflicting interests between agricultural land use and drinking-water function (Figure 2), showing that the sustainable protection of the resource would require a substantial change in land use. The information generated by the NSS and these conclusions were accepted by all stakeholders involved. Subsequently, the spatial bottom-up planning process became organized in a different setting. Agricultural representatives, recreational businessmen, and the municipality were included in the discussions on the regional planning process for the reconstruction of intensive livestock farming under the Reconstruction Law. In the context of the local process, no compensation could be found for possible agricultural losses that might occur in trying to meet the demands of the drinking-water company. In the regional process, however, such compensating measures could be generated. The water company and the province presented a land use change in the form of a policy document that would lead to the sustainable protection of drinking-water abstraction following the adapted land use scenario (Figure 4).

Discussion

This section discusses the validation of the NSS results in terms of (1) the calculated results and (2) their impact on the outcome of the spatial planning process. The discussion of the calculated results addresses the outcome of the model in terms of the effects of land use on the quality of shallow groundwater, the difference between the calculated and the measured quality of the abstracted groundwater, and the consequences of the scenarios for the quality of abstracted groundwater.

Discussion of the calculated results

Quality of shallow groundwater

The uncertainties in the calculated nitrate concentrations in the shallow groundwater below the forested areas are notable especially up to the moment the soil matrix is assumed to be saturated. This saturation was not measured in the study area, but obtained by experimental results from another similar sandy area in the Netherlands. The initial carbon/nitrogen (C/N) ratio of the organic layer was considered relevant for N leaching from forests at a regional scale (Gundersen et al., 1998; Bredemeier et al., 1998). This dependency was not taken into account in our calculations because the C/N ratio of the organic layer and the changes in this ratio over

periods of several decades are unknown. As a consequence, the accumulation of nitrogen, which is a function of the C/N ratio, could not be calculated either. De Vries et al. (2003) considered the uncertainty in the accumulation of nitrogen in non-agricultural soil to be the main factor when calculating the inflow of nitrogen in groundwater. The impact of these uncertainties on the calculated nitrate concentration may be up to a few tens of milligrams of nitrate. This maximum uncertainty corresponds to the variation in concentrations measured in the observation wells in the study area and in national monitoring studies (Frapporti, 1994; Broers, 2002). The impact of these uncertainties on the calculated concentration of the abstracted groundwater was considered to be limited since the contribution of nitrogen input from the forested area is small compared to the total nitrogen input in the study area (see Table 2).

The uncertainty of the calculated nitrate concentrations in shallow groundwater below agricultural areas is mainly related to the N application. Although manure production is known rather accurately by means of the annual livestock count, variations in manure composition and distribution may result in considerable differences in the actual concentrations in shallow groundwater underneath individual fields. This variation ranged from several hundreds of kilograms of N up to 1000 kg N on maize land in the early 1980s and resulted in large variations (min-max) in nitrate concentrations in soil moisture under grassland and maize (4–1246 and 8–800 mg NO_3 L^{-1}, respectively) (Van Beek et al., 1994). The effects of these variations will decrease rather than increase at the regional level, because individual concentrations are averaged out across wider areas (Thorsen et al., 2001). In addition, the model description of the processes contained uncertainties. The sum of the N uptake by crops and the denitrification in unsaturated zone, calculated by subtracting the N leaching calculated by SPREAD from the N input (Table 2), was compared with the ranges presented by De Vries et al. (2003). SPREAD calculated the ranges for uptake and denitrification to be 143–480, 124–319, and 156–286 kg N ha^{-1} yr^{-1}, respectively, for grassland, maize land, and arable land. De Vries et al. (2003) reported averages (and ranges) for N uptake when N was not limiting crop growth for a dry sandy soil to be 240 (150–300), 155 (125–175), and 110 (50–150) kg N ha^{-1} yr^{-1} for grassland, maize land, and arable land, respectively. The denitrification factors presented by De Vries et al. (2003) were 0.30–0.60 for grassland and 0.20–0.50 for arable and maize lands. The resulting maximum denitrification factor that we calculated from the SPREAD results ranged from 0.29 to 0.50 for grassland, 0.29 to 0.36 for maize land, and 0.33 to 0.61 for arable land. These factors lie within the range reported by De Vries et al. (2003) and Kroeze et al. (2003). Therefore, the N leaching calculated by SPREAD was well within the ranges presented by De Vries et al. (2003).

From our analysis, it can be concluded that despite the uncertainty within the main agricultural land uses, i.e., grassland and maize land, distinctive differences exist between agricultural land use and forested areas. The NSS can, therefore, be considered sufficiently sensitive to reveal of the effects of conflicting land uses and is thus useful in the planning process by formulating land use scenarios.

Quality of abstracted groundwater

Ideally, physically based transport models are applied to model reactive groundwater transport and all of the parameters are independently obtained in the field. This can be achieved for systems that are rather homogeneous in their spatial characteristics and have a temporally constant input and where a limited number of processes are operational. In comparison, this can rarely be reached when models are used for management support at a regional or larger scale, especially when a three-dimensional modeling approach is necessary as in this study. Two kinds of approaches are now recognized: calibration and use of best-available information. In the first approach, the values for process parameters are obtained by means of calibration. Keating and Bahr (1998), Brun et al. (2002), and Prommer and Stuyfzand (2005) used this approach in their three-dimensional, local field studies where groundwater transport occurred within a single geological stratum and a limited number of process parameters needed to be calibrated. One problem with calibration is that non-unique calibration solutions may exist even for relatively simple field systems (Saaltink et al., 2003). The second kind of approach is based upon the use of best-available information (Brusseau, 1991) and is adequate for management support. Suitable models have a simple model structure that satisfies the needs of the policy-maker, while still ensuring that model parameters retain physical significance (Quinn, 2004). Historical modeling with a predictive approach was performed in the present study using independently estimated or measured parameters to test the performance of the model (Kent et al., 2000). This approach is more useful for underconstrained problems as

in the Holten area, where non-unique calibration solutions certainly exist and unrealistic values for parameters may be obtained.

The calculated concentrations of abstracted groundwater were validated by the historical concentration measurements of abstracted groundwater: the NSS output was compared with measurements for the period 1960–1998 (Figure 3). Considering the historical trends in N application and the observed nitrate concentrations in the abstraction wells, it was concluded that the increase in nitrate in both the measured and calculated data was mainly due to the increase in the agricultural use of nutrients in the recharge area and that the increased atmospheric deposition and N saturation of forest soil had less influence. The continuous increase in N application (Table 2) in the agricultural areas resulted in a continuously increasing calculated curve for abstracted groundwater, while the average level of the measured concentrations remained rather stable during the period 1974–1988, although strongly fluctuating from year to year. The observed and calculated nitrate concentrations differed on average 45% during the period 1960–1998. The calculated concentrations were lower than the measured nitrate concentrations in the period 1960–1976 and higher for the period 1976–1997.

The main reason for the observed fluctuations in the nitrate concentrations in abstracted groundwater was the local heterogeneity of the groundwater flow pattern. This pattern was induced by the water company through the differences in pumping hours of the individual abstraction wells and the installation of deep abstraction wells, which changed the groundwater flow paths and geohydrochemical characteristics of the groundwater flowing towards the abstraction site. The actual annual abstraction volume during the period 1977–1998 was 2.2 10^6 m^3 yr^{-1} ±11%, while the fluctuation of the abstraction rate at the individual wells varied from 32.2 to 188.9% in the same period. No data were available on the variation before 1977. This heterogeneous abstraction pattern caused fluctuations in the nitrate concentration in abstracted groundwater due to the differences in nitrate concentrations in the groundwater originating from forested (nature) areas (40% of the recharge area) and different agricultural land uses (46% of the recharge area).

When our modeling fit is compared with other studies, it is notable that a match within 50% of the observed values for most observation points was not reached in any three-dimensional field modeling study, even those in which process parameters were calibrated (e.g., Keating and Bahr, 1998; Brun et al., 2002; Prommer and Stuyfzand, 2005). Objective criteria were not used and the reproduction of patterns was generally felt to be satisfactory. The difference between the measured and the modeled values was often several factors when individual points were compared. This discrepancy is attributed to, for example, errors in mass estimate (Schirmer et al., 2000) or problems with specifying the contaminant boundary condition in the source area (Brun et al., 2002).

Our approach based on 'best available information' is, therefore, satisfactory for water-management support (Quinn, 2004). The results of the NSS were considered transparent and convincing and were also accepted by the stakeholders in the local planning process. From the start, the stakeholders participated in the setup and use of the NSS, which increased their commitment to the information generated by the NSS. This method corresponds to the process of 'confidence building' to achieve model acceptance from regulators and the public in such a way that decisions can be made based on the models (Hassan, 2004). This further supports our statement that, for the sake of clarity and transparency, the application of reactive transport modeling in water-management support is preferably based on best-available information and sensitivity analysis than on calibration.

Scenarios

The NSS was used for the formulation of an adapted land use scenario. The impact of the reduction of the N application by changing the land use - adapted land use relative to the base scenario – was calculated to be 16% in 2050 (Figure 5) and 18% in 2100. Despite the limited amount of agricultural land use that was changed into nature (Figure 4), this significant reduction could be obtained using the explicit spatial information on areas having the highest N leaching and combining this information – infiltration rates and residence times – with the calibrated geohydrological model.

The scenarios showed that the results of the NSS were sensitive to changes in land use and/or changes in N application within the various land use areas. Despite the differences between the calculated and the measured concentrations, consistent and reproducible use of input data provided quantitative insight into the impact of measures such as adapted land use.

Discussion of the impact on the planning process

The planning process originally focused on the protection of drinking-water abstraction by changing land use. This change required measures within the types of land use as well as changes from one land use type to another. The NSS supported this process by setting up an adapted land use scenario and providing quantitative information on the impact of the proposed land use change and change in N application and on the quality of the abstracted groundwater. Implementing this scenario should decrease the conflicts between the various types of land use. Based on this insight, the need for a substantial change in the actual land use pattern in order to protect the abstraction site became a shared notion among the stakeholders. Since the local process did not have space with which to compensate the farmers, compensation was included in the regional planning process: the farmers were given land outside the recharge area of the drinking-water abstraction. The water company and the province were the leading players in the local planning process and part of a large group, including socioeconomic actors, in the regional Reconstruction process aimed at a new balance between land use functions in a rural area. They successfully advocated the need for safe and reliable drinking water and proposed the policy option that corresponded to the adapted land use scenario. Formalization of the adapted land use scenario was the end point of the planning process. Nevertheless, groundwater quality as well as the agreements made between the stakeholders regarding changes in land use and N application are still being monitored in an ongoing groundwater-protection process.

The validation of its applicability in the planning process showed that the NSS is an adequate instrument for solving groundwater conflicts that arise during land use management: land use was changed according to the information provided by the NSS in the spatial planning process near the village of Holten. The application of the NSS in more local or regional planning processes is, however, required to review the applicability of the system and its advantage in solving groundwater conflicts and protecting groundwater quality by spatial planning compared to the current system of environmental regulations.

Conclusions

A negotiation support system (NSS) is an adequate tool for a bottom-up spatial planning process to visualize and quantify conflicting interests that arise during the land use functions agriculture and drinking-water abstraction. An NSS that is suitable on regional scale can be constructed to test groundwater quality. Our NSS provided information on the impact of various land use functions on the nitrate concentration of shallow groundwater and allowed the quantification of the measures taken to change that impact. Combining this information with the properties of the groundwater system allowed the quantification of the impact of land use scenarios on the quality of abstracted groundwater. Despite the differences between the calculated and the measured nitrate concentrations, the sensitivity and resolution of the NSS in distinguishing between land use functions were sufficient to carry out scenario analyses that were accepted by the stakeholders during the planning process. It was concluded from the validation of its applicability in the planning process that the NSS is an adequate instrument for solving groundwater conflicts and protecting groundwater quality during land use management: land use was changed according to the information provided by the NSS, thus supporting an effective groundwater-resource management.

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Chapter 4

Stochastic uncertainties and sensitivities of a regional-scale transport model of nitrate in groundwater

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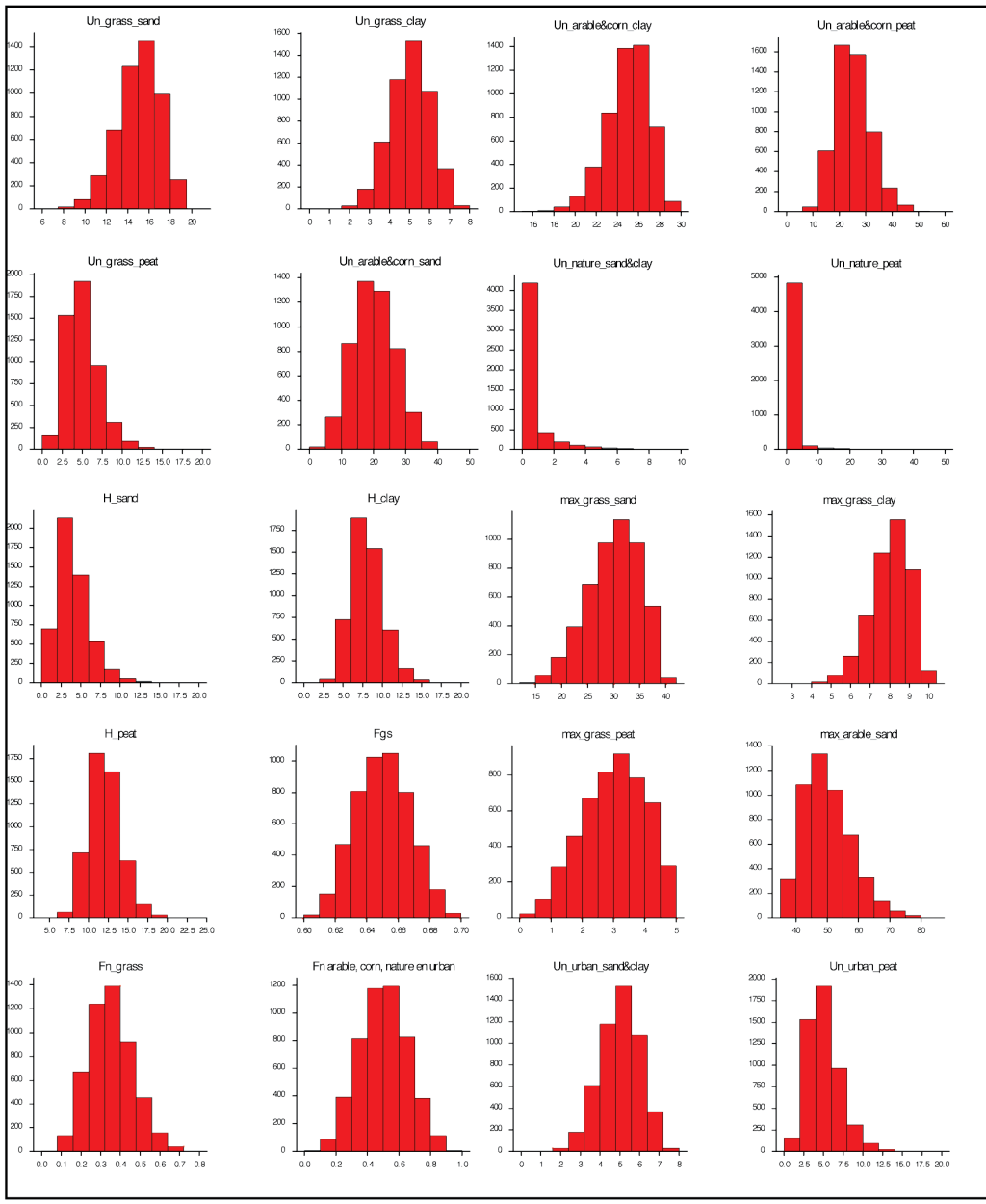

Examples of some probability density functions of the input parameters.

Stochastic uncertainties and sensitivities of a regional-scale transport model of nitrate in groundwater

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Abstract

Groundwater quality management relies more and more on models in recent years. These models are used to predict the risk of groundwater contamination for various land uses. This paper presents an assessment of uncertainties and sensitivities to input parameters for a regional model. The model had been set up to improve and facilitate the decision-making process between stakeholders and in a groundwater quality conflict. The stochastic uncertainty and sensitivity analysis comprised a Monte Carlo simulation technique in combination with a Latin Hypercube Sampling procedure. The uncertainty of the calculated concentrations of nitrate leached into groundwater was assessed for the various combinations of land use, soil type, and depth of the groundwater table in a vulnerable, sandy region in the Netherlands. The uncertainties in the shallow groundwater were used to assess the uncertainty of the nitrate concentration in the abstracted groundwater. The confidence intervals of the calculated nitrate concentrations in shallow groundwater for agricultural land use functions did not overlap with those of non-agricultural land use such as nature, indicating significantly different nitrate leaching in these areas. The model results were sensitive for almost all input parameters analyzed. However, the NSS is considered pretty robust because no shifts in uncertainty between factors occurred between factors towards systematic changes in fertilizer and manure inputs of the scenarios. In view of these results, there is no need to collect more data to allow science based decision-making in this planning process.

Introduction

Groundwater quality management relies more and more on models in recent years (e.g. Thorsen et al., 2001; Refsgaard et al., 2005). These models are used to predict the risk of groundwater contamination for various types of land use. Advanced, physically based models could be used, but require dense data input which is sparsely available, particularly on regional scale (Thorsen et al., 2001). Regional models have been developed (e.g. Tiktak, 1999; Beekman, 1998) which use more generic data inputs, such as groundwater level maps, regional manure application data and soil types.
Both approaches, i.e., the advanced physically based model and regional model, have uncertainties that need to be addressed in decision-making. Transparent environmental decision-making implies that policy-makers and regulators have insight into the uncertainty and reliability of the input parameters and output variables of a model as a kind of 'certification of modeling results' (Anderson and Bates, 2001). In addition, it means that the model is diagnostic for water management: different scenarios yield significantly different model results.

The uncertainty of environmental models complicates their use as prognostic tools (see, e.g., Freeze, 1975; Summers et al., 1993; Zhang et al., 1993; Tiktak, 1999; Thorsen et al., 2001; Refsgaard et al., 2005; Blasone et al., 2008). The sources are (1) the uncertainty of the input data both time-varying (such as climate data and N-application data) and constant (e.g., soil physical characteristics) and (2) an inadequate model structure (process descriptions) (Thorsen et al., 2001).

This paper presents an approach to assess uncertainties and sensitivities to input parameters for a regional model to improve and facilitate the decision-making process between stakeholders and in a groundwater quality conflict. It demonstrates that detailed models with highly accurate but costly input requirements are not always required to help stakeholders to reach science based decisions.
Van den Brink et al. (2008) applied a regional model to identify the best course of action to resolve a conflict between land use and groundwater management. The model under consideration is a Negotiation Support

System (NSS), set up to resolve a stakeholder-accepted conflict in the Holten region, the Netherlands. The NSS comprises of a series of nitrate-leaching models for the unsaturated zone with various kinds of land use and a three-dimensional groundwater transport model for calculation of nitrate transport from the groundwater table to a drinking water abstraction. The aims of the NSS are (1) to explain the complexity of the groundwater system in terms of recharge area and residence times, (2) to rephrase the problem so that it illustrates the unique role of each stakeholder and the interrelationship between the concerned parties, and (3) evaluate scenarios. The two former aims have been met (Van den Brink et al., 2008), but the latter has not been verified. It requires that the NSS can distinguish between various land use functions for which the uncertainty of the results must be known.

The uncertainty and sensitivity analysis of the NSS addressed three questions: (1) what is the uncertainty of the calculated concentrations of the nitrate leached into groundwater? (2) which input parameters explain the largest variation in the model output? and (3) how does the uncertainty of leached nitrate translate into the uncertainty of the concentration in the abstracted groundwater? The answers to these three questions were used to determine whether the model could significantly distinguish between the scenarios. The stakeholders proposed scenarios of land use changes.

Figure 1. Study area of the Holten case in the province of Overijssel, the Netherlands.

This analysis was considered representative for drinking water abstractions below vulnerable sandy soils with various kinds of land use in the infiltration area.

Materials and methods

Study site

Geohydrological and geochemical setting
The Holten region is located in the eastern part of the Netherlands, in the province of Overijssel (see Figure 1). The region enjoys a rural setting with a balance between small municipalities, farms/dairies, and open public lands. The northern and southernmost parts of the area consist of ice-pushed ridges covered with deciduous forest and pines. Nature and recreational functions dominate here. The central-southern valley is mainly used for intensive agriculture. The subsoil can be considered a single aquifer consisting of three distinctly different geological layers upon a geohydrological basis at approximately 90 meters below the soil surface. Groundwater abstraction started in this area in 1960 with 10 wells and most abstraction screens were placed in the second, most permeable, (sub)oxic layer. Five new wells were installed in the third, i.e. deepest, anaerobic layer in 1985 in order to avoid well clogging due to mixing of oxygen-rich and iron(II)-rich groundwater. The local water company, in cooperation with the provincial authority, initiated a local planning process in response to the mounting concerns about nitrate concentrations in abstracted and shallow groundwater.

Negotiation Support System (NSS)
The NSS operates on an annual time step encompassing the period 1950–2050 and includes a 10-year starting-up period to simulate the initial distribution of nitrate in groundwater as a function of land use (1950–1960), a 39-year validation period (1960–1998), and a 52-year planning horizon (1999–2050). The model comprises two key systems, the groundwater and nitrate budgets (see Appendix 1 for more detail). Basic elements of the groundwater budget includes net annual recharge as a function of the vegetation type, depth to the groundwater table, soil type, and the annual abstraction rate at each well. The nitrate budget tracks the primary sources, including fertilizers, forest-soil leachate, soil-mineral leachate, and the primary sinks represented by crop uptake, denitrification and abstraction.

Planning process
The NSS supported the local planning process by providing a tool with which to explore alternative land use scenarios and providing quantitative information on the impact of the proposed land use change and the change in N-application on the quality of abstracted groundwater. Various policy fields were involved: environmental and water management, management of natural heritage and land use planning. The stakeholders in the planning process were farmers, the water supply company, water board, the province, the township of Holten, recreation entrepreneurs, and organisations managing the natural heritage. The farmers have been represented by three people representative for the interest of the five farms entirely dependent economically on agricultural activities in the capture zone of the groundwater abstraction site. The water supply company has been represented by three people responsible for water quality issues, spatial planning issues and communication with farmers respectively. The water board has been represented by one person. The province has been represented by one person responsible for groundwater quality. The township of Holten has been represented by one person responsible for spatial planning. The recreation entrepreneurs have been represented by two people, running a camping site and an area with recreation homes. The regional departments of the ministries managing the natural heritage and rural area have been represented by one person respectively.

Several land use scenarios were considered by Van den Brink et al. (2008): the 'base' and 'adapted land use' scenarios were further developed to identify the best course of action to resolve the stakeholder-accepted conflict. The base scenario calculates autonomous development, assuming no changes in land use pattern. The adapted land use scenario is identical to the base scenario until 2005, at which time the adapted land use scenario is assumed to be implemented. The adapted land use scenario was defined by the water company and the provincial government based on: (1) the calculated quality of the shallow groundwater in 2005 obtained from the base scenario and (2) the residence times of the groundwater toward the abstraction site calculated with the

calibrated geohydrological flow model. No scenarios were further developed, because implementing the adapted land use scenario would be adequate for a sustainable protection of the drinking water abstraction.

The output generated by the NSS was organized in the form of maps and time series as requested by the stakeholders. This information visualized and quantified the conflicting interests between agricultural land use and drinking-water function, indicating that sustainable protection of the resource would require a substantial change in land use. The changes in nitrate concentration due to the adapted land use scenario are significant as they are realized in an area with rather short residence times and were acceptable to the stakeholders. As a result, the need for a substantial change in the actual land use pattern to protect the abstraction site became a shared notion among the stakeholders. In this way information from the NSS was used in the bottom-up spatial planning process as the technical basis for management decision-making (c.f. Simonovic, 2002). More details on the model and the planning process can be found in van den Brink et al. (2008).

Uncertainty and sensitivity of the calculated concentration of nitrate leached into groundwater

The uncertainty and stochastic sensitivity analysis was based on a Monte Carlo approach in combination with a Latin Hypercube Sampling (LHS) procedure applied to the input parameters. The uncertainty due to the model structure was assumed to be negligible compared to the uncertainty due to the input parameters, based on previous experience with the same or similar nitrate leaching models applied on comparable time and spatial scales (Beekman and Vogelaar, 1997; Beekman, 1998; Otte, 1998; Vink and Schot, 1999). The Monte Carlo LHS approach consisted of six steps: (1) select input data for inclusion in the analysis, (2) assign probability density functions (PDFs) to each input factor, (3) generate an input matrix, (4) calculate nitrate leaching, (5) analyze the model output (uncertainty analysis), and (6) assess the influence of each input factor on the model output (sensitivity analysis).

Step 1: select input data

Two types of input data were used in our model: input parameterizing the leaching processes (see Appendix 1) and input describing nitrate sources (e.g., fertilizers, atmospheric deposition and basic leaching) and sinks (crop uptake and denitrification). They play different roles in the planning process. The leaching parameters are considered constant, while the source parameters may be affected by land use planning. We conducted the analysis using 41 input parameters (see Appendix 2). The base scenario was used to describe the autonomous developments regarding the implementation of Dutch policy on mineral management and local agricultural practice and developments as provided by the stakeholders. The 2005 data set was used because the output calculated with these data showed and quantified the conflicting interests between agricultural land use and drinking-water function. The uncertainty of the quality of the abstracted groundwater was analyzed for the situation in which the nitrate concentration of the groundwater system was in equilibrium with the nitrogen input (N input) from land use, i.e. a stationary situation with a constant nitrogen load in time at the surface.

Step 2: assign PDFs

Key literature sources of the uncertainties of the input parameters were the review papers by de Vries et al. (2003) and Kroeze et al. (2003). The model parameters that were used to describe the empirical relationships within the NSS were based on Beekman (1998). PDF characteristics were established for all input parameters (see Appendix 2).

Step 3: generate an input matrix

The input matrix was generated using the LHS procedure for all combinations of model inputs and model parameters. This procedure uses a probabilistic sampling procedure that provides better coverage of the input distributions and has been shown to be more efficient than the random sampling of the standard Monte Carlo analysis (Helton and Davis, 2003). The input matrix in the present study consisted of 1000 samples. No correlations between model inputs and parameters or other restrictions on the sampling were necessary.

Step 4: calculate nitrate leaching

The nitrate concentration of the shallow groundwater (mg NO_3 L^{-1}) in the Holten area was calculated for each Monte Carlo-LHS sample. Table 1 presents an overview of the results of the different combinations of soil, land use, and groundwater table depth.

Step 5: uncertainty analysis
The mean, standard deviation, coefficient of variation, and 95% confidence interval (i.e., p = 0.05 and 0.95 quantile of the nitrate concentration) were used as measures of the uncertainty of the model output. These statistics were calculated from the 1000 samples of the Monte Carlo-LHS step.

Step 6: sensitivity analysis
Assessment of the relative importance of the individual input parameters on the output variables was conducted using variance decomposition. The variance of the model output can be seen as the specification of the uncertainty. The top marginal variance (TMV) of an input is the variance reduction that would occur if the input would become fully known (Jansen et al., 1994). It is also called the first-order sensitivity index or correlation ratio (Brus and Jansen, 2004). In a regression-based sensitivity analysis, the relationship between the model output (y) and model inputs ($x_1 \ldots x_k$) is approximated by a regression relation. Input data explaining more than 10% of the TMV were selected. In the present analysis, we used a linear regression model. The quality of the approximation was measured using R^2 adjusted for sample size and number of parameters, i.e., the percentage of variance accounted for in the full linear model, and was successful if R^2 was close to 100%. The sensitivity analysis was performed with USAGE 2.0, a collection of GenStat algorithms for sensitivity and uncertainty analyses (Goedhart and Thissen, 2002; GenStat Committee, 2006).

The sensitivity of the model output for the individual input parameters was expressed by the TMV within the combinations of land use (grassland, maize, arable, nature, urban, and recreation), soil type (sand, clay, and peat) and groundwater level classification (see Appendix 1, Table 4). The groundwater level classification used is a Dutch classification for the depth of the Groundwater table below the surface, Gt (Van Heesen, 1970; Knotters and Van Walsum, 1997). Around Holten, groundwater table depth classes are present between Gt3 (shallow) and Gt8 (deep).

Uncertainty of the calculated nitrate concentration in groundwater abstracted
The uncertainty of the nitrate concentration in the abstracted groundwater was calculated as a function of the uncertainty of the nitrate concentration in shallow groundwater and the uncertainty of the fraction of groundwater abstracted from the third anaerobic layer (see description of study site).

The average nitrate concentration and standard deviation of the various combinations of land use, soil type, and groundwater level classification – determined by stochastic uncertainty analysis – was weighted spatially according to,

$$\mu_X = \sum_{i=1}^{n} \frac{X_i A_i}{A_{tot}} \qquad (1)$$

where μ_X is the average nitrate concentration in shallow groundwater (mg NO_3 L^{-1}), X_i is the nitrate concentration for combination i of land use, soil type, and groundwater table class (mg NO_3 L^{-1}), A_i is the surface area associated with combination i (m^2) and A_{tot} is the total surface area (m^2) =$\sum A_i$.

The uncertainty of the nitrate concentration in abstracted groundwater was calculated from the variances in concentration in shallow groundwater and reduction factor, where the co-variances were assumed to be zero (Mood et al., 1974):

$$\mathrm{var}(Xf) = \mu_X{}^2\,\mathrm{var}(f) + \mu_f{}^2\,\mathrm{var}(X) + \mathrm{var}(X)\,\mathrm{var}(f) \qquad (2)$$

where X is the nitrate concentration in shallow groundwater (mg NO_3 L^{-1}), f is the fraction of abstracted anaerobic groundwater from third layer (-), μ_z is the average of z and var(z) is the variance of z.

Results

This section is divided into three parts: (1) uncertainty and (2) sensitivity analyses of nitrate leaching from the unsaturated zone, and (3) uncertainty analysis of the nitrate concentration in abstracted groundwater. The

uncertainty of the calculated nitrate concentration in shallow groundwater is expressed as the standard deviation. The 95% confidence interval is calculated as a function of soil type, groundwater level classification, and N-application.

Table 1. Calculated uncertainty in output variables for each combination of land use (lu), soil (s), and groundwater level classification (Gt). The area within the study area, mean, standard deviation, and 95% confidence interval are given for each combination.

Output variable	Area (m^2)	Mean (mg NO_3 L^{-1})	Sd	Sd (%)	Lower (mg NO_3 L^{-1})	Upper (mg NO_3 L^{-1})
lu001s1Gt3	211762	12.03	4.80	39.9	5.77	21.06
lu001s1Gt5	643685	60.06	23.81	39.6	28.03	102.77
lu001s1Gt6	1457687	72.35	27.04	37.4	35.33	124.34
lu001s1Gt7	407203	90.38	32.06	35.5	45.73	150.05
lu001s1Gt8	1087931	114.49	39.80	34.8	59.47	187.83
lu001s3Gt3	2087760	4.10	1.13	27.6	2.43	6.17
lu002s1Gt3	31978	13.58	3.66	27.0	8.09	19.86
lu002s1Gt6	12778	81.34	17.53	21.6	53.92	110.11
lu003s1Gt3	67662	8.18	2.21	27.0	4.99	12.20
lu003s1Gt5	11080	40.90	10.92	26.7	23.90	60.33
lu003s1Gt6	267130	49.13	11.09	22.6	32.50	67.84
lu003s1Gt7	33243	61.40	11.85	19.3	43.47	82.45
lu003s1Gt8	41089	77.77	13.97	18.0	54.94	100.59
lu003s3Gt3	230630	6.13	1.68	27.4	3.65	9.11
lu004s1Gt5	7739	5.45	1.94	35.7	2.58	9.09
lu004s1Gt6	46500	6.53	2.11	32.3	3.40	10.56
lu004s1Gt7	43602	8.18	2.48	30.3	4.46	12.69
lu004s1Gt8	173866	10.35	3.03	29.2	5.80	15.67
lu005s1Gt5	18299	12.36	3.68	29.8	6.96	19.15
lu005s1Gt6	11897	14.82	3.76	25.4	9.12	21.38
lu005s1Gt7	30389	18.52	4.10	22.2	12.03	25.62
lu005s1Gt8	649832	23.46	4.98	21.2	15.95	31.98
lu005s3Gt3	28241	3.22	0.86	26.8	1.96	4.77
lu006s1Gt8	45890	55.51	14.29	25.7	33.72	79.75
Vulnerable sandy soils						
lu101s1Gt3	30224	9.63	3.74	38.9	4.83	16.7
lu101s1Gt5	284290	48.06	18.65	38.8	23.59	81.46
lu101s1Gt6	2094426	57.88	21.08	36.4	29.99	97.65
lu101s1Gt7	1381742	72.31	24.96	34.5	39.31	119.45
lu101s1Gt8	2686584	91.60	30.95	33.8	50.53	149.64
lu102s1Gt6	120073	71.96	15.22	21.2	47.86	96.61
lu102s1Gt7	8852	90.10	16.77	18.6	65.23	119.00
lu102s1Gt8	104239	114.11	19.72	17.3	83.55	148.06
lu103s1Gt5	87485	33.59	9.00	26.8	19.44	49.49
lu103s1Gt6	361638	40.34	9.17	22.7	26.78	56.06
lu103s1Gt7	234511	50.42	9.78	19.4	36.01	66.79
lu103s1Gt8	1150898	63.86	11.51	18.0	45.37	82.56

lu001 = grassland, lu002 = arable land, lu003 = maize land, lu004 = nature, lu005 = urban, lu006 = recreation; lu101 = grassland, lu102 = arable land, lu103 = maize land at vulnerable sandy soils; s1 = sandy soil, s3 = peat soil; Gt = groundwater class. The groundwater level classification and leaching fraction is: Gt1 = 0, Gt2 = 0.05, Gt3 = 0.10, Gt4 = 0.40, Gt5 = 0.50, Gt6 = 0.60, Gt7 = 0.75, Gt8 = 1.0, see Appendix 1, table 4.

Uncertainty analysis

The aim of the uncertainty analysis is to quantify the overall uncertainty of the model output as a result of the uncertainties of the model input. The results in terms of statistical data are given in Table 1. The variation of the soil types in the study area was so slight that the impact of soil type on output variables could not be conducted (see Table 1). For soil type sand and groundwater table depth class Gt6 (intermediate depth), the relative uncertainty of leached nitrate differed between land uses, increasing from arable at 21.6% via maize, urban and nature, to grassland at 37.4%. In addition to having the largest relative uncertainties, the land use functions grassland and nature also covered the largest areas within the study area. The absolute uncertainties of the calculated leaching increased from nature at 2.1 mg NO_3 L^{-1}, via urban, maize and arable, to grassland at 27.0 mg NO_3 L^{-1}. The land use function grassland had by far the largest absolute uncertainty.

For soil type sand and land uses grassland and maize, the relative uncertainties varied with groundwater level classifications, increasing with decreasing groundwater table depth (i.e. an increasing impact of denitrification). The absolute uncertainties showed an opposite pattern. The relative uncertainties increased from 34.8% (groundwater table depth class Gt8 = large depth) to 39.9% (Gt3 = small depth) for grassland–sand and from 18.0% (Gt8) to 27.0% (Gt3) for the combination sand and maize. In contrast, the absolute uncertainties decreased from 39.8 mg NO_3 L^{-1} (Gt8) to 4.8 mg NO_3 L^{-1} (Gt3) for grassland on sand and from 14.0 mg NO_3 L^{-1} (Gt8) to 2.2 mg NO_3 L^{-1} (Gt3) for maize on sand.

Since 2001, the provincial authorities have allocated 'vulnerable sandy soils', where an additional reduction in the application of manure was implemented (Van den Brink et al., 2008). Although the nitrate leaching at the vulnerable sandy soils for the corresponding combinations of land use, soil type and groundwater table depth is less as result of a reduced N-application, the relative uncertainty of this concentration is identical.

Sensitivity analysis

The results of the sensitivity analysis are shown in Table 2, with the top marginal variance (TMV) being presented as the percentage of total variance. The adjusted R^2 was at least 94%, indicating that nearly all variance in the output was accounted for and that there was no indication of an interaction between the model inputs. The TMVs given in Table 2 show the uncertainties caused by a particular input parameter.

The uncertainty in the available nitrogen in the growing season *Z* is important for the variation in nitrate leaching in agricultural land uses while and the total nitrogen input in the winter period *Y* is important for the variation in nitrate leaching in non-agricultural land uses (see Appendix 2; table A.5 in Appendix 1 gives an overview of the symbols used). The variance explained by *Z* was up to 82% in the case of agricultural land use and negligible for non-agricultural land uses. The variance of *Y* was negligible for the agricultural land uses and only up to 22% for non-agricultural land use (Table 2). The uncertainty of the model parameter denitrification factor (*D*) was important for each land use. This parameter reduces the nitrate leaching by reducing the N-surplus. The N-surplus is the difference between the primary sources, such as fertilizers, sewage leakage and atmospheric deposition, and the primary sinks represented by crop uptake. The reduction depends only on the groundwater table depth class (Gt). *D* explained up to 55% of the variance in nitrate leaching for both agricultural and non-agricultural land uses. Differences in the top marginal variance among the various land uses with the same groundwater level classification were caused by a different weight of that parameter in the N-surplus and not by differences in the uncertainty of *D*. The *D* PDF uncertainties were 20% for all groundwater table depths.

For the calculations of agricultural land use, the model parameters basic leaching (*B*) and the maximum leaching percentage for a specific combination of soil and land use (ξ) were important, like *D*. *B* explained up to 44% of the variation in nitrate leaching where ξ explained up to 29%. The uncertainty of basic leaching (*B*) was less for grassland (0–20 kg N ha^{-1} yr^{-1}) than for arable and maize lands (0–40 kg N ha^{-1} yr^{-1}), because grassland disturbs the soil structure less than the cultivation of arable land and maize. De Vries et al. (2003) consider a net mineralization/immobilization of zero for non-peat soils for all agricultural land uses. This uncertainty in basic leaching explained an important part of the variance seen for maize. The impact of the uncertainty was greater for maize than for arable land, because more N was applied to arable land ($F + M = 100 + 160$ kg N ha^{-1} yr^{-1}) compared to maize ($F + M = 100 + 30$ kg N ha^{-1} yr^{-1}). *F* indicates the nitrogen application by

Table 2. Top marginal variance of the different input variables and model parameters is given as % of the total variance in the uncertainty of the combinations of land use (lu), soil (s), and groundwater level classification (Gt).

Output variable	N-application in growing season *(Z)*	N-application in winter *(Y)*	Basic leaching *(B)*	Organic matter content soil *(S)*	Maximum leaching percentage (ζ)	Leaching fraction in winter (u_w)	Denitrification factor *(D)*	R^2adjust based on a linear fit
lu001s1Gt3	61	0	0	0	11	0	23	93
lu001s1Gt5	60	0	0	0	10	0	22	91
lu001s1Gt6	70	0	0	0	13	0	14	90
lu001s1Gt7	79	0	0	1	14	0	4	92
lu001s1Gt8	82	0	0	1	15	0	0	92
lu001s3Gt3	6	0	7	3	29	1	52	92
lu002s1Gt3	19	0	8	0	17	0	55	92
lu002s1Gt6	32	0	8	0	19	1	32	93
lu003s1Gt3	16	0	20	0	8	2	53	93
lu003s1Gt5	15	0	20	1	5	2	54	90
lu003s1Gt6	24	0	30	1	8	4	35	92
lu003s1Gt7	33	0	39	2	12	4	12	93
lu003s1Gt8	38	0	44	2	15	5	0	94
lu003s3Gt3	4	0	34	1	4	2	52	94
lu004s1Gt5	0	8	12	24	0	20	32	83
lu004s1Gt6	0	11	12	31	1	27	16	84
lu004s1Gt7	0	11	18	34	0	30	7	85
lu004s1Gt8	0	13	18	36	1	33	0	85
lu005s1Gt5	0	9	4	10	0	33	47	91
lu005s1Gt6	0	13	5	14	0	43	29	88
lu005s1Gt7	0	16	8	18	0	52	8	90
lu005s1Gt8	0	19	7	19	0	59	0	90
lu005s3Gt3	0	4	12	6	0	22	55	90
lu006s1Gt8	1	22	0	4	1	74	0	93
Vulnerable sandy soils								
lu101s1Gt3	59	0	0	0	9	0	24	93
lu101s1Gt5	59	0	0	0	9	0	23	90
lu101s1Gt6	70	0	1	1	12	0	14	92
lu101s1Gt7	78	0	1	1	13	0	4	92
lu101s1Gt8	81	0	1	1	13	0	0	92
lu102s1Gt6	28	0	11	0	18	1	33	92
lu102s1Gt7	36	0	18	0	31	1	13	93
lu102s1Gt8	41	0	21	0	35	1	0	93
lu103s1Gt5	8	0	29	1	3	3	54	92
lu103s1Gt6	14	0	42	2	5	4	35	93
lu103s1Gt7	19	0	56	3	7	5	13	94
lu103s1Gt8	22	0	64	2	9	6	0	95

lu001 = grassland, lu002 = arable land, lu003 = maize land, lu004 = nature, lu005 = urban, lu006 = recreation; lu101 = grassland, lu102 = arable land, lu103 = maize land at vulnerable sandy soils; s1 = sandy soil, s3 = peat soil

fertilizer while *M* indicates the nitrogen application by manure (see Appendix 1). The uncertainty of the parameter determining maximum leaching percentage for a specific combination of soil (ζ) was different for grassland (10-40% on sandy soils), arable land (35-100% on sandy soils) and maize land (30-70% on sandy soil). This uncertainty range fits within the range reported by De Vries et al. (2003).

For the calculations on non-agricultural land uses, the model parameters leaching fraction in the winter (u_W) and organic matter content (*S*) were important, like *D*. u_W explained 20 to 74% of the variation in nitrate leaching, *S* explained up to 36%. u_W accounted for an uncertainty of 20-40% for all non-agricultural land uses. The uncertainty of the winter leaching explained up to 74% of the variance because the winter leaching comprised a substantial amount of the nitrogen budget. The uncertainty of *S* contributed only to the variance in nitrate leaching when the mineralization of organic material contributed substantially to the nitrogen balance. This was only the case for nature and urban land uses. Basic leaching (*B*) for nature and urban resulted in a TMV of 12 – 18%, despite the mean value (μ) of 0 kg N ha^{-1} yr^{-1} with a range of 0-10 and 0-5 kg N ha^{-1} yr^{-1} for nature and urban, respectively. The impact of the uncertainty of this particular model parameter on the total uncertainty was noticed because the absolute range was substantial with regard to the yearly N-input of nature and urban land use, i.e., 31.25 (20.31+10.94) and 36 (18+18) kg N ha^{-1} yr^{-1}, respectively (see Appendix 2, table 1). The uncertainty of *B* was not substantial for the N-balance of recreation land use and explained just a small part of the variance.

The additional reduction in the application of manure in 'vulnerable sandy soils' did not affect the outcome of the sensitivity analysis. The TMV values are almost identical to the values of the corresponding combinations of land use, soil type and groundwater table depth (see Table 2). Slight differences in the TMV between these combinations were caused by a slight shift in the N-balance and not by differences in the uncertainty of the dominant parameters. No shifts in uncertainty between factors occurred towards systematic changes in fertilizer and manure inputs of the scenarios. This indicates that the NSS is pretty robust towards these changes.

Uncertainties with regard to nitrate concentration in abstracted groundwater

The average fraction of groundwater abstracted from the third anaerobic layer was 0.584 and the standard deviation was 0.090. The spatially weighted average nitrate concentration that leached to the groundwater was 40.6 NO_3 L^{-1} and the standard deviation was 10.8 mg NO_3 L^{-1}. Therefore, the nitrate concentration in abstracted groundwater can be estimated by combining both terms: 40.6 x (1-0.584) = 16.9 mg NO_3 L^{-1}. The associated standard deviation is 7.3 mg NO_3 L^{-1}, i.e., 43%, using equation 2.

Discussion

Ideally, physically based transport models can be applied to model reactive groundwater transport and all parameters can be obtained independently in the field. This can be achieved for cases that are rather homogeneous in their spatial characteristics and have a temporally constant input and where a limited number of processes are operational. In reality, such an application and acquisition can rarely be achieved when models are used for management support at a regional or larger scale. In the approach we used, modeling was carried out based upon the best-available information and PDFs were assessed using a subjective method (Brusseau, 1991; Refsgaard et al., 2005; Thorsen et al., 2001).

The sensitivity analysis was performed to test the applicability of the NSS in groundwater planning. This was addressed using the following questions: (1) can the NSS be used as vehicle with which to explore alternative land use scenarios and providing quantitative information on the impact of the proposed land use change and the change in N-application on the quality of shallow groundwater? and (2) what does the sensitivity of the individual parameters mean for the utilization of the NSS in groundwater planning?

Usefulness to establish differences between land use and set up of scenarios

Shallow groundwater
The differences in nitrate leaching between the various land uses were significant – the confidence intervals of agricultural land use and nature did not overlap at all within a given soil type and groundwater level classification (see Table 1). The largest relative uncertainties for all combinations of land use, soil type, and groundwater level classification were calculated for grassland and nature. These relative uncertainties increased when denitrification increased, but were not affected by a reduction of the N-application as was the case in areas with vulnerable sandy soils. Despite the sometimes large absolute uncertainties for, for example, grassland and arable land, there was no overlap in confidence intervals of the nitrate concentrations between agricultural and nature land use functions within a given soil type and groundwater level classification. Thus, changes in land use function will also lead to distinctive changes in nitrate concentrations. In view of this significant distinction, there is no need for a further reduction of the overall uncertainty of the output by the minimization of the uncertainty of the input parameters. The additional effort will not contribute to the improvement of the prediction or provide more insight into the differences between the scenarios. The NSS is, therefore, considered sensitive enough to reveal the effects of conflicting land uses and is, thus, useful in the planning process by formulating land use scenarios.

Abstracted groundwater
The uncertainty of 43% in the nitrate concentration in the abstracted groundwater explained with the 2005 data set strongly resembles the uncertainty of 45% calculated for the period 1960–1998 (Van den Brink et al., 2008). The latter uncertainty was derived by comparing the nitrate concentration data in abstracted groundwater with the nitrate concentrations calculated with the NSS. The NSS calculations have not been fitted with abstraction data or data on the shallow groundwater (Van den Brink et al., 2008). Both uncertainties are not, however, directly comparable because one refers to a random year (2005) and the other to a period of years (1960–1998). The groundwater quality data on the abstracted groundwater and data on the pumping hours contain no additional information with which to further reduce this uncertainty. So, calibration is not very useful, in addition to the fact that the regional modeling concerns an underconstrained problem (c.f. Van der Grift and Griffioen, 2008).

Usefulness in groundwater planning
The PDF of the input parameters was based on both literature empirical data. The uncertainties of the input parameters do not limit the use of the NSS in the Holten study. A further reduction of the uncertainty of input parameters demands either a larger literature review or new experiments. The uncertainty of the input parameters can possibly be decreased. This particularly concerns *the available nitrogen in the growing season Z*, the most sensitive parameter for the land use types showing the highest N-application. The uncertainty of this parameter can, in principle, be reduced through measurements like setting up a nutrient balance per parcel or agricultural farm. This approach was used by the water company in our study area. The results however show that the N-application on the parcels of participating farms seemed inconsistent keeping in mind crop production and regional agricultural practices (Vitens, unpublished data). Based on this, the water company concluded that regional agricultural statistics form the most trustworthy source of information to estimate the N-application at a regional scale.

Because a stochastic uncertainty and sensitivity analysis does not provide any information on the tenability of the structure of the conceptual model, we evaluated the tenability using a pedigree analysis (Refsgaard et al., 2006). Refsgaard et al. (2006) mentioned aspects that need to be scored in this analysis. They are (1) supporting empirical evidence, (2) theoretical understanding and representation of the understood underlying mechanisms, (3) plausibility, and (4) colleague consensus. First, the NSS showed a good fit for the modeled concentrations based on a large series of direct measurements, even without any calibration. These measurements, however, only reflected the quality of the abstracted groundwater, i.e., the output side of the modeling process. Second, the process descriptions were founded on a well-established theory using model equations to reflect process details and empirical equations based on basic literature and experiments. Third, the process descriptions, therefore, can be considered reasonably plausible and, last, acceptable to all but a few colleagues. All four aspects

thus got positive scores. In addition to the results of the stochastic uncertainty analysis, the pedigree analysis confirmed the conclusion that the NSS is a useful instrument in the context of groundwater planning.

Conclusions

The analysis of the uncertainties of the nitrate leaching from various land uses showed that the Negotiation Support System (NSS) can significantly distinguish between land use functions for our case study. In view of this significant distinction, there is no need for a further reduction of the overall uncertainty of the output by additional data collection on input parameters to make sound science based decisions.
The sensitivity analysis showed that the NSS was most sensitive for almost all included input parameters. However, the NSS is considered robust because no shifts in uncertainty between factors occurred between factors towards systematic changes in fertilizer and manure inputs of the scenarios.
A pedigree analysis confirms that the NSS is a suitable diagnostic and prognostic tool for land use planning and protection of a groundwater abstraction sites.

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Chapter 5

Cooperative modeling: Linking science, communication and groundwater planning

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The cooperative modeling process (cartoon: Derk de Vries).

Issue Paper

Cooperative modeling: Linking science, communication and groundwater planning

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Abstract

Equitable allocation of groundwater resources is a growing challenge due both to the increasing demand for water and the changing values placed on its use. While scientists can contribute to a technically defensible basis for water resource planning, this framework must be cast in a broader societal and environmental context. Given the complexity and often contentious nature of resource allocation, success requires a process for inclusive and transparent sharing of ideas complimented by tools to structure, quantify, and visualize the collective understanding and data, providing an informed basis of dialogue, exploration and decision-making. Ideally, a process that promotes shared learning leading to cooperative and adaptive planning decisions. While variously named, mediated modeling, group modeling, cooperative modeling, shared vision planning, or computer-mediated collaborative decision-making are similar approaches aimed at meeting these objectives. In this paper we frame 'cooperative modeling' in the context of groundwater planning and illustrate the process with two brief examples.

Introduction

The solution of environmental problems is no longer seen as the exclusive responsibility of government; rather, quasi-governmental organizations, the business community, civic organizations, and citizens are demanding a greater voice in resource planning (e.g., Driessen and Glasbergen, 2002). Emergent forms of direct participation (e.g., interactive governance) by interested citizens and other stakeholders in processes of collective decision-making are reducing the distance between government and their citizenry. The value of collaboration, whereby various stakeholders work with policy-makers to address a particular issue has been well documented (Connick and Innes 2003; Spash, 2001; Claussen, 2001; Susskind et al., 2001). Such collaboration has increasingly included model building to inform the decision process. In fact, the use of models in open decision-making processes is not a new concept as there are case studies dating back to at least 1961 (Rouwette et al., 2002). While variously named, mediated modeling (van den Belt, 2004), group modeling, cooperative modeling (Tidwell et al., 2004), shared vision planning (Palmer et al., 1999), computer-mediated collaborative decision-making (Kreamer and King, 1988), the basic idea remains the same–it is a process for involving stakeholders in the conceptualization, specification and synthesis of knowledge and experience into useable information (i.e., model) for the express purpose of addressing a complex problem.

Advantages reported in support of cooperative modeling include that it provides participants with system insight, scoping analyses and education toward a common understanding of the issues, thus improving the odds that results from the collaborative effort will be implemented in decision-making (Palmer et al., 1993; Vennix, 1996; Rouwette et al., 2002; Van den Brink et al., 2003; Van den Belt, 2004). Participant values and preferences become better defined, while knowledge levels and consensus on mitigating approaches improve (Costanza and Ruth, 1998; Rouwette et al., 2002; Van den Belt, 2004). Resulting synergies enable the group to develop new policy options in deadlocked processes (Van Eeten et al., 2002). Finally, modeling conducted in a cooperative context provides value by way of structuring group thinking and dialogue, ultimately leading to group learning (Van den Belt, 2004).

Criticisms have also been levied against public participation processes. One argument cites the opportunity for special interest groups to gain disproportionate influence over the decision process (Coglianese, 1999; Kenney, 2000). Under representation by disenfranchised groups can lead to inequities in water allocation (World Bank, 2003). Lord (1986) and Ingram and Schneider (1998) recognize that value disagreements are often masked by factual disputes in public debate. The factual disputes then become the focus of the process and the situation of adversarial science or 'dueling experts' arises. Cooperative modeling efforts may also encourage unrealistic expectations; that is no level of modeling can make up for lack of understanding of basic principles–an inadequate model will never provide a sound basis for decision-making or course of action. Finally, collaborative processes are blamed for taking longer and costing more to reach a decision, although this point is questionable if the issue results in litigation.

Prior to engaging in any cooperative modeling process careful attention must be paid to the decision landscape. Toward this end, our objective is to identify key considerations in the design and implementation of a cooperative modeling project, specifically framed in the context of groundwater planning. Given the breadth of this subject our intent is simply to raise awareness, while steering the interested reader to the literature for details on 'how to'.

We begin by discussing the problem domain of groundwater planning; particularly, the unique characteristics and challenges it poses for cooperative modeling. We then define and discus three key considerations in the design of a cooperative modeling project: public participation, modeling, and their fusion. Finally, we provide a couple of brief case studies that highlight these design points.

Groundwater planning

Although not entirely unique, there are several features inherent to groundwater systems that present real challenges for resource planning. Uncertainty in the quantitative characteristics of the groundwater system and potable resource in storage is a leading source of contention. Uncertainty stems from the general difficulty in measuring broad distributed characteristics of the system such as basin bathymetry, aerial recharge, river/aquifer interaction and evapotranspiration losses. Additionally, our view of the groundwater system is limited; that is, boreholes, which only sample a small fraction of the entire aquifer, are often our only portals to measuring key petrophysical and water quality characteristics. Further complicating matters is the spatial and temporal heterogeneity inherent to the physical characteristics of the groundwater system as well as in land use function and management. While aquifer characterization technology is rapidly improving, costs to adequately resolve key measures within the context of resource planning are generally prohibitive; particularly, at levels capable of illustrating the role of individual stakeholders within the problem and the interrelation between parties. Ultimately, the danger is uncertainty leading to dueling models and polarization of views by competing interests.

Another challenge relates to the 'mysterious' nature of groundwater systems. The poorly informed my come with visions of underground rivers and lakes, as well as ideas of 'infinite groundwater supplies'. While the better informed may hold more physically based mental models, their conceptualizations are often biased toward a particular interest. Thus, the challenge is to help all participants to learn toward a shared view of the groundwater system, which in turn is scientifically defensible.

Not only are groundwater systems complex in their own right, but the dynamics of these systems are inextricably linked to many other physical and social systems. Resource planning requires an analysis adjusted not only to the nature of the groundwater issues faced, but must also consider social, economic and legal constraints (Alley and Leake, 2004). In this broader context, decisions transcend direct technical arguments to include value judgements. Thus, equitable resource planning defies myopic, piecemeal approaches, but rather requires the fusion of knowledge and experience distributed across disciplinary and constituency domains.

Cooperative modeling

Before we tackle structuring of a cooperative modeling project, we first need to consider when such an approach is appropriate. Although not a comprehensive list, we offer four considerations toward this decision. First, there must be willingness on the part of key interest groups to cooperate in the planning process. In situations involving entrenched, high-intensity conflict, the stakes are often described in narrow terms, thus leaving little room for dialogue and compromise (Van den Belt, 2004). Second, each interest group needs equal access to the planning process; that is, participation cannot be unequally constrained by proximity, financial resources, or the like. Third, the entity responsible for conducting the cooperative modeling process must be trusted by all interests. The entity must be viewed as an unbiased facilitator working toward the collective benefit of the region. Finally, the availability of financial resources must be factored into the decision.

Public Participation

In a perfect world all interested parties would be invited to participate in a cooperative modeling process; however, this is rarely practical. The challenge arises because the cooperative modeling process strives for broad participation, while meeting efficiency and productivity is best achieved in groups limited to 10-30 participants (Van den Belt, 2004). Fortunately, participation can be balanced with efficiency through careful structuring of the public participation process.

'Circles of influence' (IWR, 2002) is one such approach that organizes participation based on level of interest and ability to engage. The approach can be envisioned as a series of concentric circles with the innermost circle encompassing the 'support team'. By support team we mean the personnel responsible for physically constructing the model, process facilitation, and project management. The next larger circle includes the most interested stakeholders who through a combination of time, energy, and aptitude are involved in the technical work. This 'cooperative modeling team' directly engages in the development and/or application of the model and thus develops a detailed knowledge of the technical process. Ideally each participant acts as the liaison for a broader interest group. A third outer circle is drawn to encompass all other stakeholders who desire only a peripheral role in the process. The fourth and final circle contains the political decision-makers that provide direction to the effort. This circle overlays the three concentric circles with information exchange and dialogue occurring directly with each.

Here, an interest group or individual stakeholder is anyone with a stake in the decision. In this context an interest group might include government organizations, public service organizations, non-government organizations, or business organizations, while an individual stakeholder might include representatives of a private business, technical subject experts, or a concerned citizen. Involvement in the outer third circle is simply a self selection process as such participation is open to all. Participation in the cooperative modeling team generally occurs by a hybrid self-selection and appointment process. Ideally, each interest group would elect a representative, who in turn is willing and able to participate. The guiding principle in participant selection is to keep the process as open as possible while working to keep the representation of the individual interests balanced.

Beyond the cast of stakeholders and interest groups is the support team. This is the group that prepares the meetings, guides the discussion, synthesizes group thinking, and simultaneously translates the results into a model. This involves the four distinct roles of facilitator, modeler, process coach and gate keeper (Richardson and Andersen, 1995). The gatekeeper is the project champion who is the promoter and energy behind the project. Although an individual can fulfill more than one role, it is important to recognize the need for each and deliberately assign responsibility to skilled participants.

It is important to establish ground rules at the beginning of the cooperative modeling process to provide a structure for group dynamics (Wondolleck and Yaffee, 2000). Participants must agree on rights and responsibilities; particularly, attendance and participation expectations. Behavior guidelines should be set. Clear rules must also be set on how group decisions will be handled—by consensus, majority vote, or some hybrid. Such guidelines are necessary before contentious decisions are encountered. It is also important to define expectations for the modeling exercise; specifically, the purpose and role the model will play in any future decision process (e.g., Tidwell et al., 2004).

Modeling

Collaboration is pursued with the goal of developing broad consensus within a decision context; however, consensus in the absence of scientifically sound understanding is of limited value. Modeling plays a central role in establishing a sound and informed basis for decision-making and provides a vehicle for communicating technical results.

In cooperative modeling, it is modeling that connects the decision process with science. Specifically, modeling forces participants to confront facts. 'Science has developed specific methods and peer review processes to maintain as objective a view as possible, but in other environments this conscious effort is often lacking' (Van den Belt, 2004). The process of jointly developing and exercising a model also helps the group to better appreciate system complexities and understand its cause and effect relations (Vennix et al., 1997). Additionally, models provide a uniform basis for comparing competing alternatives and assessing tradeoffs. The aim of cooperative modeling is not to provide a scientific evaluation of a groundwater issue, but to identify the best course of action to resolve stakeholder conflict (van den Brink et al., 2008).

Models can be used in a variety of modes within the decision process. At one end of the spectrum models are used only in an illustrative capacity to help participants visualize key aspects of the system. At the opposite end, models are used to quantitatively distinguish between competing management scenarios. Other roles might include system insight, scoping analyses, education, and outreach.

Selecting an appropriate model for a cooperative modeling project is really no different than that encountered with other modeling exercises. Careful attention must be paid to matching model fidelity and resolution with model purpose, available data, and modeling resources (e.g., Bear et al., 1992; Anderson and Woessner, 1992). What is different is consideration of the means by which project participants interact with the model. Participants need an environment in which they can query the model and receive rapid feedback to promote dialogue and shared understanding. Participant interaction can be indirect where experts are responsible for manipulating and running planning scenarios requested by the team. Alternatively, the modeling framework can be designed for direct interaction where participants are able to set-up and run planning scenarios themselves.

To achieve this level of interactivity generally requires use of more than just a model; that is more than a mathematical representation of a real-word system. Rather, the model needs to be accompanied by tools to help stakeholders access information and improve understanding. In particular, a model interface is required, which is the portal by which the expert or participant structures model input to simulate future desired conditions. An interface is also needed to assist with the presentation and visualization of model results. Other useful tools include database management functions and decision analysis support. Model interactivity is also enhanced where model results can be accessed rapidly. This can be accomplished in a couple of ways. First, application of models with short (tens of seconds) computation times. Second, planning scenarios can be simulated prior to a public meeting and stored in a database coupled with a user interface.

Decision support systems (DSS), provide a valuable modeling framework with which to enhance model interactivity. While defined in many different ways, a DSS can be taken as 'a computer-based system that aids the process of decision-making' (Finlay, 1994). Basic DSS components often include a user interface, database, simulation and analysis tools, and the overarching system network architecture (Power, 1997). The required level of DSS complexity simply depends on the functionality required to connect stakeholder and science in the context of the particular planning issue.

Negotiation support systems (NSS) and planning support systems (PSS) are types of DSS with application to a particular problem. The spectrum of NSSs include (Starke and Rangaswamy, 1999) (1) expert systems that use accumulated knowledge to aid stakeholders preparing for negotiations, (2) a system that combines technologies for individual decision support (DSS) and group decision support (GDSS) and facilitates the actual negotiation process in multiagent settings (Nandalal and Simonovic, 2003), and (3) 'autonomous agents' that are programmed to negotiate on behalf of their human principals. The real advantage of NSSs is that they aid negotiators in overcoming their cognitive limitations and in identifying

their (and others') real interests rather than focusing on negotiating positions (see also Fisher and Ury, 1983). A PSS generates information that can be communicated to and among the stakeholders and used in the planning process (Geertman and Stillwell, 2003).

System dynamics provides yet another platform for cooperative modeling (Van den Belt, 2004; Winz and Brierley, 2007). The process by which system dynamics models are conceptualized and quantified lends itself well to cooperative modeling (e.g., Sterman, 2000). The value of system dynamics is in capturing the feedback and time delay between multiple interacting systems, which is key to effective resource planning. Additionally, available commercial software packages provide an integrated software environment, including object-oriented modeling, database connection, user interface development, and decision analysis tools.

Fusion

Above we considered public participation and resource modeling within the context of cooperative modeling. Now we turn our attention to the fusion of these key elements; specifically, we consider when and how to merge public participation with modeling.

Transparency and trust are key objectives of cooperative modeling. Building a model from the ground up improves transparency and trust in the resulting tool. Engaging stakeholders in the actual development of the model helps improve their understanding of the system, data sources, key assumptions and limitations. Alternatively, the modelers benefit from the experiential knowledge of the stakeholders. The broader perspective afforded by cooperation also benefits the overall modeling and decision process. However, it is not always necessary for stakeholders to be engaged in the model development stage. Where tools exist that are widely trusted, stakeholder involvement can be limited to model application and analysis exercises.

Stakeholder participation primarily occurs through attendance of group meetings. As a very rough rule of thumb, a cooperative modeling project will take 60-80 hours of group meeting time (Van den Belt, 2004); however this figure can vary significantly depending on the complexity of the issue. These meetings are used to collaboratively define the purpose, scale and scope of the model; establish decision variables and metrics; conceptualize and structure system cause and effect relations; quantify key processes; and, specify model data and parameter values. 'Coding' of the model can be performed in the meetings (as is often done with system dynamics modeling) or performed between meetings by members of the support team. Meeting time should also be scheduled for calibration, testing, and application of the model. Frequency and duration of the meetings will depend largely on the level of participant engagement and their schedules. Cooperative modeling exercises can be pursued in a compact manner where stakeholders devote several consecutive days, or meetings can be dispersed in time meeting monthly or bimonthly for several hours at a time.

Finally, in structuring a cooperative modeling process one should carefully consider the nature of the problem at hand and the underlying source of tension. There are three general groupings, the choice of which will influence the fusion of the participant, modeling, and decision process:

- Complexity of the policy field: policy fields are characterized by a complex web of actors serving their interest in a 'multi-actor governance', 'multi-level governance' and 'multi-sector governance' environment. In such a context, cooperative modeling is helpful in surfacing relevant and adequate information and putting it into the hands of stakeholders and decision-makers (Sehlke and Jacobson, 2005);
- Complexity of the physical system: physical systems that have cause and effect separated by time and or space, i.e., dynamic complexity (Senge, 1994). For such systems cooperative modeling can improve awareness and understanding of the problem, assist in rephrasing of the problem such that it illustrates the unique role of each stakeholder within the problem, and illustrate the interrelation between concerned parties and their shared fate in the dilemma;
- Uncertainty in the process: the course and outcome of a (negotiation) process should leave enough space to identify the best course of action rather than to communicate the solution of a major actor.

It is also often advantageous to structure the cooperative modeling process toward investigating future trends in key system behavior rather than attempting to predict the exact future state or scientific evaluation of a system (Nandalal and Simonovic, 2003; Van den Brink et al., 2008).

Case studies

There are several examples of cooperative modeling in the context of environmental resource planning, including assessing the effects of sheep grazing on sage grouse populations (Van den Belt, 2004); energy use in iron and steel production (Costanza and Ruth, 1998); air quality issues (Stave, 2002); sustainability of Arctic communities (Nicolson et al., 2002); park management (Videira et al., 2003); water planning (Moxey and White, 1998; Van Eeten et al., 2002), and groundwater management (McPhee et al., 2004). Here, we offer two additional case studies that address issues important to groundwater planning and that illustrate several of the cooperative modeling design aspects noted above. The first example addresses coupled land use-groundwater quality management for a community in the Netherlands, the second example considers sustainable water planning for a three-county region in north central New Mexico, U.S.A. While these examples are similar in their use of cooperative modeling, they differ in scope, geography, modeling platforms and the collaborative framework.

Groundwater Quality Management

The Holten region is located in the eastern part of the Netherlands in the province of Overijssel. The region enjoys a rural setting with a balance of small municipalities, farms/dairies, and open public lands. The local water company in cooperation with the provincial authority (responsible for groundwater quality and with a clear mandate to resolve the conflict) initiated a local planning process in response to mounting concerns over nitrate concentration in abstracted and shallow groundwater. Drivers for this local process were requirements of the EC Nitrate Directive (EC, 1991). In addition to this local planning process the province is formulating a regional plan in response to the Dutch legislation for restoration of areas impacted by intensive livestock farming (Reconstruction Law). The purpose of this legislation is to reach a new balance between various social, economic and ecologic functions in rural areas. To reach a final agreement, assistance of procedures or methodologies acceptable to all concerned parties is very much needed (Nandalal and Simonovic, 2003).

In support of these planning processes, Royal Haskoning and TNO Geological Survey of the Netherlands (i.e., the support team) were requested to develop a negotiation support system (NSS) to assist in solving disagreements among stakeholders. The NSS was used to model and evaluate the interrelationships of different components and activities within the groundwater-land use system and can be considered an object oriented modeling approach (Nandalal and Simonovic, 2003). Purposes of the NSS were to identify the best course of action to resolve the stakeholder-accepted conflict between land use management and groundwater quality; rephrase the problem such that it illustrates the unique role of each stakeholder and the interrelation between concerned parties; and explain the complexity in the groundwater system in terms of recharge area and residence times.

In order to build acceptance and confidence in the NSS, parties including water boards, provincial authorities and water companies (i.e., the cooperative modeling team), participated in a series of 2 daylong workshops to define the requirements and select the modeling tools of the NSS. Upon its completion, a spatial bottom-up planning process was organized at the local level. This involved the water company, provincial government and local stakeholders (i.e., all other and possibly conflicting stakeholders). Several meetings were held over a 9 month period of time to explore alternative land use scenarios aimed at reducing nitrate loading to the groundwater system. A broader set of agricultural representatives, recreational entrepreneurs, and civil servants of the municipalities became part of a regional planning process.

Several land use scenarios were considered, of which the 'base' and the 'adapted' land use were developed to identify the best course of action to resolve the stakeholder-accepted conflict. The base scenario calculates the autonomous development assuming no changes in the land use pattern. The adapted land use scenario is identical to the base scenario until 2005 at which time the adapted land use scenario was assumed to be implemented. The adapted land use scenario was defined by the water company and the provincial government based on: (1) the calculated quality of the shallow groundwater in 2005 obtained from the base scenario and (2) the residence times of the groundwater toward the abstraction site calculated with the calibrated geohydrological flow model. Experiential knowledge of the stakeholders was used to set parameter values reflecting historical, current and future nutrient application for the three agricultural land uses (grassland, maize, and arable land).

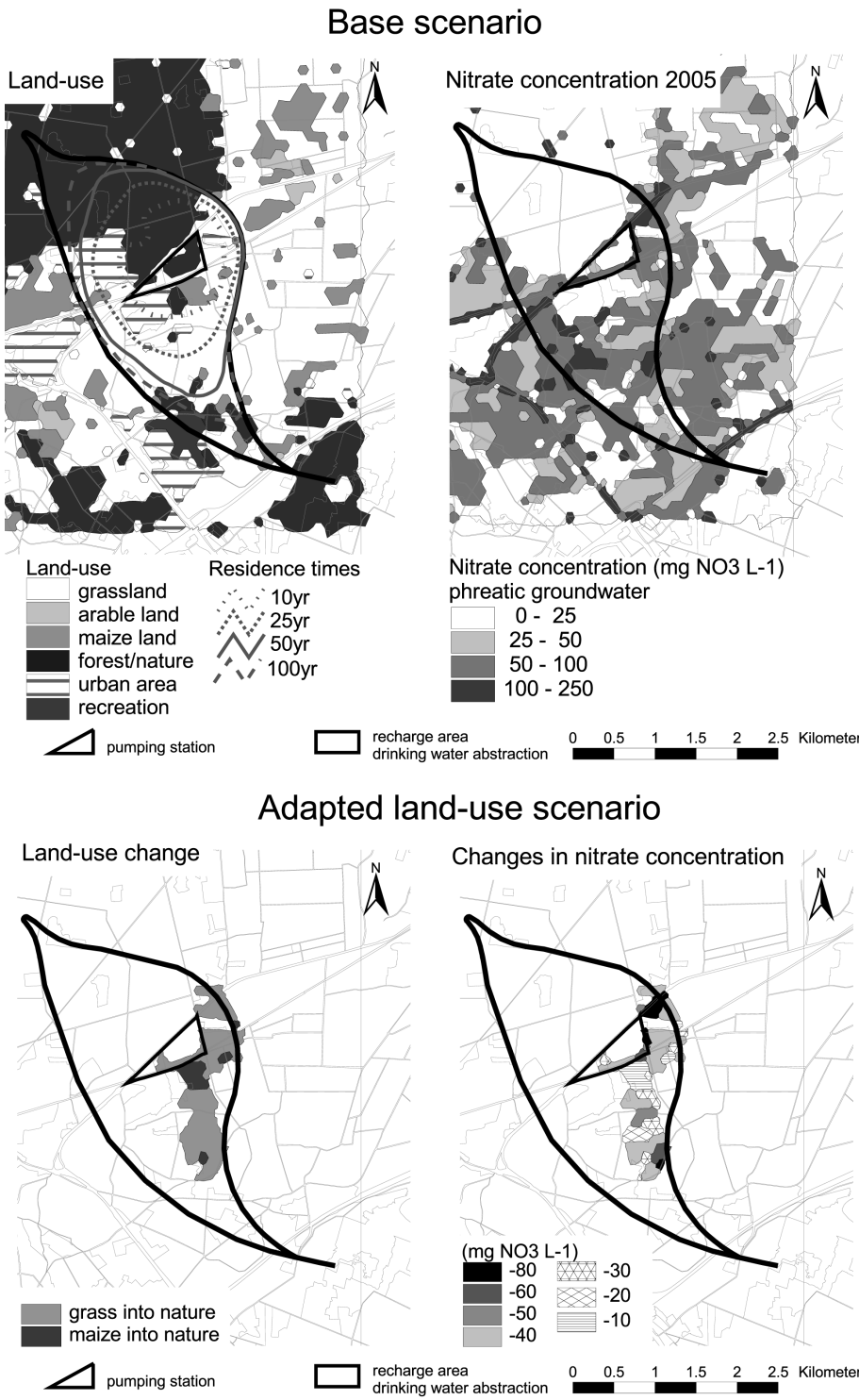

Figure 1. Land use, residence times, and calculated nitrate concentrations (mg NO_3 L^{-1}) in shallow groundwater in the base scenario (2005) and the changes in land use and calculated nitrate concentration in the adapted land use scenario.

The focus of the local planning process was the protection of drinking water through modification of land use practices. The NSS supported this process by providing a vehicle with which to explore alternative land use scenarios and providing quantitative information on the impact of the proposed land use change and the change in N-application on the quality of abstracted groundwater. The output generated by the NSS was organized in the form of maps and time series as requested by the stakeholders. This information visualized and quantified the conflicting interests between agricultural land use and drinking-water function, indicating that sustainable protection of the resource would require a substantial change in land use. The changes in nitrate concentration due to the adapted land use scenario are significant as they are realized in an area with rather short residence times (Figure 1). Maps of shallow groundwater quality contributed not only to the quantification of the land use conflict, but also helped to overcome the difference in planning horizon of the stakeholders and the time scale of the geohydrological system. Based on this insight, the stakeholders realized their own role as well as their combined role with other stakeholders in these conflicts. As a result, the need for a substantial change in the actual land use pattern to protect the abstraction site became a shared notion among the stakeholders. In this way information from the NSS was used in the bottom-up spatial planning process as the technical basis for management decision-making (c.f. Simonovic, 2002).

While formalization of the adapted land use scenario was the end point of the local planning process, issues still remained. Proposed land use changes were largely focused on agricultural practices; but, the local planning process did not have the means to compensate the farmers. To resolve this issue attention was directed at the regional planning process, which provided a compensation mechanism for farmers located outside the drinking-water abstraction recharge area. Although the water company and the province were leading actors in the local planning process, they were just a part of a larger group (including socio-economic actors) in the regional Reconstruction process. This process included political decision-makers. By building on experience gained in the local planning process, the water company and the province were able to have a strong voice in regional planning. Specifically, insight into the relation between land use, groundwater residence times and groundwater quality as provided by the NSS helped push the local planning process scenarios to the forefront of the regional planning process. Cooperative modeling proved helpful for integrating complex hydrologic data with other information (e.g., policy, regulatory, and management criteria) in defining a course of action to resolve stakeholder conflict (c.f., Sehlke and Jacobson, 2005). A complete description of this case study can be found in Van den Brink et al. (2008).

Regional Water Planning

The New Mexico Office of the State Engineer (NMOSE) initiated a statewide water planning process in the mid-1990s in response to mounting concerns over water issues in the state. Sixteen planning regions were formed, each with the responsibility of preparing a 50-year water management plan that balances sustainable water supply with projected demand in a publically and politically acceptable manner. One of the sixteen regions, the Middle Rio Grande (MRG), adopted a cooperative modeling process to support their water planning efforts (Tidwell et al., 2004). This region encompassed three counties in arid north-central New Mexico that are centered on the Rio Grande and include Albuquerque, the principal urban center in the state.

Public participation within the cooperative modeling process was largely structured around the 'circles of influence' approach discussed above. In the inner circle was the support team which consisted of two modelers from Sandia National Laboratories and a professional facilitator provided by the Utton Transboundary Resources Center of the University of New Mexico. In the next circle were members of the cooperative modeling team (CMT). Team members were appointed by the Mid Region Council of Governments (MRCOG), the local government association with oversight responsibility for the planning process and the Middle Rio Grande Water Assembly (MRGWA), the volunteer organization that spearheaded the planning. Team members were selected according to their willingness to participate and the need to balance the interests of local government, irrigators, environmentalists, urban developers, and regional water managers. The third circle of participants was made up of other MRGWA and MRCOG members not on the cooperative modeling team, the interested public, and local/state/federal water agency representatives.

The support team in close collaboration with the CMT worked to develop a decision support model to aid the water planning process. The model was developed from the ground up and took approximately 18 months to complete. During this period of time the support and cooperative modeling teams met on a bi-weekly basis for two to three hours per meeting. During these meetings team members helped conceptualize the physical system, identify data sources, quantify causal relations, design the interface, and review the model. Coding and calibration of the model was largely performed outside the meetings by the support team modelers. In this way the cooperative modeling team was in full control of the design and implementation of the model and had full knowledge of the limitations, assumptions and uncertainties inherent to the model and associated data. The outer circle of participants also assisted with model development. On a quarterly basis public meeting were held to discuss progress on the model and to solicit feedback. Additionally, numerous meetings were held with local, state, and federal water agency personnel to benefit from their expertise, data, and review of the process.

The decision support model was tasked to quantitatively evaluate alternative water conservation strategies, engage the public in the planning process, and explain complexities of the regional water system. Additionally, the model was designed to be fully accessible to the public; that is, operate on a PC without the aid of an expert modeler. Toward these needs, a decision-support model was created within a system dynamics framework using the object oriented modeling package Powersim Studio 2003. Basic model elements include surface water and groundwater supplies balanced against municipal, agricultural, evaporative, and riparian demand. Also included in the model are 24 water conservation strategies that the public identified through community meetings held as part of the planning effort. The model is accessible through a user friendly interface that spans approximately 80 computerized pages including pictures, explanatory text, 66 slider bars and buttons for exploring alternative water conservation strategies, and graphs and tables for visualizing simulation results. Simulations take less than 10 seconds to complete.

In spring 2003, a working model was delivered to the MRGWA and used to develop scenarios that integrate various combinations and intensities of the 24 conservation alternatives. Ultimately, a series of five 'scenarios' intended to represent perspectives from each of five constituency groups (agriculture, environment, urban, specialists, managers) were developed. These scenarios were then vetted with the public to gather their preferences and perspectives. Working closely with the MRCOG, the MRGWA used the decision-support model to combine the individual scenarios along with public feedback into a 'preferred scenario' which then became the basis for the regional water management plan. Once the public had commented, the plan was finalized and submitted. The NMOSE formally accepted the plan in August 2004. Currently, some articles of this plan are in the process of being implemented while others are being studied/discussed in more detail.

Post-project interviews were performed with the general public and CMT members to gauge perceptions on modeling, identify strengths and weaknesses of the cooperative modeling process, and determine if and how the model facilitated the planning process. Results overwhelmingly support using models in developing solutions to complex public policy issues and using models to educate and engage the public in such processes. CMT team members believed that it was appropriate to use the model in planning and that the planning process would have been more difficult without it. This is not to say the process did not face its share of challenges. Specific lessons learned include the need for expert facilitation in the collaborative process, transparent and clear communication throughout the process, and careful definition of the role of modeling in the planning process. Additional details can be found in Cockerill et al. (2004 and 2006).

Conclusions

Water touches our lives in many ways from basic physical needs, the food we eat, the economy, and the environment in which we live. Management of this tightly coupled system benefits from the sharing of knowledge and experience across disparate disciplines and constituencies. To assist in the sharing of information and ideas, models are needed to structure group dialogue and thinking toward shared learning. Models also help ground the process with a scientifically informed basis for decision-making. In this paper we present a general framework for cooperative modeling; specifically, a process for involving stakeholders in the conceptualization, specification and synthesis of knowledge and experience into useable information (i.e., model) for the express purpose of addressing a complex problem. This approach is not predicated on a single decision process or modeling platform; rather, it encompasses a continuum of operational modes that can be tailored to the unique needs of an individual problem.

To demonstrate this approach, two examples in the context of groundwater planning are given, which are similar in their use of cooperative modeling, yet differ in scope, geography, modeling platforms and the collaborative process. These examples show that beyond a model's technical contribution there is the added value of improved awareness and understanding of the problem, a rephrasing of the problem such that it illustrates the unique role of each stakeholder within the problem, and illustrates the interrelation between concerned parties and their shared fate in the dilemma. Cooperative modeling therefore provides a vehicle for identifying the best course of action to resolve groundwater conflict.

While cooperative modeling holds promise, it should not be assumed to be the solution to all environmental conflict. Where there is no willingness to cooperate, litigation may be the only alternative. Even where cooperative modeling approaches are employed, some level of continued conflict should be expected. In fact, some conflict is healthy as it suggests that the hard issues are being addressed and that competing views are being expressed and discussed. While cooperative modeling does not preclude conflict, it does provide a structured process for transparently and scientifically dealing with the conflict.

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Chapter 6

General discussion

General discussion

Introduction

This PhD study is focused on the relationship between land use and groundwater quality. Land use in the Netherlands is very intense and different types of land use occur within relatively short distances of each other. Land use effects on groundwater quality may result in conflicts between various land uses and between land use and groundwater functions such as drinking water abstractions (e.g. Lake et al., 2003). As a result, various stakeholders, policy fields and institutions are involved in groundwater planning.

For groundwater planning to be successful, stakeholders should be encouraged to identify and discuss ideas and planning alternatives. Such a stakeholder driven planning process needs to be supported by technical tools such as computer simulation models that quantify and visualize relationships between land use and groundwater quality. In this way the dialogue among the stakeholders and governmental authorities is facilitated, which can be the basis of common decision-making (Tidwell and Van den Brink, 2008). In this way, the models are helpful tools to structure the group dialogue in a quantitative manner by connecting the decision process with scientific information.

The central part of my PhD study regarded the development of a so-called Negotiation Support System (NSS). This is a computer simulation model that analyses different scenarios of multiple land uses with respect to their impact on groundwater quality. The NSS also shows how these impacts may impose conflicts between the various land use types. The NSS was designed to identify the best course of action to resolve a conflict between land use planning and a groundwater abstraction in Holten, Overijssel, the Netherlands. An important aspect of the NSS was that it was developed in a cooperative manner. Both the development and application of the NSS were organized in an actual planning process including active participation of a relatively broad group of stakeholders representing the various interests and policy fields. In addition to the analyses as presented in the individual chapters, I here will discuss the results of the cooperative modeling approach from the perspective of 'adaptive governance' (e.g. Bruntland, 1987; Dorland and Jansen, 2006; Driessen and Leroy, 2007, Folke et al., 2005). This concept is especially useful to view the role of technical knowledge in social processes (Regeer and Bunders, 2007).

Adaptive governance of social-ecological systems

Adaptive governance considers both the social and ecological dimension of natural resource management to enable effective management of ecosystems (Carpenter and Folke, 2006). Berkes and Folke (1998) use the term 'social-ecological systems' instead of ecosystems to emphasize humans as inextricable part of nature. They stress that the delineation between social and ecological systems is artificial and arbitrary and that focusing only on the ecological side as a basis for sustainable decision-making may lead to narrow or incomplete conclusions (Folke et al., 2005). In the transformation from a non-sustainably to a sustainably governed social-ecological system four steps are distinguished (Olsson et al., 2004): (a) declining values are perceived by individuals; (b) the system is prepared for a change; (c) an opportunity for change – a 'window of opportunity' - is offered or created; and (d) improvement towards sustainability is achieved.

To summarize the role of scientific or technical information in adaptive governance, Regeer and Bunders (2007) introduce a conceptual model distinguishing three 'modes': 'mode-0', 'mode-1', and 'mode-2'. (Table 1).

Scientific knowledge development is considered an autonomous process in mode-0, while it is related to social problems in mode-1. Mode-2 or transdisciplinary knowledge development comprises an intensive cooperation between the scientific community and society. A specific aspect of mode-2 science is transdisciplinary: a form of learning and problem-solving in which various parts of society and scientists cooperate. This implies that the knowledge part is dependent on the social context: context-bound knowledge.

Table 1. Scientific knowledge development in mode-0, mode-1, and mode-2 (Regeer and Bunders, 2007).

	Relationship between knowledge and practice	Assumed role of scientific knowledge-(development)	Type of knowledge
Mode-0	Separate Science and practice are (almost) free entities (independent)	Autonomous More scientific knowledge leads to more progress	Mono-disciplinary knowledge Emphasis on exact (beta) sciences.
Mode-1	Cooperation Cooperation between science and practice. No change in the ways either 'side' works	Instrumental Development of policy-relevant knowledge and/or applied knowledge leads to solving social problems	Mono-, multi-, and interdisciplinary knowledge
Mode-2	Co-production Practice and science are both actively looking for the best manner in which to design and manage complex processes of change	Transdisciplinary Scientific knowledge (mono-, multi-, and interdisciplinary) plays a part in the common solubility process, which is part of scientific knowledge development	Mono-, multi-, and interdisciplinary knowledge Also knowledge from experience

Adaptive governance of groundwater systems

Groundwater is a very interesting, but also complicated case for adaptive governance, as groundwater is a physical entity for which effects of human interventions often occur long after the cause and also at different locations (Senge, 1994). Groundwater quality may in fact react so slowly to environmental changes and disturbances that a sense of urgency is missing there where it is necessary. Therefore, adaptive groundwater governance should aim at predicting and timely adapting human activities to the groundwater system rather than restore groundwater quality. In this section I will discuss the two studied cases of groundwater management and how the role of scientific knowledge in these cases fits in the different 'modes' of the model of Regeer and Bunders (2007).

Groundwater management in Halásztelek (mode-1)

The Hungarian government asked for an analysis of the extent to which the use of fertilizers in the upland area in the town Halásztelek, south of Budapest, would form a threat to the drinking-water supply. Therefore, we developed a model (FLUNIT) to quantify the effects of the use of fertilizers on the nitrate concentrations in abstracted groundwater. The model output provided quantitative insight in transport processes in groundwater and the cause-effect chain in groundwater quality (see chapter 1 of this thesis). The knowledge was developed in cooperation with scientists and policy makers from the Hungarian ministry responsible for groundwater management. Other actors – municipality, agriculture and water company – provided data on groundwater quality or agricultural practice, but did not participate in the modeling process. Knowledge of the relationship between land use and groundwater quality produced by this model was mainly diagnostic, it only assesses the cause-effect relationship.

The role of scientific knowledge and the way in which it was acquired has a number of mode-1 characteristics. The methodology was developed and applied by institutes and scientists in answer to a social issue. The presupposition of the modeling process was that the development of policy relevant knowledge would lead to the solving of the social issue. Solving the issue was, however, not a part of the technical process. Further, there was no (learning) process in which there was a collective search for a solution. Probably, therefore the results of the FLUNIT modeling have not been used in the actual groundwater planning process.

Groundwater planning in Holten (mode-2)
The groundwater conflict in Holten was caused by rising nitrate concentrations in abstracted and shallow groundwater which threatens a drinking water abstraction. The approach taken to solve this conflict has a number of adaptive groundwater governance characteristics. The local water company in cooperation with the provincial authority initiated a local planning process. To support this process with scientific information, the local water company and province requested a model that quantifies the relationship between land use and groundwater quality. This has become the Negotiation Support System as described in chapter 3 of this thesis. The NSS was set-up in the form of a cooperative modeling approach to link the physical aspects of the groundwater conflict to the social context. The cooperation with stakeholders offered insight in the requirements of the NSS from the perspective of the stakeholders. In this way, the cooperative modeling resulted in a mutual learning process in which alternative land use scenarios were defined to solve the conflict. This NSS-process resulted in a so-called 'adapted land use scenario' for the Holten area that was acceptable to the stakeholders. Then, a window of opportunity arose in the form of the regional plan in response to the Reconstruction Law (Driessen and De Gier, 2004). The national government created this law in response to the 1999 swine-flu epidemic. The adaptive land use scenario was suitable to be embedded in this regional plan. In this way, the acceptation of this scenario led to a transformation toward sustainable groundwater management. The farming crisis on pig disease rather than the occurrence of groundwater contamination itself offered the window of opportunity for the implementation of the adapted land use scenario.

The role of scientific information in the Holten case has typical mode-2 characteristics. The NSS was designed together with stakeholders based on: (1) the requirements regarding the leaching models, (2) the requirements regarding the type and format of information to be generated and (3) detailed information regarding the agricultural use of nutrients in the region. Therefore, this knowledge can be considered context-bound knowledge. The learning process benefited from four explicit choices made with respect to the role of knowledge in the decision-making process. These were: (1) the model – the NSS - was initially based on existing and available knowledge instead of the development of new knowledge, (2) thereafter, several periods of active knowledge input from the stakeholders were created during the process, instead only informing them, (3) information on the groundwater system was made understandable and accessible for all stakeholders and (4) the results were presented in a way that the various interests could be recognized.

Limitations of the Negotiation Support System
The way in which the results of the Negotiation Support System (NSS) contributed to the policy decision is directly related to the reliability and applicability of the information generated by the NSS. This reliability is in turn related to the reliability and availability of input data. Groundwater is, however, a 'hidden' resource for which data acquisition is difficult and expensive due to its inaccessibility. Upon result, little data is available for groundwater planning in specific areas. This forms an important limitation for the use of the NSS in groundwater planning. The modelling both in Halásztelek and Holten was hampered by limited data on (1) the spatial distribution of nitrate in soil and groundwater at the start of the calculations; (2) the vertical distribution of the aquifer layers and their geochemical reactivity and (3) groundwater quality data of the nitrate concentrations in shallow groundwater. It is likely that this limitation is general because environmental geochemical data is not widely available.

Another limitation is the applicability of the NSS for problems with multiple contaminants. The present NSS can be used to quantitatively describe the impact of any contaminant of which the leaching model is available. But the NSS cannot compare these impacts over areas of different sizes. For example, it cannot be used for answering questions like 'Is a twofold exceedance of the MTBE standard in 1 hectare worse than a 1.5-fold exceedance of the nitrate standard in 20 hectares?' The present NSS can neither be used to quantify the impact of land use functions on groundwater quality integrated for all contaminants and is thus unusable for the quantitative support of a balanced approach per land use or planning process. This would be an important improvement of the NSS. Despite these limitations however, the NSS did provide the necessary information regarding the impact of land use on groundwater quality. The stakeholders could then weigh this information together with other aspects – like costs and social impact – in their negotiation as part of the decision process.

Perspectives: groundwater planning at national and European scale

Various studies have shown that the impact of human activities on groundwater quality is still increasing at a national scale, despite all kinds of protection regulations and measures (e.g., Duenk et al., 2003). In 2003, the Netherlands Ministry of Housing, Spatial Planning and the Environment has recognized these facts and formulated a 'change of course' which was accepted by regional authorities and water companies (Duenk et al., 2003). This 'change of course' entailed of: (1) groundwater governance by means of environmental planning, i.e., protecting groundwater by changing land use functions into groundwater friendly land use functions, and (2) expanding groundwater friendly land use management within current land use functions. This change of course had been accepted by the stakeholders involved in groundwater protection, i.e., provinces, association of Netherlands municipalities and water companies. However, this did not lead to a transformation toward sustainable groundwater management (Netherlands Ministry of Housing, Spatial Planning and Environment, internal documents, 2006; VEWIN, 2006).

In 2006 the national groundwater protection policy was analyzed again. It was concluded that several national stakeholders lack necessary knowledge on the function of groundwater (Driessen and Van Rijswick, 2006; Van den Brink and Buitenkamp, 2007). This lack of knowledge concerns both the awareness of land use related deterioration of groundwater quality as such and the knowledge necessary to quantitatively describe the impact of the spatial developments on groundwater. Moreover, Wuijts et al. (2008) concluded that the policy instruments were adequate, but the lack of knowledge hampers to effectively meet the requirements of the European Water Framework Directive (EC, 2000) and the European Groundwater Daughter Directive (EC, 2006). Maybe, a window of opportunity seems to be missing at this national scale, which might be needed to enable the transition toward sustainable groundwater management. The implementation of the European Water Framework Directive (WFD) and the European Groundwater Daughter Directive (GDD), both of which demand certain results with regard to groundwater quality, might be a new impetus to a transition toward sustainable groundwater planning, as groundwater management in the Netherlands will increasingly be affected by the WFD and GDD and the White Paper on European Governance (EC, 2001). These directives demand that: (1) stakeholders are involved in the water resources management process, (2) the impact of alternative measures intended to improve the ecological status in the hydrological system is studied, and (3) a better involvement of stakeholders and openness of the process exists at all stages of decision-making. In addition, the Sixth Community Environment Action Programme (EC, 2002) requires that '*full consideration shall be given to ensure that the Community's environmental policy-making is undertaken in an integrated* [social-ecological] *way and to all available options and instruments, taking into account regional and local differences, as well as ecologically sensitive areas, with an emphasis on the best available scientific evidence, and the further improvement of scientific knowledge through research and technological development*.' Science and society have to cooperate to make the management of our environment sustainable. Cooperative modelling both as tool and as process should be part of this.

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Samenvatting

In de toegepaste wetenschap wordt kennis ontwikkeld om bij te dragen aan het oplossen van maatschappelijke vraagstukken. Denk hierbij aan natuurwetenschappelijke kennis over antropogene invloed op natuurlijke hulpbronnen met als doel deze bronnen te beschermen tegen overexploitatie of verontreiniging. Het blijkt echter vaak moeilijk om deze kennis adequaat in de praktijk te implementeren bij het beheer van die hulpbronnen. In mijn promotie onderzoek heb ik me gericht op de vraag hoe natuurwetenschappelijke kennis beter gebruikt kan worden bij het oplossen van maatschappelijke vraagstukken. Hierbij heb ik me met name gericht op de relatie tussen landgebruik en grondwaterkwaliteit en hoe begrip van deze relatie kan bijdragen aan grondwaterplanning: het mede beheren van de grondwaterkwaliteit via het landgebruik.

Landgebruik in Nederland is zeer intensief waardoor verschillende vormen van landgebruik op relatief korte afstanden van elkaar vóórkomen. Hierdoor kunnen effecten van landgebruik op de grondwaterkwaliteit leiden tot conflicten. Zo kan bijvoorbeeld agrarisch landgebruik leiden tot een conflict met een grondwaterfunctie als de drinkwaterwinning. Ook kunnen de gewenste grondwaterstanden ten behoeve van bijvoorbeeld landbouw en stedelijke functies leiden tot verdroging van grondwaterafhankelijke natuurgebieden. De uitdaging van grondwaterplanning bestaat er dan ook uit grondwater optimaal te beheren in een context met verschillende actoren, instituties en beleidsvelden. Om dit te kunnen doen, heb ik in dit onderzoek computermodellen ontwikkeld en aangepast waarmee de relatie tussen landgebruik en grondwaterkwaliteit geanalyseerd en voorspeld kan worden en die daarmee gebruikt kunnen worden in het grondwaterbeheer. De bruikbaarheid van deze modellen is getest door deze modellen toe te passen in concrete projecten in het grondwaterbeheer.

In eerste instantie werd een model ontwikkeld door een grondwaterstromingsmodel en een nitraatuitspoelingsmodel te koppelen aan een Geografisch Informatie Systeem (GIS). Met dit model zijn de effecten geanalyseerd van agrarisch landgebruik op de kwaliteit van grondwater van de drinkwatervoorziening in Halásztelek, Hongarije. Het doel van de modellering was het in beeld brengen van de effecten van het gebruik van meststoffen in de landbouw op de nitraatconcentratie in het onttrokken grondwater voor drinkwater. De uitkomsten gaven kwantitatief inzicht in de belangrijkste processen in de oorzaak en gevolg relatie tussen landgebruik en grondwaterkwaliteit, maar leidden niet tot een verandering in het grondwaterbeheer.

Vervolgens werd een vergelijkbaar model ontwikkeld samen met een brede groep van betrokken actoren om op die manier de toepasbaarheid van de modelresultaten te vergroten. Dit model werd ontwikkeld in het kader van een grondwaterconflict in Holten, Overijssel, Nederland. De actoren vertegenwoordigden specifieke belangen in dit conflict, zoals de landbouw, waterwinning, recreatie, natuurbeheer, de provincie als grondwaterbeheerder en de gemeente Holten (nu gemeente Rijssen-Holten) waar alle ruimtelijke belangen afgewogen worden in een bestemmingsplan. De modelopzet werd ten opzichte van het eerder ontwikkelde modelconcept uitgebreid om naast de effecten van landbouw ook de effecten van andere functies als stedelijk gebied, recreatie en natuur op de grondwaterkwaliteit te kwantificeren. Hierdoor werd de rol van iedere actor in het grondwaterconflict inzichtelijk gemaakt en werd het model in het lokale planproces bruikbaar als onderhandelingsondersteunend instrument (Negotiation Support System, NSS). Vanwege de grote heterogeniteit binnen stedelijke gebieden ten aanzien van actoren en activiteiten werd het effect van stedelijk landgebruik op de grondwaterkwaliteit via een inverse benadering gemodelleerd. Deze inverse benadering kwantificeert het effect van landgebruik op basis van een statistische analyse van grondwaterkwaliteitgegevens in plaats van het berekenen van de grondwaterkwaliteit vanuit het landgebruik via ruimtelijke deterministische relaties.

Tijdens de ontwikkeling en toepassing van het NSS met de actoren werden scenario's opgesteld en geanalyseerd om zodoende verschillende alternatieven te verkennen om het conflict op te lossen. Dit resulteerde in een algemeen geaccepteerd 'aangepast landgebruik scenario' dat geïmplementeerd werd in een regionaal ontwikkelingsplan (Reconstructieplan). Met dit 'aangepast landgebruik scenario' wordt het grondwater meer duurzaam beheerd door het landgebruik af te stemmen op het grondwatersysteem en de drinkwaterwinning als

grondwaterfunctie. Met een gevoeligheidsanalyse werd aangetoond dat het model voldoende nauwkeurig onderscheid maakt tussen de verschillende landgebruikfuncties om de scenarioanalyses uit te voeren. Ook bleek uit deze analyse dat natuurwetenschappelijk onderbouwde beslissingen genomen kunnen worden die zijn gebaseerd op dit model en de beschikbare informatie.

Met dit promotieonderzoek heb ik laten zien dat technische informatie kan bijdragen aan het oplossen van maatschappelijke conflicten zoals een grondwaterconflict, maar ook dat dit vereist dat (1) de technische informatie acceptabel is voor actoren en (2) dat deze acceptatie in sterke mate vergroot kan worden door interactie en samenwerking met de actoren tijdens de opzet van het model, de keuze van de scenario's en de analyse en interpretatie van de resultaten in het licht alternatieven voor de ruimtelijke planning. Dit onderzoek leerde mij, dat een dergelijke aanpak via zogenaamde coöperatieve modelering niet alleen de 'kwaliteit' van het planningsproces verbetert, maar vice versa, dat hierdoor ook de kwaliteit van het technische proces verbetert.

Curriculum Vitae

Cors van den Brink is op 4 juli 1962 geboren in Ermelo. Van 1974-1981 rondde hij achtereenvolgens de HAVO en het VWO af aan het Christelijk College Groevenbeek, voordat hij in 1981 begon aan de brede milieukunde opleiding Rollecate in Deventer. Deze opleiding bleek dermate breed, dat hij besloot zich te verdiepen in een onderdeel daarvan. Dit heeft hij gedaan door van 1982-1988 aan de Landbouw Universiteit Wageningen bodemkwaliteitsbeheer te studeren. Hierbij lag de nadruk op het kwantitatief beschrijven van het gedrag van stoffen in bodem en grondwater.
Sinds 1989 werkt hij als adviseur bij Iwaco / Royal Haskoning. Van 1989-1994 heeft hij zich met name gericht op het ontwikkelen van rekeninstrumenten als FLUNIT en SORWACO ter ondersteuning van projecten in kader van grondwaterbeheer en –sanering. Van 1995-2000 heeft hij zijn werkterrein verbreed met beleidsmatige aspecten bij onder meer de beleidsvernieuwing bodemsanering en grondwaterbescherming. Na 2000 is zijn focus verder verbreed waarbij de maatschappelijke context een steeds prominentere rol speelde. Het promotieonderzoek is in deze periode is uitgevoerd. In 2005 heeft hij samen met Stefan Ouboter de overall prijs gewonnen van de ‘Prijsvraag grondwaterbeheer Zwolle’.

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Appendix 1:
Descriptions of the calculation of the nitrate concentration in shallow groundwater

Introduction

The NSS used in this study was made up of three components: the Modular Dynamic Spatial Analysis Tool (MD-SAT); the Integrated Transport Model (ITM) and TriShell (Van den Brink et al., 2008). MD-SAT calculates the nitrate concentration of shallow groundwater, ITM calculates the quality of the abstracted groundwater and TriShell serves as the user interface for the NSS. The calculations of the nitrate concentration in shallow groundwater are based on the current and historical load, land use type, and soil characteristics. MD-SAT combines various existing simulation programs, each specific for the calculation of the quality a particular aqueous component of the shallow groundwater and a particular land use type. These programs are all one-dimensional, vertical transient descriptions. Within MD-SAT, leaching from agricultural lands was calculated using the program SPREAD (Beekman, 1998). Leaching from forest was calculated with a combined empirical and mass balance approach using SPREAD and the model based on Boxman (2002). Leaching from urban and recreational areas was calculated by an input-output model based on Grimmelikhuyse et al. (1998).

Components of MD-SAT

Agricultural lands: description of the model SPREAD (after Beekman, 1998)
The module SPREAD quantifies the dominating processes of the nitrogen cycle in the unsaturated zone using empirically derived relations and external factors. Nitrogen leaching out of the root zone U_r (kg N ha^{-1} yr^{-1}) is commonly described by the independent leaching terms (e.g., Van Drecht 1993):

$$U_r = U_b + U_w + U_g \qquad (1)$$

where U_b is the basic amount of nitrogen leaching (kg N ha^{-1} yr^{-1}), U_w is the amount of nitrogen leaching during the winter period (kg N ha^{-1} yr^{-1}), and U_g is the amount of nitrogen leaching during the growing season (kg N ha^{-1} yr^{-1}).

Basic leaching
The basic leaching U_b is a function of soil type, land use and mineralisation as function of the organic matter content of the soil:

$$U_b = B + f_b S \qquad (2)$$

where B is the basic leaching as a function of the soil type and land use (kg N ha^{-1} yr^{-1}; see Table A.1), f_b is a constant (≈0.17) and S is the organic matter content of the soil (%).

Table A.1. Basic leaching B as a function of soil type and land use (kg N ha^{-1} yr^{-1}).

Land use	Sand	Clay	Peat
Grassland	15	5	5
Arable land	20	25	25
Maize land	20	25	25
Nature	0	0	0
Urban	5	5	5

Leaching during the winter period
The leaching during the winter period U_w is a function of soil type and land use:

$$U_w = u_w Y \tag{3}$$

where Y is the nitrogen load in the winter period (kg N ha^{-1} yr^{-1}), u_w is the fraction of the nitrogen load in the winter period leaching out of the root zone as a function of soil use (-; see Table A.2). The latter is based on field experiments (Ten Broeke and De Groot, 1995; Lammers et al. 1983; Kolenbrander, 1981; Kroes et al., 1990). Y describes the total nitrogen input during the winter period (kg N ha^{-1} yr^{-1}), which is a function of manure application and atmospheric deposition:

$$Y = m_w M + (1 - f_g)CA \tag{4}$$

where m_w (-) is the fraction of manure applied during the winter period (being 0.2 from 1950 through 1988 and 0.0 from 1988 onwards), M is the nitrogen applied via manure (kg N ha^{-1} yr^{-1}), f_g (-) is the duration of the growing season as a fraction of the year (in this case: 0.65), C (-) is the roughness factor of the land cover with regard to atmospheric deposition (which is 1.0 for agricultural land), and A is the atmospheric deposition of nitrogen (kg N ha^{-1} yr^{-1}).

Table A.2. Leaching fraction during the winter season u_w as a function of land use.

Land use	Leaching fraction
Grassland	0.35
Arable land	0.50
Maize land	0.50
Nature	0.50
Urban	0.50

Leaching during the growing season
The leaching during the growing season U_g is the amount of N leaching as a function of the nitrogen available during the growing season:

$$U_g = u_g Z \tag{5}$$

where Z is the available nitrogen in the root zone during the growing season (kg N ha^{-1} yr^{-1}) and u_g is the leaching fraction (-) based on field experiments (Van Dijk, 1985; Kolenbrander, 1981). The available nitrogen in the root zone during the growing season is calculated as:

$$Z = F + (1 - m_w) f_n M + f_g CA \tag{6}$$

where F is the amount of nitrogen applied as fertilizer (kg N ha^{-1} yr^{-1}), M is the amount of nitrogen applied via manure (kg N ha^{-1} yr^{-1}), A is the atmospheric deposition of nitrogen (kg N ha^{-1} yr^{-1}), and f_n (-) is the fraction of nitrogen effective during the growing season. The fraction f_n is a function of the volatilization of ammonium just after the manure is applied and the mineralization velocity of the organic matter. The organic matter that is not mineralized during the growing season is assumed to contribute to the organic matter content of the soil. The empirical relation for this f_n and carryover from season to season is calculated as:

$$f_n = \frac{f_o M_o + (1 - f_e) M_m}{M_o + M_m} \tag{7}$$

where M_o is the amount of organic nitrogen in manure (g kg^{-1}), M_m is the amount of mineral nitrogen (g kg^{-1}), f_o (-) is the fraction of organic matter that is mineralized during the growing season, and f_e (-) is the fraction of

nitrogen emission during and after manure application. The parameters are specific for manure types and taken from handbooks (e.g., Rinsema, 1985).

The leaching fraction, u_g, during the growing season is:

$$u_g = \frac{\zeta}{1 + e^{-\alpha(Z-\beta)}} \qquad (8)$$

where ζ is the maximum leaching percentage for a specific combination of soil type and crop (-; see Table A.3). The parameters α (ha yr kg N^{-1}; see Table A.3) and β (kg N ha^{-1} yr^{-1}; see Table A.3) are empirical constants for various combinations of soil type and crop.

Table A.3. Constants regarding the relative N leaching as a function of soil type and crops.

Crops	Sand			Clay			Peat		
	α	β	ζ	α	β	ζ	α	β	ζ
Grassland	0.01	400	30	0.01	250	8	0.1	200	3
Maize land	0.004	300	55	0.005	150	15	0.05	150	6
Arable land	0.001	400	50	0.005	150	15	0.05	150	6

Denitrification

In addition to the denitrification implicitly described with the empirical relations for the basic leaching, leaching during the winter period and leaching during the growing season, the leaching has to be corrected for denitrification as function of the groundwater table depth. The nitrogen flux reaching the shallow groundwater U_t is calculated from the leaching from the root zone U_r (equation 1) by accounting for denitrification in the unsaturated zone between the root zone and the shallow groundwater:

$$U_t = U_r D \qquad (9)$$

where U_t is the nitrogen flux reaching the groundwater table (kg N ha^{-1} yr^{-1}) and D is the correction factor for the denitrification as a function of the depth of groundwater table (-; see Table A.4).

Table A.4. Correction factor for denitrification below the root zone *D* as function of the groundwater table (Steenvoorden et al., 1997).

Groundwater table (Dutch classification)	MHG (cm to soil surface)	MLG (cm to soil surface)	*D* (-)
I	< 20	< 50	0.00
II	< 40	50 – 80	0.05
III	< 40	80 – 120	0.10
IV	> 40	80 – 120	0.40
V	< 40	> 120	0.50
VI	40 – 80	> 120	0.60
VII	80 – 140	> 160	0.75
VIII	> 140	> 160	1.00

MHG = mean high groundwater table (groundwater table in the spring)

MLG = mean low groundwater table (groundwater table in the autumn)

Nature: description of input-output balances

The estimation of the nitrogen leaching by the forests is calculated as the input-output balance and divided into a summer and a winter period. The uptake of the forest is averaged over the entire calculation period to be 6.7 kg N ha^{-1} yr^{-1} as the average value for a forest with different tree species and ages (De Vries e.a., 2003). The input is dependent on the atmospheric deposition *A* and the roughness factor *C*.

Nitrogen accumulation is estimated using a 'basis accumulation' before an increase in atmospheric deposition occurred from 1970 through which the nitrogen input in the forests and the carbon/nitrogen ratio respectively increased and decreased. Since 1970, a linear increase has been assumed for nitrogen leaching because of an assumed linear decrease in nitrogen accumulation. When the nitrogen accumulation is complete, nitrogen leaching from the surface soil of the forests is estimated from the difference between the input via deposition and the output via uptake by the forest. In addition, the basic leaching as function of the organic mater content of the soils (*S*) is added to the calculated leaching. Until the year 1970, *Z* and *Y* by forest are assumed to be zero.

$$Z = Y = 0 \tag{10}$$

From 1970 till 2000, *Z* and *Y* by forest are calculated by:

$$Z = (CA\tfrac{2}{3} - L_f)(0.1 + 0.9\frac{t - 1970}{30}) \tag{11}$$

$$Y = (CA\tfrac{1}{3})(0.1 + 0.9\frac{t - 1970}{30}) \tag{12}$$

where 2/3 and 1/3 refer to the length of the summer and winter period respectively, *t* is the year between 1970 and 2000 and the second part of the equations represents a linear transition from a value of 0.1 in 1970 to 1.0 in 2000.

After the year 2000, *Z* and *Y* by forest are calculated by:

$$Z = CA\tfrac{2}{3} - L_f \tag{13}$$

$$Y = CA\tfrac{1}{3} \tag{14}$$

Urban and recreational land use: description of input-output balances

The estimation of the N application by urban and recreation areas is calculated as an input-output balance in which the input terms are atmospheric deposition and leakage from sewers for urban areas and direct discharge of waste water into the soil (i.e., without any purification) for recreation areas. Leakage from sewers in urban areas is estimated to be 36 kg N ha^{-1} yr^{-1}, while the extra application from direct discharge in the recreation areas is estimated to be 120 kg N ha^{-1} yr^{-1} in the summer period and 40 kg N ha^{-1} yr^{-1} in the winter period (Grimmelikhuyse et al., 1998):

– Nitrogen application urban area summer (T_S): (36 kg N ha^{-1} yr^{-1})/2
– Nitrogen application recreation area summer (R_S): urban application + 120 kg N ha^{-1} yr^{-1}
– Nitrogen application urban area winter (T_W): (36 kg N ha^{-1} yr^{-1})/2
– Nitrogen application recreation area winter (R_W): urban application + 40 kg N ha^{-1} yr^{-1}

The total nitrogen input in summer *Z* and winter *Y* by urban areas are calculated by:

$$Z = T_s + A/2 \tag{15}$$

$$Y = T_w + A/2 \tag{16}$$

For recreational areas the nitrogen input in summer *Z* and winter *Y* are calculated by:

$$Z = R_s + A/2 \quad (17)$$

$$Y = R_w + A/2 \quad (18)$$

From 2005 onwards, it is assumed that the total of the atmospheric deposition and leakage from sewers for urban areas will be reduced to 36 kg N ha^{-1} yr^{-1}. The N-input by recreation areas is then 196 kg N ha^{-1} yr^{-1}.

In addition, the basic leaching as function of the organic matter content of the soils *S* is added to the calculated leaching.

Table A.5. Symbols used in the equations of appendix 1.

Symbol	Description
α	empirical parameter for different combinations of soil type and crop (agricultural land use)
β	empirical parameter for different combinations of soil type and crop (agricultural land use)
ζ	maximum leaching percentage for a specific combination of soil type and crop (agricultural land use)
A	atmospheric deposition of nitrogen
B	basic leaching as a function of the soil type and land use
C	roughness factor of the land cover with regard to atmospheric deposition as a function of land use
D	denitrification factor for unsaturated zone as function of the depth of the groundwater table
e	base of natural logarithm (approx. 2.72)
F	amount of N applied as fertilizer
f_b	constant for basis N leaching in relation to organic matter content
f_e	fraction of nitrogen emission during and after manure application
f_g	duration of the growing season as a fraction of the year
f_n	fraction for effective N related to N applied via manure in the growing period
f_o	fraction of organic matter that is mineralized during the growing season
L_f	nitrogen uptake by forest
M	N applied via manure
M_m	amount of mineral nitrogen in manure
M_o	amount of organic nitrogen in manure
m_w	fraction of manure applied during the winter period
R_s	nitrogen load in summer for unsewered recreational areas
R_w	nitrogen load in winter for unsewered recreational areas
S	organic matter content of the soil
t	year between 1970 and 2000
T_s	nitrogen load in summer for urban areas
T_w	nitrogen load in winter for urban areas
U_b	basic amount of N leaching
U_g	amount of N leaching during the growing season
u_g	leaching fraction for the growing season
U_r	N leaching out of the root zone
U_t	N flux reaching the groundwater table
U_w	amount of N leaching during the winter period
u_w	fraction of N load in the winter period leaching out of the root zone
Y	total N input in the winter period
Z	available N in the root zone during the growing season

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Appendix 2:

Probability density functions

Introduction

Insight is necessary into the uncertainties of the mean values of the input parameters in order to execute a stochastic sensitivity and uncertainty analysis. The input parameters are listed in the tables A.1 – A.8. It is assumed that the uncertainties and type of distributions are time-independent. The mean values used for the calculations, deviations, and type distributions are then inventoried from the input parameters. The probability density function is subsequently calculated using this information. This distribution illustrates all of the possible results and accompanying probabilities of a certain parameter value and shows the probability density of a certain parameter value. Finally, 1000 samples are compiled using the Latin Hypercube Sampling method. The symbols used in this appendix can be found in table A.5 in appendix 1.

Input parameters and uncertainties

The input parameters can be found in the tables below. The mean values used for the calculations (mean or μ), the deviations (sd or σ), and the type of distributions are given in each one. Different types of distributions can be discerned:

- Normal: distribution is from -∞ to +∞ (μ is centered in the infinite interval);
- Beta: distribution around a certain μ with a set minimum and maximum (μ is located at a random spot in a finite interval);
- Gamma: distribution that is bound at one end (often 0).

The uncertainty of the time dependent parameters for the year 2005 is described by the two independent parameters atmospheric deposition (*A*) and fertilizer application *F* and manure application (*M*). The distribution is obtained for these input parameters (see Table A.1).

Table A.1. Time-dependent parameters for the year 2005.

Parameter	Mean (kg N ha^{-1} yr^{-1})	sd	Distribution (expected)	Uncertainty	Reference
Fertilizer *F* and manure application *M*	1		Normal	15%	Bleeker and Erisman, 1996; professional judgement
Atmospheric deposition *A*	25		Normal	15%	Bleeker and Erisman, 1996; professional judgement

For each type of land use the actual N-applications in the growing season (*Z*) and in the winter period (*Y*) are derived by combining the N-application from atmospheric deposition *A* and the fertilizer and manure application multiplied with the distribution of *A* and *F* plus *M*: (see equation 4 and 7 of appendix 1 respectively.

The calculated values for each type of land use for *Z* and *Y* are listed in table A.2.

Table A.2a. N-application during the growing season *Z* for the year 2005 (regular land use and in vulnerable sandy soils).

Landuse	Mean (kg N ha^{-1} yr^{-1})	Sd (kg N ha^{-1} yr^{-1})
Grassland	396.25	59.33
Arable	276.25	41.36
Maize	146.25	21.90
Nature	20.83	2.50
Urban	18.00	2.69
Recreation	138.00	20.66
vulnerable sandy soil - Grassland	356.25	53.34
vulnerable sandy soil - Arable	236.25	35.37
vulnerable sandy soil - Maize	106.25	15.91

Table A.2b. N-application during the winter period *Y* for the year 2005 (regular land use and in vulnerable sandy soils).

Parameter	Mean (kg N ha^{-1} yr^{-1})	Sd (kg N ha^{-1} yr^{-1})
Grassland	8.75	1.31
Arable	8.75	1.31
Maize	8.75	1.31
Nature	10.42	1.25
Urban	18.00	2.70
Recreation	58.00	8.69
vulnerable sandy soil - Grassland	8.75	1.31
vulnerable sandy soil - Arable	8.75	1.31
vulnerable sandy soil - Maize	8.75	1.31

Table A.3. Basic leaching.

Land use – soil combination	Mean (kg N ha^{-1} yr^{-1})	Sd (kg N ha^{-1} yr^{-1})	Distri-bution	Min (kg N ha^{-1} yr^{-1})	Max (kg N ha^{-1} yr^{-1})	Uncertainty (expected)	Reference
Grassland-sand	15	2	Beta	0	20	0-15	Van Drecht, 1991
Grassland-clay	5	2	Beta	0	8	0-5	Van Drecht, 1991
Grassland-peat	5	2	Beta	0	200	5-200	After De Vries e.a. 2003
Arable-sand	20	2	Beta	0	45	0-45	Van Drecht, 1991 (45); De Vries e.a. 2003 (0)
Arable- clay	25	2	Beta	0	30	0-25	Van Drecht, 1991
Arable-peat	25	2	Beta	0	200	5-200	Van Drecht, 1991
Maize-sand	20	2	Beta	0	45	0-45	Van Drecht, 1991
Maize-clay	25	2	Beta	0	30	0-25	Van Drecht, 1991
Maize-peat	25	2	Beta	0	200	0-200	Van Drecht, 1991
Nature-sand	0	2	Beta	0	10	0-10	De Vries e.a. 2003
Nature-clay	0	2	Beta	0	10	0-10	De Vries e.a. 2003
Nature-peat	0	2	Beta	0	150	0-150	De Vries e.a. 2003
Urban-sand	5	2	Beta	0	8	0-5	Professional judgement
Urban-clay	5	2	Beta	0	8	0-5	Professional judgement
Urban-peat	5	2	Beta	0	150	0-150	Professional judgement

Table A.4. Organic matter content soils *S*.

Soil type	Mean (%)	sd	Distribution	Min (%)	Max (%)	Uncertainty (expected)	Reference
sand	4		Normal	0	8		De Vries (1999)
clay	8		Normal	4	12		De Vries (1999)
peat	12		Normal	8	80		De Vries (1999)

Table A.5. Duration of the growing season f_g.

Parameter	Mean (-)	Sd	Distribution	Min (-)	Max (-)	Uncertainty (expected)	Reference
f_g	0.65		Normal	0.60	0.70		Professional judgement

Table A.6. Maximum relative N leaching ζ.

Land use – soil combination	Mean (-)	Distribution	Min	Max (-)	Uncertainty (-)	Reference (expected)
grassland-sand	30	Beta	10	40	10-40	Beekman, 1998; after De Vries e.a. 2003
grassland-clay	8	Beta	1	10	1-10	Beekman, 1998; after De Vries e.a. 2003
grassland-peat	3	Beta	0	5	0-5	Beekman, 1998; after De Vries e.a. 2003
arable-sand	50	Beta	35	100	35-100	Beekman, 1998; after De Vries e.a. 2003
maize-sand	55	Beta	30	70	30-70	Beekman, 1998; after De Vries e.a. 2003
maize-clay	15	Beta	0	30	0-30	Beekman, 1998; after De Vries e.a. 2003
maize-peat	6	Beta	0	20	0-20	Beekman, 1998; after De Vries e.a. 2003

The parameters α (ha yr kg N^{-1}) and β (kg N $ha^{-1}yr^{-1}$) from the description of u_g determine the shape of the curve. It is assumed that they are more or less typical for a combination of crop and soil and thus do not vary (see Appendix 1, Table A.3).

Table A.7. Leaching fraction in winter u_g as function of land use.

Landuse	Mean (-)	Sd	Distribution	Min (-)	Max (-)	Uncertainty (expected)	Reference
Grassland	0.35	0.11	Gamma	0	1	20-40%	After Kroeze e.a. 2003
Arable	0.50	0.15	Gamma	0	1	20-40%	After Kroeze e.a. 2003
Maize	0.50	0.15	Gamma	0	1	20-40%	After Kroeze e.a. 2003
Nature	0.50	0.15	Gamma	0	1	20-40%	After Kroeze e.a. 2003
Urban	0.50	0.15	gamma	0	1	20-40%	Professional judgement

Table A.8. Correction factor for denitrification below root zone as function of the groundwater table.

Groundwater table (Dutch classification)	Mean (-)	sd	Distribution	Min (-)	Max (-)	Uncertainty (expected)	Reference
I	0.01°		Beta	0	0.012	20%	After De Vries e.a. 2003
II	0.05		Beta	0.04	0.06	20%	After De Vries e.a. 2003
III	0.10		Beta	0.08	0.12	20%	After De Vries e.a. 2003
IV	0.40		Beta	0.32	0.48	20%	After De Vries e.a. 2003
V	0.50		Beta	0.4	0.6	20%	After De Vries e.a. 2003
VI	0.60		Beta	0.48	0.72	20%	After De Vries e.a. 2003
VII	0.75		Beta	0.6	0.9	20%	After De Vries e.a. 2003
VIII	0.95°		Beta	0.76	1.0	20%	After De Vries e.a. 2003

° The mean for ground water table class I and VIII is not equal to the value used in the deterministic calculations (cf. table A.4 in appendix 1) because the range of possible values lies entirely on one side of the deterministic values.

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Technical and scientific knowledge can help sustainable groundwater planning, provided that the information is acceptable to stakeholders. This acceptability can be greatly improved by interaction and cooperation with the stakeholders in the set-up of the model, the choice of scenarios and the analysis of the results towards spatial planning alternatives.

Cors van den Brink