EUROHARP 1-2003

Review and Literature Evaluation of Quantification Tools for the Assessment of Nutrient Losses at Catchment Scale

Editors
Oscar F. Schoumans, ALTERRA, the Netherlands
Martyn Silgram, ADAS, United Kingdom

EUROHARP Report no 1
Review and Literature Evaluation of Quantification Tools of Nutrient Losses

Schoumans, O.F. & Silgram, M. (eds.), 2003. Review and literature evaluation of nutrient quantification tools. EUROHARP report 1-2003, NIVA report SNO 4739-2003, Oslo, Norway, 120 pp.

A publication developed within Work Packages 3 and 4 of the EUROHARP project: ″Application and evaluation of phosphorus and nitrogen loss quantification tools" .

Published by:	Norwegian Institute for Water Research (NIVA), Norway
Editors:	Oscar Schoumans, ALTERRA, The Netherlands - Martyn Silgram, ADAS, United Kingdom
Authors:	Hans Estrup Andersen, NERI, Denmark - Steven Anthony, ADAS, United Kingdom - Berit Arheimer, SMHI, Sweden - Alan Barr, KMM, Ireland - Horst Behrendt, FV-IGB, Germany - Fayçal Bouraoui, EC-JRC, Italy - Helene Ejhed, IVL, Sweden - Piet Groenendijk, ALTERRA, The Netherlands - Michel Jeuken TERRA, The Netherlands- Holger Johnsson, SLU, Sweden - Brian Kronvang, NERI, Denmark- Guillaume Le Gall, BETURE-CEREC, France - Alison Murdock, KMM, Ireland - Antonio Lo Porto, IRSA-CNR, Italy - Laura Price, ADAS, United Kingdom - Oscar Schoumans, ALTERRA, The Netherlands - Martyn Silgram, ADAS, United Kingdom - Robert Smit, ALTERRA, The Netherlands - Ekaterini Varanou, NTUA, Greece - Ulrike Zweynert, FV-IGB, Germany.
Number Printed:	60
ISBN:	82-557-4411-5
Pages:	120
Date of printing:	November 2003
The Project:	The EC funded EUROHARP project encompasses 22 research institutes from 17 European countries (2002-2005). The overall objective of the EUROHARP work is to provide end-users with guidance for an appropriate choice of quantification tools to satisfy existing European requirements on harmonisation and transparency for quantifying diffuse nutrient losses, e.g. to facilitate the implementation of the Water Framework Directive and the Nitrates Directive. The project includes both the assessment of the performance of individual models and the applicability of the same models in catchments with different data availability and environmental condition throughout Europe. The basis for the performance and applicability studies is the compilation of a harmonised GIS/database for all catchment data and the analysis of these data (trends, watercourse retention).
Key words:	Model assessment. Nitrogen. Phosphorus. Diffuse losses. Water Framework Directive. Nitrates Directive.

The report may also be downloaded from the EUROHARP web site: www.euroharp.org.

Preface

In the OSPAR "HARP" guidelines, several guidelines were described which focused on estimating the contribution of nutrient sources to surface waters. However, no single method could be agreed for estimating diffuse losses from agricultural land to surface waters or in-stream retention of nutrients because of fundamental differences in the methodologies used in individual countries. In order to compare these different approaches, the EUROHARP project was developed, at OSPAR's request. The resulting EC Framework V project began in January 2002. This report represents the first deliverable from that project.

One of the aims of EUROHARP is to improve transparency by reviewing different modelling methods, compare and contrast the differing approaches, and consider the potential capability of these different type of models ("quantification tools") in a scientific evaluation.. This preliminary scientific review has been undertaken *before* model results are available, and is intended to provide information concerning the strengths, weaknesses, capabilities and potential limitations of different models predicting nitrogen and phosphorus loss from agricultural land to surface waters. This assessment included consideration of the boundary conditions, process description, and the pathways that are taken into account by each model. In this report the outcome of the intercomparison is described as a result of the work of Work Package 3 (WP3) focusing on phosphorus and Work Package 4 (WP4) focusing on nitrogen.

This intercomparison was undertaken during the first 18 months of the study and the progress and outcome was discussed by representatives from all participating model institutes in several project meetings:

- Berlin, FV-IGB, 13-14 April 2002
- York, ADAS, 6-7 November 2002
- Wageningen, Alterra, 11-12 February 2003

This document is not intended as a comprehensive description of every facet of each model, but rather an overview of the main model elements and an intercomparison of the approaches used. Although every effort has been made to ensure the accuracy of the information presented in this document, including the participation of representatives from each modelling institute, readers are urged to consult the original published sources and named institute contacts cited at the end of this report if they require definitive descriptions of individual models. Any errors or omissions should be brought to the attention of the report's editors.

The final scientific evaluation of the actual *performance* of each model will be made in later stages of the project once each model has been applied to each of the three "core" catchments and the statistical performance criteria have been calculated. The results of this application will be published in 2004 in a future report based on outputs from Work Packages 3 and 4. The applicability of the quantification tool in additional catchments will be tested within Work Package 5. These results will be published in 2005.

Contents

Executive Summary 5

1. Introduction 8
2. Scientific and operational details 10
3. General description of quantification tools 13

3.1 General description 14
3.2 Overview of pathways and processes described 50
3.3 Model comparison 53
   3.3.1 Boundary conditions 53
   3.3.2 Nutrient inputs and land use management 53
   3.3.3 Plant growth and crop uptake 54
   3.3.4 Hydrology 56
   3.3.5 Soil chemical processes 64
   3.3.6 Biochemical soil processes 66
   3.3.7 Approaches used for lumping soil processes 72
   3.3.8 Model Output 80

4. Evaluation 81

4.1 Spatial and temporal resolution 81
4.2 Pathways 82
4.3 Processes and modelled nutrient species 83
4.4 Cost implications 84
4.5 Potential suitability for scenario analysis 85
4.6 Applicability 86

5 Potential strengths and weaknesses of the quantification tools 89
6 References 93

ANNEX A Hydrological pathways 102
ANNEX B Short model information 107

Executive Summary

The enrichment of fresh water systems with nutrients is acknowledged as a major problem in many European countries. In the HARP guidelines methodologies have been described to assess the contribution of nutrient pollution of river basins by different sources. However, the contribution of diffuse nutrient losses caused by agricultural activities is not well understood and not well defined. The Water Framework Directive demands the implementation of measures in order to reach the defined targets for water bodies. Furthermore, a monitoring network has to be set up, to follow the effectiveness of measures on the quality of water bodies. In order to set up an effective measurement program and monitoring programme, it is important to determine the contribution of different sources and to understand dominant transport pathways. Models can be very helpful in defining the sources of nutrient pollution and the magnitude of relevant pathways. However, the capability and applicability of different models under different European conditions has not been thoroughly quantified.

In EUROHARP, a EC Framework V project, which started in 2002 with 22 partners in 17 countries across Europe, a detailed intercomparison of contemporary catchment-scale modelling approaches is being undertaken to help characterise the relative importance of point and diffuse pollution in surface freshwater systems. Several stages have been defined within the project. First of all a review and literature evaluation of each nutrient "quantification tool" (this document) has been undertaken. Thereafter, the work focuses on three core catchments in order to determine the capability of models to predict diffuse losses from agricultural land. The three core-catchments, from North to South, are: Vansjo-Hobol (Norway), Yorkshire Ouse (England), and Enza (Italy).

The nutrient quantification tools (models) involved, differ profoundly in their complexity, level of process representation and data requirements. The methods range from data oriented models (empirical and statistical models) to process oriented (deterministic) models. This report covers the intercomparison of the nitrogen and phosphorus quantification tools, based on literature study, a review of the models by model owners and the outcome of discussions of several workshops.

Nine quantification tools are involved in this study: NL-CAT (a combination of the models ANIMO/SWAP/SWQN/SWQL), REALTA, N-LES CAT, MONERIS, TRK (a combination of the models SOILNDB/HBV-N), SWAT, EveNFlow, NOPOLU, and Source Apportionment.

For the intercomparison of the quantification tools 15 different aspects were considered, including (1) Original purpose/status and history of the model (maturity), (2) Dependencies on previous models (scientific evolution), (3) Review of pathways and processes described by the quantification tools, (4) Scientific description of the processes involved, (5) Spatial resolution and discretisation (horizontal and vertical), (6) Temporal resolution and discretisation, (7) Forms of nutrient losses described by the quantification tool, (8) Data requirements, (9) Operational experience and skills requirement of users, (10) Participation in previous model comparison studies, (11) Sub-modules that can be independently checked, (12) Existing sensitivity analysis, (13) Cost indication (based on work load to set up and apply the quantification tool), (14) Capability to evaluate nutrient and watershed management strategies (scenario analysis) and (15) Applicability (climate, land use etc)

The horizontal spatial resolution between the models increases from about 0.1 km^2 to 50 km^2:
N-LES CAT < NL-CAT=SWAT = TRK < EveNFlow = SA < NOPOLU = REALTA < MONERIS

With respect to the temporal resolution, all quantification tools are able calculate *annual* nutrient losses (N and/or P) from agricultural land to surface waters (the major objective of this study to compare the quality of this assessment by the different methodologies). Only four models are able to produce the temporal dynamics of nutrient losses to surface waters (daily loads: SWAT, TRK, NL-CAT and EveNFlow), which is of great value when considering for example, frequency of exceedance of threshold values for nitrate concentration or the seasonality of eutrophic status (e.g. Nitrates Directive).

An important limitation of four models (REALTA, NOPOLU, N-LES CAT, SA) is that they are not able to quantify the water flow by different pathways by themselves, but they need measured flow data for each of the pathways or sometimes combined information of the measured flows of these pathways. In contrast, MONERIS needs only the total river flow. Other models (e.g. EveNFlow) model the water balance and river flow explicitly. In these cases, achieving an adequate representation of the water balance, and the timecourse of water flows is the most important first target, with the satisfactory representation of chemical signatures a secondary stage.

The level of detail in representations of individual nitrogen processes in the soil decreases in the following order:
NL-CAT > TRK (SOILNDB) > SWAT>> EveNFlow > MONERIS > N-LES CAT > NOPOLU > SA
With respect to phosphorus the comparable order is:
NL-CAT > SWAT>> MONERIS > TRK = NOPOLU > SA

The source apportionment method (SA) and the REALTA and NOPOLU quantification tools do not consider soil processes, but can nonetheless serve as “broad brush” tools to assess pollutant loads at catchment level. In the N-LES CAT model, which is a statistical relationship between on the one hand nitrogen input, crop, soils, and climate characteristics, and on the other hand measured nitrate concentrations leaching out of the root-zone, the internal nutrient processes are implicitly taken into account. For all these four models (SA, REALTA, NOPOLU and N-LES CAT) soil processes are lumped and implicitly derived from measured monitoring data. *In those cases direct extrapolation to other soil, climate, or hydrological conditions may not be possible.*

Within MONERIS, net mineralisation and immobilisation is ignored and the net N surplus (input minus harvest offtake) is assumed to be released as dissolved inorganic nitrogen. With respect to phosphorus no sorption and desorption mechanisms are taken into account, with an overall equation used to describe the relationship between P content of the soil and the P concentration in soil solution.

Within EveNFlow a module estimates the mass of nitrate present in the soil at the onset of winter drainage that is vulnerable to leaching. The calculation is based upon empirical relationships between soil nitrogen supply and the nutrient balance under conventional cropping and grazing regimes, with coefficients associated with different land uses and animal types. EveNFlow uses a meta-model to estimate nitrate losses as a function of rainfall and soil water content in relation to these potential nitrate losses.

The models SWAT, NL-CAT and TRK (SOILNDB) have a detailed representation of nutrient dynamics in soils. In SWAT all processes (plant growth, mineralization, immobilisation, denitrification, sorption and desorption) are modelled, but for each process a lumped equation is used. TRK (SOILNDB) and NL-CAT are more comparable in their approach for nitrogen processes. However, for phosphorus differences are quite noticeable because TRK uses a different approach based on a empirical (statistical) relationship for Swedish conditions.

Based on the workload needed to apply the model on one new catchment, the amount of man-months increases from about 0.5 man-months up to 3 man-months per catchment per nutrient. For the nitrogen quantification tools the amount of workload increases from
SA < MONERIS < N-LES CAT = EveNFlow = TRK =SWAT < NLCAT
For phosphorus the total workload increases in the following order:
SA < NOPOLU = REALTA < TRK < MONERIS < SWAT < NLCAT

Although the costs of applying the quantification tools differ substantially, the most suitable model for a particular application will depend on the purpose of the study (e.g. identify risk areas, detailed quantification of partitioning of losses from land, scenario analysis etc.) and the quality (accuracy and precision) needed from model results ("quality" versus "cost"). With respect to the quantification tools in this study, this review shows that the possibilities for scenario analyses tend to increase as the complexity of the model increases, but so too does the relatively high costs associated with setting up these more complex models. A summary table showing potential strengths and weaknesses of each model is included at the end of this report.

A table was also compiled with initial impressions regarding the potential suitability of each model for application to different catchment types covering a range of climate, soils and land use. At this stage, no single model initially appeared well suited for application to all the different European catchment typologies. This initial assessment has increased transparency – giving modellers a clear understanding of each other's approaches, assumptions, capabilities and limitations – and has enabled an initial view to be formed regarding the potential strengths and weaknesses of individual approaches. The next stage is to review actual model performance against measured river flow and water quality in each study catchment. This work, which will be concluded during 2004 and 2005, will enable the performance of each model to be assessed in three different catchment types, and enable a ranking of all models to be calculated for each of three "core" catchments studied. The ultimate outputs at the end of the project will include recommendations for model selection depending on catchment typology, and a robust assessment of the strengths, limitations, and cost-effectiveness of different approaches for modelling the diffuse agricultural contribution of nitrogen and phosphorus to surface freshwater systems.

1. Introduction

Several different types of quantification tools for nutrient losses to river basins have been developed during the last decade within European countries (Kronvang et al., 1995; Arheimer and Brandt, 1998; Krysanova *et al*., 1999; Behrendt & Bachor, 1998; Behrendt et al. 2000; *Kronvang et al*.,1999B) and outside Europe (Beasley *et al*., 1980; Leonard *et al*., 1987; Arnold *et al*., 1990; Arnold *et al*., 1993). These quantification tools were established for different regions and different tasks. They differ in their complexity, their resolution in time and space, and they need different levels of detail in terms of data requirements (Figure 1). In this study the term quantification tool is used, because a number of these quantification tools consist of some individual models/modules which are separately described, and because the approaches vary e.g. from a very simple difference method to complex mechanistic models.

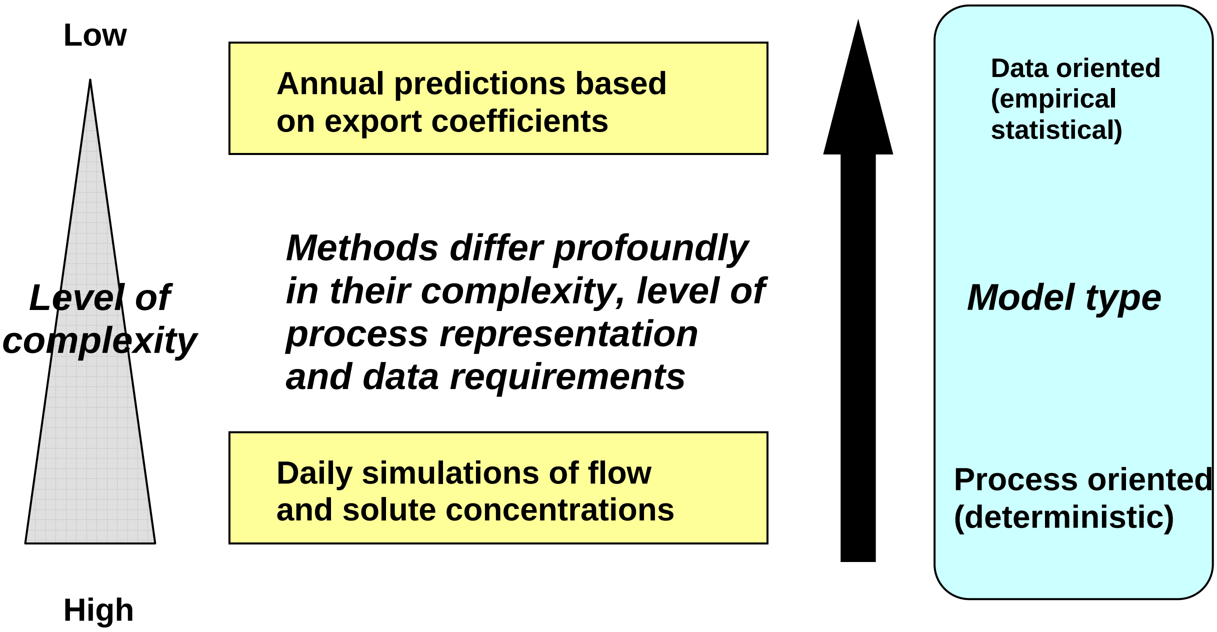


Fig. 1 A general relation between the complexity of models (left), model type (right) and the generated output.

The quantification tools used within this study have often been applied at different scales and cover a wide range from spatially lumped static quantification tools to fully distributed process orientated dynamic quantification tools. The nutrient quantification tools are able to describe either parts or all of the different processes that govern nutrient cycling at catchment scale. Moreover, many quantification tools have only been applied to a specific part of Europe, which means that they may not be able to handle the gradient in climate (e.g. frozen soils), hydrology (shallow groundwater), land use and/or agricultural practices existing in other parts of Europe. Problems with the acquisition of input data to the different models can also severely limit their application to different parts of Europe.

Process-orientated dynamic quantification tools normally require large amounts of input data at a very detailed temporal and spatial scale. In many cases, such detailed data may not be available, at least not at the larger scale, requiring some assumptions or default values to be made, or transfer functions developed. Empirical and quasi-empirical approaches, such as statistical models, may in such cases be viable alternatives. Even in this category there is a large variability in complexity (e.g. Grimvall & Stålnacke, 1996; Caraco & Cole, 1999). However, many statistical based models have the limitation that they may not be able to describe the dynamics in the fluxes. This trade-off between the complexity and applicability of these two approaches has been discussed by several authors (e.g. de Vries, 1994) and is an important consideration during the EUROHARP project. In recent years, initiatives have studied the linkage between more dynamic models and pure statistical ones. For example Lidèn *et al.* (1999) showed that the export coefficients from the Swedish HBV-N model (Arheimer & Brandt, 1998) were very similar to the export coefficients derived from the statistical MESAW-model (Grimvall & Stålnacke, 1996).

One of the major aims of the EU-project EUROHARP is to determine the performance and potential capability of these different type of quantification tools by means of a scientific evaluation and a 'practical' test by comparing the results of the quantification tools on the measured data of three core catchments. The scientific evaluation is important because many factors determine the phosphorus and nitrogen loss from agricultural land to surface waters and therefore, end-users should be aware of the limitations are of each quantification tool. Furthermore, most quantification tools are used to predict the effect and impact of measurements on the nutrient losses (scenario-analysis), e.g. new manure strategies, different type of land management, land use changes etc. Also from this point of view it is important to understand to what extend the quantification tools are capable of predicting changes in nutrient losses.

EUROHARP aims to provide end-users (national and international environmental policy-makers) with a thorough scientific evaluation of contemporary quantification tools and their ability to estimate diffuse nutrient losses to surface freshwater systems and coastal waters. EUROHARP focuses on an objective assessment of the accuracy, strengths and weaknesses, cost-effectiveness and practicability of each tool, and include guidance on suitability for application to different catchment types, and responsiveness to changes in land use and land management. The project aims to provide results that will help managers of river basin districts in watershed planning, as well as assisting institutes that report to policy makers on the contribution of nutrient losses from agricultural land to surface waters. EUROHARP will help such end-users decide which quantification tools are most appropriate for their catchment or river basin in order to obtain accurate results at affordable cost, based on the available source data. In order to achieve this goal a scientific intercomparison of the different conceptual structures and boundaries, data requirements, levels of complexity, underlying assumptions, and temporal and spatial resolution in quantification tools currently used for estimating nutrient losses at catchment scale by European policymakers is necessary.

In this report the results of a scientific pre-evaluation study are described. Chapter 2 explains the approaches used in this review. Chapter 3 includes a short description of the quantification tools. In Chapter 4, the boundary conditions and restrictions of the quantification tools are summarised. Finally, in Chapter 5 the conclusions of this study are given including the perceived or "potential" strengths and weaknesses of the different quantification tools.

2. Scientific and operational details

The methodologies that are currently used for quantifying *diffuse* P losses have been developed at a national level within Europe, and differ profoundly in (i) their level of complexity, (ii) their representation of system processes and pathways, and (iii) resource (data and time) requirements. They range from complex, process-based models - which typically have demanding data requirements - to semi-empirical (conceptual) meta-models with some export coefficients, and approaches based on mineral balances and source apportionment. With many nations using varying approaches, there is now an urgent need for an intercomparison of these contrasting methodologies in order to form an objective judgement of their performance under different agricultural, geophysical and hydrological conditions throughout Europe.

Based on a discussion at a workshop in Berlin (17-18 April 2002), with all modellers of the EUROHARP project, the following scientific details were selected for the intercomparison of the quantification tools.

1) Original purpose/status and history of the model application (maturity)
2) Dependencies on previous models (scientific evolution)
3) Review of pathways and processes described by the quantification tools
4) Scientific description of the processes involved
5) Spatial resolution and discretisation (horizontal and vertical)
6) Temporal resolution and discretisation
7) Forms of nutrient losses described by the quantification tool
8) Data requirement
9) Operational experience and skills requirement of users
10) Participation in previous model comparison studies
11) Sub-modules that can be independently checked
12) Existing sensitivity analysis
13) Cost indication (based on work load to set up and apply the quantification tool)
14) Capability to evaluate nutrient and watershed management strategies (scenario analysis)
15) Applicability

These factors are discussed below.

1) Original purpose/ status and history of the model application (maturity)

Since the original purpose underlying the development of each model may differ, it is important to know these differences in order to understand the assumptions that have been made in each modelling approach. Furthermore, this will provide information on the scope, applicability and capability to evaluate water and nutrient management strategies for each model considered.

2) Dependencies on previous models

Pat of the quantification tools may have been derived from modules in other models. In this way the quantification tools have often evolved based on already peer-reviewed models.

3) Review of pathways and processes described by nutrient quantification tools

Nutrient loads of surface waters from non-point sources, mainly agriculture and nature, is caused by transport of different forms of nutrients over and through the soil to surface waters. Since a lot of quantification tools were developed for specific situations/circumstances (e.g. just for applications within a nation) simplifications were made from that perspective. However, from an European point of view it is important to understand which pathways and forms of nutrient losses are described by each of the nutrient quantification tools. This information will be used to identify some of the restrictions of the nutrient quantification tools (applicability; see also point 9)

4) Scientific description of processes
Since the biological, chemical and physical interaction of nutrients in soil is rather complex and difficult to (understand and) describe, many model developers have made appropriate simplifications or assumptions. In order to assess the capability of nutrient quantification tools to evaluate nutrient and watershed management strategies (scenario analysis; see also point 14) information should include the extent to which the quantification tools are able to describe the impact of different strategies on nutrient losses to surface waters.

5) Spatial resolution and discretisation (horizontal and vertical)
This factor covers the way in which the horizontal as well as the vertical (profile) discretisation is handled. Some quantification tools have limits on the smallest "unit" that can be modelled, and/or the range of catchment sizes for which the approach is valid.

6) Temporal resolution and discretisation
Some models only describe the mean annual or seasonal nutrient loss while others describe the dynamics in smaller timesteps (e.g. daily).

7) Forms of Nutrient losses
Nutrient losses from agricultural land to surface waters contain different forms/species of phosphorus and nitrogen e.g. the bioavailability of phosphorus in surface waters depends on the distribution of P-forms of the total load of P. Within this study, phosphorus is considered as soluble inorganic P, soluble organic P, particulate P, and total P; while nitrogen is considered as NO_3, NH_4, organic N and total N components.

8) Data requirement
Since the original aim of the quantification tools differ, the type as well as the amount of data differs remarkably. With regard to data requirements, the following type of data will be distinguished: management (fertilisation/crops), soil physical and biochemical characterisation, water balance.

9) Operational experience and skills requirement of users
This information is needed in order to determine if watershed managers will be able to use the quantification tool themselves, or whether applications and the processing of results should be conducted by independent experts.

10) Participation in previous model comparison studies
If available, results of earlier model comparison studies will be mentioned.

11) Sub-models that can be independently checked
Most models contain different modules and each module has their own functionality. Some of these modules/functions can be considered separately (e.g. water balance), which assists in the identification of sources of model error. This point is also related to point 2.

12) Existing sensitivity analysis
If available, detailed reported sensitivity analysis will give additional information about the most important input parameters of the model. Such work shows that the model has been tested for many different combinations of parameter settings and a large number of different values. An awareness of the most sensitive parameters assists in model applications as modellers are able to focus efforts on the accurate identification of the most sensitive model parameters.

13) Cost indication
The quantification tools can be classified in terms of complexity. Often it is the application of data-based models, such as dynamic process orientated tools, which require the greatest workload (through from data collection, processing, parameterisation, and calibration)

compared to simpler statistical approaches. As time is money, there is therefore a cost implication associated with selecting a particular model which may be a factor in model selection. We provide an indication of the total months of workload needed to apply the quantification tool for a particular "new" catchment.

14) Capability to evaluate nutrient and watershed management strategies (scenario analysis)
The capability of quantification tools to determine the effects of different types of measures will be considered based on the mathematical description of the processes described in the tools. The measures that will be looked at include: nutrient management, land use changes and changes in watershed management.

15) Applicability
The potential applicability of the quantification tool to different environments will be considered by the model owner. This will be a qualitative indication because the "applicability" issue will be examined in greater detail later in Work Package 5 in the EUROHARP project.

3. General description of quantification tools

Within the EUROHARP project one of the primary strategic objectives is the validation and intercomparison of catchment quantification tools on nutrient losses ranging from statistical models (such as export coefficient models or load oriented models) to data-based models (such as process oriented models). The nine nutrient quantification tools that are subject to comparisons and applied on European catchments in EUROHARP are listed in Table 1, together with the name of the modelling institute.

Since the results of the application of the quantification tools will be compared against monitoring data, applications of each quantification tool will need to take into account: point source inputs, natural background losses and retention within the surface waters. The EUROHARP expert group on retention (work package 5) will provide estimates of the latter.

Table 1: *Quantification tools and modelling institute*

QT no.	Name of the tool	Modelling institute
1	NL-CAT (ANIMO/SWAP/SWQN/SWQL)	ALTERRA
2	REALTA	KMM
3	N-LES CAT	NERI
4	MONERIS	FV-IGB
5	TRK (SOILNDB/HBV-N)	SLU / SMHI
6	SWAT	EC-JRC / NTUA / IRSA-CNR
7	EveNFlow	ADAS
8	NOPOLU	IFEN / BETURE-CEREC
9	Source apportionment	NERI

Of the models studied, modelling tools 1, 5 and 6 are amongst the most data-hungry models (highly process orientated). These models typically divide a catchment into unique combinations of land use, level of nutrient input, slope, soil type, hydrological situation/drainage system, and then consider them as homogeneous plots. The location and area of each is known, and the quantification tool is applied to each plot. In the process-orientated tools, the dynamics of the fate of nutrient inputs in the soil are modelled in a two or three-dimensional way, often on a daily basis. All major biological and chemical processes that occur in soils are taken into account (e.g. mineralization / immobilisation; phosphorus (de) sorption). Based on the representation of system processes, nutrient concentrations are calculated. The water flow and particulate flow is modelled (runoff, erosion, subsurface runoff/leaching) in order to assess the total nutrient load to surface waters. In quantification tool 5 detailed process descriptions are made for a number of representative “type fields” with generalised parameterisation, and the results are then transposed to all arable land after classification in a GIS. This reduces the input data demand. In some quantification tools for phosphorus, the processes are only described in detail in the topsoil, since runoff and erosion are the major sources of diffuse pollution. In other quantification tools the leachate is (conceptually) mixed with the leachate of other plots in order to estimate groundwater pollution and the nutrient input to surface waters. Some of the quantification tools take all deeper layers separately into account.

The quantification tools 3, 4 and 7 do not attempt a comprehensive representation of all individual system processes. Instead, they simulate losses by using a series of simpler conceptual, (semi-)empirical or statistical functions. Such tools comprise functions which may retain a physical basis (e.g. soil field capacity) and or may use empirical coefficients that have been found to reproduce observed field and river measurements. These tools may include parameters such as nutrient surplus, nutrient status of the soil, soil type, land cover, precipitation or net precipitation surplus and slope. Most of these tools have component relationships to estimate retention in surface waters in order to estimate the nutrient load at a specific monitoring station. Most of the time these models require less input data then the highly process orientated models, although there are exceptions e.g. quantification tool 3 requires the most detailed input concerning field activities and N-input.

The quantification tools 2 and 8 are relatively low in data input requirements and can be described as balance approaches or risk assessment approaches. With respect to the balance approach (QT 2) most of the complex biochemical reactions in soils are lumped into one retention coefficient for different types of soils and different levels of nutrient status. Most of the time nutrient losses by different pathways are a fraction of the nutrient input or related to the nutrient status of the soil. The risk assessment approach (QT 8) uses categories of risk areas within the catchment based on local circumstances (e.g. slope, soil type, crop type, fertiliser input). Each risk category needs data of the surface water quality in one specific area within the catchment. This value is extrapolated to areas with the same risk class.

The source apportionment quantification tools (QT 9) is the simplest balance method to quantify diffuse nutrient losses and is the common approach proposed in the OSPAR HARP guidelines. In this case the diffuse nutrient pollution is calculated by simply deducting point source contributions from the total measured outlet of nutrients after correction of nutrient retention in surface waters. This methodology does not identify the area or source of the diffuse contribution – which would be needed in order to target mitigation options.

3.1 General description

QT 1 - NL-CAT:

In the Netherlands process oriented models play an important role in the assessment of pollution and the evaluation of intended measures, because trends in water quality parameters as a consequence of fertilisation reduction or water management strategies can be predicted. The far-reaching effects of the intended fertilisation measures on agricultural production justify a thorough examination of the relationship between environmental compartments.

In regions with shallow groundwater tables and water discharge towards surface water, residence times are strongly influenced by the drain spacing and the depth of the local flow system. A sound description of the link between the local system and the regional system is of great importance for water quality simulations, because the greater part of the final discharge concentration depends on processes within the upper layer of the soil system. In the relation between groundwater and surface water pollution, the representation of the hydrological system is of utmost importance. Mechanistic sub-models for water and nutrient behaviour are required because of the combined impact of seasonal variations in meteorology, hydrology, and the timing of fertiliser applications which govern the leaching of N and P to surface waters.

For national evaluation ANIMO is part of a model chain called STONE (Dutch acronym). In this model chain the model SWAP (Soil-Water-Plant-Atmosphere; Van Dam, 2000) is used to generate hydrological input for ANIMO and the model CLEAN is used to generate the manure and fertiliser input for ANIMO over a long-term period. With this model chain the diffuse non-point nutrient losses from agricultural land and nature areas to groundwater and surface waters are modelled. For national studies more general models are used to predict the impact of these nutrient losses from land to the surface waters, together with other nutrient (point) sources, on the chemical and ecological water quality in lakes and the main streams and to predict the nutrient load to the sea.

For regional model application, e.g. catchment scale, more or less the same model chain is used, only the manure and fertiliser information is generated in more detail (using local expert judgement) and other models are used for modelling the retention in surface waters within the catchment and the ecological quality of these surface waters (mainly ditches-streams). To calculate the water distribution, different models have been used in the past in the Netherlands (e.g. SIMWAT; DUFLOW and WATDIS). For the EUROHARP project the model WATDIS (WATer DIStribution model; Smit et al., 1995) is slightly adapted and used to calculate the actually realised distribution of water within a catchment. This Surface Water Quantity Model is called SWQN and is described by Smit et al. (2003). Regarding the modelling of the surface water quality within (large) catchment the model SWQL (Groenendijk and Jeuken, 2003) is used to estimate retention and ecological impact in surface waters. This model version is a simplification of the NUSWA (NUtrient modelling in Surface Waters) model (Van der Kolk et al., 1995). Finally, this whole model chain, together with a discretisation procedure to subdivide the area in homogeneous sub regions is called NL-CAT (Nutrient Losses on CATchment scale; Figure 2). Most of the time also a model for the quantification of fertiliser additions in relation to (international) market structure, fertiliser restrictions and directives is used in order to obtain nutrient inputs to agricultural land. However in this study these data will be gathered for the catchments.

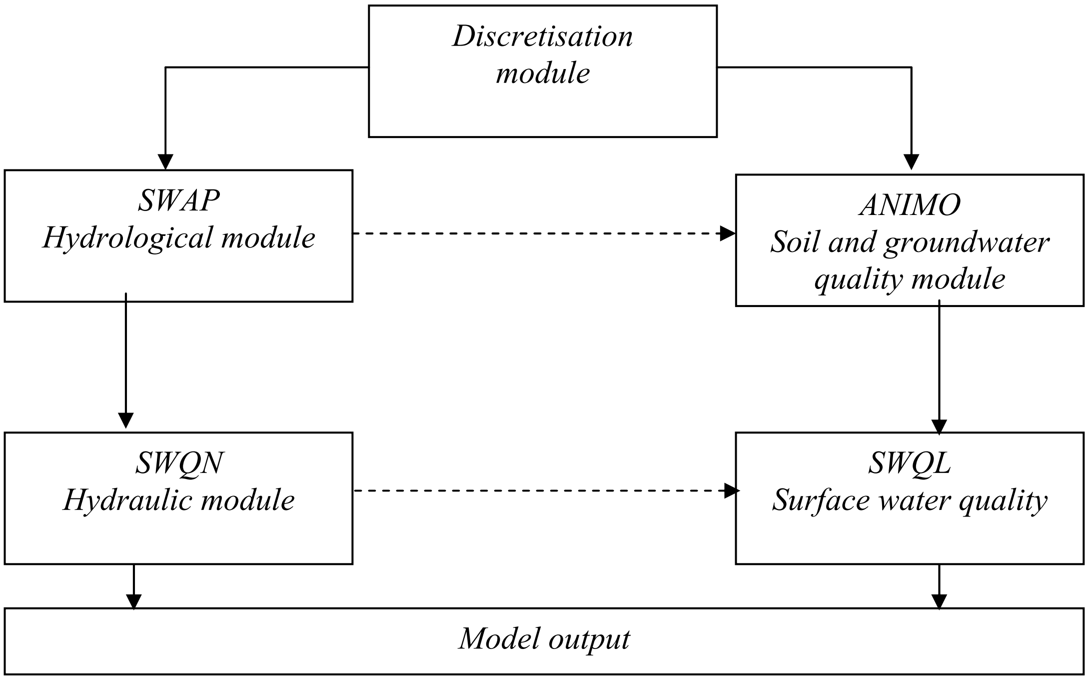


Fig. 2 Model components of the quantification tool NL-CAT

In this paragraph the sub models of NL-CAT are generally described. Since, the scope of the EUROHARP project is the quantify the diffuse pollution from land to surface waters, only the sub-models for water and nutrient behaviour in soils are described in more detail.

Soil and groundwater quantity modelling (SWAP)
Water discharge to groundwater and surface water is schematised by a pseudo-two-dimensional flow in a vertical soil column with unit surface. The ground level provides the upper boundary of the model and the lower boundary is at the hydrological basis of the system defined. The lateral boundary consists of one or more different drainage systems. The position of lower and lateral boundaries depends on the scale and type of model application.

Hydrological data, such as water fluxes and the moisture content of the distinct soil layers, are supplied by an external field plot model (Feddes *et al.*, 1978, Van Dam *et al.*, 1997) or a regional groundwater flow model (Querner & Van Bakel, 1989). The schematisation of the soil profile and the main terms of the water balance for a particular drainage situation are depicted in Figure 3.

In regions with high groundwater levels and water discharge towards surface water, residence times are strongly influenced by the size and depth of the drainage system. In non-point water quantity models, the extent of water flows to each of the drainage systems must be calculated by using drainage formulae applicable to the local flow.

In the non-point water quality models, regional spatially distributed patterns of soil type, land use and hydrology are schematised by a number of homogeneous subregions. The size of a subregion depends on the heterogeneity of these factors and on the ultimate goal of the model application. The boundary between local and regional flow can be defined as the depth below which no discharge to local surface water occurs. Above this depth, the greater part of the precipitation surplus flows to water courses and other drainage systems. This depth depends on the deepest streamline discharging water to the drainage systems.

Once the regional and local flow have been segregated by the position of the boundary surface, the streamline pattern within the top system is schematised into vertical fluxes between soil layers and into lateral fluxes in the saturated zone. Information on water discharges and drainage distances is used to simulate residence times of water and solute in the saturated zone.

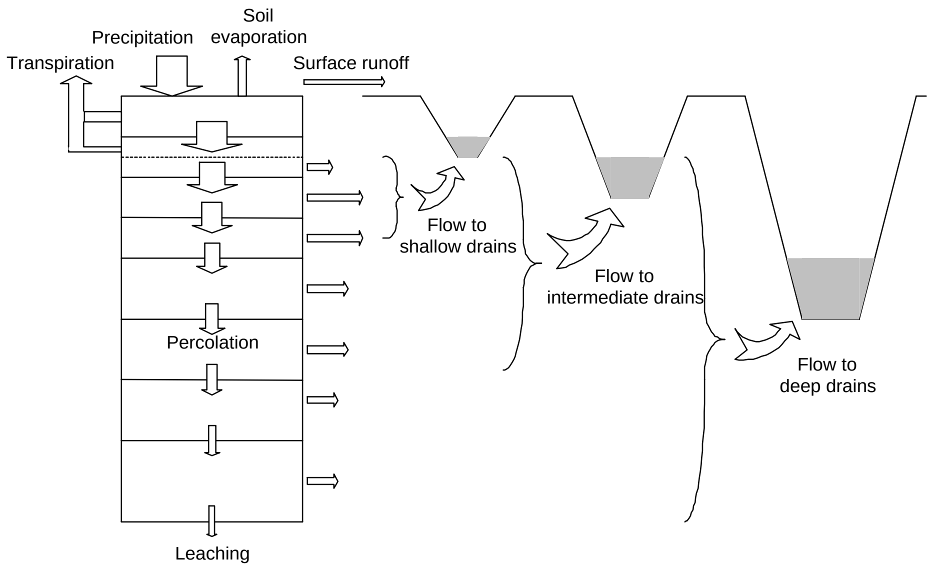


Fig. 3 Scheme of water flows in a soil profile and the main terms of the water balance.

Soil and groundwater quality modelling (ANIMO)

The Dutch quantification tool, called ANIMO, aims to quantify the relation between fertilisation level, soil management and the leaching of nutrients to groundwater and surface water systems for a wide range of soil types and different hydrological conditions. The model was developed in 1985 to evaluate nitrogen losses (Agricultural NItrogen Model; Berghuis -Van Dijk, 1985). In the early nineties phosphorus behaviour was also described and parameterised (Schoumans, 1995; Schoumans and Groenendijk, 2000) and the phosphorus cycle (organic and inorganic) was implemented (Groenendijk and Kroes, 1995). From that moment the model was called *Agricultrural NutrIent Model*. The model ANIMO is a functional model incorporating simplified formulations of processes. The organic matter cycle plays an important role for the assessment of long term effects of land use changes and fertilisation strategies. The upper and horizontal boundary systems of the model are the surface of agricultural land (where the nutrient inputs take place) and the edge of the field/plot (horizontal nutrient out flow). The lower boundary system is, most of the time very low (e.g. 7-15 m below surface level). Therefore, only retention in the soil is modelled.

This Dutch soil and groundwater quality quantification tool ANIMO, focuses on the following processes:

- additions (fertiliser, manure, crop residues, atmospheric deposition),
- mineralization of nutrient compounds in relation to formation and decomposition of different types of organic matter as organic fertilisers, root residues, yield losses and native soil organic matter;
- volatilisation (CO_2, NH_3, N_2, N_2O),
- nitrification of NH_4 and denitrification of NO_3;
- sorption onto and diffusion within soil particles, described by a combination of instantaneous and time dependent sorption and chemical precipitation of phosphates;
- uptake by the vegetation;
- transport of dissolved organic and inorganic nutrients with water flow to deeper soil layers and to adjacent surface water systems; and
- overland flow of dissolved organic phosphorous, inorganic phosphate and particulate phosphate with water flow to adjacent fields (runoff and erosion)

In the most recent version of ANIMO (version 4.0; Groenendijk and Roelsma, 2002) also two other important processes are described:

- (preferential) macro-pore flow
- snow melting

ANIMO comprises description of the organic matter cycle, the nitrogen cycle and the phosphors cycle since these cycles are interrelated in most of the modern farming systems and in soil bio-chemistry.

Figure 4, 5 and 6 shows, respectively, the pathways of carbon, nitrogen and phosphorus losses from agricultural land to surface waters and C, N and P processes implemented in ANIMO.

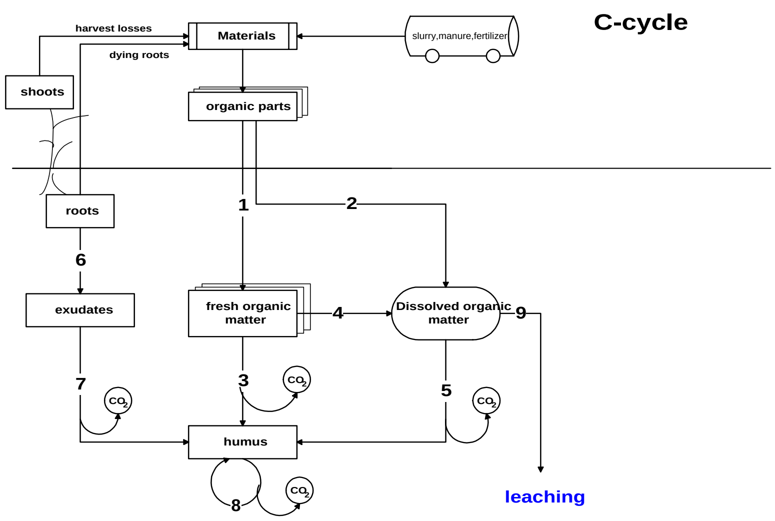


Fig. 4 Relational diagram of the organic matter cycle described in the ANIMO-model

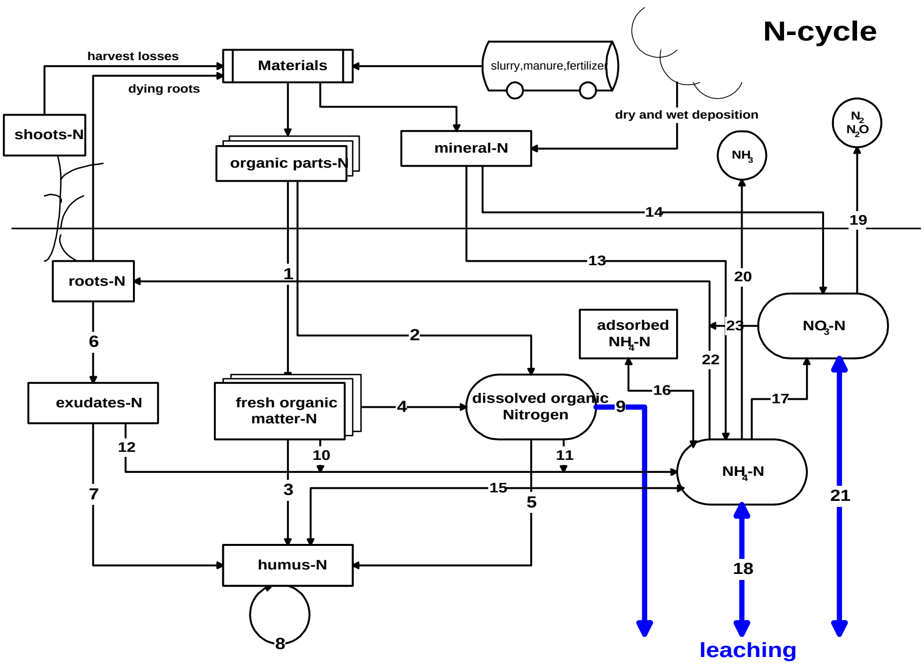


Fig. 5 Relational diagram of the nitrogen cycle described in the ANIMO*-model*

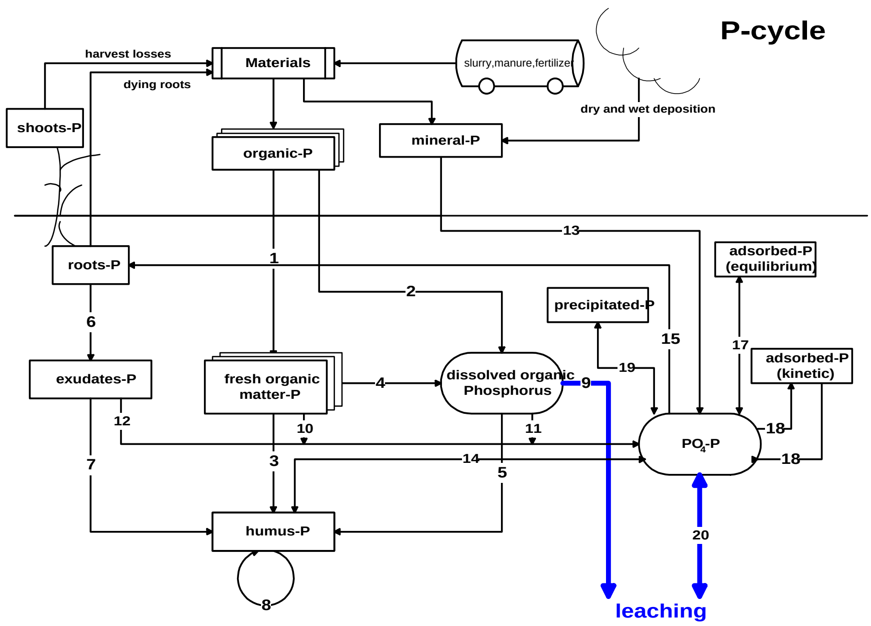


Fig. 6 Relational diagram of the phosphorus cycle described in the ANIMO*-model*

Nutrient losses from land to surface waters
Transport routes from agricultural land are related to surface runoff, leaching to groundwater and leaching to surface water systems (Figure 7).

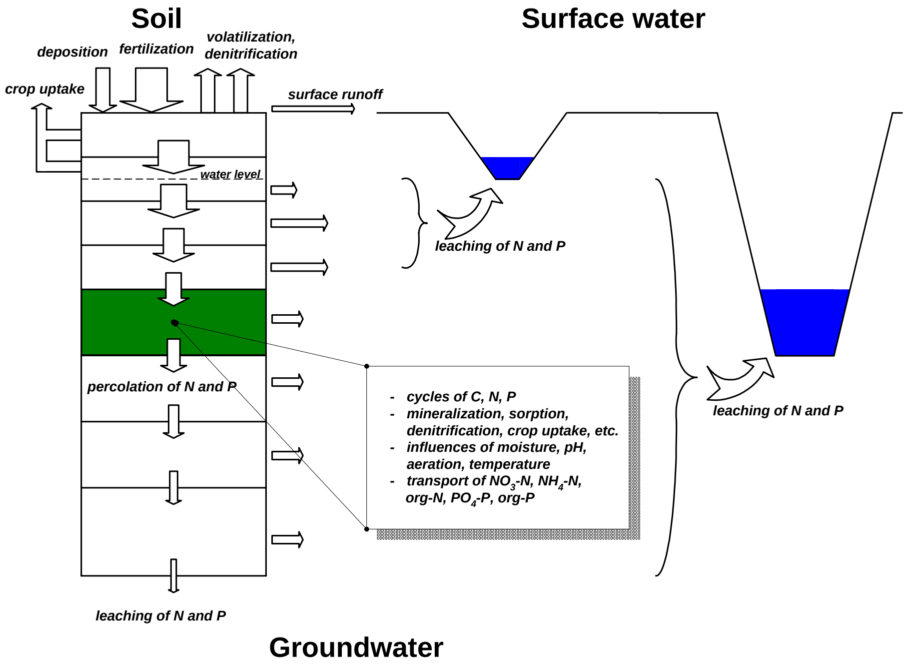


Fig. 7 Transport routes and nitrogen and phosphorus related processes included in the ANIMO model

Surface water quantity model (SWQN)
Within the EUROHARP project the surface water module SWQN will be used. The SurfaceWater module is a distributed surface water quantity model and is based on the description of 1-dimensional flow in linear surface watercourses. The model uses a network based on nodes with connections between them. The nodes contain a certain volume of water based on the actual water level and the dimensions (e.g. length, width and slope) of the canals connected to it. The connections can be defined as open watercourses with a certain resistance, or as a structure (e.g. weir, culvert, pump) with specific parameters. The specifications of structures can be changed in time by providing structure control time series. Water flow between the nodes is calculated as a linear function of the water level difference during the distinguished time steps and the calculated resistance of the connections. Simulation results are redirected to CSV-files to enable easy post processing. Optionally SWQN can send the results to input files for the next step in the model chain: SWQL (also called NuswaLite).

Surface water quality model (SWQL)
The surface Water Quality Model SWQL (NuswaLite; Jeuken and Groenendijk, 2003) calculates the retention and the ecological effects of nutrients in a river basin. The model is a simplification of the NUSWA (NUtrient modelling in Surface Waters) model (Van der Kolk et al., 1995). The model describes the dissolved organic and mineral fractions of nitrogen and phosphorus concentrations in a network of nodes. Also two fractions of living biomass are considered: a floating fraction, which can be transported with water flow, and an immovable fraction having roots in the sediment. Biomass is considered to have a fixed nutrient ratio, so no separate pools of nitrogen and phosphorus in biomass are defined. Besides inflow, outflow (not for immobile biomass) and loading (not for biomass), the following processes are taken into account (Figure 8):

- Growth of biomass with linked uptake of nutrients and limited by solar radiation and nutrient availability
- Death of biomass which adds to the organic nutrient pools
- Degradation of organic nutrients to their mineral forms
- Denitrification of mineral nitrogen
- Linear sorption of mineral nutrients to the sediment

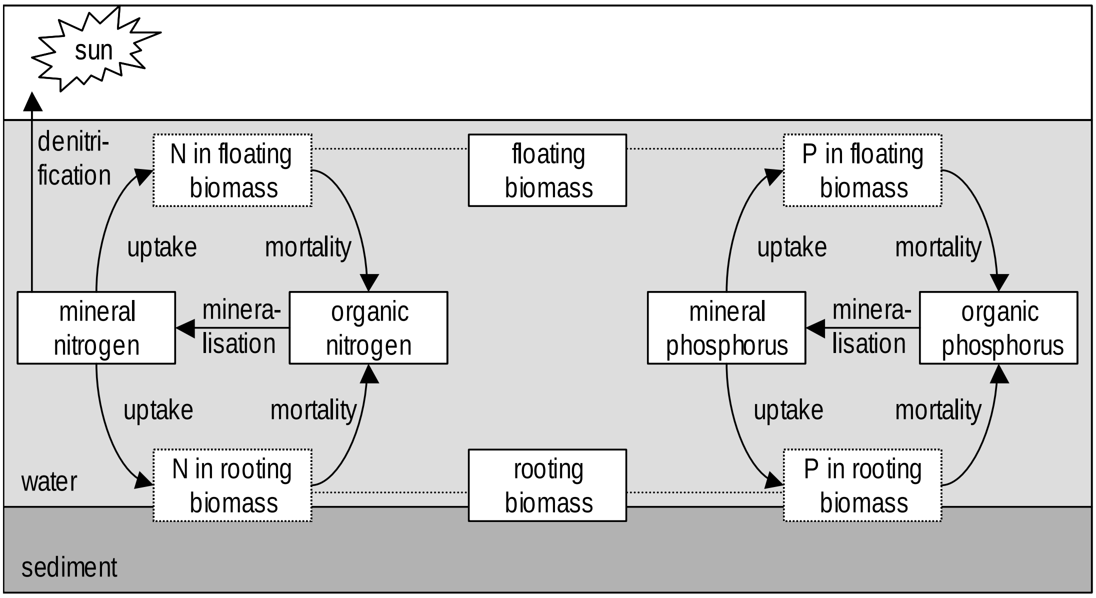


Fig.8 Relational diagram for the nutrient cycles as described in NuswaLite

The set of equations describing these processes is solved using a numerical finite difference solution technique. The time variable is solved analytically which enables the use of large time steps (usually limited to one day due to variability of boundary conditions). Input consists of a network layout and a water balance (as could be provided by SWQN or any other hydraulic model), nutrient loading from various sources (e.g. leaching as calculated by ANIMO or point sources), environmental conditions (e.g. temperature and global radiation), initial conditions and parameter settings.

Applications
Over the last two decades ANIMO/SWAP has been used as a leaching module in several studies at catchment and national scales for the purpose of ex-ante evaluation with respect to fertilisation policy (RIVM, 2000; RIVM, 2001; RIVM, 2002; Boers et al, 1997; Schoumans et al., 2002).

Data Requirement
Geo referenced input data
- ❑ Topography (DEM; layout of surface water system)
- ❑ Met. data
- ❑ Land use
- ❑ Fertiliser / manure application
- ❑ Hydrology
- ❑ Soil
- ❑ Groundwater quality conditions
- ❑ Stagnant surface water conditions
- ❑ Atmospheric conditions

- ❑ Point sources (sewage treatment plants, etc.)

Agronomic definitions
- ❑ Vegetation / crops
- ❑ Fertilisation: definition number of fertiliser / manure types
- ❑ Erosion/Tillage practice
- ❑ Historical data regarding land use and fertilisation (20 - 50 years; sub divided into 5 periods)

Calibration / validation data (time series) at catchment outlet and for sub-watersheds
- ❑ Hydrology: water levels; discharges; groundwater levels
- ❑ Groundwater quality
- ❑ Surface water quality

Operational experience and skills required for users
The NL-CAT model suite comprises a number of complex process oriented models. The main users group consists of applied scientists. Some modules are also used for educational purposes (SWAP model). The most successful applications are expected when a team of professionals (GIS, hydrology, soil science, agronomy) co-operate together in model applications. In the Netherlands, model runs for ex-ante evaluations in the framework of national policy making are conducted through team work.

Participation in previous model comparison studies
A comparison of simulation results of five nitrogen models using different datasets on field scale has been reported for the EC (Vereecken et al., 1991; Soil and Groundwater Research Report II, Nitrate in Soils, Final report of contracts EV4V-0098- NL and EV4V-00107-C, Commission of the European Communities).

Sub-modules that can be checked independently

Model part	Module	Possibilities to check
Soil water balance	SWAP 3.0	Groundwater levels, drain discharges
Nutrient leaching	ANIMO 4.0	Crop uptake, nitrate concentrations in groundwater, Mineral N, P-status of soil, P-contents in soil, N & P discharges tube drains
Surface water quantity	SWQN	Water levels, water discharges at different points
Surface water quality	SWQL	Time series on N & P concentrations at outlet and outlets of sub watersheds.

Sensitivity analysis
Based on sensitivity analysis of different model parameters (Monte Carlo simulations, Groenenberg et al., 2000) it was concluded that variation is N losses to surface waters was highly determined by the N surplus (total N input minus N harvest) and the parameters dealing with organic matter transformations, aeration, temperature and pH. For phosphorus losses the most important parameters were the process parameters of the phosphate sorption reaction in soils (affinity of the soils to sorb phosphate and phosphate sorption capacity of the different soil types).

Cost Indication
For each catchment about 4-6 man-month of workload is necessary to predict the nutrient losses, N as well as P, from rural areas to surface waters. Most of this time is needed for data collection and parameterisation.

Capability to evaluate nutrient and watershed management strategies
The ANIMO model aims to quantify the relation between fertilisation level, soil management and the leaching of nutrients to groundwater and surface water systems for a wide range of soil types and different hydrological conditions. Therefore, nutrient losses to the environment are simulated, with an emphasis on nitrogen and phosphorus leaching to groundwater and surface water systems, as influenced by:

- soil type and climate
- fertilisation
- agricultural practise
- water management

Currently, the model is primarily used for the ex-ante evaluation of fertilisation policy and legislation at regional and national scale.

The hydrological module SWAP3.0 allows for adjusting the boundary conditions and driving forces enabling the simulation of scenarios regarding:

- climate change
- groundwater withdrawal
- water conservation and weir management
- measures taken to combat desiccation
- land use change (e.g. afforestation)

QT2 – REALTA:

The Irish model, called REALTA, uses a self-developed procedure for estimating phosphorus losses from agriculture based on actual measurements obtained from catchment monitoring and management systems. The procedure takes on board detailed knowledge of physical conditions and farming practices in the catchment. Percentage loss figures, initially derived from detailed agricultural studies at mini-catchment and sub-catchment level are linked to an agricultural risk map. Estimated nutrient percentage loss figures can be applied to the total agricultural import to produce an overall estimate for the total agricultural nutrient losses to surface waters. This procedure determines the P loss based on:

- a potential P risk map of the catchment derived by ranking and weighting important geographically distributed input parameters (such as fertiliser and manure loading, soil P levels, runoff risk parameters); and
- the relationships that were derived between the percentage agricultural P loss rates at mini-catchments and sub-catchments and the agricultural risk category.

The results of water quality monitoring programme confirmed the strong correlation between the areas identified as being high or very high potential risk and poor water quality.

Step I) Development of the Potential Agricultural Risk Map

A ranking scheme is developed whereby each of the phosphorus loss indicators is subdivided into zones of relative risk, each of which has a numerical value for scoring purposes. The relative importance between factors is also represented by a further scoring system or 'weighting'.

A 'score' or 'rank' for a given combination of factors affecting loss and transport of phosphorus is developed in two steps:

1. Multiply the weight of each factor by the relative risk associated with the magnitude of each factor; and
2. Sum all of the products derived in Step 1.

The resulting composite map establishes the range of potential agricultural risk areas across the River Basin District.

Step II) Calibration of the Potential Agricultural Risk Map

The potential agricultural risk map is calibrated on an annual basis by the physical measurement of in-stream phosphorus loadings in selected agricultural areas. These physical measurement results are then extrapolated across each of the main subcatchments to enable the quantification of the annual phosphorus export rate from the River Basin Districts.

The application of the model therefore requires a limited programme of physical in-stream measurements in small agricultural areas each year to take account of annual variations in hydrological conditions, farm management practices, and the associated impact on agricultural losses to water.

Step 3: Extrapolation to overall catchment

This uses relationships derived between percentage agricultural loss rates calculated at mini-catchment and subcatchment level and agricultural risk category. Percentage loss factors are applied to the overall catchment using the agricultural risk map

Agricultural nutrient loss rates were extrapolated to the overall catchment as follows:

1. An Agricultural Risk map was developed for the catchment using the Geographical Information System (GIS) to investigate the relationship between a set of agricultural indicators and water pollution potential;

2. Relationships were derived between the percentage agricultural loss rates calculated at mini-catchment and subcatchment level and the agricultural risk category;

3. The percentage loss factors derived from step 2 were applied to the overall catchment using the agricultural risk map.
4. The estimated N and P percentage loss for each of the subcatchments were applied to the total agricultural N and P import from chemical fertiliser usage and pig slurry production.

Figure 9 gives an overview of the methodology to estimate phosphorus losses from agricultural land to surface waters with the model REALTA.

Data Requirement
The main model input parameters, ranked in order of their importance (highest to lowest) are as follows:
(i) Organic Fertiliser Loading; Land Use; Runoff Risk to Surface Waters.
(ii) Soil Phosphorus Levels
(iii) Mineral Fertiliser Loading

Operational Experience and Skill Requirement of Users
REALTA is a simple load-oriented model. The model is essentially a Geographical Information System (GIS) based risk analysis. The operational experience of the user is specifically related to the application and development of relatively simple GIS modelling techniques. To date the modelling has only been carried out using SPANS GIS (Canadian software), however, the modelling techniques can be carried out using ArcView. General water quality background information is also required by the user, along with the understanding and ability to manipulate point source data.

Participation in Previous Model Comparison Studies
The REALTA model was developed in Ireland with the aim of using existing detailed monitoring data to quantify diffuse sources from a predominantly grassland, agricultural catchment. Information from the Lough Derg and Lough Ree Catchment Monitoring and Management System was used to develop the risk-based assessment as a means of quantifying diffuse sources from the catchment for the purpose of implementing the HARP Guidelines. The model has only been used on the Lough Derg and Lough Ree Catchment and has not been used in any other modelling studies.

Sub-modules that can be independently checked
The REALTA model is a simple load oriented model which does not include modules or sub-modules.

Existing Sensitivity Analysis
There have not sensitivity analysis carried out.

Cost
It is estimated that the REALTA model requires 2-3 man-months to ‘set up’ and apply to each catchment.

Capability to evaluate nutrient and watershed management strategies (Scenario Analysis)
The REALTA model can calculate the P load reduction expected as a result of abatement measures in priority areas.

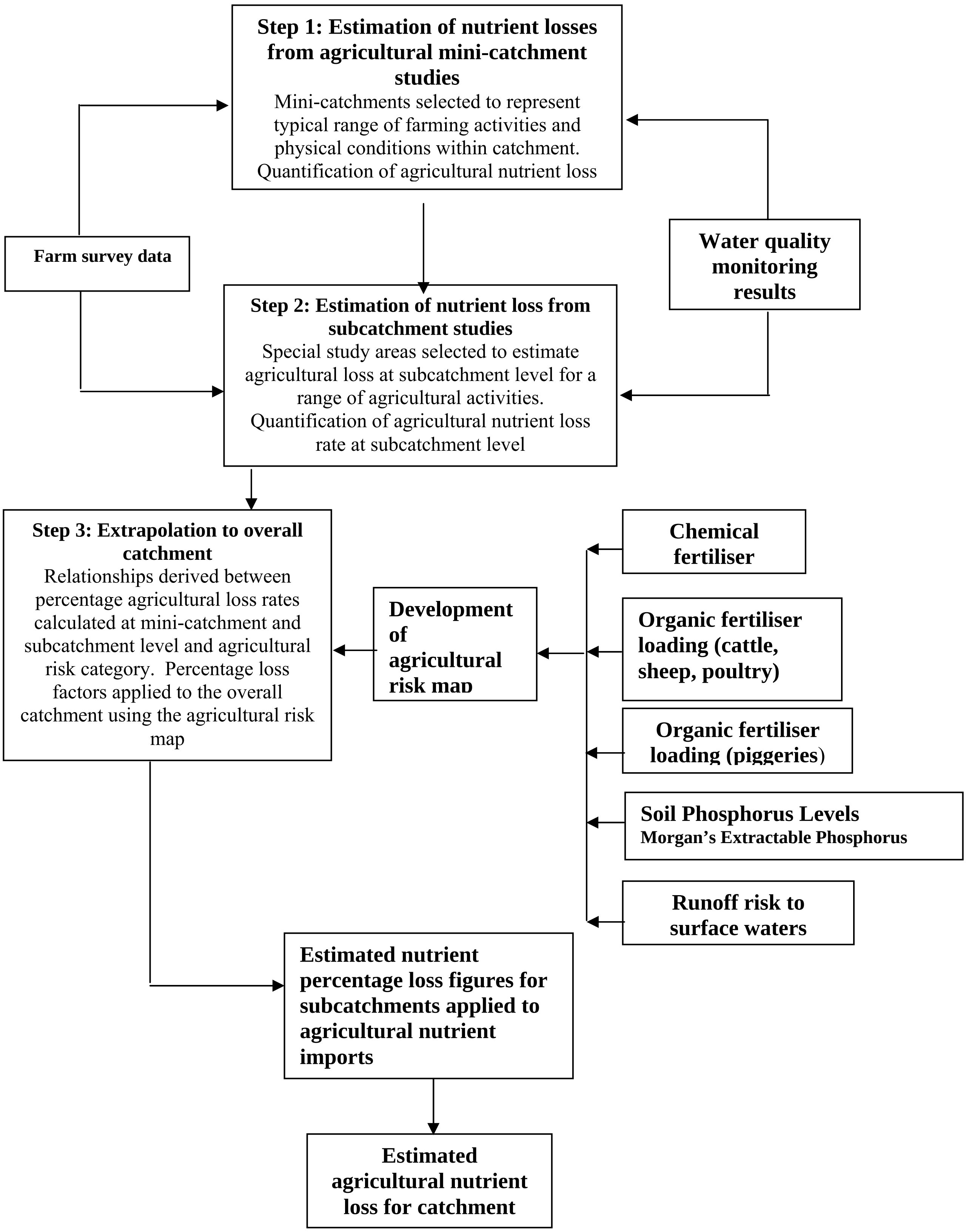


Fig. 9 Components of the quantification tool REALTA

QT 3 – N-LES CAT:

The Danish N-LES CAT is an empirical-conceptual methodology for calculation of annual values of nitrogen losses at the catchment scale. The core of the concept – the N-LES model for arable land - needs input on crop rotations, soils and nitrogen input as well as on the amount of water percolating through the root zone. The root zone water balance is calculated by a precipitation-evaporation model, EVACROP, on a daily basis. EVACROP comprises conceptual models for describing vegetation and for calculating the water balance. The model has modest requirements for data. Daily values of precipitation, temperature and potential evapotranspiration area required. The most important soil and crop parameters must also be specified. The N-LES model comprises a combination of additive and multiplicative effects. N-LES was developed based on 600 observations of annual leaching of nitrogen from the root zone from both experimental fields and fields in normal agricultural production in Denmark. The model explained 68% of the observed variation. The systematic effects included in the model are: level of total nitrogen added in the crop rotation; fertilisation in spring; autumn fertilisation; nitrogen left by grazing animals; effect of ploughing-in of grass; soil type (clay and humus content); water percolation through the root zone, and crop type. N-LES has been used in Denmark since 1992 as a tool for evaluating the effect of policy measures for assessing diffuse nitrogen pollution from agricultural production.

In the N-LES CAT concept (Figure 11) a catchment is divided into a number of subcatchments. N-LES is run for each subcatchment on a number of representative combinations of land management, soils and climate. Root zone leakage concentration from non-arable land is included, but needs to be input to the model. Retention during subsurface transportation (groundwater) is calculated using a calibration procedure, while retention in surface waters is an input to the model (calculated separately in EUROHARP). Retention in groundwater is calculated in the following way: The river hydrograph obtained at the subcatchment outlet is divided into three flow components, (i) Q_{95}, the discharge which is exceeded 95% of the time, and which depicts the deepest groundwater, (ii) $Q_{OF,}$ derived by using the BFI-index (Centre for Ecology and Hydrology, UK), and which is the most quickly responding flow component, and (iii) Q_{IF}, which is the difference between the BFI-derived slowest responding component and Q_{95}. Q_{OF} is assigned a nitrate concentration which is an area-weighted mixture of root zone leakages from arable and non-arable land. Q_{95} is assigned the nitrate concentration measured in deep groundwater in the subcatchment. Q_{IF} is initially assigned the same nitrate concentration as Q_{OF} and subsequently a retention constant is calibrated against measured river nitrate load. For predictive purposes the concentration of the Q_{95} flow component and the subsurface retention are assumed constant.

Data requirement

The following data information is needed to set up the model:

- level of total-nitrogen added in the crop rotation; fertilisation in spring; fertilisation in autumn.
- nitrogen left by grazing animals;
- nitrogen fixation by leguminous plants;
- timing of ploughing-in of grass;
- soil type;
- water percolation through the root zone;
- crop type (main crop and winter or catch crop)

Operational experience and skills of users

Hydrological and agronomic insight is required in preparing data for the sub-models, however, submodels are well-described and easy to run.

Participation in previous model comparison studies

Comparison between N-LES and DAISY on a Danish dataset has been made (Thirup, 2000).

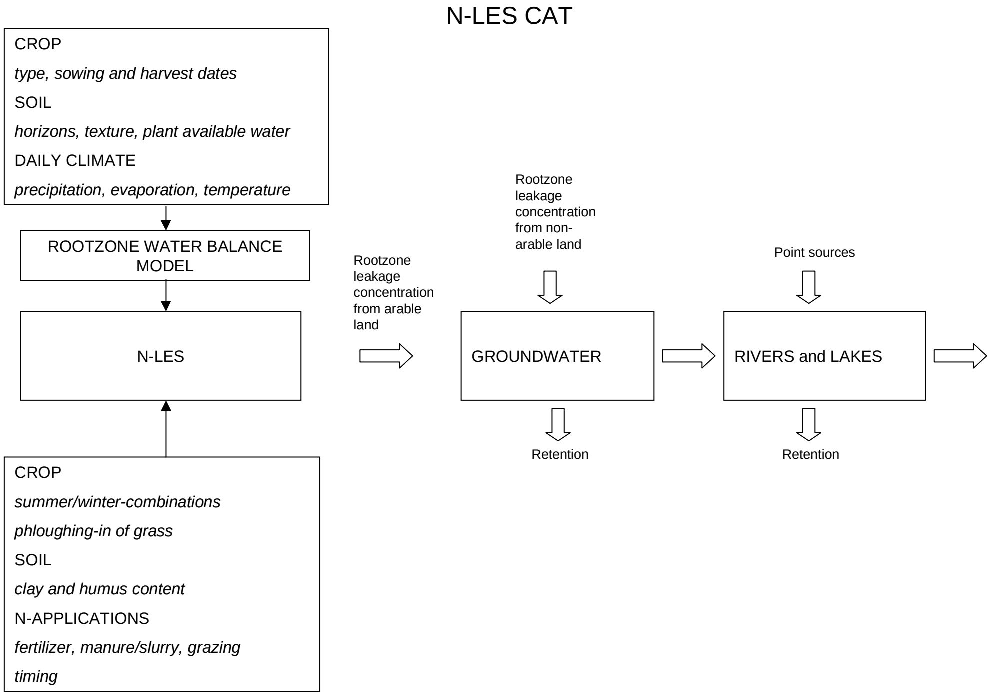


Fig. 11 Overall scheme of Quantification Tool N-LES CAT

Sub-modules that can be independently checked
The hydrological sub-module EVACROP calculates evapotranspiration and percolation out of the root zone. N-LES calculates the nitrate-concentration in the water leaving the root zone.

Existing sensitivity analysis
A sensitivity analysis is being undertaken during 2003.

Cost indication
Dependent on available data. An expectation of 1-2 months per catchment.

Capability to evaluate nutrient and watershed management strategies (scenario analysis)
Good – as long as the scenario is not extended beyond the validity of the model.

QT 4 - MONERIS:

The model MONERIS (**MO**delling **N**utrient **E**missions in **RI**ver **S**ystems) was developed for the investigation of the nutrient inputs via various point and diffuse pathways in German river basins. The basis for the model is data on runoff and water quality for the studied river catchments and also a Geographical Information System (GIS), in which digital maps as well as extensive statistical information are integrated.

While the point inputs from municipal waste water treatment plants and from industry are directly discharged into the rivers, the diffuse entries of nutrients into the surface waters represent the sum of various pathways which have been realised over the individual components of the runoff. The distinction of these individual components is necessary because both the concentrations of materials and the processes are at least clearly distinguished from one another. As a consequence, there are at least four different paths to consider (Figure 10):

- Direct nutrient input on the water surface area by atmospheric deposition,
- Nutrient input into the river systems by surface runoff,
- Nutrient input via interflow which represents a fast subsurface flow component and
- Nutrient inputs via base flow (groundwater) realised by the slow subsurface flow component.

This distinction is not sufficient for the material inputs coupled to surface runoff and the interflow. With surface runoff, inputs of dissolved substances via surface runoff and entries of bound nutrients and suspended particulate matter via erosion must be distinguished. Further it should be considered that the processes coupled to surface runoff depend on the nature of the area. As a result, surface runoff from paved urban areas must be separately quantified.

Interflow can originate both under natural conditions and through human activities. In particular, inputs from tile drainage must be considered separately. The quantification of the input of substances via natural interflow and the drains is particularly complex. During this study, an attempt will be made to estimate the proportion of tile-drained areas in the German catchment areas. However, regionalized estimates of nutrient inputs via natural interflow could not yet be carried out because hydrological models for the calculation of the interflow share of the total runoff are not available for all German river basins.

In addition to the inputs from the tile-drained areas, all other subsurface flows will be summarised in the groundwater inputs. Estimates for the following specific inputs (Figure 10) are possible for the catchment areas considered:

- Point sources
- Atmospheric deposition
- Erosion
- Surface runoff
- Urban areas
- Tile drainage areas
- Groundwater

To quantify and forecast nutrient inputs in relation to their cause requires knowledge of transformation and retention processes. This is often not possible through detailed dynamic process models because the current state of knowledge and existing databases are limited for medium and large river basins. Therefore, existing approaches of macro-scale modelling are used, and if necessary attempts will be made to derive new applicable conceptual models for the estimate of nutrient inputs via the individual diffuse pathways.

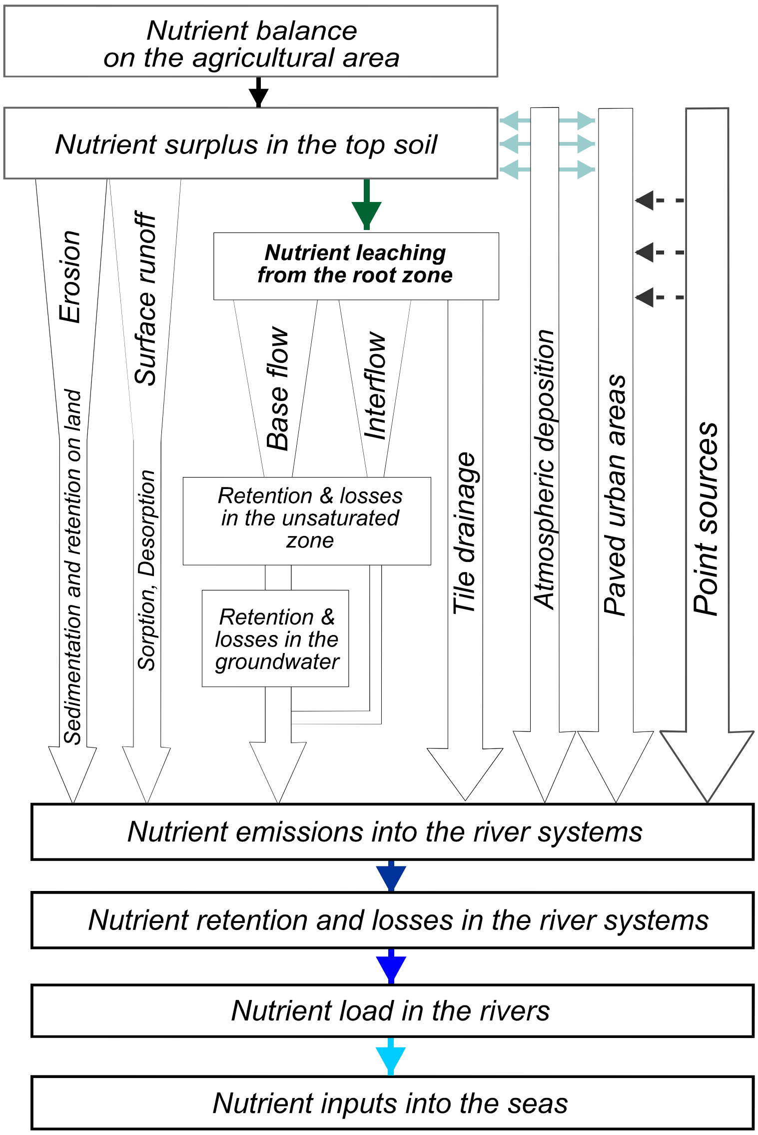


Fig. 10 Pathways and processes considered in the model MONERIS.

WERNER & WODSAK (1994), BEHRENDT (1996A), BRAUN ET AL (1991), PRASUHN & BRAUN (1994) and PRASUHN ET AL (1996) have already successfully undertaken the estimation of losses from land to water bodies, not only for administrative units but also for river catchments. With a view to the Water Framework Directive, this study also focuses on the estimate of nutrient inputs in river catchments, where a size of about 1,000 km^2 has been chosen for the lower limit of the investigated river basins. This has, in contrast to the preceding Germany studies (e.g. HAMM, 1991), the advantage that measurements for these rivers can be used as a control of results and on the other side that maps can be presented showing the regional differences in the nutrients inputs in all German river basins. These allow the derivation of regionally different measures for the reduction of nutrient inputs.

Data requirement

Meteorological time series, measured flow data, land use map, soil map, topography, statistical data about agriculture, sewer systems, treatment plants, water quality measurements (historical data: precipitation, surplus)

Operational experience and skills requirements of users
Basic knowledge in a GIS system and knowledge about spreadsheet analysis. The model was successfully used for the evaluation of the nutrient input in different water basins (Behrendt et al., 1999; Behrendt et al., 2002a; Behrendt et al., 2002b).

Participation in previous model comparison studies
The model is applied within the EU-project BUFFER and STREAMS for about 10 further small catchments in Europe and in DANUBS for the whole catchment of 803000 km² as well as in 6 case studies. Within the EUROCAT project the application of the model to the catchments of Po, Axios and Vistula is done by the responsible groups in these countries.

Sub- modules that can be independently checked
Some sub-models as a whole can be independently checked:
- N-concentrations in groundwater (with regional groundwater data and measurements in rivers during low flow)
- N-concentrations in tile drained areas (measurements and literature)
- N/P concentrations in urban systems (measurements of N and P in sewer systems and overflow (combined/separate sewer)
- Erosion (sediment transport measured (above of a critical discharge)
- Retention of TP, DIN and TN in the surface waters of a catchment

Existing sensitivity analysis
A sensitivity analysis has not been carried out.

Cost indication
About 1-2 man-months are needed to set up the model for a new catchment.

Capability to evaluate nutrient and watershed management strategies (scenario analysis)
Different type of scenario can be analysed, for instance changes in land use, and surplus of nutrients. Changes in water flow as a result of changing the water management within the region can not be evaluated.

Applicability
It is easy to use (some diploma thesis work done) and the time requirements depend on the availability of the input data.

QT 5 – TRK (SOILNDB/HBV-N):

The Swedish TRK system has been developed to calculate gross and net load and source apportionment of nutrients for national assessments of progress towards environmental targets regarding reductions in eutrophication in surface waters. The TRK system has further been developed to provide the option for scenario analysis e.g. mitigation options at subcatchment level associated with agricultural practices. To permit assessment of the most effective measures, and to avoid the large effects due to inter-annual variations in climate, results are presented from the system as long-term climate-normalised load for a specific year. The models included in the system however, can provide a daily output resolution. The results from the system have been used for international reports on the transport to the sea, for assessment of the reduction of the anthropogenic load on the sea and for guidance on effective measures for reducing the load on the sea on a national scale (Naturvårdsverket 1997, Brandt and Ejhed 2002).

The TRK system consists of a GIS and database that prepares input data for models included in the system, and calculates gross and net load and source apportionment. Calculations of both N and P are included from both diffuse sources and point sources, including calculations of hydrology and nitrogen retention in soils, rivers and lakes. P calculations are however limited to gross load since a P retention model is not included, but developments have been initiated.

The TRK-system includes two dynamic simulation models. Firstly, the SOILNDB model that is a one-dimension model, describing nitrogen dynamics and losses in soil profiles in arable land. Nutrient losses from arable land are calculated from areas with a unique combination of crop, soil type, region, and climate and fertiliser regime from the root zone or deeper. A method for calculating a number of leaching estimates for different typical cropping situations has been developed. Outputs as leaching (in mg/l) from different combinations of arable crops, soils and fertiliser regimes, are input data to the second model (HBV-N). In this more conceptual model, root zone concentrations are assigned to various land-use categories (i.e. from pasture, forest, and other land) to water percolating from the unsaturated zone to the groundwater. The runoff model calculates daily runoff from the various land uses in subbasins. The summarised soil leakage is mixed with load from rural households, and point sources are added to the river discharge as well as atmospheric deposition. In addition to the mixing of waters and various loads through the river network, turnover processes (retention) in the groundwater (below root-zone) and ditches, rivers and lakes are simulated both for inorganic and organic nitrogen.

Figure 12 shows the schematic processes of TRK system and includes the following steps:

1. Import spatially distributed input data to produce point and diffuse sources, hydrology and retention.
2. Preparation and coupling of distributed land-use categories to other data and subcatchments, and coupling of point sources to subcatchments using GIS;
3. Import land use to the HBV-model calculations followed by export of hydrology for the SOILNDB calculations.
4. Import agricultural data (crops, soils, practices), meteorological data and hydrological data to the SOILNDB model. Calculations and export of leaching concentrations from arable land.
5. Calculations and export of P losses from arable land using HBV hydrology, arable land area, soil data and livestock numbers in a regression model.
6. Export all compiled data of diffuse sources (leaching concentrations and land-use area) and point source discharge to HBV and HBV-N models. Calculation of nitrogen transport and retention in soils, rivers and lakes.
7. Import retention from HBV & HBV-N model calculations.
8. Compilation of gross and net load and source apportionment.

The results are presented in the GIS, and source apportionment is made for each sub-basin as well as for the whole river basins.

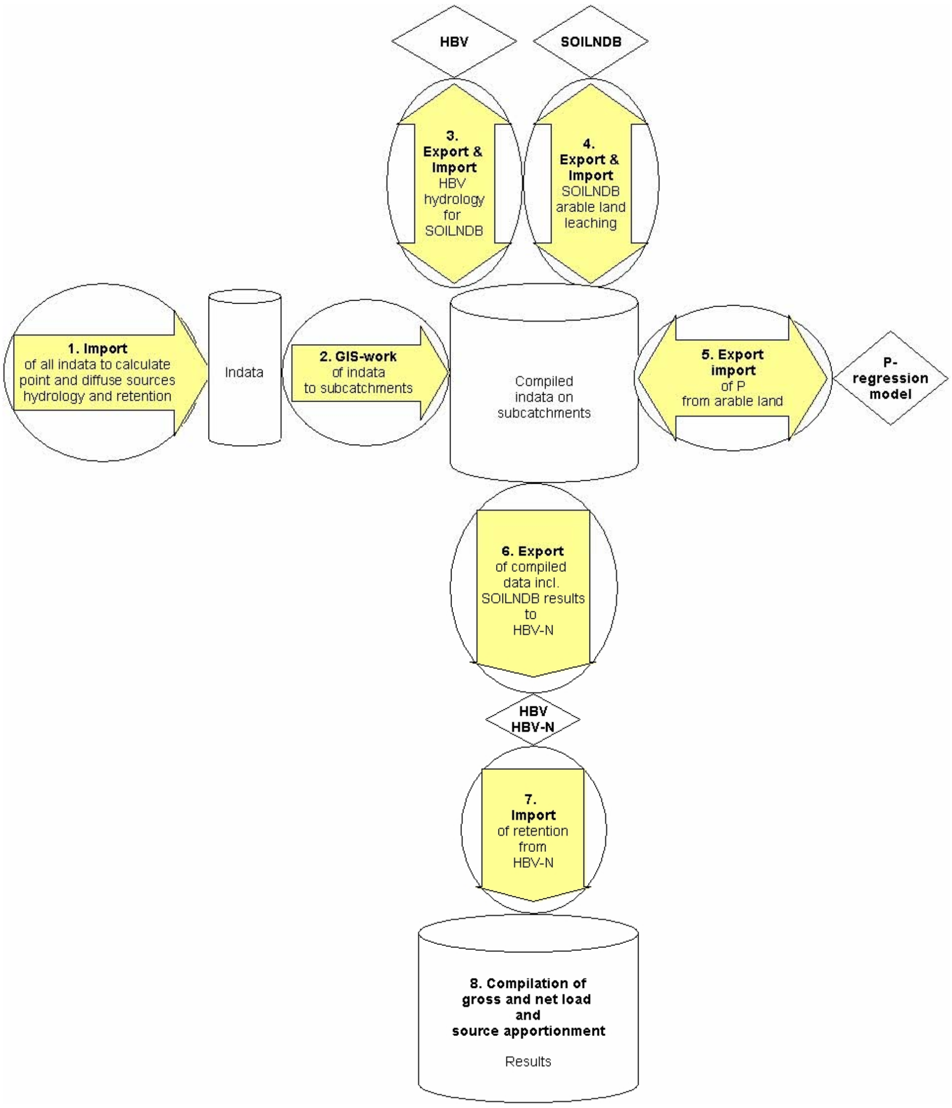


Fig. 12 Pathways and processes described in the Quantification Tool TRK

SOILNDB: N- leaching from arable land

Generalised N root-zone leaching estimates for arable land are calculated using the SOILNDB modelling tool (Johnsson et al., 2002). The method is based on calculating a number of standard N leaching rates (i.e. nitrogen leaching from the root zone for a specified year if the weather and harvest would have been normal) for a number of combinations of soils, crops and fertilisation forms and regions (catchment, area etc.). For this calculation the following is used: SOILNDB, a crop rotation generator, long-term meteorological data, agricultural statistics of crops and area distribution, standard yields, normal fertilisation rates and crop management information. Leaching is simulated for a large number of years using the meteorological timeseries to get acceptable mean values of the standard leaching rates for the different crop-soil combinations. Thus, leaching estimates are normalised with respect to year to year variation in weather conditions and crop production. The method of calculating leaching estimates was developed by Hoffmann & Johnnsson (1999) and Johnsson & Hoffmann (1998) and has been further developed by Johnsson & Mårtensson (2002). The system has been used for calculating leaching estimates for combinations of different climates, soil textural classes, crops, organic matter classes and fertilisations regimes in the Nordic countries and Sweden (Johnson & Hoffmann, 1996; Johnsson & Hoffmann, 1998, Johnsson & Mårtensson, 2002).

SOILNDB is a management oriented modelling tool based on the one-dimensional SOIL-SOILN models describing N dynamics and losses in arable soils, a parameter database and parameter estimation algorithms. The soil N model, SOILN (Johnsson et al., 1987) is coupled in series with the soil water and heat model, SOIL (Jansson & Halldin, 1979; Jansson, 1991). SOIL provides driving variables for the SOILN model, i.e., infiltration, water flow between layers and to drainage tiles, unfrozen soil water content and soil temperature. The SOIL model includes snow dynamics, frost, evapotranspiration, infiltration, surface runoff and drainage flows as well as water uptake by vegetation. The SOILN model includes the major processes determining inputs, transformations and outputs of N in arable soils: inputs of fertiliser and deposition; mineralisation dependent on soil temperature and moisture; decomposition to CO_2, humus and recycling within the pool; soil temperature function, Q10, for regulation of all biological processes; plant uptake from empirical functions; denitrification dependent on soil temperature, soil oxygen status and soil nitrate content (Figure 13). Nitrate transport is calculated as the product of water flow and nitrate concentration in the soil layer. Ammonium is considered to be immobile in the soil profile. Gross load from arable land is calculated using spatial distribution of crops and soil types.

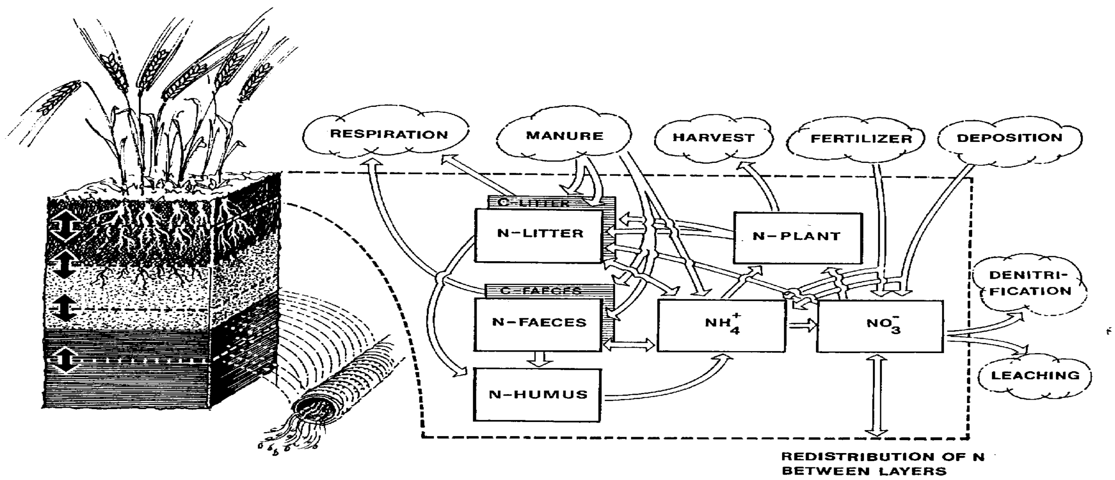


Fig. 13. Structure of the nitrogen model SOILNDB showing state variables (boxes) and flows (arrows) included in the model. The structure is replicated for each layer. Areas within the dotted line represent the top layer of the soil. Layers beneath have the same structure but have no direct input through fertilisation and deposition (Johnsson et al., 1987).

<u>HBV: Catchment modelling of water discharge</u>

The HBV model (Bergström, 1976 and 1995; Lindström et al., 1997) is a conceptual, continuos, dynamic and distributed rainfall-runoff model. When applying the model the catchment is divided into several coupled sub-basins. The daily water balance is calculated for each sub-basin using daily precipitation and temperature data from climate stations. It provides daily values of spatial precipitation, snow accumulation and melt, soil moisture, groundwater level, and finally, runoff from every sub-basin, and routing through rivers and lakes. The model is calibrated and validated against observed time-series. The HBV model has been applied in more than 40 countries over the world and is used operationally in the Nordic countries. Normalised water flow is based on an average from 10-20 years of daily modelling.

<u>HBV-N: N transport and retention</u>

The HBV-N model simulates N transport, residence and retention in groundwater, river and lake systems at the catchment scale (Figure 14). The N model, is based on the HBV-model and has separate routines for daily simulations of inorganic and organic N (Arheimer and Brandt, 1998 and 2000). The soil leakage from different land uses is mixed with discharge from rural households in the groundwater. Concentration variations in the local runoff, due to biological and chemical processes in e.g. open ditches and riparian zones, are described with simple functions mainly based on temperature, concentration and hydrology. The local N runoff is then mixed with contributions from upper sub-basins and lake water. In the river and lake routines, N atmospheric deposition on the water surface and load from industry and treatment plants are included. N retention is calculated in rivers and lakes. The inorganic N may be reduced due to denitrification, sedimentation and biological uptake, while organic N may increase due to biological production or decrease by sedimentation and mineralisation. These processes are also simulated with simple conceptual functions. The N routine is calibrated and validated against observed time-series. N transport and retention are normalised from temporal weather and flow variations using averages from 10-20 years of daily modelling.

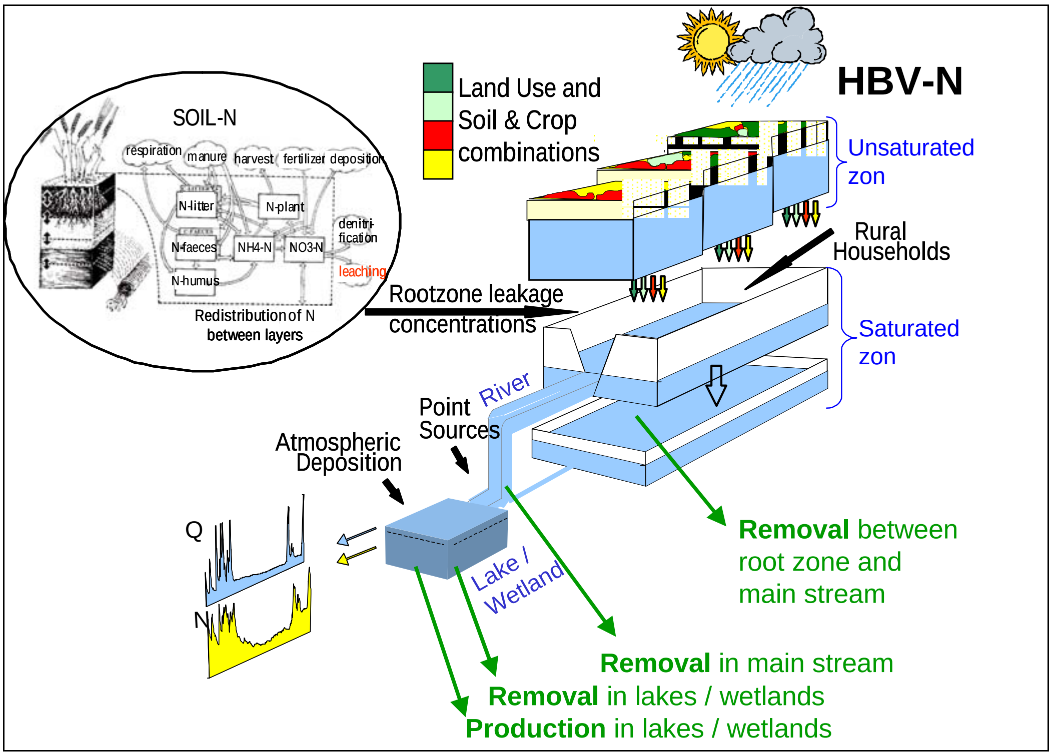


Fig. 14. Structure of the HBV-N model showing the water and nitrogen mixing in a basin. Retention and nitrogen transformation occurs in the groundwater, in rivers and in lakes.

P-leaching from arable land:
P transport is based on water discharge simulated by HBV linked to multiple regression models. Four parameters influence the P concentration from arable land; livestock density, P concentration in topsoil, duration of high water flow and soil specific area.

So, the Swedish TRK system consists of GIS and database setup preparing input data for included models and producing gross and net load and source apportionment of nutrients on subcatchment level for national applications regarding long-term normalised nutrient data valid a specific year. The TRK system includes several models, of which the most notable are the models SOILNDB (for root zone leaching of nitrogen), HBV (for water balance and discharge) and HBV-N (for nitrogen transport and retention in catchments). The scientific evaluation below (8-15) is focused on these three model components of the TRK system.

Data requirements

General TRK: Land cover data including paved surface area, soil texture data, Soil USDA class data, crop area, phosphorus soil data, livestock density, runoff data from HBV, N deposition, leaching data from SOILNDB for arable land and leaching average data from long-term measurements regarding other land-use, point source position and discharge data, percentage of separate sewer for paved surfaces, rural household position and discharge, retention in % from HBV-N. Data are compiled at subcatchment level.

SOILNDB: meteorological data, soil type (texture) and average soil organic matter, crop distribution, crop management and yield, N fertilisation and manuring, N fixation rates in ley, deposition rates, non-existent crop sequence combinations.

HBV: subbasin division and coupling, altitude and land cover distribution, time-series of precipitation and temperature (time-series of observed water discharge at some site).

HBV-N: results from HBV and SOILN, crop and soil distribution, leaching concentrations from other land use, location and emissions from point sources and rural households, lake depths and atmospheric N deposition (time-series of observed riverine N concentrations in some site).

Operational experience and skills requirement of users

Overall TRK: Advanced GIS knowledge and basic database skills.

SOILNDB: Basic knowledge in soil science, agriculture (crop management) and agricultural water quality management. Training in using the SOILNDB modelling system.HBV: Two weeks of training for model setup and applications. Basic knowledge in hydrology.

HBV-N: Two weeks of training for model setup and applications. Basic knowledge in hydrology (and limnology).

Participation in previous model comparison studies

SOILNDB: SOILN model: Comparison of models for nitrogen turnover in the soil-plant system (De Willigen, 1991), Comparison of agroecosystem models (Diekkruger et al, 1995), Comparison of soil nitrogen models (Wu & McGechan, 1998; McGechan & Wu, 2001).

HBV: Intercomparison of 10 models of snowmelt runoff in 6 catchments (WMO 1986), Simulated real-time intercomparison of 14 hydrological models in 3 catchments (WMO 1992), Comparison with the Xinanjiang model (Zhang and Lindström (1996), HYRROM, SMAR, ARNO (Bruen, 1999).

HBV-N: Compared with MESAW (Lidén et al. 1999), MONERIS (Fogelberg, 2003); Model results compared to previous results of various models (Arheimer and Brandt, 1998).

Sub-modules that can be independently checked

General TRK: Included models.

SOILNDB: SOIL-SOILN models: soil water flow, soil heat flow, soil frost, snow accumulation/melting, evapotranspiration, soil N mineralization, denitrification, N transport.

HBV: Precipitation interpolation (Johansson, 2002), Snow accumulation and melt (Brandt and Bergström 1994; Sandén and Warfvinge 1992; Turpin et al. 1999), Evapotranspiration (Eklund, et al. 2000), Soil moisture accounting (Andersson 1988, Andersson and Harding, 1991; Sandén and Warfvinge 1992), Recharge and discharge of the saturated zone (Bergström 1976, Bergström and Sandberg 1983, Lindström et al. 1997; 2000), Pathways and travel times by using stable isotopes (Lindström and Rodhe 1986, Lindström 2000), Integrated internal model validation of snow depth, groundwater, soil frost depth (Lindström et al., 2002)

HBV-N: N discharge, turnover, and concentration in individual water bodies, such as groundwater, rivers, wetlands and lakes (Arheimer 1998, Arheimer and Brandt 1998).

Existing sensitivity analysis

Overall TRK: Analysis is performed within the component models SOILNDB, HBV, HBV-N and the phosphorus regression model. Results of net load of nitrogen from the overall TRK-system are compared to monitoring data.

SOILNDB: Sensitivity analysis of different parameters in the SOILN model has been done by e.g. Larocque and Banton (1994); Silgram (1997); Tychon et al. (1998); Wu et al. (1998). Many applications and tests of the model, where simulated output is compared with measurements, also include partial analysis of parameter sensitivity (SOILNDB: Larsson & Johnsson, 2003; SOILN: Alvenäs & Marstorp, 1993; Aronsson & Torstensson, 1998; Bergström & Jarvis, 1991; Bergström & Johnsson, 1988; Bergström et al., 1991; Blombäck et al., 1995; Blombäck & Eckersten, 1997; Borg et al., 1990; Eckersten & Jansson, 1991; Gustafson, 1988; Jansson & Andersson, 1988; Jansson et al., 1989; Jansson et al., 1987; Johnsson, 1991; Johnsson et al., 1987; Johnsson et al., 1991; Lewan, 1993; Lewan, 1994; Torstensson. & Johnsson, 1996; Ragab et al., 1996; Kätterer et al., 1999; Ulén, 1998; Torstensson & Aronsson, 2000).HBV: Parameter sensitivity (Lindström and Harlin 1992, Harlin and Kung 1992, Seibert 1997, Lidén and Harlin 2000), Extrapolation analysis (Harlin 1992).

HBV-N: Parameter sensitivity (Arheimer and Wittgren 1994, Arheimer 1998). Impact of hydrological model structure and calibration (Pettersson et al., 2001).

Phosphorus regression: Parameter sensitivity (Ulén, B., Johansson, G. & Kyllmar, K. 2001).

Cost indication (based on work load to set up and apply the quantification tool)

Whole TRK: 0.5-4 months depending on data quality.

SOILNDB: 1-4 months depending on data quality, etc.

HBV: for an experienced modeller about 2 weeks, if required database is available.

HBV-N: for an experienced modeller about 2 weeks , if required database is available.

Capability to evaluate nutrient and watershed management strategies (scenario analysis)

General TRK:The model components in TRK are constructed to be used in scenario analyses and several scenario studies have been performed (Nitrogen from land to sea, Naturvårdsverket 1997).

SOILNDB: Impact of changes in agricultural management practices on nitrogen leaching from arable land (e.g. Johnsson, 1991, Hoffmann & Johnsson, 2000; Hoffman et al, 2000, Granlund et al, 2000), climatic change on N losses from arable land (Kallio et al., 1997), impact of changes in atmospheric N deposition on N losses form forests using SOIL/SOILN (Silgram, 1997).

HBV: Impact on water discharge from: Forest clearcut (Brandt et al. 1988), Soil drainage (Iritz et al. 1994, Johansson and Seuna 1994, Andersson and Arheimer 2001), Climate change (Bergström et al. 2001, Gardelin et al. 2001), Wetland constructions (Arheimer and Wittgren 1994).

HBV-N: Impact on N load through: Constructed wetlands (Arheimer and Wittgren 1994; 2002), Changes in national arable leaching (Arheimer and Brandt 2000), Historical changes in human impact and climate (Andersson and Arheimer 2001; 2003), Evaluation of environmental goals (Wittgren et al, 2000) and Remedial measures to reduce coastal eutrophication (Arheimer et al., 2003).

Applicability

The TRK system is constructed to be relatively easy to apply, by using general regional parameter data. The models included have been in use for decades.

SOILNDB: Relatively simple to apply in relation to the complexity of the models included (see Johnsson et al, 2002). Limited data requirement – only simplified input data required, e.g., the model has been applied for calculations of nitrogen root zone leaching losses for all arable land in southern Sweden in 1985 and 1994 (Johnsson & Hoffmann, 1998) and all arable land in the whole of Sweden in 1995 and 1999 (Johnsson & Mårtensson, 2002) and for all arable land in an small agricultural catchment (Hoffmann, M. & Johnsson, 2003).

HBV: The model is simple, and has been applied in some 40 countries, in all parts of the world. The model runs under a Windows graphical user interface (IHMS), and a new modern interface will be available in 2003.

HBV-N: The model has been applied in some 3500 catchments in southern Sweden (Arheimer and Brandt, 1998), and in some 1000 catchments across the whole country (Brandt and Ejhed 2002). The model has also been applied to Matsalu River in Estonia (Lidén at al., 1999). The model runs partly in the Windows graphical user interface (IHMS).

QT 6 - SWAT:

The USA model SWAT a three-dimensional / continuous time watershed model that operates on a daily time step at basin scale (Fig 15). The major objective of the model is to predict the long-term impacts in large basins of management and also timing of agricultural practices within a year (i.e., crop rotations, planting and harvest dates, irrigation, fertiliser, and pesticide application rates and timing). It can be used to simulate at the basin scale water and nutrient cycles in landscapes where the dominant land use is agriculture. It can also help in assessing the environmental efficiency of best management plans and alternative management policies. The chemicals considered in the model include nutrients (N-based, P-based, O-based and algae) and pesticides.

In order to apply SWAT, each watershed is discretised into sub-watersheds for which the top surface corresponds to the upper boundary. The lower boundary is represented by the top of the deep aquifer (several metres). The losses (water, sediment, and nutrients) for a specific sub-watershed are computed at the sub-watershed outlet. The point sources and the losses for each sub-watershed are then routed through a channel network where retention and transformation of nutrients is simulated. The model takes into account not only the retention taking place in the soil, but also the retention occurring in the river system.

The hydrology in the model is based on the water balance equation comprising surface runoff, precipitation, evapotranspiration, infiltration and subsurface runoff. Evapotranspiration can be calculated by the Priestley-Taylor method or Penman-Monteith method. Precipitation can be estimated using a weather generator included in SWAT; however, measured time series can also be used, thereby reducing uncertainties. For calculation of the infiltration, the soil profile is represented by up to 10 layers, a shallow aquifer and a deep aquifer. When the field capacity in one layer is exceeded, the water is routed to the next soil layer. If this layer is already saturated, a lateral flow occurs. Bottom layer percolation goes into the shallow and deep aquifers. Water reaching the deep aquifer is lost, but a return flow from the shallow aquifer due to the deep aquifer saturation is added directly to the subbasin channel. Runoff volumes are computed by the SCS Curve Number Method. Surface runoff is estimated as a non-linear function of precipitation and a retention coefficient. Also the Green & Ampt approach is available. SWAT also incorporates models to predict channel losses, runoff in frozen soils, snow melt, or capillary rise. A simplified EPIC model is used to simulate crop growth (e.g. wheat, barley, alfalfa, corn) using unique sets of parameters for each crop. Natural vegetations (i.e. forest, grass, pasture) are also included in the crop database.

Once all hydrological processes are calculated for an homogeneous part of the subbasin, the resulting flows are considered to contribute directly to the main channel. SWAT includes a routing module based on the ROTO model. This routing procedure moves downstream the water budget taking into account how subbasins and reservoirs are connected.

Sediment yield is determined for each subbasin with the Modified Universal Soil Loss Equation, including runoff, soil erodibility, slope and crop factors.

Nutrient loading to the channel is calculated from the concentrations in the upper soil layer and the runoff volumes. Use of P and N by crops is estimated by using a supply and demand approach. The nitrogen module also includes processes like mineralisation, denitrification, and volatilisation. Phosphorus association with the sediment phase is also considered in the phosphorus module. Both modules are based on the CREAMS model. After considering the N and P dynamics, the chemicals are also routed into the subbasin channels.

Data requirements

SWAT is a comprehensive model that requires a diversity of information in order to run. Related data refer to both required input and optional input. SWAT requires a number of years as a warming up period.

Operational experience and skills requirements of users
Basic GIS/ArcView knowledge is required to set up the relevant ArcView map themes and data files in the model. Some hidden bugs in the interface however, require some experience or the assistance of experienced users. Also the interface relays heavily on dbf files that are easily corrupted by Microsoft Excel.

Participation in previous model comparison studies
In the framework of the FP4 CHESS project an inter-comparison exercise was run to compare SWAT to the ICECREAM field scale model developed in Finland on the basis of the USEPA GLEAMS model.

Sub-modules that can be independently checked
The in-stream water quality module is explicitly on/off switchable. Since the model outputs comprise actual ET, Runoff, Percolation and Plant growth, it is possible to check these sub modules separately.

Existing sensitivity analysis
So far, no published sensitivity analyses have been identified.

Cost indication (based on work load to set up and apply the quantification tool)
As a rough estimate, 3-4 months are requested to setup the SWAT database (starting from available data) and run, calibrate and validate the model.

Capability to evaluate nutrient and watershed management strategies (scenario analysis)
SWAT is a river basin, or catchment scale model developed to predict the impact of land management practices on water, sediment and agricultural chemical yields in large, complex catchments with varying soils, land use and management conditions over long periods of time. Different scenarios can refer to changing climate, land use, agricultural management, water management and structural BMP implementation. The model is physically based and computationally efficient, uses readily available inputs and enables the users to study long term impacts.

Applicability
SWAT can be used to simulate a single watershed or a system of multiple hydrologically connected watersheds.

Figure 16 and Figure 17 shows the transformation of the N and P processes in the soil which are described in the SWAT model.

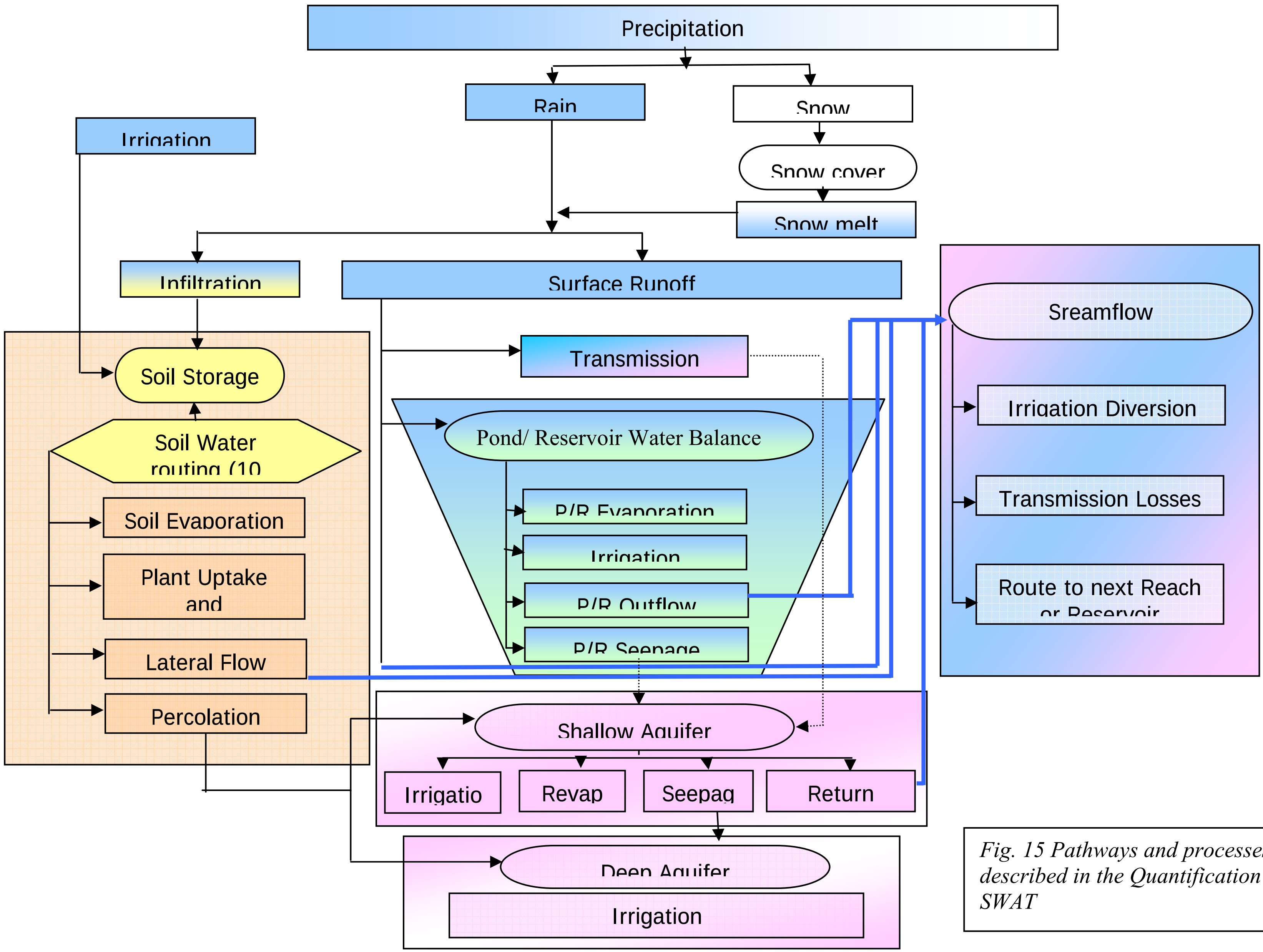


Fig. 15 Pathways and processes described in the Quantification Tool SWAT

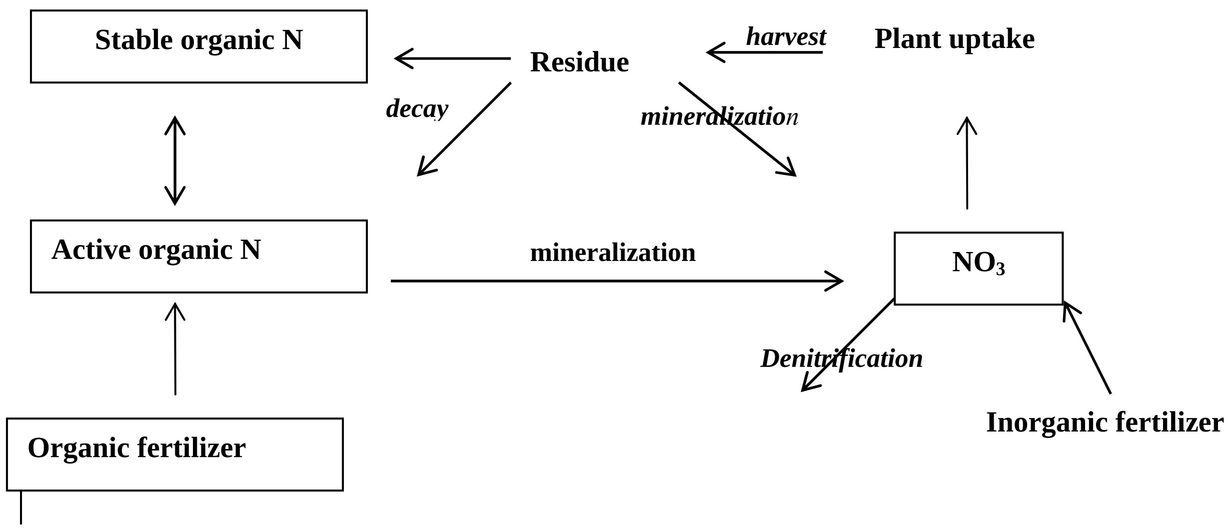


Fig. 16 Nitrogen processes described in the Quantification Tool SWAT

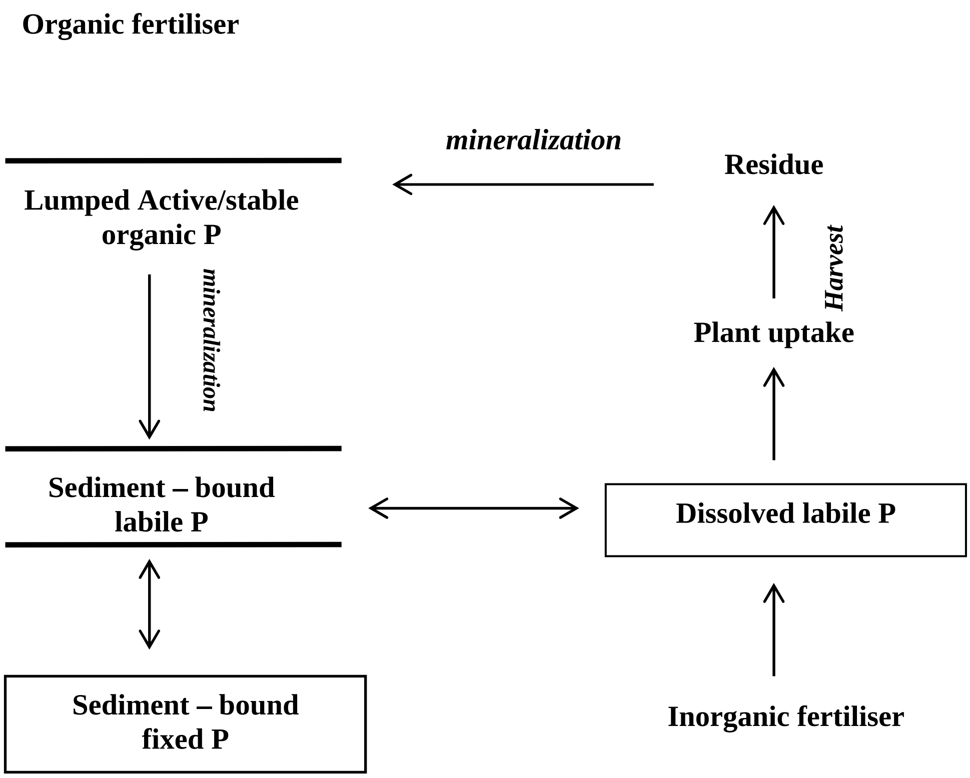


Fig. 17 Phosphorus processes described in the Quantification Tool SWAT

QT 7 – EveNFlow

The quantification tool EveNFlow is a catchment model that simulates the loss of nitrate in soil drainage and the routing of leachate through a catchment system. The system developed uses statistical data on land use, farming practices, climate and soil characteristics as inputs, collated at a spatial resolution of one square kilometre as a National Environment Database. The model was developed to provide a robust estimates of inorganic nitrogen fluxes and concentrations in river waters, primarily originating from agricultural land, for any catchment within England and Wales. In addition, one of the core principles of the model was that it should be suitable for integration with national databases such as those held within the MAGPIE Decision Support System (Lord & Anthony, 2000). The system is intended to work in two modes: the national mapping of annual total nitrate losses at a spatial resolution of 1 km^2; and the simulation of daily river flow and nitrate concentrations at the mouth of river catchments that are between 100 and 2,000 km^2 in area.

EveNFlow expects an input daily time series of soil drainage and estimates of the autumn soil nitrate content that is at risk of leaching in the following winter for each crop and soil combination within a study catchment. Drainage time series can be provided by *capacity* based evapo-transpiration models. Autumn soil nitrate content is calculated using a N model that integrates field observations and modelling expertise in a simplified framework that expresses potential nitrate losses as a function of a nutrient balance on a per capita (livestock) and per hectare (cropping) basis. There is no explicit representation of the soil nitrogen cycle. However, crop growth is represented using simple growth models from MORECS version 2 (Hough *et al.*, 1996) to provide information on Leaf Area Index for the calculation of canopy resistance, root depth and crop height.

EveNFlow integrates a nitrate leaching model that calculates the proportion of nitrate lost as a function of soil field capacity and hydrologically effective rainfall - with a conceptual model of lateral hillslope and groundwater transfers based upon TOPMODEL theory, parameterised from hydrological indices of observed river flows. The hydrological indices are currently obtained from a database of soil hydrological characteristics that enables the spatially distributed modelling of contributions to river flows from individual soil associations. Nitrate losses from a river system by denitrification and plant uptake may be taken account of by a physically structured empirical model that relates rate of removal to water temperature, nitrate concentration, surface bed area, and rate of flow. The retention module may require calibration of a parameter controlling the efficiency of biochemical removal of nitrate.

EveNFlow is a semi-distributed model with five modular components. The components of EveNFlow incorporate a number of simple *meta-models* that are adapted to the scale and information content of the environment database. The model concerns only diffuse inputs, effluent contributions to the river nitrate load are estimated either on the basis of catchment population figures and per capita estimates of effluent volumes and nitrogen load, or information on licensed dry weather flow discharges.

Component 1 is a soil nitrate model that simulates the soil crop interaction that control the mass of nitrate present in the soil at the onset of winter drainage that is vulnerable to leaching. The model comprises elements of the NITCAT (Lord, 1992), N-CYCLE (Scholefield *et al*., 1991) and MANNER (Chambers *et al*., 1999) field scale models of nitrogen cycling under arable and grassland.

Component 2 is a soil drainage model. The model comprises elements of MORECS and IRRIGUIDE evapotranspiration models and can be driven by data derived from a stochastic weather generator. Alternatively observed, interpolated data may be used.

Component 3 is a leaching function that predicts the cumulative proportion of available nitrogen that is leached as a function of rainfall and soil water content. The model was derived from the SLIM and SACFARM models (Addiscott and Whitmore, 1991).

Component 4 is a drainage routing model based upon a one-dimensional form of TOPMODEL (Beven *et al,* 1995). The model simulates the river hydrograph and mixes rapid and slow soil drainage derived from different depths in the soil profile. The model is parameterised from soil HOST class (Boorman *et al*, 1995).

Component 5 concerns nitrate retention. Retention in aquifers or the vadose zone is currently not simulated, but can be by application of denitrification rate parameters from de Witt (2001) to the deepest soil water store in the routing model. The retention in the river is calculated on a daily basis using empirical relationships between discharge and channel geometry to estimate the proportion of nitrate removed by bed processes.

Figure 18 shows the pathways of nitrogen and losses from agricultural land to surface waters and nitrogen processes implemented in this quantification tools EveNFlow.

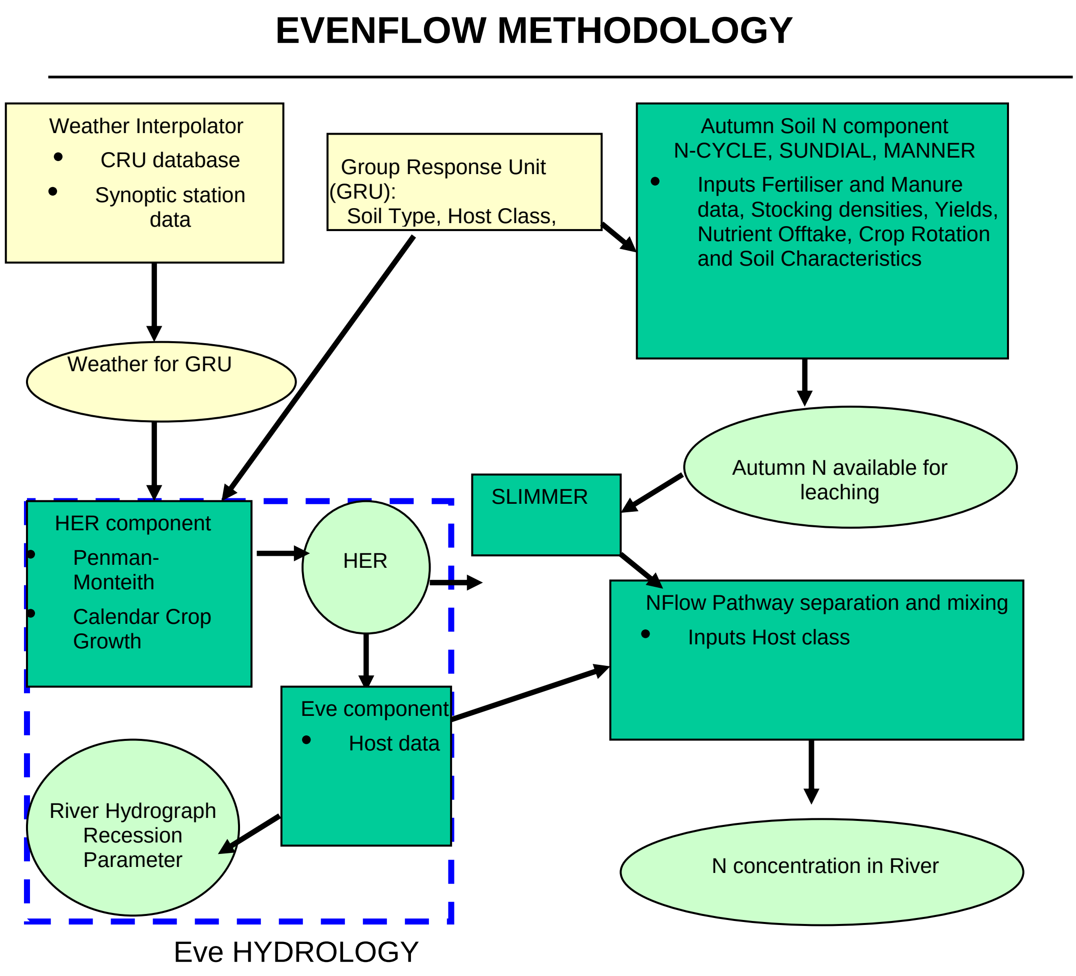


Fig. 18 Pathways and processes described in the Quantification Tool EVEN-FLOW

Data requirements
Soil Nitrate: component 1:
- For current UK practice, autumn soil nitrogen contents can be estimated from a table of standard coefficients knowing the crop areas and animal numbers. Changes in agricultural practice require re-derivation of these coefficients using data on crop types and yields, fertiliser and manure management, soil characteristics stock densities and mean climate data.

Soil Drainage: component 2
Soil type and characteristics, daily weather data, crop type.
Soil Leaching: component 3
HOST class, soil type and characteristics.
Drainage routing: component 4
Host Class.
Nitrate Retention: component 5
River network, river bed characteristics, point source inputs.

Operational experience and skills requirement of users
EveNFlow is a recently developed model and therefore currently only suitable for use by model developers.

Participation in previous model comparison studies
None

Sub-modules can be independently checked
The water balance and hydrograph can be independently checked.

Existing sensitivity analysis
None yet

Cost indication (time needed for application)
Due to the continuing developments of this model, application is time-consuming as there is currently no automated procedures or front-end. EveNFlow therefore contrasts with other models (e.g. ANIMO, SWAT) which have been available for a longer period and hence have had greater resources directed at their development and refinement.

Potential capability for use in scenario analyses
Would be responsive to changes in land use and livestock numbers/management. The model has no explicit representation of soil processes (e.g. mineralization) or crop growth to climatic variables. The drainage (HER) values are responsive to weather variations. Therefore any inter-annual variations in flow reflect changes in drainage characteristics. A separate snow-melt function is being developed for this project. There is no explicit link between temperature and soil/plant processes. The model assumes fields are managed according to recommended agricultural practice.

Applicability
EveNFlow is a young model still in development, and is currently only applied by researchers involved in its continued development. The model has been developed for use in the UK, and may therefore be difficult to apply to warmer climates. In order to be applied to a non-UK catchment, the model requires expert guidance on soil, land use and livestock characteristics. The model has modest data requirements, can be considered as a meta-model, and is unsuitable for considering more subtle changes in land management without additional information to support re-parameterisation.

QT 8 - NOPOLU:

This agricultural non-point emissions module is a part of a package built around a comprehensive catchment description database. The database is designed to handle all items and their relationships that are relevant to process data and produce information for issues related to catchments. Non-point emission modules are separated according to the sector responsible for emissions. Industrial and urban non-point sources are processed in a classical way using areas of concern and emission factors, and modified by run-off.

The module calculating nutrient surpluses from agricultural origin using two sets of models:

1) The first model calculates the surplus of nutrients in line with the DPSIR (EEA) conceptual framework. Its methodology is based on improving the statistical data by using land cover information. The core model is derived from the soil-surface balance model used by Eurostat to make its yearly surplus calculations. Great concern is attached to sustainability. All model features are designed in a way that variables (that change with time) are obtained from statistical systems or permanent monitoring. Constants are obtained from different sources, including research programmes.
2) The second part is a transfer model. It separates the background noise (constant per ha of soil type per catchment) from the surplus related transfer, addressed by a power function depending on the surplus value and the soil characteristics. Incorporation of actual run-off as a transfer factor, with the related issues of year-to-year management of the remaining fraction of the surplus, is under consideration.

Both parts of the model are designed in such a way that they are a) fully comparable with other sources of information and b) managed by scenarios that allow the use of better or local data. It allows aggregation/disaggregation between administrative / catchment / other territorial units. The use of CORINE *land cover* proved useful in yielding stable responses from statistics aggregated at a pseudo NUTS4 (clusters of 4 to 6 NUTS5) level to NUTS3 level.

The methodology developed especially for that purpose makes a throughout use of the CORINE land cover layer to standardise the transfer of information between the administrative and the catchment layers. These layers represent the source of data on the one hand and the target for results on the other hand. The newly implemented agricultural model is presently based on the French official fertilisation model agreed by EUROSTAT. The CORINE land cover layer is used to distribute the statistical data available in agricultural census files (administrative level, data not geo-referenced) on the most likely real area that belongs to the catchment belonging to the same administrative area. Phosphorus calculations are currently being verified, and some technical coefficients are not currently available.

Custom-built links are used between, for example, the CORINE Land Cover codes (which are unique at the European scale) and agricultural census codes (country dependent). To improve the versatility of the agricultural modelling system, NOPOLU2 handles regional tables. These tables allow consideration of the crop and land cover relationship in a given region, as well as different fertiliser and yield values. This may be the case for example for maize crops in harvested areas compared to the same land use in hunting areas where the crop is grown to feed game birds.

Figure 20 shows the input balance (for example for nitrogen) used within NOPOLU. A part of this surplus is assumed to transport to the surface waters (Figure 21)

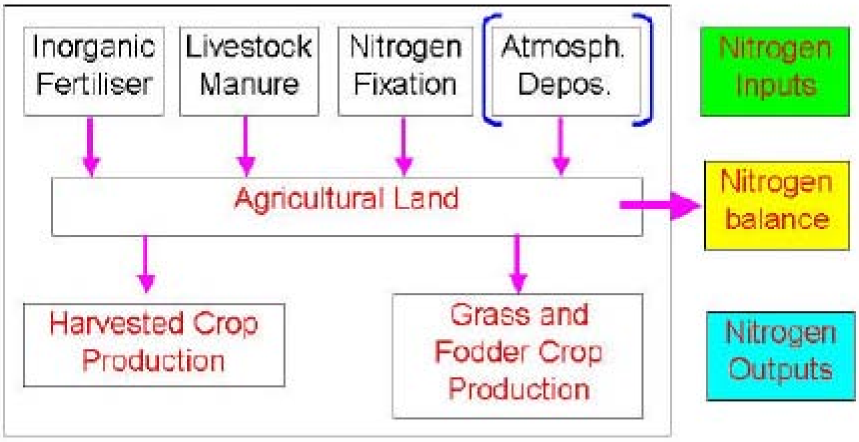


Fig. 20 Nitrogen input balance used within NOPOLU.

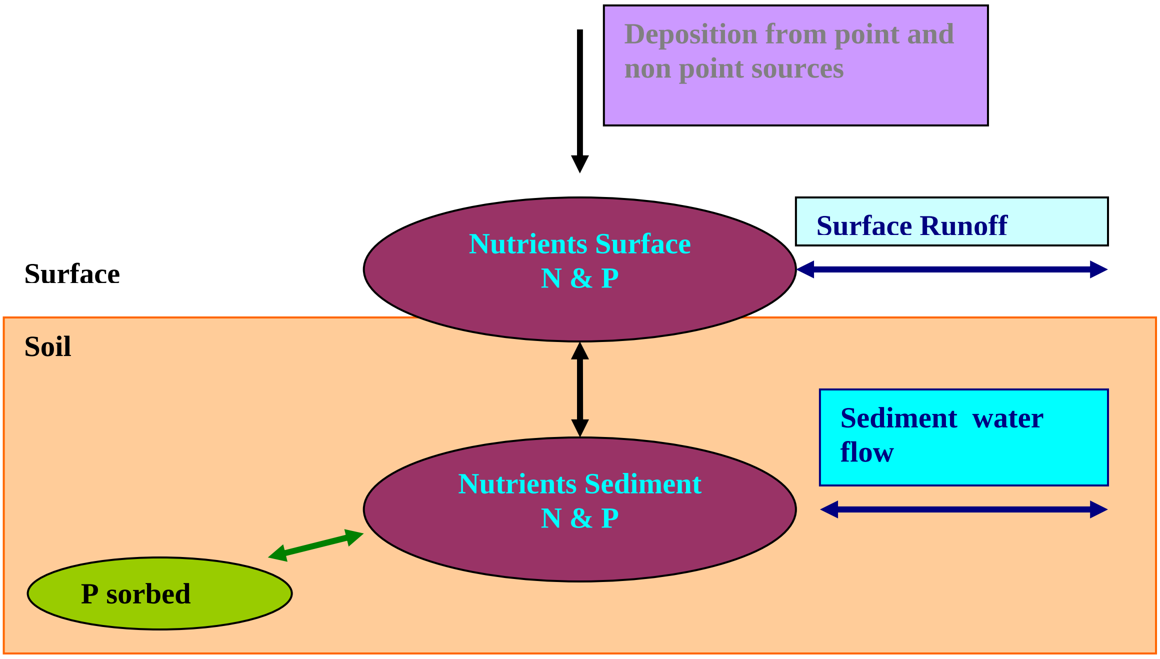


Fig. 21 Pathways and processes described in NOPOLU

The NOPOLU System2 transfer model is based on statistical analysis of observed concentration data. There is no detailed modelling of the nutrient runoff from the field and its transport in surface waters. The runoff and transport parameters are estimated from statistical data including:

(1) The observed flow rate and the measured agrochemical concentrations in the variety of watersheds scattered over the whole basin;

(2) The DEM from which a watershed morphometry can be calculated;

(3) Parameters that describe the climatic conditions of the sampled watershed.

Data requirements
- Layers: administrative, catchments boundaries, Corine land cover
- Agricultural Census
- Crop and livestock technical coefficients
- Soil and hydrology coverage
- Corine land cover and agricultural uses
- Atmospheric deposition coverage

Operational experience and skills requirements of users
The setting up of a model application requires standard skills in data collection and manipulation. Once this has been done, using the model is straightforward using a "point and click" menu driven system. Calibration requires agronomic and soil science knowledge.

Participation in previous model comparison studies
The whole NOPOLU river system model (including hydraulics, O_2 cycle, N & P, eutrophication) has been compared with other similar available tools for the French Ministry of Environment. NOPOLU and PEGASE were the final modelling tools recommended.

Sub modules that can be independently checked
None.

Existing sensitivity analysis
No particular study focused on sensitivity analysis has been performed. The parameters sensitivity have been analysed during calibration phases of the applications.

Cost indication
The cost depends mainly on the spatial availability of regional (local) specific coefficients to be entered. The global coast is low compare to the scale (spatial and time) covered. About 1 man-months of workload is needed to set up the model for a catchment.

Capability to evaluate nutriment and watershed management strategies (scenario analysis)
You can easily compare two scenarios that can be build upon:
- Modification of sources
- Change in fertiliser uses
- Watershed land use evolution

Applicability
The model can be use at different (small to very large scale) with easily available data.

QT 9– Source apportionment (implemented Danish approach):
Nutrient losses to freshwater are often greater than the measured nutrient transport due to the retention and cycling of the nutrient in lakes and rivers, and deposition during flooding riparian areas. Retention plays an essential role for the amount and the composition of nitrogen and phosphorus fluxes through river systems. Empirical sub-models for shallow lake retention of nitrogen and phosphorus have been developed based on mass-balances for a great number of lakes in Europe. The empirical retention models for lakes are quite simple since they only require information about the hydraulic loading, water temperature and nitrogen and phosphorus loading, respectively, and an estimate of the phosphorus pool in lakes.

Nutrient losses from diffuse sources such as agricultural land, forest and pristine area are estimated as the difference between the gross transport (calculated retention added to the measured transport) and the measured emission from point sources. The nutrient loss from scattered dwellings is included in the diffuse sources. Using estimates of atmospheric deposition on surface water and figures for natural background losses, the losses from agricultural land can be estimated.

The nutrient load from agricultural areas includes the potential load from scattered dwellings entering the surface freshwater system. The model accumulates the uncertainty factors with respect to the total nutrient load from agriculture. Therefore, monitoring is performed in several small (5 to approx. 60 km^2) agricultural catchments (more than 150), with low input from point sources. Figures from these catchments are used to estimate diffuse losses to primary recipients, as retention in surface water is very low in small catchments with very few lakes. Further, monitoring results from these small agricultural catchments are used to calculate the flow-weighted concentrations (the annual transport of a nutrient divided by the annual runoff of water) or area coefficients that are used to estimate diffuse losses in non-monitored catchments.

To calculate nutrient losses from agriculture (F) to coastal waters, the following variables must be determined:
1) Total load of a particular nutrient to coastal waters consisting of the load from monitored catchments (L_m) and from unmonitored catchments (L_u).
2) Point source nutrient emission to freshwater from sewage treatment plants, industrial plants, fish farms and urban storm water runoff consisting of the discharge from monitored (P_m) and unmonitored catchments (P_u).
3) Losses from scattered dwellings ($S_m + S_u$).
4) Losses from background/natural areas (B).
5) Retention in lakes (R_l) and rivers (R_r).
6) Atmospheric deposition on freshwater (D).

The nutrient losses to freshwater from agricultural areas (A) in a specific catchment are then calculated as:

$$A = (L_m+L_u) - (P_m+P_u) - (S_m+S_u) - B + (R_l+R_r) - D \quad (1)$$

This equation represents basically the same principle that is described in the OSPAR HARP guideline concerning source apportionment (Guideline 8) although in (1) both monitored and unmonitored catchments are included, as is also the input from atmospheric deposition on freshwater and scattered dwellings. If only the monitored parameters are used equation (1) gives the source apportionment for a monitored catchment.

Data requirements
Requires only one year of data of nutrient emissions from different point sources in the catchment (STWs, industrial plants, scattered dwellings, urban runoff etc.), total nutrient retention in surface waters in the catchment and the transport/export of nutrients at the outlet from the catchment to give an estimate of the eutrophication pressures in the catchment.

Operational experience and skills requirement of users
Many years of operational experience throughout Europe and low demands of skills for end-users.

Participation in previous model comparison studies
Has participated in former tool comparison studies (HELCOM).

Sub-modules that can be independently checked
No sub-modules that can be independently checked.

Existing sensitivity analysis
Not possible because it is a load oriented approach without parameterisation.

Cost indication
Very low in work load and hence costs to apply on a catchment (work load in days).

Capability to evaluate nutrient and watershed management strategies (scenario analysis)
Not appropriate for scenarios and land management strategies

Applicability
Very applicable for all European catchments.

3.2 Overview of pathways and processes described

In Table 2 an overview is given of the characteristics, pathways and processes described by the different Quantification Tools. A division is made into:
- Spatial resolution
- Nutrient input and management
- Water balances and pathways
- Soil physical/chemical/biochemical processes
- River flow and prediction of stream concentrations
- Intermediate output

In paragraph 3.3 each of these themes will be discussed separately.

Table 2: *Pathways and processes described by the quantification tools*

Model pathway, process or characteristic	**NL-CAT**	**REALTA**	**N-LES**	**MONERIS**	**TRK**	**SWAT**	**EVEN-FLOW**	**NOPOLU**	**SOURCE AP.**
QT number	1	2	3	4	5	6	7	8	9
Spatial and temporal resolution of application									
- Vertical boundaries (m); [Field (FD)]	1	10-15	FD	50	1	1	1	10-15	1
- Vertical boundaries (m); [Root Zones (RZ)]	*	N	RZ	*	1.5	*	3	N	-
- Internal timestep for calculation (Hour, Day, Year)	D	Y	Y	Y	D/H	H	D	Y	Y
- Temporal resolution of output (Day, Year)	D	Y	Y	Y	Y	D	D	Y	Y
Nutrient Inputs and Management									
- Atmospheric deposition	Y	N	Y	Y	Y	Y	I	Y	Y
- Fertiliser additions	Y	Y	Y	Y	Y	Y	Y	Y	N
- Livestock density / manure additions	Y	Y	Y	Y	Y	Y	Y	Y	N
Method of manure application	Y	Y	Y	N	Y	Y	Y	N	N
- Plant nutrient cycle/uptake	Y	N	Y	Y	Y	Y	Y	Y	N
- Land management practices	Y	N	Y	Y	Y	Y	Y	N	N
- N fixation (legumes)	Y	N	Y	Y	N	Y	I	Y	N
- Non-agricultural land	Y	N	Y	Y	Y	Y	Y	Y	Y
- Anthropogenic effects (point sources and water transfer)	Y	Y	Y	Y	Y	Y	Y	Y	Y

Keys:
(Y)es, (N)o,
(E)xplicit, (I)mplicit,

Table 2: *Pathways and processes described by the quantification tools (Continued)*

Model pathway, process or characteristic	**NL-CAT**	**REALTA**	**NLES**	**MONERIS**	**TRK**	**SWAT**	**EVEN-FLOW**	**NOPOLU**	**SOURCE AP.**
QT number	1	2	3	4	5	6	7	8	9
Water balance and pathways									
Rainfall interpolation corrections for alt. (e.g. to grid)	N	N	N	Y	Y	Y	Y	N	N
Frost and snow	Y	N	Y		Y		Y	N	N
Anthropogenic effects (point sources and water transfer)	Y	Y	Y	Y	Y	Y	Y	Y	N
Canopy interception	Y	N	Y	N	Y	Y	Y	N	N
Evapotranspiration	Y	N	Y	I	Y	Y	Y	Y	N
Overland flow									
Hortonian overland flow	Y	N	N	Y	Y	Y	Y	N	N
Saturation excess	Y	N	N	Y	Y	N	Y	N	N
Subsurface drainage volume									
- Routing: Preferential flow	Y	N	T	N	N	Y	T	N	N
- Routing: Matrix flow (Interflow)	Y	N	T	T	Y	Y	T	N	N
- Routing: Tile drainage	Y	N	N	Y	T	Y	Y	N	N
- Groundwater input/loss	Y	N	N	T	T	Y	Y	N	N
- Shallow (S) and/or deep (d) groundwater	Sd	N	N	S	Y	Sd	S	N	N
- Measured flow used to calculate water balance		Y	Y	Y			N	Y	
- Model prediction of river hydrograph	Y	N	N	N	Y	Y	Y	N	N
- Travel time	Y	N	N	Y	Y	Y	Y	N	N
Soil physical/chemical/biochemical processes									
- N and P mineralization/immobilisation	Y	N	N	I	Y	Y	N	N	N
- Linked to C cycle	Y	N	N	N	Y	N	N	N	N
- P sorption/desorption	Y	N	N	I	N	Y	N	N	N
- P precipitation	Y	N	N	N	N	Y	N	N	N
- Nitrification	Y	N	N	N	Y	Y	I	N	N
- Denitrification	Y	N	N	Y	Y	Y	I	N	N
- Ammonia volatilisation	Y	N	N	I	Y	Y	I	N	N
- Erosion (gross/net)	Y	N	N	I	Y	Y	N	N	N
- Sediment delivery function	N	N	N	Y	N	Y	N	N	N
- Enrichment ratio	N	N	N	Y	N	Y	N	N	N
- 1, 2 or 3D solute transport processes	1,3	N	N	N	1	3	1	N	N
- Implicit lumping of processes	N	Y	Y	Y	Y for P	N	Y	Y	Y

Keys:
(Y)es, (N)o,
(E)xplicit, (I)mplicit,
(T) Combined

Table 2: *Pathways and processes described by the quantification tools (Continued)*

Model pathway, process or characteristic	**NL-CAT**	**REALTA**	**NLES**	**MONERIS**	**TRK**	**SWAT**	**EVEN-FLOW**	**NOPOLU**	**SOURCE AP.**
QT number	1	2	3	4	5	6	7	8	9
River flow and prediction of stream concentrations									
- Model prediction of river hydrograph	Y	N	N	N	Y	Y	Y	Y	N
- Hydrograph separation approach	N	N	Y	N	N	N	N	N	N
- Instream retention (streams and rivers)	Y	N	N	T	Y	Y	Y	Y	Y
- Retention in lakes	N	N	N	T	Y	N	N	N	Y
- Retention below the root zone	Y	N	Y	Y	Y	Y	N	N	N
- Load and/or concentration emission from land to water bodies (excluding retention in the surface waters)	LC	L	LC	LC	LC	LC	LC	LC	LC
Soluble inorganic P	Y	Y	N	N	N	Y	N	N	N
Dissolved organic N/P	Y	N	N	N	Y	N	N	N	N
Particulate organic N/P	Y	N	N	N	N	Y	N	N	N
Particulate inorganic P	N	N	N	N	N	N	N	N	N
Total P	Y	Y	N	Y	Y	Y	N	Y	Y
Suspended solids	N	N	N	Y	N	Y	N	N	N
Nitrate-N	Y	N	Y	N	N	Y	Y	N	N
Ammonium-N	Y	N	N	N	N	Y	N	N	N
Nitrite-N	N	N	N	N	N	Y	N	N	N
DIN (dissolved inorganic nitrogen)	Y	N	N	Y	Y	Y	N	N	N
Total nitrogen	Y	N	N	Y	Y	Y	N	Y	Y
Intermediate Output									
Runoff	Y	N	N	N	Y	Y	N	N	N
Root zone	Y	N	Y	Y	Y	Y	N	N	N
Subsurface	Y	N	N	N	Y	Y	N	N	N
Groundwater/base flow	Y	N	Y	Y	Y	Y	N	N	N

Keys:
(Y)es, (N)o,
(E)xplicit, (I)mplicit,
(T) Combined
(L)oad and/or (C)oncentration

3.3 Model comparison

The quantification tools are now discussed with respect to each of the following topics:
- Boundary conditions / spatial and temporal resolution
- Data inputs nutrient inputs and land use management
- Plant growth / uptake representation
- Hydrological aspects (canopy, surface runoff, soil water balance, surface water systems)
- Soil chemical processes (e.g. P sorption)
- Soil biochemical/physical processes (e.g. mineralization)
- Lumped process description
- Model output:

3.3.1 Boundary conditions

The horizontal spatial resolution between the models differs profoundly. The SWAT and NL-CAT model are able to estimate the losses at 0.01 km^2 (1 ha) up to 100 km^2, while MONERIS has a relative high lower limit of 50 km^2 (5000 ha). The other models are in between these values (ca. 1 km^2 to large basins). These differences result from the different objectives of the models. The models with a high resolution are mainly developed to evaluate the effect of agricultural management practices or detailed (sub-)catchment management. The models with a low resolution are often derived from river basin or watershed management perspectives.

The vertical spatial resolution (discretisation of the soil) is described quite differently. The lower boundary of the SWAT and NL-CAT models is represented by the top of the deep aquifer (several metres) and subdivided into ca. 10 layers, while for N-LES the lower boundary is the root zone (and consists of one layer, however the root zone is divided into horizons when running the water balance submodel). Three models do not use any vertical discretisation, namely REALTA, NOPOLU and the SA approach. The models TRK, MONERIS and EveNFlow are focussed on the nutrient losses from land to surface waters from the upper 1 to 3 m of the top of the soil.

With respect to the temporal resolution of the models there are also differences. Although the internal timestep can be quite small for some models (SWAT and NL-CAT during rainfall events), many of the models have a temporal resolution of the output of one year. Only SWAT, NL-CAT, TRK and EveNFlow can produce output at a daily timestep for comparison with measured river data (e.g. rising and recession limbs of a hydrograph). In addition to this, the TRK model can produce *normalised* annual losses (based on an average of a period of 20 year of meteorological data). This means that many of the models can be validated only on an annual basis. As a consequence, annual output models will carry inevitable limitations in terms of their potential responsiveness to changes in land use management.

3.3.2 Nutrient inputs and land use management

- Agricultural nutrient input
All models (except SA) evaluate the impacts of manure (or livestock density) and fertiliser additions on the nutrient losses to surface water. The REALTA model takes into account the annual net P input by fertiliser and manure and the annual plant uptake is directly subtracted from the P input.

In ANIMO three forms of organic nutrient forms are distinguished
- Fresh organic N and P as a product of crop residues/roots
- Root exudates produced during growing season

- Organic N and P of animal waste (different forms of material)

Each organic material (e.g. pig and poultry manure) is defined by a distribution of typical organic N and P fraction. Each organic N and P fraction has its own decomposition rate. Most of the time the added organic material is defined by three fractions (direct soluble, fast and slow decomposable organic material). In the SOILNDB model (part of TRK) two of these threes organic nutrient forms are modelled (not the exudates). In many of the other models all organic nutrient forms are assumed to be fresh organic N and P, and are described as a total amount of organic N and P input from manure applications.

All models, except REALTA, take some account of atmospheric deposition. Since REALTA is a phosphorus model and atmospheric deposition of phosphorus is quite small (e.g. about 0.4 kg P per ha) in relation to the level of manure and fertiliser input, this simplification is not important.

With respect to nitrogen input, two of the N-models (N-LES CAT and MONERIS) use net NH_4 input, which means that these models do not calculate NH_3 volatilisation. In MONERIS NH_3 volatilisation is considered to be constant for each animal. Many of the other N models consider (implicitly or explictly) the NH_3 volatilisation which reduces the N available for leaching from manures. However, the TRK method can also use net NH_4 as input, which is preferred if available. Many of the N models take into account N fixation either by modelling (ANIMO) or as an input value (other models). For EveNFlow, N fixation is implicitly taken into account, because measurements of soil mineral N in the autumn are needed as an input value for this model.

The models REALTA, MONERIS and NOPOLU do not represent land management practices, such as ploughing. MONERIS has an option in the erosion and surface runoff module, where ploughing and conservative management (non-ploughing) can be considered, especially for scenario calculations. Within N-LES-CAT only the effect of ploughing in of grass is considered.

- *Diffuse input / losses from "nature" areas (forest areas including clear-cuts, wetland, urban paved surfaces etc.)*

The nutrient losses from nature areas are taken into account by all models. Only SWAT and NL-CAT calculates the N and P losses while the other models use standard figures for these areas.

- *Deposition on lakes and rivers*

The nutrient input from deposition on lake and river surfaces is included in most models as input values.

- *Point sources including rural households.*

These data are input values for all methodologies. Most of the time it is not an input value for the agricultural module (e.g. SOILNDB (TRK), NL-CAT and N-LES CAT), but is represented in the catchment hydrology module (like HBV-N (TRK), SWQN and Danish retention routine).

3.3.3 Plant growth and crop uptake

The environmental impact of nutrient inputs to agricultural land is highly dependent on the amount of harvest offtake from the fields, because this process is the most important process in immobilising the nutrients available for leaching.

In the models Source Apportionment (SA), REALTA and N-LES CAT, crop uptake and harvest are not issues that need to be described. In SA, agricultural losses are not modelled but calculated from measured balances. The REALTA model is a risk-based approach, where levels of risks are assigned to measured monitoring data. The N-LES CAT method is a statistical relationship between on the one hand nitrogen input, crop, soils, and climate characteristics, and on the other hand measured nitrate concentrations leaching out of the root-zone.

The EveNFlow model uses annual figures for nutrient offtake from the field as a result of harvest offtake (arable) or grazed grass (pasture). The MONERIS model calculates the total uptake of nutrients and total nutrient inputs by deposition, livestock and mineral fertiliser as well as N-fixation based on the annual agricultural statistics according to the method of OECD and Bach et al. (1998).

The models SWAT, SOILNDB (TRK) and NL-CAT are able to calculate total nutrient uptake and the offtake (export) of nutrient by harvest. The plant growth is modelled, mainly based on a nutrient demand function for optimal growth at a certain time during the year, the nutrient status of the plant at that moment and a function that can reduces nutrient uptake under specific circumstances (e.g. water deficit in dry periods). Within the SWAT model the same approach is used for grass production and plant growth (arable land). In the text boxes the different approaches by these three models are described in more detail.

The models SWAT, TRK (SOILNDB) and NL-CAT are able to calculate total nutrient uptake and the offtake (export) of nutrient by harvest. The plant growth is modelled, mainly based on a nutrient demand function for optimal growth at a certain time during the year, the nutrient status of the plant at that moment and a function that can reduce nutrient uptake under specific circumstances (e.g. water deficit in dry periods). Within the SWAT model the same approach is used for grass production and plant growth (arable land). In the text boxes the different approaches by these three models are described in more detail.

SWAT approach

Crop use of N is estimated using a supply and demand approach. The daily (day i) crop N demand can be computed using the equation

$$UND_i = (C_{NB})_i B_i - (C_{NB})_{i-1} B_{i-1}$$

where UND_i is the N demand of the crop in kg/ha, C_{NB} is the optimal N concentration of the crop, and B is the accumulated amount in kg/ha. The optimal crop N concentration is computed as a function of growth stage using the equation

$$C_{NB} = 4.0\,(bn) + 1.54\,(bn)\exp(-bn\,B_1)$$

where bn is a crop parameter expressing N concentration and B_1 is the fraction of the growing season. The value of B_1 is estimated as a function of heat units

$$B_{1,i} = \sum_{k=1}^{i} \frac{HU}{PHU}$$

where HU is the daily heat units in °C above the crop's base temperature and PHU is the potential heat units to mature the crop in °C.

The crop is allowed to take N from any soil layer that has roots. Uptake starts at the upper layer and proceeds downward until the daily demand is met or until all N has been depleted. If the soil cannot supply the daily N demand for legumes, the deficit is attributed to N fixation.

NL-CAT (ANIMO) approach

The nutrient uptake by arable crops has been described by a simple model. The nutrient demand has been defined by considering two phenological stages. During each period the concentration in the transpiration flux resulting in optimum growth is defined as:

$$c_{opt} = \frac{U^*}{\sum_{t_1}^{t_2} q_{tr}} \qquad (1)$$

where c_{opt} is the optimal uptake concentration (M L^{-3}), U^* is the reference cumulative uptake within the phenological stage (M L^{-2}), Σq_{tr} is the expected cumulative transpiration flow (L) and t_1 and t_2 are the first date and last date of the stage considered.

The expected optimal cumulative uptake and cumulative transpiration flow are defined by the user in the model input files. For years with higher or lower transpiration rates, the total crop uptake will increase or decrease proportionally. Under optimal circumstances, the plant uptake parameters σ_{NO3}, σ_{NH4} and σ_{PO4} are defined as:

$$\sigma_{NO3} = \frac{c_{opt,NO3}}{c_{NO3}(t_0)} \quad and \quad \sigma_{NH4} = \frac{c_{opt,NH4}}{c_{NH4}(t_0)} \quad and \quad \sigma_{PO4} = \frac{c_{opt,PO4}}{c_{PO4}(t_0)} \qquad (2)$$

The nutrient uptake by grassland is compared to arable crops more complex:
The concept of the supply potential is based on the assumption where the total uptake is determined by the sum of passive flow with the transpiration stream and a diffusive flow. Most crops can develop an internal nitrate concentration in the plant liquid. The concentration gradient between the nitrate concentration in roots and the concentration in soil water is considered as a driving force for nitrate uptake.

TRK (SOILNDB)

Plant uptake of nitrogen is calculated from time-dependent empirical function requiring parameter values specific for the crop and site concerned. A logistic growth curve is used to define a potential uptake demand during the growing season, which is distributed in the soil profile according to an assumed root distribution. Nitrogen uptake is reduced when the demand exceeds the available mineral N in the soil (given as a fraction of the total mineral N in soil). At harvest and ploughing the roots and harvest residues are incorporated into the soil litter pool.

$$N_{pl} = \frac{p_{ua}\, p_{uc}\, \frac{p_{ua} - p_{ub}}{pub}\, e^{-p_{uc}\Delta t}}{(1 + \frac{p_{ua} - p_{ub}}{pub}\, e^{-p_{uc}\Delta t})^2}$$

where p_{ua}, p_{ub}, p_{uc} are paramters and Δt is the time since the start of growth.

3.3.4 Hydrology

Four of the nine quantification tools do not have a hydrological module (REALTA, NOPOLU, N-LES CAT, NOPOLU and Source Apportionment model). The hydrological module of MONERIS is limited to the separation of the total measured runoff from a catchment into the discharges from the different pathways. But all these models need this input data from other models or from measured data. So, they are not discussed within this section.

Catchment discretisation

NL-CAT (ANIMO) and SWAT

A two step approach is applied to discretise a watershed:

1. a topographic discretisation where the watershed is divided into subbasins. This step serves to the basis for the routing of water and pollutant through the watershed.
2. divides each subbbasin into homogeneous hydrological response units (HRU) obtained by overlying the soil and land use maps. Different criteria can be used to obtain the HRU. Using a dominant criteria, each HRU is characterised by a unique combination of soil and land use

Each subbasin is associated with a channel segment, while the HRUs within a subbasin have no spatial links to each other. The hydrologic simulation proceeds in two steps. The first step corresponds to the land phase of the hydrologic cycle and controls the amount of water, sediment, nutrient and pesticide loading to the main channel in each subbasin. The second step corresponds to the routing phase of the hydrologic cycle, which can be defined as the movement of water, sediments, etc. through the channel network. Only SWAT considers sediment transport.

EveNFlow

EveNFlow divides the river catchment into a number of response groups based on a hydrological classification of soils. Each group is sub-divided into a number of diffuse sources of nitrate based on land use. The diffuse sources are the basic unit of calculation for land drainage and nitrate losses. The outputs from each diffuse source are area weighted to provide a timeseries of inputs to the river system for each group. The group inputs are distributed between the adjacent river reaches in proportion to reach length.

The soil hydrological attributes of each group determine the plant available water and the rate at which effective rainfall is delivered to the river reaches. The soils present in each group are identified from the national SEISMIC database that describes the percentage of each 1 km^2 area occupied by individual soil series (Hallet *et al.*, 1994). The soils are grouped according to the Hydrology of Soil Types (HOST) classification. The HOST classification consists of 29 conceptual models that describe the dominant pathways of water movement through the soil and substrate to the river system (Boorman *et al.*, 1995).

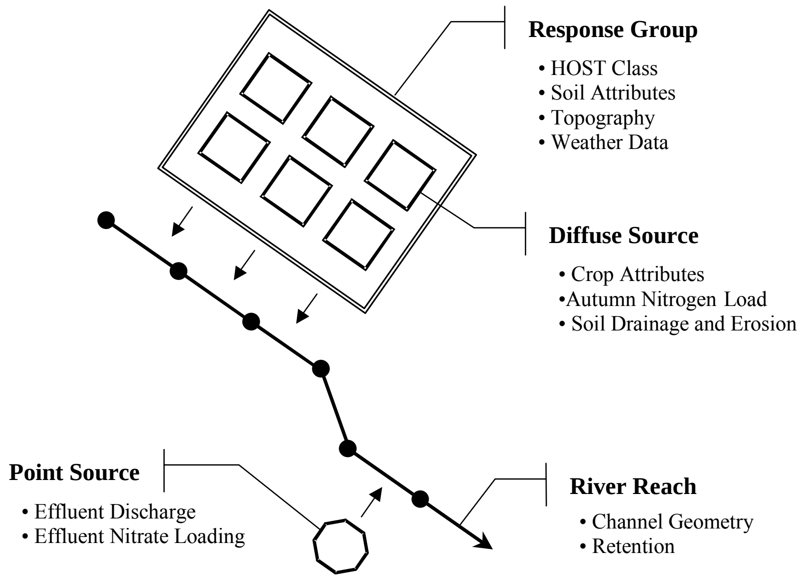


Figure 22: EveNFlow data structures and their attributes.

The EveNFlow model requires as input a description of the river network topology to enable routing of diffuse runoff and calculation of retention during the transit time to the catchment outlet. The network is normally derived from Ordnance Survey mapping at a scale of 1:250000 and is discretised into reaches of 100-250 m length. The catchment boundaries and an estimate of mean slope for each response group area are determined from a raster digital elevation model with a cell size of 50 m.

Calculation of drainage for each diffuse source within a response group requires daily time series of observed weather. Soils of each HOST class generally occupy a specific altitude and climate range within a catchment. Therefore, a single time series for each response group is generated by spatial interpolation from monitoring stations within the catchment. The method of interpolation uses triangulation to determine which observing stations contribute to the time series estimate for each group. The observed data at these stations are expressed as a percentage of long-term monthly mean observations, and the percentages interpolated to the group centroids by an inverse-distance formula. The interpolated percentages are multiplied by the long-term monthly mean observations at the group centroid to recreate a weather timeseries.

TRK (HBV)

The HBV model (e.g. Lindström et al., 1997) is a conceptual, continuous, dynamic and semi-distributed model. Calculations are made for elevation zones in coupled subbasins within a catchment. Soil moisture, snow distribution and redistribution is considered within each elevation zone. The routing between subbasins can be described by the Muskingum method or simple time lags. Each one of the subbasins has individual response functions. Finally, transformation of runoff is taking place after water routing through the lake according to a rating curve.

Canopy and snow

For a scientific review of hydrological models regarding their capability to generate appropriate hydrological input for leaching models and water quality models for the different climatic conditions in the EC, the description of water flows in the vegetation and at the land surface can be of special interest. Snow fall, snow melt, canopy interception and compaction of the snow cover govern a major part of the water cycle during winter time in the Northern regions and total rainfall, rainfall intensity, canopy interception and plant evaporation determine the water cycle to a large extend in the semi-humid areas and Southern regions of Europe. Therefore the capabilities of models to cope with canopy related factors are considered in the EUROHARP project.

NL-CAT (ANIMO)

The ANIMO model does not have a water balance simulation routine and thus requires data to be delivered by a water quantity model applied in advance. In the Euroharp project, these data are generated by the SWAP3.0 model (Kroes et al., 2003). The SWAP module comprises representations of:

- snow accumulation, snow melt with air temperature as driving force
- snow sublimation
- canopy interception with the Leaf Area Index as driving force
- potential soil evaporation and potential plant evaporation described by the Penman equation
- evaporation from open water. Ponding occurs when infiltration is limited or by inundation

The actual soil evaporation and plant evaporation is calculated as a function of potential rates, plant development stage and soil moisture conditions. The model runs on a daily basis.

EveNFlow

Given daily inputs of rainfall, potential evapotranspiration, and crop state parameters (including crop type and root depth), EveNFlow calculates the canopy interception of rainfall and the actual transpiration on a daily basis. PET is calculated using the Penman-Monteith equations and AE is calculated using the method given by Bailey and Spackman (1996). A new simple function to represent the effect of snow has been developed specifically for this project.

TRK (HBV)

The HBV model is the rainfall – runoff component of the TRK model, and it includes numerical descriptions of hydrological processes at the catchment scale. A routine for snow accumulation and melt is part of the model. Calculations are made for elevation zones in coupled subbasins within a catchment. The elevation zones are used for the snow and soil moisture routines only. The classes of land use are normally open areas, forests, lakes and glaciers. It is possible to use different parameterisation for the routines of soil moisture, runoff response and the interception storage capacity for different vegetation zones, but the ratios between the values for forested and non forested areas are kept constant. A geostatistical method is used for optimal interpolation of precipitation and temperature. For potential evapotranspiration, the model uses monthly data of long term mean, usually based on the Penman formula, which may be adjusted for temperature anomalies. Evaporation is also considered from water stored by interception. The snowmelt routine of the HBV model is a degree-day approach, based on air temperature, with a water holding capacity of snow which delays runoff. Melt is further distributed according to the temperature lapse rate. Snow distribution is considered as well as redistribution within each elevation zone. Soil frost can be computed from air temperature and snow conditions.

SWAT

The model computes evaporation from soils and plants separately as described by Ritchie (1972). Potential soil water evaporation is estimated as a function of potential evapotranspiration and leaf area index. Actual soil water evaporation is estimated by using exponential functions of soil depth and water content. Plant transpiration is simulated as a linear function of potential evapotranspiration and leaf area index. Potential Evapotranspiration is assumed to be unaffected by micro-climatic processes such as advection or heat-storage effects. The model offers three options for estimating potential evapotranspiration: Hargreaves, Priestley-Taylor, and Penman-Monteith.

Surface runoff and erosion

NL-CAT (ANIMO)

The SWAP3.0 model calculates surface runoff when:

- The rainfall intensity exceeds the infiltration capacity of the soil, whereas the infiltration capacity is governed by soil moisture conditions and soil hydraulic characteristics.
- The phreatic groundwater level rises to a level higher than the soil surface.

When the water ponding exceeds a certain defined threshold level, surface runoff occurs according to a non-linear conceptual relation with ponding depth. Parameterisation of this relation is done on the basis of calibration and extrapolation of effective parameters is conducted by taking into account the driving forces in the USDA-SCS curve number model. For solute transport simulations, the surface runoff is partitioned into overland flow and interflow. Sediment transport by erosion is not simulated but nutrient displacement by erosive transport is taken into account by multiplying the overland transport with an empirical factor which is related to rainfall intensity.

EveNFlow

In EveNFlow the magnitude of surface runoff, as a proportion of annual rainfall, is estimated as a function of the mean rainfall intensity and topsoil air capacity (Kirkby, 1976). The timing of surface runoff is determined by a dynamic version of the USDA-SCS curve number model, constrained so that annual total surface runoff is equal to that predicted by the Kirkby equation. Surface runoff can also occur by saturation excess. Soil characteristics are used to estimate the mean proportion of effective rainfall that is saturation excess surface runoff. The timing of this runoff is determined by a function of the catchment water store.

TRK (HBV)
Overland flow and surface runoff are components in the water balance which are described implicitly in the runoff response relation. The overland flow component can be determined from a subroutine based on either HBV groundwater conditions or the USDA-SCS curve number method. Erosion is not described in HBV.

SWAT
Using daily rainfall amounts, the model simulates surface runoff volumes and peak runoff rates for each HRU. Surface runoff is predicted for daily rainfall by using the SCS curve number equation (USDA-SCS, 1972). Erosion and sediment yield are estimated for each HRU with the Modified Universal Soil Loss Equation (MUSLE) (Williams, 1975). The hydrology model supplies estimates of runoff volume and peak runoff rate which, with the subbasin area, are used to calculate the runoff erosive energy variable. The crop management factor is recalculated every day that runoff occurs. It is a function of above-ground biomass, residue on the soil surface, and the minimum C factor for the plant. Other factors of the erosion equation are evaluated as described by Wischmeier and Smith (1978).

Soil water balance

NL-CAT (ANIMO)
The ANIMO model does not comprise a water balance simulation routine and thus requires data to be delivered by a water quantity model applied in advance. The SWAP3.0 model is used to generate hydrological information to be used as input for ANIMO. Simulations are run on a daily time step. SWAP3.0 calculates vertical water transport in the root zone, the unsaturated zone and the upper groundwater zone based on a discretisation of the Richards equation. The user should define the depth of the groundwater zone which is to be considered in the SWAP model. Options are available to define a geohydrological stratification of the upper groundwater zone. Drainage to different types of surface waters, interflow and root extraction for plant evaporation are defined as lateral sinks of each compartment. Root extraction is defined as a function of crop development stage, potential evaporation, soil moisture suction, rooting depth and plant parameters. Flow to drainage systems is calculated as a function of groundwater elevation and so called drainage resistances which can be derived by well known drainage formulae (Hooghoudt, Kirkham, Glover-Dumm). Drain tubes are considered to be a special type of surface water to each drainage flow may occur. Bypass through macropores and macropore flow is described by process oriented relations. Macropores are influenced by swelling and shrinking of the soil and thus by the soil moisture status. The soil moisture flow is hampered when the soil is frozen. Soil temperatures are calculated by the SWAP3.0 model using the air temperature as a boundary condition.

EveNFlow
EveNFlow uses a soil water balance to describe the water balance of the system. The model uses the hydrological year for its simulations. On the first day, the Soil moisture deficit (SMD) is assumed to be zero. SMD_k is calculated according to the following equation:

$$SMD_k = SMD_{k-1} + AET_k - P_k^* - Runoff$$

Where:
SMD_{k-1} is the soil moisture deficit (mm) estimated the previous $k\text{-}1^{th}$ day

AET_k	is the actual evapotranspiration (mm) from the crop for the k^{th} day
P_k^*	is the daily precipitation minus the interception (mm) for the k^{th} day
Runoff	is the surface runoff (mm).

Drainage or Hydrologically Effective Rainfall (HER) which is the input to the EveNFlow leaching function and drainage routing model occurs when the soil moisture content exceeds the field capacity value.

TRK (HBV)

The soil moisture routine of the HBV model initially emanates from the oversimplified bucket approach, but with the very important additional condition that the water holding capacity of the soil in the basin has a statistical distribution. This leads to a contributing area concept concerning runoff generation. Only those "buckets" that have reached their field capacity will contribute to runoff in the event of rain or snowmelt. It is very important to note that this approach thus implicitly accounts for the sub-basin or sub-grid variabilities in both soil water holding properties and input in the form of rain or snowmelt, without explicit separation of the two.

The parameter values of the model thus reflect the physical properties of the ground as well as their statistical distribution and also the random character of the input. The values of the parameters in different basins will therefore be identical as long as the basinwide distribution functions are the same. The routine will then be independent of, or at least only mildly sensitive to, scale. It is similar to the cumulative distribution function used for soil moisture saturation in the ARNO rainfall-runoff model (Todini, 1995), an approach that has also found its way into climate modelling (Dümenil and Todini, 1992) where sub-grid variability is a critical issue.

The runoff generation function of the HBV model covers a wide range of soil conditions with only two empirical parameters, FC and BETA. FC corresponds to the maximum basinwide water holding capacity of the soil and BETA describes how the runoff coefficient increases as this limit is approached. BETA is thus more an index of heterogeneity than of soil properties in the basin. A BETA value of zero implies that the basin is entirely lacking in water-holding capacity in the soil, whereas a high BETA value indicates such homogeneous conditions that the whole basin may be regarded as buckets that overflow simultaneously when their field capacity is reached.

SWAT

The amount of water entering the soil profile is calculated as the difference between the amount of rainfall and the amount of surface runoff. The redistribution component uses a storage routing technique to predict flow through each soil layer in the root zone. Percolation occurs when field capacity of a soil layer is exceeded and the layer below is not saturated. The flow rate is governed by the saturated conductivity of the soil layer. Movement of water from a subsurface layer to an adjoining upper layer may occur when the water content of the lower layer exceeds field capacity. The upward movement of water is regulated by the soil water to field capacity ratios of the two layers.

Lateral subsurface flow is a streamflow contribution which originates from below the surface but above the water table. Lateral subsurface flow in the soil profile (0-2m) is calculated simultaneously with redistribution. A kinematic storage model, taken from Sloan et al. (1983), is used to predict lateral flow in each soil layer. The model accounts for variations in conductivity, slope and soil water content. It also allows for flow upward to an adjacent layer or to the surface.

Return flow is defined as the volume of streamflow originating from groundwater. The model partitions groundwater into two aquifer systems: a shallow, unconfined aquifer which contributes return flow to streams within the watershed and a deep, confined aquifer which contributes return flow to streams outside the watershed. Water percolating past the bottom of the root zone is partitioned into two fractions—each fraction becomes recharge for one of the aquifers.

Surface water system

NL-CAT (ANIMO)
The surface water quantity is simulated using a distributed model which uses a network of nodes with connections between them. The nodes contain a certain volume of water based on the actual water level and the dimensions (e.g. length, width and slope) of the links connected to it. The connections can be defined as open watercourses with a certain resistance, or as a structure (e.g. weir, culvert, pump) with specific parameters. The specifications of structures can be changed in time by providing structure control time series. Water flow between the nodes is calculated as a linear function of the water level differences during timesteps and the calculated resistance of the connections. The surface water quality module NuswaLite (Jeuken and Groenendijk, 2003) simulates the retention and the ecological effects of nutrients in a river basin. An overview of the processes described are depicted in Figure 8.

EveNFlow
Daily time series of infiltration excess runoff and soil drainage for each diffuse source are routed to simulate the total river hydrograph and the proportions of flow derived from the saturation excess and macropore pathway components of soil drainage. The model is constructed so that is may be parameterised in catchments where observed flow data are unavailable. Parameterisation of the model requires information on the areas of soils of each HOST class within a catchment or long-term estimates of the Base Flow Index (BFI) from observed flows (Boorman *et al.*, 1995; NERC, 1998). The BFI is conceptualised as a measure of the proportion of flow that travels via the deeper, slower routes to the river system.

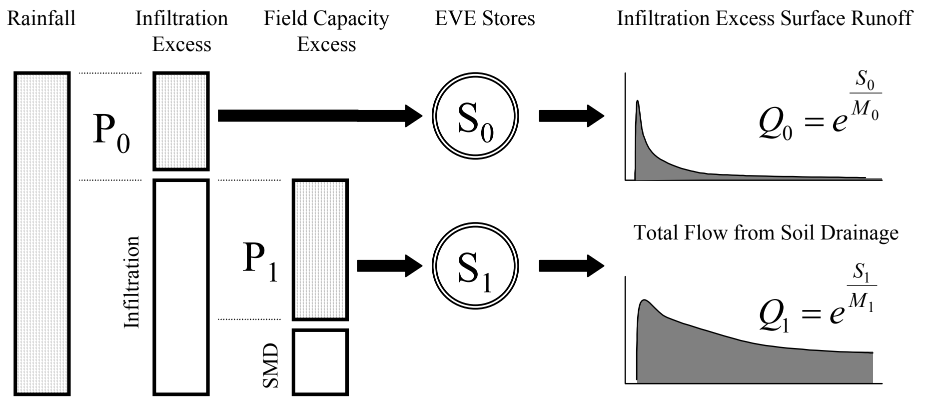
The flow methodology is based upon an exponential model of the drainage from a non-linear catchment soil water store or reservoir, as derived by Kirkby (1975), in which the instantaneous rate of discharge from the store is calculated as:

$$Q_t = Q_0 \cdot e^{\left(\frac{S_t}{M}\right)}$$

where Q_t is the rate of discharge at time t, Q_0 is the rate of discharge when the soil store is saturated, S_t is the catchment soil moisture store, and M is the master recession constant, representing the rate at which the soil store empties and hence the recession of the river hydrograph.

The store is representative of an exponential decline in lateral transmissivity of the soil with depth (Beven *et al.*, 1994). Thus, the change of the rate of discharge is nonlinear with respect to the rate of change of the catchment store. This nonlinear description enables the model to represent the delivery of water to a river channel via rapid flow routes such as macropores.

In EveNFlow, each diffuse source within a response group is represented by two soil water stores (Figure 23). The first is driven by additions of infiltration excess surface runoff, and the second by additions of soil drainage. Routed flow from the diffuse source is the sum of the predicted flows from each store.



P_0 : Proportion of rainfall that becomes infiltration excess surface runoff
P_1 : Proportion of infiltrating rainfall that becomes soil drainage after replenishing any soil moisture deficit

Figure 23: Schematic of rainfall separation into infiltration excess surface runoff and soil drainage in EveNFlow

The total flow from a response group is calculated by area weighting of the sum of infiltration excess and soil drainage derived flows for each diffuse source within the group. River flow at any point on the river network is then calculated by area weighting of the simulations for the response groups located upriver of the reach.

TRK (HBV)

The runoff generation routine is the response function which transforms excess water from the soil moisture zone to runoff. It also includes the effect of direct precipitation and evaporation on areas which represent lakes, rivers etc. The function consists of one upper, non-linear, and one lower, linear, reservoir. This is the origin of the quick (superficial channels) and slow (base-flow) runoff components of the hydrograph. Level pool routing is performed in lakes located at the outlet of a subbasin. The division into submodels, defined by the outlets of major lakes, is thus of great importance for determining the dynamics of the generated runoff. The routing between subbasins can be described by the Muskingum method (e.g., Shaw, 1988) or simple time lags. Each one of the subbasins has individual response functions.

Precipitation on lakes will be the same as for a non-forested zone at the same altitude and will be added to the lake water regardless of ice conditions in the same way for both rain and snow. Evaporation from lakes will equal the potential evaporation but can be modified by a parameter and will occur only when there is no ice. Transformation of runoff is taking place after water routing through the lake according to a rating curve. If no specific rating curve for the lake is given as input, the model will assume a general rating curve.

SWAT

SWAT uses Manning’s equation to define the rate and velocity of flow. Water is routed through the channel network using the variable storage routing method or the Muskingum river routing method. Both the variable storage and Muskingum routing methods are variations of the kinematic wave model. SWAT models four types of water bodies: ponds, wetlands, depressions/potholes, and reservoirs. Ponds, wetlands, and depressions / potholes are located within a subbasin off the main channel. Water flowing into these water bodies must originate from the subbasin in which the water body is located. Reservoirs are located on the main channel network. They receive water from all subbasins upstream of the water body. The model incorporates a simple mass balance module to simulate the transport of sediment into and out of water bodies. Sediment processes modelled in ponds, wetlands, reservoirs, and potholes are identical. When calculating sediment movement through a water body, SWAT assumes the system is completely mixed. A simple empirical model to predict the trophic status of water bodies is available. For studies that require detailed modelling of lake water quality, SWAT has been linked to distributed lake water quality models such as WASP.

3.3.5 Soil chemical processes

Table 2 has shown that six of the nine models (REALTA, N-LES CAT, MONERIS, EveNFlow, NOPOLU and SA) do not explicitly describe the internal soil physical/(bio)chemical turnover processes such as (de)sorption, precipitation, mineralization/immobilisation, (de)nitrification, and ammonia volatilisation. Therefore, these models are not discussed within this section.

The models NL-CAT and SWAT both describe the N and P processes, while the TRK (SOILNDB) model only describes N turnover. Recently, developments to build in a P module in the TRK system have been initiated.

Since the inorganic P pool in the soil is often 70-90% of the total amount of P accumulated in soils, it is very important how inorganic chemical reactions are described as this will determine the inorganic P concentrations leaching through the soil to surface waters. Only two models consider these chemical transformations: SWAT and NL-CAT.

NL-CAT (ANIMO)

In ANIMO three pools of inorganic phases are described. The behaviour of inorganic P is based on a soil physical/chemical description of inorganic P forms in soils and their transformations. The inorganic P pools are:

1. Adsorbed P at the surface of aggregates such as Al and Fe (hydr)oxides (directly available mineral P) called Q
2. Diffused P in aggregates and adsorbed or precipitated in the amorphous/micro crystalline aggregates (slowly available mineral P) called S
3. Precipitated P (directly or slowly available mineral P) called PREC

1) The *Langmuir* isotherm is derived from the assumption of a homogeneous monolayer of adsorbate on the adsorbent. The Langmuir equation is used to describe instantaneous sorption of phosphates to soil constituents (Van der Zee, 1987; Schoumans, 1995).

$$X_{e,PO4} = X_{e,\max,PO4} \frac{K_L \; c_{PO4}}{1 + K_L \; c_{PO4}} \qquad (3)$$

2) The difference between the equilibrium concentration that is reached in the steady state situation and the actual solid phase concentration is considered as the driving force for mass transfer. The ANIMO-model describes the rate dependent phosphate sorption to soil constituents by considering three separate sorption sites (Schoumans, 1995):

$$\rho_d \frac{\partial X_{n,PO4}}{\partial t} = \rho_d \sum_{i=1}^{3} \frac{\partial X_{n,PO4,i}}{\partial t} = \rho_d \sum_{i=1}^{3} (k_{ads,i}, k_{des,i}) \left(K_{F,i} \, c_{PO4}^{N_{PO4,i}} - X_{n,PO4,i} \right) \qquad (4)$$

3) Phosphate precipitation takes place when the concentration of the bulk solution exceeds a defined equilibrium concentration c_{eq}. The precipitation reaction is modelled as an instantaneous reaction. The reaction occurs immediately and complete when the solute concentration exceeds the equilibrium concentration c_{eq}. The precipitated minerals dissolve immediately when the concentration of the water phase drops below the buffer concentration. When the store of precipitated minerals has been exhausted, the term $\partial X_{p,PO4}/\partial t$ equals zero. In most of the application of the ANIMO model for Dutch sandy soils, the parameterisation of the model has

been restricted to the instantaneous precipitation formulation. For establishing the equilibrium concentration, the following relation between *pH* and c_{eq} has been utilised:

$$c_{eq} = 0.135 \bullet 3^{5-pH} \approx 10^{-0.447\ pH\ +\ 1.516} \qquad (5)$$

Schoumans and Groenendijk (2000) have shown that, based on this soil chemical/physical approach, it is possible to model also soil P test values that are commonly used for fertiliser recommendations. So, labile P, also called plant available P, can also be modelled based on these kinetics, which is interesting from an agricultural point of view (management practice).

SWAT

The main difference between the ANIMO and SWAT model is that within the SWAT model no distinction is made between precipitated P (third pool; PREC) and the slow diffusion of phosphorus in aggregates (second pool; S). So two pools are described:

1. Adsorbed P at the surface of aggregates like Al and Fe (hydr)oxides (directly available mineral P) called PAI
2. Diffused P in aggregates and adsorbed or precipitated in the amorphous/micro crystalline aggregates (slowly available mineral P) in combination wit a slow precipitation reaction (called)

1) The equilibrium surface reaction is described with:

$$P_{sol,act} = P_{solution} - \min P_{act} \bullet \frac{pai}{1-pai} \qquad if\ \ P_{solution} > \min P_{act}\ \frac{pai}{1-pai}$$

$$P_{sol,act} = 0.1\left(P_{solution} - \min P_{act} \bullet \frac{pai}{1-pai}\right) \qquad if\ \ P_{solution} < \min P_{act}\ \frac{pai}{1-pai}$$

where $P_{sol,\ act}$ is the amount of phosphorus transferred between the soluble and active mineral pool, $P_{solution}$ is the amount of phosphorus in solution (kg P/ha), $minP_{act}$, is the amount of phosphorus in the active mineral pool, and *pai* is the phosphorus availability index. The pai of a soil layer is defined as

$$pai = \frac{P_{solution,f} - P_{solution,i}}{fert_{\min P}}$$

where $P_{solution,f}$ is the amount of phosphorus in solution after fertilisation and incubation for a period of 6 months at 25^{o}C, $P_{solution,i}$ is the amount of phosphorus in solution before fertilisation, and $fert_{minP}$ is the amount of soluble P fertiliser added to the sample

When $P_{sol,\ act}$ is positive, phosphorus is being transferred from solution to the active mineral pool. When $P_{sol,\ act}$ is negative, phosphorus is being transferred from the active mineral pool to solution. Note that the rate of flow from the active mineral pool to solution is 1/10th of the rate of flow from solution to the active mineral pool. In fact there the "fast" adsorption/desorption reaction is time dependent and in fact not an equilibrium reaction.

2) Slow sorption/precipitation reaction

SWAT simulates slow phosphorus sorption by assuming the active mineral phosphorus pool is in slow equilibrium with the stable mineral phosphorus pool. At equilibrium, the stable mineral pool is 4 times the size of the active mineral pool.

$$P_{act,sta} = \beta_{eqP}\,(4\min P_{act} - P_{act,sta}) \qquad if\ \min P_{act,sta} < 4\min P_{act}$$

$$P_{act,sta} = 0.1\,\beta_{eqP}\,(4\min P_{act} - P_{act,sta}) \qquad if\ \min P_{act,sta} > 4\min P_{act}$$

where $P_{act,\mathrm{sta}}$ is the amount of phosphorus transferred between the active and stable mineral pools, β_{eqP} is the slow equilibration rate constant, $minP_{act}$ is the amount of phosphorus in the active mineral pool, and $minP_{sta}$ is the amount of phosphorus in the stable mineral pool.

3.3.6 Biochemical soil processes

With respect to biochemical soil processes two major processes will be distinguished and described: (1) organic nutrient cycle and (2) (de)nitrification. As noted before, only three models have an explicit mathematical description of these processes: SWAT, NL-CAT and TRK (SOILNDB).

Organic nutrient cycle

SWAT
Within the SWAT model two sources of mineralization are considered: fresh organic N pool, associated with crop residue and microbial biomass, and the stable organic N pool, associated with the soil humus. Mineralization from the fresh organic N pool is estimated with the equation

$$RMN_{\ell} = (DCR_{\ell})\,(FON_{\ell})$$

where RMN is the N mineralization rate in kg/ha/day for fresh organic N in layer l, DCR is the decay rate constant for the fresh organic N, and FON is the amount of fresh organic N present in kg/ha. The decay rate constant is a function of C:N ratio, C:P ratio, composition of crop residue, temperature, and soil water:

$$DCR_{\ell} = 0.05\ (CNP_{\ell}\ )\ \sqrt{\left(\frac{SW_{\ell}}{FC_{\ell}}\right) \bullet TF_{N\ell}}$$

where CNP is a C:N and C:P ratio factor and FC is the soil water content in mm at field capacity. Organic N associated with humus is divided into two pools--active and stable--by using the equation

$$ON_{a\ell} = (RTN_{\ell})\,(ON_{\ell})$$

where ON_a is the active or readily mineralised pool in kg/ha, RTN is the active pool fraction (set at 0.15), ON is the total organic N in kg/ha, and the subscript l is the soil layer number. Organic N flux between the active and stable pools is governed by the equilibrium equation

$$RON_{\ell} = BKN\,(ON_{a\ell}\ (\frac{1}{RTN_{\ell}}) - ON_{s\ell})$$

where RON is the flow rate in kg/ha/d between the active and stable organic N pools, BKN is the rate constant ($\approx 10^{-5}\cdot day^{-1}$), ON_s is the stable organic N pool, and subscript l is the soil layer number. The daily flow of humus related organic N (RON) is added to the stable pool and subtracted from the active pool.

Only the active pool of organic N is subjected to mineralization. The humus mineralization equation is

$$HMN_{\ell} = (CMN)(ON_{a\ell})(SWF_{\ell} \bullet TF_{N\ell})^{0.5}$$

where HMN is the mineralization rate in kg/ha/day for the active organic N pool in layer l and CMN is the humus rate constant (≈0.0003 day^{-1}). To maintain the N balance at the end of the day, the humus mineralization is subtracted from the active organic N pool; the residue mineralization is subtracted from the FON pool; 20% of RMN is added to the active ON pool; and 80% of RMN is added to WNO3 pool.

The daily amount of immobilisation is computed by subtracting the amount of N contained in the crop residue from the amount assimilated by the micro-organisms:

$$WIM_{\ell} = (DCR_{\ell})(FR_{\ell})(0.016 - c_{NFR})$$

where WIM is the N immobilisation rate in layer l in kg/ha/day; 0.016 is the result of assuming that C=0.4 FR, that C:N of the microbial biomass and their labile products = 10, and that 0.4 of C in the residue is assimilated; and c_{NFR} is the N concentration in the crop residue in g/g. Immobilisation may be limited by N or P availability. If the amount of N available is less than the amount of immobilisation predicted, the decay rate constant is adjusted with the relationship

$$DCR'_{\ell} = \frac{0.95\, WNO3_{\ell}}{FR_{\ell}(0.016 - c_{NFR})}$$

where DCR′ allows 95% use of the available NO_3-N in soil layer l. A similar adjustment is made if P is limiting. The crop residue is reduced by using the equation

$$FR_{\ell} = FR_{o\ell} - (DCR'_{\ell})(FR_{o\ell})$$

where FR_o and FR are the amounts of residue in soil layer l at the start and end of a day in kg/ha. Finally, the immobilised N is added to the FON pool and subtracted from the WNO3 pool.

ANIMO

Materials can vary strongly in quality. Each material consists of a fixed number classes, to be to be defined by the model user. This allows the mathematical simulation of empirical decomposition curves as given by Kolenbrander (1969) or Janssen (1986). When appropriate parameter sets are chosen for combination of class-fractions and first order rate constants of a certain organic material, the ANIMO concept is able to reproduce equivalent relations to these empirical approaches (Rijtema et al., 1997). Decomposition of fresh organic materials is described by:

$$\frac{d\,M_i(t)}{d\,t} = f_{i,1}\ OM_1(t_0)\,e^{-k_1(t-t_0)} + f_{i,2}\ OM_2(t_0)\,e^{-k_2(t-t_0)} + \bullet + f_{i,fn}\ OM_{fn}(t_0)\,e^{-k_{fn}(t-t_0)} + \bullet$$

Fresh organic materials and dissolved organic matter are applied as instantaneous pulse-type doses. The organic part of the applied substance is divided over fresh organic matter and dissolved organic matter.

Residual root materials of arable crops are added to the soil layers of the root zone at the end of the growing season. During the growing season, the growth and maintenance of the root system produces dead root cells and hair roots. These materials are defined as root exudates which are described by a separate pool with fixed nitrogen and phosphorus contents. Dry matter production of arable crops is defined as input to the ANIMO model, but for dry matter production and nutrient uptake of grassland the model comprises a dynamic sub-model. In this sub-model grassroots die continuously throughout the year. Dead roots are considered as a composition of two classes of fresh organic material and the division over these materials is calculated from the defined nutrient fractions and the actual nitrogen fractions of the remaining plant parts. Grazing losses and harvest losses of grass shoots are treated in a similar way, but are added to the top layer only.

The input of fresh organic matter to the soil system occurs by additions of manure, root materials, grazing and harvest losses and any other organic materials defined by the model user.

Decomposition of fresh organic materials results in dissimilation of organic carbon, solubilisation and transformation to the humus/biomass pool. Decomposition of dissolved organic compounds results in dissimilation and transformation to the humus/biomass pool. The humus/biomass pool decomposes to a residual fraction, accompanied by partial dissimilation of these residues. This residual material has been lumped with the humus/biomass pool, so only net dissimilation of this pool has been taken into consideration.

Production of exudates is considered for arable crops only. It has been formulated proportionally to the root mass increase. The root growth characteristic has to be defined as model input and from these data, the model calculates the increase of root mass during the simulation timestep. Within the simulation timestep, the root growth is assumed constant. On the basis of scarce literature data Berghuijs-van Dijk et al., (1985) derived an exudate production of 41% of the gross dry matter production of roots.

Production of humus/biomass results from the decomposition of fresh organic matter, dissolved organic matter, exudates and an internal turnover of humus. The assimilation process is accompanied by a dissimilation which requires most of the organic material for energy supply of the living biomass. The assimilation ratio a is taken constant for all organic matter pools. No separate production of humus/biomass as a result of humus/biomass turnover has been formulated explicitly, because the residual humus/biomass material has been lumped with the total humus/biomass pool and the rate constant has been formulated for the net decomposition.

As a result of organic matter dissimilation, part of the organic nitrogen is transformed into the mineral status. Another part of the organic nitrogen remains in the organic status in dead humic components. On the other hand, part of the mineral nitrogen can be immobilised through the biomass-synthesis in the living biomass. Depending on the assimilation ratio and the ratio between nitrogen content in parent fresh organic material and the nitrogen weight fraction of the humus/biomass pool, the transformation yields or requires the mineral of the nutrients. The net mineralisation of phosphorus:

$$R_{p,PO4} = \frac{1}{\Delta t} \int_{t_0}^{t_0+\Delta t} \Big(\sum_{fn=1}^{nf} (f_{P,fn} - a\, f_{P,hu})\, f_{hu}\, k_{fn}\, OM_{fn}(t) + (f_P^L - a\, f_{P,hu})\, k_s (\theta\, c_{OM}) + (f_{P,ex} - a\, f_{P,hu})\, k_{ex}\, EX(t) + f_{P,hu}\, k_{hu}\, HU(t) \Big)\, d\, t$$

.

When the right hand side of the equation takes a negative value, mineral phosphorus is immobilised.

Response functions on process rates
Transformation rate coefficients, decomposition of fresh organic materials, dissolved organic matter, exudates and humus biomass and the nitrification rate coefficient are defined by a reference value k_{ref}. Environmental influences are taken into account by multiplication factors for reduced aeration at wet conditions, drought stress at dry conditions, temperature and pH. For organic transformation processes:

$$k = f_{ae,OM}\, f_T\, f_\theta\, f_{pH}\, k_{ref}$$

All of these coefficients are non linear descriptions (e.g. for temperature the Arrhenius equation (Groenendijk en Kroes, 1995)

TRK (SOILNDB)

Three organic matter pools are defined: two fast cycling pools (litter and faeces), where litter represents an organic matter-microbial biomass complex receiving fresh organic material and faeces represents manure-derived faeces, and a slow cycling pool (humus) composed of stabilised decomposition products (Fig. 24). mineralisation of humus nitrogen is calculated as a first-order rate process controlled by a specific mineralization constant and response functions for soil temperature and moisture. Decomposition in the two organic carbon pools (litter and faeces) are the main controls on N mineralization from these sources.

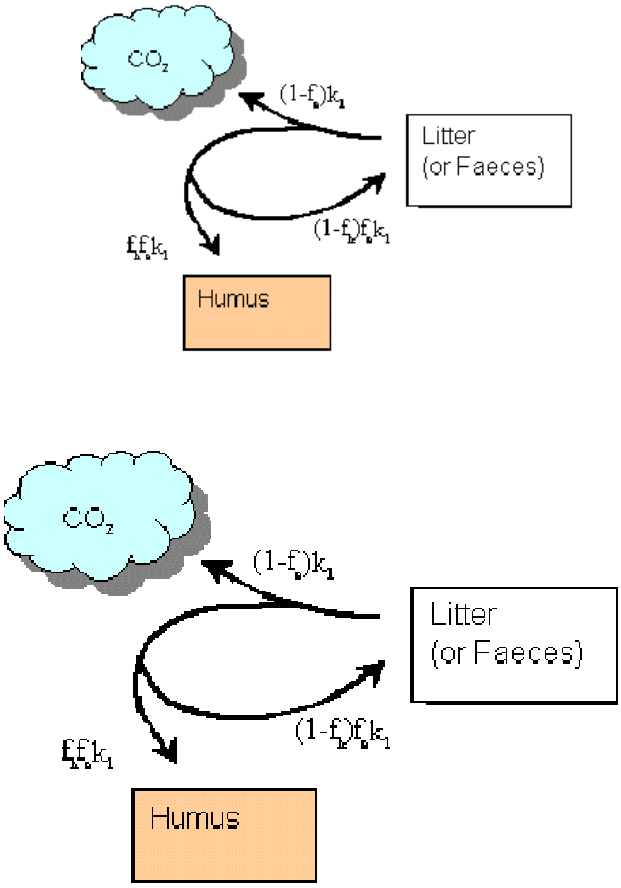


Fig. 24 Flow diagram showing the relative decomposition products formed

Decomposition of soil litter carbon and faeces carbon is calculated as:

$$C_{decomp,i} = k_{decomposition}\ f_T\, f_\theta\, C_i$$

where, i = litter, faeces.

$$C_{litter->co_2} = (1 - f_{e,l})\ C_{decomp}$$

$$C_{litter->humus} = f_{e,l}\ f_{h,l}\ C_{decomp}$$

where, f_e determines the fraction of carbon mineralised, and f_h determines the carbon flux to humus. Corresponding nitrogen flows are calculated assuming a constant carbon-nitrogen ratio of decomposed biomass and humification products. Nitrogen humification is calculated as:

$$N_{litter->humus} = C_{litter->humus}\ /\ cn$$

where cn is the C-N ratio of decomposer biomass and humified products.

The net mineralization of litter and faeces nitrogen is determined by balance between the release of nitrogen during decomposition and the nitrogen immobilised during microbial synthesis and humification:

$$N_{i\leftrightarrow NH_4^+}(z) = \left[\frac{\mathrm{N_i}(z)}{\mathrm{C_i}(z)} - \frac{f_e}{r_o}\right] C_{i(d)}(z)$$

where, i = litter, faeces. A Q_{10} expression is used for the soil temperature response function and regulates all biological processes in the model. The effect of soil moisture on biological activity (all processes except denitrification) is calculated based on the assumption that the activity decreases on either side of an optimum soil moisture content range.

Denitrification in soils

SWAT

As one of the microbial processes, denitrification is a function of temperature and water content. The equation used to estimate the denitrification rate is

$$DN_\ell = WNO3_\ell\ (1 - \exp[-1.4\,(TF_{N\ell})(C_\ell)]\ ),\quad SWF \geq 0.95$$

DN = 0. , SWF < 0.95

DN is the denitrification rate in layer l in kg/ha/day, TF_n is the nutrient cycling temperature factor, C is the organic carbon content in %, and SWF is the soil water factor. The temperature factor is expressed by the equation

$$SWF_\ell = \frac{SW_\ell}{FC_\ell}$$

where SW is the soil water content in layer l and FC is the field capacity in mm.

ANIMO

In the ANIMO model, it is assumed that denitrification is governed by soil organic matter respiration, the aeration status of the soil and nitrate availability. If the carbon content of organic material is taken as 58% on dry weight basis, it follows that the nitrate demand for denitrification can be expressed by a zero-order consumption term:

$$R_{p,den} = -0.58 \frac{24}{30} \frac{14}{12} f_{hetero}\ respiration\ rate$$

The factor f_{hetero} has been introduced to account for the reduced organic matter transformation rates when only nitrate oxygen is available. In many field validations and regional applications, a value 0.5 has been assumed for f_{hetero}. In case the nitrate concentration limits the decomposition of organic materials under anaerobic conditions, the following first order rate expression has been defined:

$$R_{d,den} = k_{den}\, \theta\, c_{NO3}(t)$$

where k_{den} is the first order rate constant to be defined as model input (d^{-1}). Determining which process rate limiting is done by computing both alternatives. The process leading to the highest nitrate concentration at the end of the time interval is subsequently selected by the model.

The partitioning between the aerobic soil fraction and the anaerobic soil fraction is determined by the equilibrium between oxygen demand for organic respiration processes plus nitrification and the oxygen supply capacity of the soil air and soil water system. Both the vertical diffusion in air filled pores and the lateral oxygen diffusion in the soil moisture phase are taken into consideration. The aeration fraction f_{ae} depends on a number of factors:

- Oxygen demand, as a result of organic transformations and nitrification. Oxidation of other reduced components (e.g. sulphur) have been ignored;
- Soil physical characteristics;
- Hydrological conditions (partitioning between soil moisture and soil air).

The aeration factor f_{ae} has been formulated as an multiplicative factor. At f_{ae} = 1, organic transformation and nitrification processes are optimal (Fig. 25). For sub-optimal conditions (f_{ae} < 1), the diffusive capacity of the unsaturated zone is insufficient to fulfil the oxygen requirement. In situations where partial anaerobiosis occurs, the oxygen demand for the organic transformations is met by atmospheric oxygen as well as by nitrate-oxygen. The nitrification rate will be sub-optimal. Under these conditions, the available nitrate will be reduced partial or complete (denitrification). Under unfavourable wet conditions the upper layers consume all oxygen which can enter the soil profile by diffusion and the atmospheric oxygen will not penetrate into the lower part of the unsaturated zone.

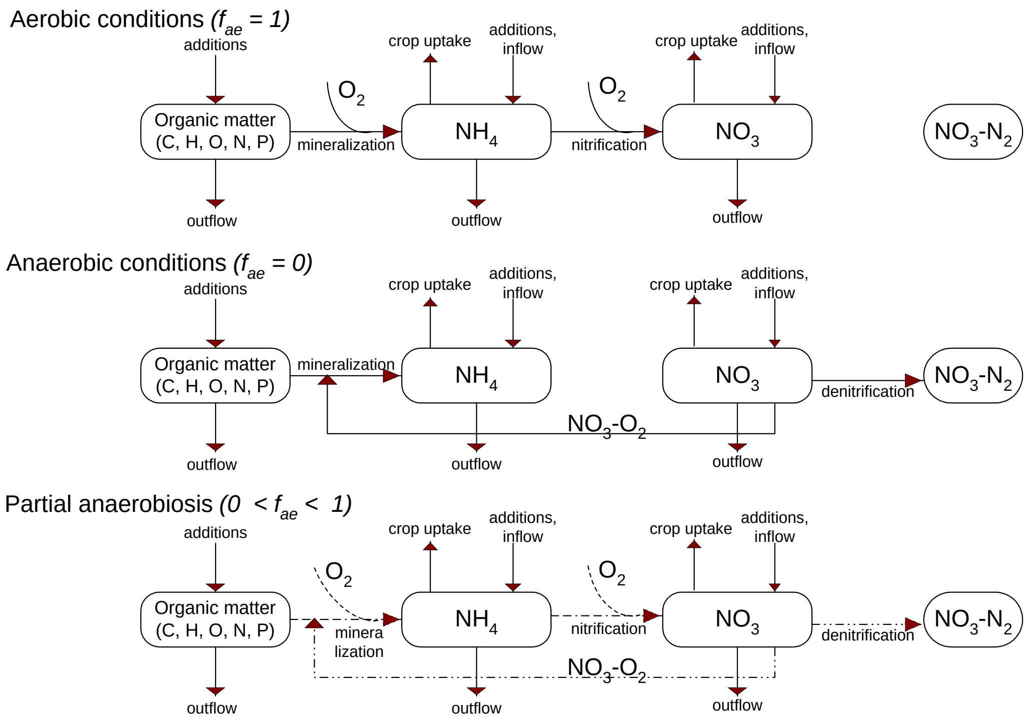


Fig. 25 Atmospheric oxygen and nitrate oxygen related processes in the ANIMO *model*

TRK (SOILNDB)

Denitrification is calculated as a first order rate process controlled by a potential rate and response function accounting for the effects of soil temperature, soil oxygen status and soil nitrate content. The effect of oxygen status is indirectly expressed as a function of soil moisture content, which increases linearly from zero at a threshold water content and reaches a maximum of 1.0 at saturation. The effect of nitrate is given as a Michaelis-Menten type expression, i.e. a hyperbolic function controlled by a half-saturation constant.

$$N_{den} = f(T)\, f(\theta)\, f(N_{NO_3})\, d_{dist}(\Delta z)\, d_{pot}$$

$$f(N_{NO_3}) = \frac{N_{NO_3}}{N_{NO_3} + d_{Nhalfsat}}$$

where d_{pot} is a parameter f(T), f(θ) and $f(N_{NO3})$ are response functions for soil temperature, soil moisture and nitrate concentration in the soil. A coefficient, $d_{dist}(\Delta z)$ adjusts the potential denitrification rate for each soil layer.

3.3.7 Approaches used for lumping soil processes

REALTA

With respect to REALTA, which is a risk assessment approach, no soil lumping procedures are really necessary.

A digital information system is used comprising a series of databases coupled with a Geographical Information System. Information is held on the following topics:

- Geology/geomorphology

- Land-use
- Soil characteristics
- Hydrology and hydrometry
- Agriculture and forestry
- Peat milling operations
- Fishery resource
- Municipal, industrial and other significant discharges
- Recreation and amenity resources
- Nature conservation and cultural heritage
- River and lake water quality

The Geographical Information System (GIS) has been used to investigate the relationship between a set of agricultural indicators and water pollution potential. Variation in both physical (land) characteristics and usage (management) practices are considered to influence the risk of nutrient loss to surface waters. The factors considered in evaluating the potential for *loss* and *transport* of diffuse nutrients from agricultural systems are:
(a) Chemical fertiliser loading
(b) Organic fertiliser loading (cattle, sheep, poultry)
(c) Organic fertiliser loading (piggeries)
(d) Soil phosphorus levels
(e) Runoff risk to surface waters

With respect to the runoff risk, the physical characteristics which influence the transport of phosphorus to surface waters (soil type and by inference drainage density, slope and rainfall) have been combined in a runoff risk map developed by Gleeson (1992). Gleeson's original eight risk classes have been simplified into high, medium, low and very low runoff risk. All other factors being equal, '*the greater risk of P-loss will coincide with those combination of factors that create a higher risk of runoff*' (Magette, 1998).

Other factors that have a significant bearing on nutrient loss from agriculture include farmyard condition and the management of landspreading activities. An equal bias for these factors has been assumed across the catchment in the absence of quantitative information of this nature on a catchment-wide basis. However, it is considered that the organic loading data, (b) and (c) above, in part reflect this variation in that greater volumes manure are generated, stored and disposed of in areas of higher stocking density.

A ranking scheme, Table 3, was developed whereby each of the phosphorus loss indicators is subdivided into zones of relative risk, each of which has a numerical value for scoring purposes. The relative importance between factors is also represented by a further scoring system or 'weighting'.

The total risk index is calculated by

$$P_{risk} = \sum_{i=1}^{5} w_i \ S(j)$$

Where, w_i is the weighting factor and S(j) is de score of the risk class within factor I (Table 3).

The result is then presented in the form of a composite map, which highlights priority areas to be at high or very high potential risk (respectively indexes 3 and 4 of four potential risk classes).

Table 3 Phosphorus Ranking Scheme

Factor	Factor Weighting	Risk Class	Score
(a) Chemical Fertiliser Loading	12	1. (0-9 kg/ha) 2. (10-11 kg/ha) 3. (12-14 kg/ha) 4. (15-19 kg/ha) 5. (20+ kg/ha)	0.8 1.6 2.4 3.2 4.0
(b) Organic Fertiliser Loading (cattle, sheep, poultry)	24	1. (0.0-1.0 LU/ha)* 2. (1.0-1.5 LU/ha) 3. (1.5-2.0 LU/ha) 4. (2.0 + LU/ha)	1.0 1.5 2.0 4.0
(c) Organic Fertiliser Loading (piggeries)	24	1. (low potential) 2. (moderately low potential) 3. (moderately high potential) 4. (high potential)	0.8 1.6 3.6 4.0
(d) Soil Phosphorus Levels**	16	1. (0-5 mg/l) 2. (6-9 mg/l) 3. (10-14 mg/l) 4. (15+ mg/l)	1.0 2.0 3.0 4.0
(e) Runoff Risk to Surface Waters	24	1. (very low risk) 2. (low risk) 3. (medium risk) 4. (high risk)	1.0 1.5 2.5 4.0

* Unit LU/ha is livestock units/hectare
** Morgan's Extractable Phosphorus

The calculated Risk index for a catchment is categorised into 4 groups (very high, high, medium and low). For each group the average phosphorus concentration in surface waters is determined within the catchment (for representative areas). Finally, this concentration is assumed to occur in all the areas with the same risk category.

Table 4 Comparison Between Identified Agricultural Risk Areas and Surface Water Quality

Risk Category	Number of Sampling Stations*	Number Satisfactory	Number Unsatisfactory	Average MRP Concentration (mg P/l)
Very High	13	7	6	0.054
High	45	27	18	0.035
Medium	125	110	15	0.019
Low	7	6	1	0.015

*Sampling stations immediately influenced by point discharges have been excluded

In Ireland, the results of the water quality monitoring programme (April 1998-March 1999) in the Lough Derg and Lough Ree catchment, confirmed a strong correlation between the areas identified as being of high or very high potential risk and poor water quality (Table 4). It is important to note that the agricultural risk map is not static, and will need to be periodically reviewed as knowledge regarding the factors influencing agricultural nutrient loss improves. The approach presented is based upon current best understanding and will benefit from the ongoing work in the agricultural mini-catchments, and from research undertaken by others.

NOPOLU
The model uses a soil surface balance type approach defined by the difference between the inputs to soil surface minus the harvest exports and gaseous N emissions emissions to the atmosphere.

Inputs	Exports
Wet and dry deposition from the atmosphere,	Harvested crop material.
Fixation by leguminous crops,	Herbage grazed

Organic wastes applied to agricultural land,
Nutrients in irrigation waters
Mineral fertilisers applied to agricultural land
Organic manure applied to agricultural land.

Gaseous emissions to the atmosphere

This model assumes that part of the net N and P surplus is directly transported to surface waters. The transfer factors are determined by a set of coefficients depending on:

- Soil type (S)
- CLC type (S)
- Slope factor (S)
- Yearly runoff factor based on rainfall patterns (max monthly average / long term average)
- Potential concentration of element N & P in groundwater and in soil
- Surplus value

The (S) symbol in the above bullet list indicates that the spatial resolution of these coefficients can be adjusted if some specific information is available in these areas.

<u>EveNFlow</u>

Several models were used to develop the NEAP-N crop and livestock maximum potential N loss coefficients, these include NITCAT for arable crops and manures (Lord, 1992); N-CYCLE for grassland and livestock systems (Scholefield *et al*., 1991); and MANNER, for the fate of manure nitrogen (Chambers *et al*., 1999). A series of more detailed field scale model runs were undertaken for a wide range of scenarios and summarised by the NEAP-N model coefficients. For grassland systems and grazing livestock (sheep, beef and dairy cattle), N loss potential coefficients are based on livestock type and numbers, and assumptions concerning the distribution and management of manures: further details and an example application are presented in Silgram *et al.* (2001). The NEAP-N coefficients are used in a module of the EveNFlow model in order to obtain estimates of the mass of nitrate present in the soil at the onset of winter drainage. This soil nitrate concentration leaching out of the profile is determined by a meta-model derived from SLIM (Addiscot and Whitmore, 1991).

<u>MONERIS</u>

Within the MONERIS quantification tool all soil processes regarding N and P are implicitly modelled. This means that reduction coefficients are used for the overall retention in soils. With respect to nitrogen only denitrification is taken in to account over the net N surplus, because net mineralization and net immobilisation are assumed to be negligible.

Nitrate concentrations in drainage waters

The calculation of nitrogen concentration in drainwater is based on the regionally differentiated N-surpluses (BACH ET AL., 1998). From the N-surpluses the potential nitrate concentration in leakage water is calculated according to FREDE & DABBERT (1998) which should correspond to the concentration in drainage water. It is assumed that the net mineralization and net immobilisation are negligibly low.

$$C_{DR_{NO3-N}} = \frac{(N_{ÜLN} - DNR) \cdot AF \cdot 100}{SW}$$

with $C_{DR_{NO3-N}}$ = nitrate concentration in drainage water [g N/l],
$N_{ÜLN}$ = nitrogen surplus of agricultural areas [kg N/ha/a],
DNR = denitrification rate [kg N/ha/a],
AF = exchange factor and
SW = leakage water quantity [l/(m^2·a)].

A denitrification rate of 30 kg N/ha/a can be used for such areas according to FREDE & DABBERT (1998). However, for some catchments, particularly in the Schleswig-Holstein coastal area, negative nitrate concentrations were calculated. To take this into account, the approach for the calculation of nitrate concentration in the seepage was modified so that the denitrification is considered in the form of a power coefficient DR, which is less than 1.

$$C_{DR_{NO3-N}} = \frac{\left(N_{ÜLN}\right)^{DR} \cdot AF \cdot 100}{SW}$$

with DR = exponent for denitrification.

The coefficient (DR) was estimated to 0.85 from a comparison of drain water concentrations.

Nitrate concentration in leakage water (non-drained areas)

The basis for the calculation of nitrate concentrations in non-drained areas is built on the defined correction factors for long-term changes in the regionally-differentiated N-surplus of agricultural land for the new and old German states and also on atmospheric deposition. Next, the average N-surplus is calculated from these three parameters on the basic outflow-carrying areas according to:

$$N_{ÜGES} = \frac{N_{ÜLN} \cdot A_{LN} \cdot LKF + N_{DEP} \cdot (A_{EZG} - A_{LN} - A_{W} - A_{URBV} - A_{GEB})}{A_{EZG} - A_{W} - A_{URBV} - A_{GEB}}$$

with
$N_{ÜGES}$ = total nitrogen surplus [kg/ha],
$N_{ÜLN}$ = nitrogen surplus of agricultural areas [kg/ha],
LKF = correction factor for the long-term changes in surpluses,
N_{DEP} = atmospheric nitrogen deposition [kg/ha],
A_{EZG} = catchment area [ha],
A_{LN} = agricultural area [ha],
A_{W} = total water surface area [ha],
A_{URBV} = impervious urban area [ha] and
A_{GEB} = mountain area [ha].

The N-surpluses thus estimated are used for the calculation of the overall potential nitrate concentrations in leakage waters for the areas contributing to base flow. For this, the first steps of the approach of FREDE & DABBERT (1998) are also used. A condition for this is that the net-mineralization and immobilisation are negligible for both time periods. Furthermore, it is assumed that there is no denitrification in the root-zone. Then, the following applies:

$$C_{SWPOT_{NO3-N}} = \frac{N_{ÜGES} \cdot AF \cdot 100}{SW}$$

with
$C_{SWPOT_{NO3-N}}$ = potential nitrate concentration in leakage water for the total area with base flow [g N/m³],
AF = exchange factor and
SW = leakage water quantity [l/(m^2·a)].

Nitrate concentration in groundwater

For the derivation of a catchment-specific model for denitrification in soil, in the unsaturated zone and in the aquifer, the potential nitrate concentration in leakage water is compared to the nitrate concentration in groundwater and retention functions were estimated.

It is assumed that the following relationship between groundwater and leakage water concentrations exists:

$$C_{GW_{NO3-N}} = \frac{1}{1 + R_{NO3-N}} \cdot C^{a}_{SWPOT_{NO3-N}}$$

with $C_{GW_{NO3\text{-}N}}$ = nitrate concentration in groundwater [g N/m³],
a = model coefficient and
$R_{GW_{NO3\text{-}N}}$ = retention or denitrification of nitrate in the unsaturated zone and in groundwater.

For $R_{GW_{NO3\text{-}N}}$, it is assumed that the retention is a function of the leakage water level and the hydrological conditions:

$$R_{GW_{NO3-N}} = k_1 \cdot SW^{k_2}$$

with k_1 and k_2 = model coefficients.

To characterise the hydrogeological conditions, two particular groups for the unconsolidated and consolidated rock region are chosen according to the hydrogeological map. For both types of rock region, one group with high permeability and another group with low water permeability is chosen.

The nitrate concentrations in groundwater can than be calculated according to Equation:

$$C_{GW_{NO3-N}} = \left(\sum_{i=1}^{4} \frac{1}{1 + k_{1i} \cdot SW^{k2i}} \cdot \frac{A_{HG\,i}}{A_{EZG}} \right) \cdot C^{a}_{SWPOT_{NO3-N}}$$

with A_{HG} = area of different hydrogeologically rock types [km²].

The coefficients a, k_1 and k_2 are determined by means of calculations of non-linear adjustment with the condition that the sum of squares should be minimal. For this, the solver unit of a spreadsheet program was used. A value of 0.627 was determined for coefficient a.

Phosphorus

It has already been reported by WERNER ET AL. (1991) that the extractable dissolved P concentration often lies in a range from 0.8 to 1 g P/m³. BRAUN ET AL. (1991) assume for the Swiss Rhine basins downstream of the lakes that the P concentration in the surface runoff is 0.5 g P/m³ for arable land and 2 g P/m³ for grassland. These results could be validated from the extensive studies on P content and P absorption capacity of soils in the northeast German flatlands (PÖTHIG & BEHRENDT, 1999). From these studies it can be derived that the water-extractable P concentration depends very strongly on the P saturation of the soil as shown in Figure 26.

Assuming that for arable areas the topsoil layer shows on average a P saturation of 90 to 95%, one can expect from the equations in Figure 26 a P concentration of ca. 1 g P/m³ for the soil solution. GELBRECHT ET AL. (1996) proved that such concentrations are usually especially during storm water events in frozen soils and in puddles of arable land. If the soil is 80 percent saturated, the value of water soluble P concentrations is reduced to 0.2 g P/m³ and with 50% saturation to around 0.05 g P/m³.

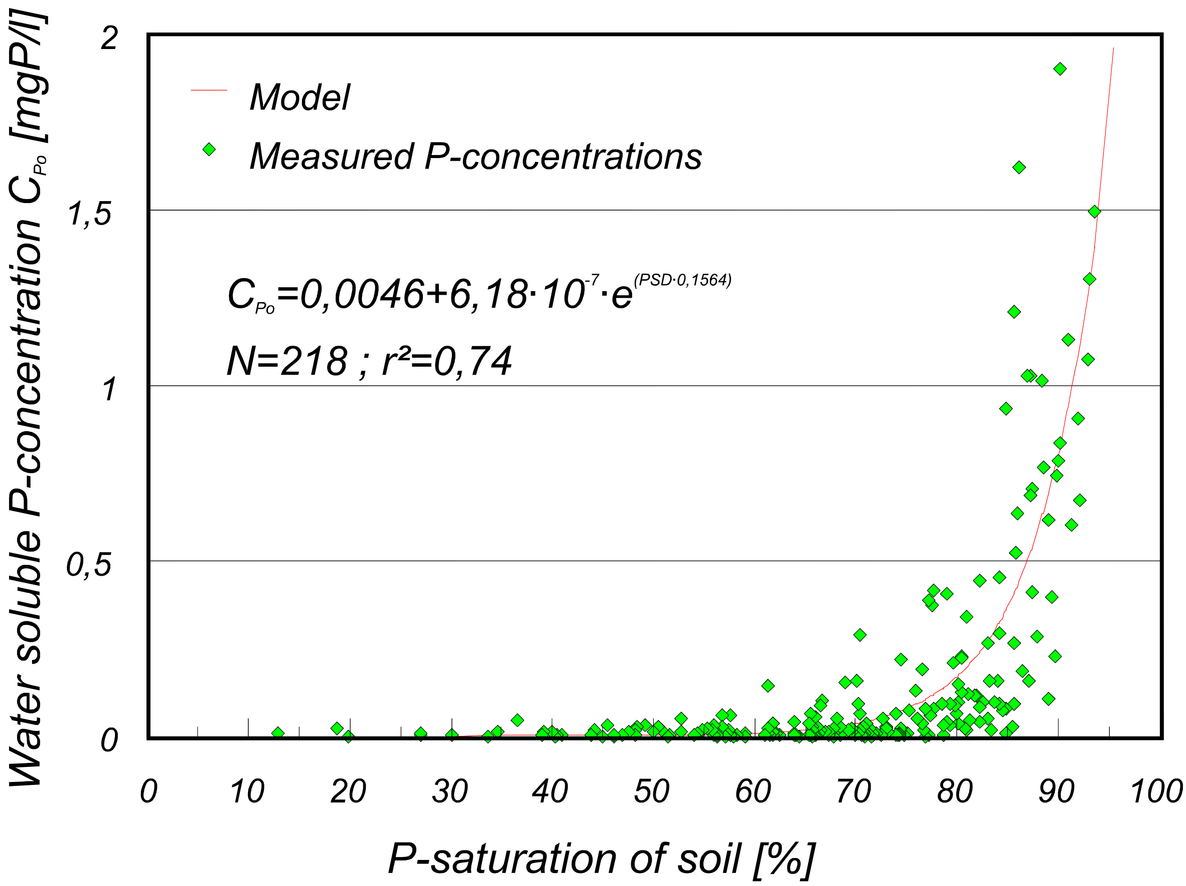


Fig. 26 Dependence of the water-extractable P-concentration on the P-saturation of soils used within MONERIS.

The values given in Table 5 for dissolved P concentrations in surface runoff from arable land, grassland and open areas are based on the assumption that in arable land, an average P saturation of the topsoil layer is around 90%. For grassland, a P saturation is assumed to be only 70% because of lower P accumulations. For the natural open areas, the P saturation in topsoil is assumed to be only 50% or lower. Where the dissolved P concentrations in the topsoil are elevated, particularly with a saturation of more than 60%, the estimates should be more detailed depending on area-specific information on the P saturation in topsoils. This would require the knowledge of the area-specific P accumulation in the last 40 to 50 years or the soil P content and its change over time and regional estimates of the P sorption capacity. However, essential data for such a regionalisation are not yet available.

Table 5 Estimation of used nutrient concentrations in surface runoff for arable land, grassland and open areas.

Use	**Nitrogen**	**Phosphorus**
	[g N/m³]	[g P/m³]
Arable land	$0.3+N_{DEP}/N_J$	0.8
Grassland	N_{DEP}/N_J	0.2
Open land	N_{DEP}/N_J	0.05

Calculation of N and P-content in topsoil and enrichment ratios

The N and P content of the topsoil are estimated together with the enrichment ratio. For the calculation of topsoil P contents, the yearly P surpluses and cumulative values are calculated on the basis of statistical information on mineral fertiliser applications, animal numbers and harvest offtakes. A non-linear regression between the estimated ER and the specific load of suspended solids explains the most of the variance of the enrichment ratio. The following model is derived for the calculation of the enrichment ratio:

$$\boldsymbol{ER_P = 18 \cdot l_{AFS}^{-0,47}}$$

where ER_P = enrichment ratio for phosphorus and
l_{AFS} = specific load of suspended solids [t/km²/a].

$$\boldsymbol{ER_N = 7{,}7 \cdot l_{AFS}^{-0,47}}$$

where ER_N = enrichment ratio for nitrogen.

<u>TRK (SOILNDB; HBV-N; P regression for arable land)</u>

In the TRK system landscape information, leaching rates and emissions are combined through GIS. The SOILNDB application results in one normalised concentration for each combination of region, soil and crop. For each subbasin, an average root-zone concentration is then calculated based on land-use information of areal crop and soil distribution. This "lumped" average leaching concentration is assigned to the water discharge from the root zone in the HBV-N catchment model.

<u>TRK (P regression, arable land)</u>

Additionally, within the TRK-system a lumped model is used to describe the phosphorus losses from arable land. This model has been developed via a multiple regression method (Ulén, B., Johansson, G. & Kyllmar, K. 2001) and has been simplified to adapt to available input data.

The transport of total phosphorus from arable land is calculated from the following regression:

TP = (-0,0803 + 0,10 x Density,LD + 0,003 x SoilSps + 0,0025 x PHClss) x Q

Where Density, LD is the livestock density, SoilSps is the soil specific surface area, PHCLss is the HCl soluble phosphorus in the topsoil and Q is the runoff from arable land. Unit of TRP TP is kg/ha/year. This regression is easy to apply, but is a simplification of all processes inducing P losses from arable land. It is unclear if it is valid outside the range and region that it was developed for.

3.3.8 Model Output

Table 6 shows which Nitrogen and Phosphorus species are modelled by the different quantification tools. All Phosphorus quantification tools are able to calculate emissions of total P concentrations. Only two models (SWAT and NL-CAT) are able to divide this total concentrations of different P species. For Nitrogen, almost all models are able to calculate the total N concentrations (except N-LES CAT and EveNFlow which focus on nitrate losses). Different nitrogen species are modelled by SWAT, NL-CAT and TRK.

Table 6 Overview of the Nitrogen and Phosphorus species modelled by the quantification tools

Model pathway, process or characteristic	**NL-CAT**	**REALTA**	**NLES**	**MONERIS**	**TRK**	**SWAT**	**EVEN-FLOW**	**NOPOLU**	**SOURCE AP.**
QT number	1	2	3	4	5	6	7	8	9
Species of N and P; concentration modelled									
Nitrogen									
- NH4	Y	N	N	N	Y	N	N	N	N
- NO3	Y	N	Y	N	Y	Y	Y	N	N
- DIN (dissolved inorg. N)	Y	N		Y^	Y				
- organic N	Y	N	N	N	Y	Y	N	N	N
- eroded particulate N	Y	N	N	N	N	Y	N	N	N
- total N	Y	N	N	Y	Y	Y	N	Y	Y
Phosphorus									
- PO4	Y	Y	N	N	N	Y	N	N	N
- organic P	Y	N	N	N	N	N	N	N	N
- eroded particulate P	Y	N	N	N	N	Y	N	N	N
- total P	Y	Y	N	Y	Y	Y	N	Y	Y

^ MONERIS (combination of NH_4 and NO_3)

4. Evaluation

This chapter considers the quantification tools in terms of:
(a) the potential ability to calculate the dynamics of different forms of nutrient losses and pathways from agricultural land to surfaces water within a catchment (focus of OSPAR HARP Guideline 6)
(b) the potential capability to evaluate the impact of nutrient management strategies, land use changes and water measures on nutrient losses (fluxes, concentrations, dynamics)

From this point of view a 'scientific' and an 'operational' evaluation is necessary, based on the description of the quantification tools in Chapter 3. The scientific evaluation focussed on the following topics: (1) spatial and temporal resolution, (2) pathways represented, (3) process and nutrient species considered. The operational evaluation focused on: (1) potential costs of application, (2) restrictions for applications (scenario analyses) and (3) applicability.

4.1 Spatial and temporal resolution

The horizontal spatial resolution of the models increases from about 0.1 km^2 to 50 km^2 in the order:

N-LES CAT < NL-CAT=SWAT = TRK < EveNFlow = SA < NOPOLU = REALTA < MONERIS

These differing spatial resolutions reflect the original focus of the models' development, which range from field/small catchment approaches to large river basin models. The models with a detailed spatial resolution are also able to describe large river basins, but for those models a smart discretisation of the whole basin is necessary in order to identify "homogeneous unique areas". Most models have discretisation routines or documented methodologies to divide the basin into homogeneous unique areas based on available maps (geo-referenced input data). As computing speed continues to increase each year, the actual computer time required to run each model for each elementary area and the river basin as a whole is becoming less important. The time needed for the discretisation and parameterisation stages of the model application therefore becomes much more important for determining the cost for each model application on a particular catchment. The assumptions made during model development (e.g. lumping of processes and parameters) strongly influence the acceptable smallest size of each unique elementary modelling unit area.

Several of the quantification tools which model nutrient losses from soils have a lower vertical boundary that reaches the uppermost boundary of deep aquifers. Other simpler approaches such as REALTA, NOPOLU and SA do not model the processes within the soil at all. Instead REALTA uses a risk ranking system and NOPOLU uses an export coefficient for specific agricultural areas. NLES-CAT estimates the nitrate losses from the root zone, where nitrate is routed to surface waters via different pathways based on a hydrograph separation technique.

With respect to the temporal resolution, all quantification tools are able calculate ***annual*** nutrient losses (N and P) from agricultural land to surface waters – a major objective of this broader study is an intercomparison of predictions using the different approaches. Only four of the models studied are capable of simulating the dynamics of nutrient losses to surface waters (daily loads: SWAT, TRK, NL-CAT and EveNFlow). Since nutrient losses are strongly influenced by intense rainfall events, it is clear that the other quantification tools are therefore not capable of simulating the impact of changes in meteorological conditions or water management on the environmental losses from agricultural land *within* an individual year (i.e.

modelling dynamic peaks and troughs in flow and concentration). For the other models studied, such as MONERIS, simulated fluxes are only generated on an annual basis.

4.2 Pathways

A important limitation of four of the models studied (REALTA, NOPOLU, NLES-CAT, SA) is that they are not able to quantify the water flow by different pathways by themselves, but they need measured flow data for each (or a total) of the simulated pathways. MONERIS does model losses via individual hydrological pathways, but requires river flow as an input.

Modelling the water flow is often one of the most complex parts of many of the models studied, and is often one of the most difficult to predict accurately on a daily basis. As both N and P are moved within the water body, it is clearly of prime importance to obtain a satisfactory simulation of river flows prior to focusing attention on the representation of nutrient losses themselves. By checking simulations of flow and concentration against measured river data, it is also possible to infer the relative importance of different sources of error in model predictions (i.e. errors associated with simulating the water balance and hydrological routing, or biogeochemical processes). Annex A contains a summary of all hydrological aspects of each of the quantification tools.

The approaches of the models which can simulate water flow (NL-CAT, SWAT, TRK, and EveNFlow) differ markedly and are discussed below.

The Swedish model TRK (HBV) has a detailed representation of modelling *snow and melting* processes. In contrast, in NL-CAT and (a new routine developed in) EveNFlow a simpler approach is used, whereas SWAT does not take into account snow/melting effects. So, from this point of view there may be limitations on some of the models in terms of their potential suitability for modelling the climate in Nordic countries.

In EveNFlow the magnitude of *surface runoff*, as a proportion of annual rainfall, is estimated as a function of the mean rainfall intensity and topsoil air capacity (Kirkby, 1976). The timing of surface runoff is determined by a dynamic version of the USDA-SCS curve number model, constrained so that annual total surface runoff is equal to that predicted by the Kirkby equation. Surface runoff can also occur by saturation excess. Soil characteristics are used to estimate the mean proportion of effective rainfall that is saturation excess surface runoff. The timing of this runoff is determined by a function of the catchment water store.

In SWAT the surface runoff is described by the USDA-SCS curve number model on a daily base. In NL-CAT, runoff occurs in situations when rainfall intensity exceeds the infiltration capacity of the soil (defined by soil moisture conditions and soil hydraulic characteristics) and when the phreatic water level rises above the soil surface (e.g. in wet polder areas). When the water ponding exceeds a certain defined threshold level, surface runoff occurs according to a non-linear conceptual relationship with ponding depth. Surface runoff is also partitioned into overland flow and interflow. Within TRK (SOILNDB/HBV-N) no distinction is made between surface runoff and root zone leaching from the soils. The overland water flow component can be determined from a subroutine based on either HBV groundwater conditions or the USDA-SCS curve number method. However, this function is normally not used in nitrogen modelling.

Soil water balance and the water drained (*drainage*) through the soil to surface waters are explicitly described by the four quantification tools NL-CAT, SWAT, TRK, and EveNFlow.

For TRK leaching occurs under circumstances when the soil moisture content exceeds the field capacity in the soil water balance. The primary hydrological unit in the semi distributed HBV model are sub-basins, which are further divided into elevation zones with separate calculations of soil moisture. Percolation and drainage to surface water is calculated by means of a system of linear and non-linear groundwater reservoirs. The function consists of one upper, non-linear, and one lower, linear, reservoir for each subbasin. These are the origin of the quick (superficial channels) and slow (base-flow) runoff components of the hydrograph.

Within EveNFlow leaching occurs under circumstances when the soil moisture content exceeds the field capacity value. The soil moisture content is recalculated by a water balance for the soil system. The amount of soil water increases or decreases based on net water input (precipitation minus actual evapotranspiration minus runoff).

Within NL-CAT, the SWAP model solves the Richards equation numerically. The method takes into account the distribution with depth of the water retention curve and the hydraulic conductivity. Preferential flow in macropores can be represented. Different types of drainage systems and water courses can be described by means of a piecewise linear relation between groundwater level and drain discharge. In NL-CAT the lower boundary of a field/plot in the distributed model consists of a boundary condition by which the influence of regional deep groundwater flow can be taken into account. For water flow, the boundary condition can either be defined as a specific potential, a flux or a mixed condition, and for solute transport the boundary condition is a concentration value.

The SWAT model does not solve the Richards equation but utilises a conceptual model for the soil water movement. SWAT considers the upper groundwater zone as a model layer. Properties of this model layer determine the response of inputs to the groundwater system. Upward seepage and exchange with vadose zone are also considered.

The NL-CAT soil moisture module describes the water movement through the soil in considerable detail, including macropore flow. The SWAT model and the TRK (HBV) model use semi-empirical relations to describe the percolation from the root zone to deeper soil layers.

4.3 Processes and modelled nutrient species

The major differences between the quantification tools are concerned with the way nutrient biological and chemical processes in soils are modelled.

The *source apportionment method (SA) and the REALTA and NOPOLU quantification tools* do not take into account any internal soil processes. In the N-LES CAT model, which is a statistical relationship between on the one hand nitrogen input, crop, soils, and climate characteristics, and on the other hand measured nitrate concentrations leaching out of the root zone, the internal nutrient processes are implicitly taken into account. For all these four models (SA, REALTA, NOPOLU and N-LES CAT) soil processes are lumped and implicitly derived from measured monitoring data. *In those cases direct extrapolation to other soil and climate or hydrological conditions may not be possible.*

With MONERIS, net mineralisation/immobilisation is assumed to be zero under all conditions. Furthermore, the net N surplus (input minus harvest) is assumed to be completely transformed into dissolved inorganic nitrogen. The reduction of the nitrate concentration due to denitrification, is lumped into one equation and has been fitted based on measurements of DIN or nitrate concentrations within German catchments. The procedure considers the residence time in the unsaturated zone and in the groundwater by taking into account the historical data on nitrogen surplus (1 to 50 years). The flow by groundwater and natural interflow is calculated from the water balance (total flow minus the flow of surface runoff, flow from tile drained area, flow from urban areas). With respect to phosphorus, no sorption or desorption mechanisms are taken into account within MONERIS.

Within EveNFlow a module estimates the mass of nitrate present in the soil at the onset of winter drainage that is vulnerable to leaching. The calculation is based upon empirical relationships between soil nitrogen supply and the nutrient balance under conventional cropping and grazing regimes. A set of baseline coefficients has been derived for the UK for maximum potential N loss for each arable crop and for each category of livestock. EveNFlow uses a meta-model to estimate nitrate losses as a function of rainfall and soil water content in relation to the potential nitrate losses.

The quantification tools SWAT, NL-CAT and TRK (SOILNDB) have specific representations of nutrient dynamics in soils. In SWAT all processes (plant growth, mineralization, immobilisation, denitrification, sorption and desorption) are modelled, but for each process a lumped equation is used. TRK (SOILNDB) and NL-CAT are more comparable in their approach (regarding nitrogen processes). With respect to the mathematical description of the N processes, the differences are probably small due to the fact that both models originate from agricultural nutrient models originally developed at field scale. However, there are still some differences between those two nitrogen approaches, but these are small compared to the differences between the other models within EUROHARP. With respect to phosphorus the differences are significant because TRK uses an empirical (statistical) derived equation for Swedish conditions because the sub-model SOILNDB is only used for nitrogen species.

The representation of individual nitrogen processes with occur in the soil therefore decreases in the following order:

NL-CAT > TRK (SOILNDB) > SWAP>> EveNFlow > MONERIS > N-LES CAT > NOPOLU > SA

With respect to phosphorus the order is:

NL-CAT > SWAP>> MONERIS > TRK = NOPOLU > SA

4.4 Cost implications

Based on the workload needed to apply a given model on one new catchment, the number of man-months of *anticipated* input required for each model application increases from about 0.5 man-months up to 3 man-months per catchment per nutrient.

For the nitrogen quantification tools the amount of *anticipated* workload increases from

SA < MONERIS < N-LES CAT = EveNFlow = TRK =SWAT < NL-CAT

For phosphorus the total workload increases in the following order:

SA < NOPOLU = REALTA < TRK < MONERIS < SWAT < NL-CAT

The actual run-time of each model by the computer is negligible compared to the total time needed for earlier stages preparing for the model run including data processing, formatting and preparation, catchment discretisation, and model parameterisation etc..

So, the SA, NOPOLU and REALTA are the least demanding in terms of time (not necessarily in terms of data costs), while the NL-CAT approach is the most demanding in terms of time input. This sequence also reflects the level of process detail described in the quantification tools (see Sections 4.1 to 4.4 above). Although the time costs of applying the quantification tools differ substantially, the most suitable model for a particular application will depend on the purpose of the study (e.g. identifying risk areas; detailed quantification of the contribution to pollutant load from different sources; scenario analysis for mitigation or climate change etc.) and the quality (accuracy and precision) needed from model results. A comparison of the *actual* costs of model application against performance for each model (i.e. cost-effectiveness) will only be possible once the model applications themselves have been completed.

4.5 Potential suitability for scenario analysis

As already mentioned in the introduction, the suitability of the quantification tools for exploring scenario analyses will be considered in detail in a later stage of the EUROHARP project. However, a preliminary view can be developed now based on the model descriptions presented in previous sections, and focusing on the potential sensitivity of different models to different management strategies, land use changes and water measures.

With respect to scenarios dealing with the nutrient management strategies, it is clear that those models that include agricultural practices such as crop-soil input, manuring, fertilisation, ploughing etc, will have the potential ability to predict the impact of land management strategies on nutrient losses to surface waters due to changes in the amounts and timings of applications of nutrient input related to fertiliser and manure. In Table 7 some initial qualitative comments are given concerning the potential suitability of different models for this type of scenario analysis. Most models are able to predict change in nutrient losses due to changes in fertiliser application or livestock numbers. However, the simpler models may not be able to consider some changes in management and therefore may have more limited potential for scenario investigations predicting the impact of land management changes on nutrient loss to surface waters.

Table 7 Potential suitability of models for three types of scenario analysis

QT	Nutrient Management	Land use Changes	Water Measures
EVENFLOW - N	o	+	+
MONERIS - N	+	+	–
MONERIS - P	+	o	–
NLCAT - N	++	++	++
NLCAT - P	++	+	++
N-LES CAT - N	+	o	–
NOPOLU - N	o	o	–
NOPOLU - P	–	–	–
REALTA - P	–	–	–
SA - N	–	–	–
SA - P	–	–	–
SWAT - N	++	++	++
SWAT - P	+	+	++
TRK - N	++	++	++
TRK - P	–	–	–

++ very suitable (e.g. dynamic effects on turnover are modelled)
+ suitable (key processes are considered, at least in a lumped manner)
o more or less suitable (e.g. only long-term effects assessed without major recalibration)
- not suitable (model does not take account of management practices)

In Table 7 a second column relates to the capability of models to describe the impact of land use changes on diffuse load from agricultural land to surface waters. An example of profound land use change might be the change from an agricultural crop type to a "nature crop type" (forest; extensive grassland) where no fertiliser and virtually no manure is applied.

A third type of scenario analysis refers to water management strategies. This concerns changes in nutrient losses from agricultural land to surface waters due to major constructions in surface waters (e.g. to control high water levels; water conservation for rewetting areas). It is clear that these measures will have also an impact on the distribution of the water flow from land to surface waters. In principle the impact of water management strategies (Table 7) on nutrient losses from agricultural land to surface waters can only be determined by models which contain a hydrological component. Four of the nine quantification tools do not have an explicit hydrological module (REALTA, NOPOLU, N-LES CAT, and Source Apportionment). These models need this input data from another model or from measured data. So, these models are not able to evaluate water management scenarios independently. However, models such as MONERIS can calculate the response of different water management by changes in the relative importance of hydrological pathways – such as changes of urban areas or areas of land with tile drains.

Table 7 shows that if the complexity of the models increases then the potential for scenario analyses also potentially increases (but the time needed for individual model applications increases too). The main differences between the quantification tools can be explained by the origins of the models. The less complex tools were often developed to calculate the contribution of the different sources on the total loss from the outlet of the catchment (SA) and to screen for high risk problem areas (agricultural "hot spots") within the catchment and (REALTA, NOPOLU). The moderately complex catchment scale models (MONERIS, EveNFlow, N-LES CAT) achieve this objective but are also sensitive to major changes in agricultural activity (all three models) or the impact of hydrology on nitrogen losses (EveNFlow only). The most complex models (TRK, NL-CAT) originate from very detailed field-scale models which represent individual processes in great detail. This level of detail provides some potential benefits in terms of potential suitability for scenario work, but carries the disadvantages of higher time inputs and greater parameterisation issues.

4.6 Applicability

The applicability of the quantification tools is the most complicated question to consider, because different criteria can be used to qualify the applicability, for example:

- is the model valid for use under the specific catchment conditions being considered?
- what is the temporal and spatial scale at which model output is required, and which chemical species need to be modelled?
- what are the resource limitations (time and data costs) on a particular study (the models vary widely in their high or low data input requirements and time needed for model applications)
- does the model itself (rather than the model results) have to be passed to a third party (as some models are currently only used by their development team)

The answers to these questions are clearly different for different purposes, so no simple, single recommendations is going to be possible. Nonetheless, once the results of applications of each model to the three core catchments have been assessed, it will be possible to provide some guidance on the relative performance and costs associated with each of the different models applied to examples of three very different catchment typologies.

Table 8 Global climatic characteristics for the six European regions

	N	M	W	S	SE	NE
Precipitation (mm/y)	300 - 3500	600 - 800	700 - 1400	400 - 1800	500-800	550 - 750
Summer temperature (^{o}C)	15	20	20	25	20	17-18
Winter temperature (^{o}C)	-5	0	5	12	0	-2 – -1
Period of frozen soils (months)	2 - 4	0 - 2	< 1	0	< 1	3

The preliminary opinions (subject to review once results are received) regarding applicability of the models to different environments are presented in Table 9. The categories for climatic conditions use subdivisions into the following regions:
- Northern Europe (No, Swe, F)
- Mid Europe (Ger, Au, Sw, Csz. Rep.);
- West Europe (UK, Ire, Dk, NL, Be, Fr);
- Southern Europe (Sp, It, Gr)
- Eastern and South Eastern Europe (HU, SK, SL, RO, HR, YU, BG, MO)
- North Eastern Europe (PL, ES, LT, LI)

Examples of the main characteristics of these regions are shown below and in Table 8.

The slope for each landscape class is defined as follows:
- Mountainous slope > 10%;
- Hilly 2-10%;
- Plains 0-2%,
- Deltas
- Riperian zone.

Regarding the drainage conditions a subdivision is made between:
- Runoff / overland flow;
- Subsurface drainage;
- Artificial drainage (tile drainage)
- Deep groundwater flow

Agricultural activity:
- Intensive : > 500 kg N/ha/y and/or > 25 kg P/ha/y
- Moderate 200-500 and/or 5-25 kg P/ha/y
- Extensive : < 200 and/or < 5 kg P/ha/y

Soil conditions
- Unstructured Deep soils;
- Unstructured Shallow soils (non permeable layer within 1-2 metre;
- Structured soils (e.g. clay and peat),

Climatic condition: Northern Europe (No, Swe, F); Mid Europe (Ger; Au; Sw, Csz. Rep.); West Europe (UK; Ire; Dk.; NL; Be; Fr); Southern Europe (Sp, It, Gr); Eastern and South Eastern Europe (HU, SK, SL, RO, HR, YU, BG, MO); North Eastern Europe (PL, ES, LT, LI)
Landscape: Mountainous; Hilly; Plains, Deltas, Riperian zones
Flow paths: Runoff; Subsurface drainage; Artificial drainage; Deep Groundwater flow
Agricultural activity: Intensive, Moderate, Extensive
Soil conditions Unstructured Deep soils; Unstructured Shallow soils; Structured soils (e.g. clay and peat)

Table 9 Overview of the tentative suitability/applicability of the quantification tools to apply the tool on different conditions that occur within Europe

	Climatic conditions							Landscape						Flow paths					Agricultural activity				Soil conditions		
	N	W	M	S	SE	NE		M	H	P	D	R		R	SS	AD	DG		I	M	E		UD	US	S
NLCAT - N	+/-	++	++	+/-	+	+/-		+/-	+	++	++	+		+/-	++	++	++		++	++	+		++	+	++
NLCAT - P	+/-	++	++	+/-	+	+/-		+/-	+	++	++	+		+/-	++	++	++		++	++	+		++	+	++
SWAT - N	+/-	++	++	+	+	+/-		+/-	++	+	+/-	+		++	++	++	++		++	++	++		++	+	++
SWAT - P	+/-	++	++	+	+	+/-		+/-	++	+	+/-	+		++	++	++	++		++	++	++		++	+	++
TRK - N	++	++	++	+/-	+/-	++		+/-	++	++	+/-	+/-		+/-	++	++	+		++	++	++		++	++	++
TRK - P	+	+/-	-/+	+/-	+/-	+/-		+/-	+	+	+/-	+/-		+	++	++	+		-	+/-	-		++	++	++
MONERIS - N	+/-	++	++	+	+	+		+	++	++	+	-		++	++	++	+/-		++	+	+		+	+	+
MONERIS - P	+/-	++	+	+	+	+		+	+	+	+	-		+	+	+	+/-		++	+	+		+	+	+
EVENFLOW N	+/-	+	+	+/-	+	+		+	+	+	-	-		+	+	+	+/-		++	++	+/-		+	+	+
N-LES CAT- N	+/-	++	+	+/-	+/-	+		+/-	+	+	+/-	+/-		-	-	-	-		++	++	+		+	+	+/-
NOPOLU - N	+/-	+	+	+	+	+		+/-	+	+	+	+		+	+	+	+		+	+	+/-		+	+	+
NOPOLU - P	+/-	+	+	+	+	+		+/-	+	+	+	+		+	+	+	+		+	+	+/-		+	+	+
REALTA - P	–	++	+/-	+/-	+/-	-		+/-	++	–	+/-	-		++	–	–	–		++	++	++		+	+	+
SA - N	+	+	+	+	+	+		+	+	+	+	+		+	+	+	+		+	+	+		+	+	+
SA - P	+	+	+	+	+	+		+	+	+	+	-		+	+	+	+		+	+	+		+	+	+

++ = very suitable
+ = suitable
+/- = uncertain
– = not suitable/applicable

5 Potential strengths and weaknesses of the quantification tools

In Table 10 an general overview is given of the strengths and weaknesses of the observed quantification tools based on the information submitted by the model owners (because they have the most experience applying the model under different situations).

Table 10. Overview of the strengths and weaknesses of the quantification tools

QT	Strengths	Weaknesses
NL-CAT: ANIMO SWAP SWQN SWQL	- Process oriented approach (predictive/responses) - Integrated catchment model approach - Integration of N and P in one tool - All the major C, N and P processes are described separately - Flexible plot based spatial schematisation - 15 years experience in the Netherlands - Validated at farm level - The ANIMO/SWAP module combination is used to underpin national legislation on manure policy - Processes in surface water (retention and ecological effects) are modelled (SWQN/SWQL)	- Input data requirements very high - Runoff/erosion module still under construction - Frozen soils not modelled - Limited validation on catchment in the Netherlands i.e. high time inputs needed (predictions should be made for catchment and river basin scale) - The model has not been tested outside The Netherlands - High skill required for successful application
REALTA	- The model has been proven to work in Irish grassland catchments. - Data requirements are limited and available for most River Basin Districts. - The model is relatively easy to use and is therefore cost effective.	- The model has not been tested outside Ireland. - In-stream and lake retention is not included. - Additional calibration data will be required for land uses and agricultural practices not found in Ireland. - A limited programme of physical in-stream measurements is required each year.
N-LES CAT	- Empirical model - Hydrological pathways described based on a river hydrograph analysis - 8 years experience in Denmark as a tool for evaluating the effect of policy measures for combating diffuse nitrogen pollution from the agricultural production, including scenario analysis - Validated at the field level - Relatively easy to set up and use	- Moderately high input data requirements concerning agricultural practices - Only valid within the range of the calibration data set. Less dynamic. - For use in other agro-climatic regions than Denmark the model needs a calibration data set from experimental fields which for some countries/regions might be lacking. - Some factors influencing nitrogen leaching – e.g. the effect of an unsuccessful harvest – are not included in the model.

MONERIS	- Integrated catchment modelling with consideration of point and diffuse sources - Enables small and large-scale applications (50-800000 km²). - Data requirements are available for most river basins in Europe - Enables scenario for measures to change point and diffuse pathways. - The modules for pathways and retention can be applied and validated independently - Transparent (4 Diploma thesis outside of the model group, application by regional groups)	- River flow not modelled (input data) - Resolution in time is limited to annual values - Model approach is conceptual, processes are only described by box models - Moderately high requirements for spatially distributed input data - No separate module for plant growth - Spatial resolution is limited: smallest area of 50km^2 means high risk pollution "hot spots" may not be identified
TRK SOILNDB HBV HBN-N	- Integrated catchment modelling - Regional parameterisation, which facilitates application - Resolution is adapted to input data - Applied at national scale (450 000 km^2; 1000 catchments) - Process-based suitable for scenarios with changes in climate and land management. - HBV and HBV-N includes an automatic calibration routine - Validated against independent measurements - Includes well documented and tested models - Enables high resolution in both time and space	- High skill level required for application - Model set-up may be time-consuming - Some internal variables are not validated (involves uncertainties) - P estimation of losses are rather simply described
SWAT	- The model describes in detail the complete N and P cycles and fate in the streams - Continuous in time and capable of simulating long periods for scenario analyses e.g. management or climate changes. - Allows point sources impact to be modelled, - Quite widely used all across the world and in Europe - Computationally efficient to operate on large basins in a reasonable time, - GUI available for ESRI ArcView® (Windows NT/ 2K) and GRASS (Unix) GIS, - Allows a flexible watershed configuration (unlimited Number of Sub-watersheds) - Very co-operative user network	- Forest growth simulation is poor - P simulation somewhat simple - Hydrological Response Units are not georeferenced within a sub-basin - Extensive input data requirements

EveNFlow	- EveNFlow is a conceptual model of moderate complexity - The data required are generally widely available - The model is validated at catchment level - In principle the hydrological component of the model does not require calibration for use in new catchments. - Water balance processes are considered in detail - There is a separate in-stream retention module which allows the direct prediction of stream nitrate concentrations at the gauging station - EveNFlow operates at daily timestep, allowing the frequency of exceedance of nitrate concentrations to be predicted (Nitrates Directive)	- The model has only recently been developed and therefore has not been applied to a large number of catchments. - EveNFlow does not explicitly model the interaction between the root zone and groundwater. - Crop growth is not subject to nutrient limitation and EveNFlow does not model weather related variation in crop yields or soil N cycling processes - At present application is restricted to the original developers
NOPOLU	- The model is easy to use, - Possibility to use statistical data at different levels of resolution - Results presented at different levels (administrative or hydrologic) - Possibility to detail results per Corine Type	- The model is still in development for the transfer of nutrients to the groundwater - Annual statistical approach, so not able to consider subtle changes in management (timing etc)
Source apportionment	- Easy and rapid to apply as a first screening tool for pressure/impact analysis in catchments. - Very transparent, user friendly and low cost Quantification Tool - Approved through inclusion in International Guidelines (cf. OSPAR; HELCOM).	- Load oriented approach so requires river monitoring data. - Edge of channel approach and thus no soil/groundwater processes. - Not suitable for scenarios or land management options. - Depending on good retention estimates in surface waters - Very simple – not capable of identifying high risk areas of agricultural pollution within catchments.

Table 9 illustrates that none of the nutrient loss models reviewed appears suitable for application to all European catchments with their specific soils, land use, climate, and hydrological conditions. The SWAT model (USA) is probably one of the most widely applied models within Europe by different users. Over the last years several workshops have been organised for SWAT-modellers from all over the world. Another model that is applied on more than ten catchments within Europe is MONERIS. The SWAT model is to a large extent a physically-based model, and can be compared with NL-CAT (N and P) and TRK/SOILNDB (N only). The NL-CAT model is even more process based than the SWAT model, since some of the soil processes are lumped within SWAT but are described separately within NL-CAT.

In summary, this review has demonstrated that the EUROHARP quantification tools differ profoundly in their approach to predict the diffuse nutrient losses from agricultural land to surface freshwater systems. This is a reflection of differences in (i) their level of complexity, (ii) their representation of system processes and pathways, and (iii) their resource (data and time) requirements. The quantification tools range from complex, process-based models - which typically have demanding data requirements - to semi-empirical (conceptual) meta-models with some export coefficients, and approaches based on mineral balances and source apportionment (Table 9). These differences between modelling approaches are also a result of the original purpose associated with model development i.e. some models were intended as catchment scale screening tools, some for more detailed policy support work (such as pressure/impact assessments), and some were developed from detailed field scale models with a highly complex representation of soil-plant system processes.

For example, the more complex process-oriented models like ANIMO (as part of NL-CAT), SOILNDB (as part from TRK) and SWAT were originally developed for field or small homogeneous plots in order to assess the impact of nutrient management strategies under different conditions on the nutrients losses to the environment (nutrient accumulation in soils, nitrate concentrations in groundwater and nutrient losses to surface waters). The field-scale model N-LES CAT has a different history, as it was developed by comparing measurements of nitrate losses with local field characteristics (a statistical approach). Later on these models have been implemented in tools for regional/catchment applications and have become "data-driven" modelling tools. However, for large scale applications all these approaches use regional parameterisation which makes them suitable for scenario analyses at catchment scale.

In contrast, other less complex models (e.g. NOPOLU, REALTA) have been primary developed at large catchment scale to identify “high risk” areas within a catchment and estimate the magnitude of nutrient losses from those areas. These models do not attempt to predict changes in soil processes or interactions with the environment, but they do provide an assessment of approximate changes in nutrient losses from agricultural land to surface water in a relatively cheap and fast way. However, such approaches often need hydrological input data from other models or from flow measurements in order to be successfully applied.

Two quantification tools lie between these more complex and less complex approaches: MONERIS and EveNFlow. Both models have lumped representations of certain processes e.g. MONERIS assumes that net mineralisation/immobilisation is negligible. MONERIS has been developed especially for medium and large catchments/basins (the smallest area for application is 50 km^2). Although MONERIS takes into account some soil processes, the model still needs hydrological input data on total flow from another model or from flow measurements within the catchment - the hydrological module is restricted to the separation of the flow between pathways. In contrast, EveNFlow uses a 1 km^2 spatial resolution and includes explicit representations of river flow (and hence concentration) on a daily basis.

The final selection of a particular model for a particular catchment will depend on the question being asked, the data availability, the resource limitations, and the physical characteristics of the catchment in question (which limit the suitability of some models). This review document – the first EUROHARP report - is intended as an initial overview of different model approaches. The report has highlighted the main approaches used in each modelling tool, including the representation of hydrological and plant-soil processes. This in itself has improved the transparency associated with the modelling exercise. Equally importantly, some initial views have been collated concerning the potential strengths and weaknesses of each approach, and the precise temporal and spatial range of each model have been noted.

The next stage of EUROHARP involves the actual assessment of the performance of each model on each of the three core catchments. These results, which will be produced during 2004 and 2005 and which will be presented in a future report, will allow the performance of each model to be compared across the different catchment types, and also permit the ranking of all models on the same catchments. The final output from the project will include a EUROHARP toolbox, summarising all results and implications in a user-friendly interface to assist end-users in the selection of appropriate models for use in particular catchment typologies for a range of policy evaluation and assessment purposes.

6 References

Addiscott, T.M. & Whitmore, A.P. 1991.Simulation of solute leaching in soils of differing permeabilities. Soil Use & Management 7(2), 94-102.

Alvenäs, G. & Marstorp, H. 1993. Effect of ryegrass catch crop on soil inorganic-N content and simulated nitrate leaching. Swed. J. Agric. Res. 23, 3-14.

Andersson, L. & Harding, R.J. (1991): Soil moisture deficit simulations with models of varying complexity for forest and grassland sites in Sweden and the U.K. Water Resources Management 5, 25-46.

Andersson, L. (1988): Hydrological analysis of basin behaviour from soil moisture data. Nordic Hydrology (19):1-18.

Andersson, L. and Arheimer, B. (2003): Modelling of human and climatic impact on nitrogen load in a Swedish river 1885-1994. Hydrobiologia (in press)

Andersson, L. and Arheimer, B., (2001). Consequences of changed wetness on riverine nitrogen - human impact on retention vs. natural climatic variability. Regional Environmental Change 2:93-105.

Anthony, S. G., Quinn, P., and Lord, E. I. 1996. Catchment scale modelling of nitrate. Aspects of Applied Biology 46, 23-32.

Arheimer, B and Brandt, M., 2000. Watershed modelling of non-point nitrogen pollution from arable land to the Swedish coast in 1985 and 1994. Ecological Engineering 14:389-404.

Arheimer, B. (1998) Riverine Nitrogen - analysis and modelling under Nordic conditions. Ph.D. thesis. Kanaltryckeriet, Motala. pp. 200.

Arheimer, B. and Brandt, M., 1998. Modelling nitrogen transport and retention in the catchments of Southern Sweden. Ambio 27(6):471-480.

Arheimer, B. and Wittgren, H. B., (1994). Modelling the effects of wetlands on regional nitrogen transport. Ambio 23(6):378-386

Arheimer, B. and Wittgren, H.B., (2002). Modelling Nitrogen Retention in Potential Wetlands at the Catchment Scale. Ecological Engineering 19(1):63-80.

Arheimer, B., Torstensson, G. and Wittgren, H.B (2003): Landscape planning to reduce coastal eutrophication: Constructed Wetlands and Agricultural Practices. Landscape and Urban Planning (in press)

Arnold, J.G., Allen, P.M. & Bernhardt, G. 1993: A comprehensive surface-groundwater flow model. Journal of Hydrology, 142, 47-69.

Arnold, J.G., Williams, J.R., Nicks, A.D. & Sammons, N.B. 1990: SWRRB - A Basin Scale Simulation Model for Soil and Water Resources Management. Texas A&M University Press, College Station.

Aronsson, H. & Torstensson, G. 1998. Measured and simulated availability and leaching of nitrogen associated with frequent use of catch crops. Soil Use and Management, 14, 6-13.

Bailey, R. J. & Spackman, E. (1996) A model for estimating soil moisture changes as an aid to irrigation scheduling and crop water-use studies: I. Operational details and description. Soil Use and Management 12, 122-128 (1996).

Beasley, D.B., Huggins, L.F., & Monke, E.J. 1980: ANSWERS: A model for watershed planning. Transactions of the ASAE. 23(4), 938-944.

Behrendt, H., Dannowski, R., Deumlich, D., Dolezal, F., Kajewski, Kornmilch, M., Korol, R., Mioduszewski, W., Opitz, D., Steidl, J. & Stronska, M. (2002): Nutrient and heavy metal emissions into the river system of Odra - results and comparison of models. Schriftenreihe des Institutes für Abfallwirt-schaft und Altlasten, Technische Universität Dres-den, Bd. 28, Vol.2, 213-221.

Behrendt, H., Huber, P., Kornmilch,M., Opitz, D., Schmoll, O., Scholz, G. & Uebe, R. (2002): Estimation of the nutrient inputs into river basins - experiences from German rivers. Regional Environemental Changes, Spec. Issue, (in print; online published).

Behrendt, H., Huber, P., Ley, M., Opitz, D., Schmoll, O., Scholz, G. & Uebe, R. (1999): Nährstoff-bilanzieru-ng -der Flußgebiete Deutsch-lands. UBA-texte, 75/99, 288 S.

Berendt, H., 1999. Comparison of different methods of source apportionment of nutrients to river basins-. Wat. Sci. Tech., 39,12, 179-187.

Bergström, G. and Graham, L.P., 1998. One the scale problem in hydrological modelling. Journal of Hydrology (211):253-265.

Bergström, L. & Jarvis, N. 1991. Prediction of nitrate leaching losses from arable land under different fertilisation intensities using the SOIL-SOILN models. Soil Use and Management, 7, 79-85.

Bergström, L. & Johnsson, H. 1988. Simulated nitrogen dynamics and nitrate leaching in a perennial grass ley. Plant & Soil, 105, 273-281.

Bergström, L., Johnsson, H. & Torstensson, G. 1991. Simulation of soil nitrogen dynamics and losses using the SOILN model. Fert. Res. 27, 181-188.

Bergström, S. (1976): Development and application of a conceptual runoff model for Scandinavian catchments. Ph.D. Thesis. SMHI Reports RHO No. 7, Norrköping.

Bergström, S. and Sandberg, G. (1983): Simulation of groundwater response by conceptual models - Three case studies. Nordic Hydrol. 14, 71-84.

Bergström, S., 1995. The HBV model. In Singh, V. P. (ed.) Computer Models of Watershed Hydrology, Water Resources Publications, Littleton, Colorado, pp. 443-476.

Bergström, S., Carlsson, B., Gardelin, M., Lindström, G., Pettersson, A. & Rummukainen, M. (2001) Climate change impacts on runoff in Sweden - assessments by global climate models, dynamical downscaling and hydrological modelling. Climate Research, Vol 16, No 2, 101-112.

Beven K, Lamb R, Quinn P, Romanowicz R, Freer J. 1994. TOPMODEL. In Computer Models of Watershed Hydrology. Singh V, (ed). Water Resource Publications; 1-43.

Blombäck, K. & Eckersten, H. 1997. Simulated growth and nitrogen dynamics of a perennial rye grass. Agricultural and Forest Meteorology 88:37-45.

Blombäck, K., Stähli, M. & Eckersten, H. 1995. Simulation of water and nitrogen flows and plant growth for a winter wheat stand in central Germany. Ecological Modelling, 81, 157-167.

Boorman, D., Hollis, J. and Lilly, A. 1995 Hydrology of soil types: a hydrologically based classification of the soils of the United Kingdom. Institute of Hydrology Report No. 126, Wallingford, Oxfordshire.

Boorman, D., Hollis, J. and Lilly, A. 1995 Hydrology of soil types: a hydrologically based classification of the soils of the United Kingdom. Institute of Hydrology Report No. 126, Wallingford, Oxfordshire.

Borg, G.C., Jansson, P.-E. & Lindén, B. 1990. Simulated and measured nitrogen conditions in a manured and fertilised soil. Plant & Soil, 121, 251-267.

Borgvang, S-A. & Selvik, J.S., eds. Development of HARP Guidelines - Harmonised Quantification and Reporting Procedure for Nutrients. SFT Report 1759/2000. ISBN 82-7655-401-6. 179 pp.

Brandt, M. & Bergström, S. (1994): Integration of field data into operational snowmelt-runoff models. Nordic Hydrol. 25, 101-112.

Brandt, M. and Ejhed, H. (2002): TRK-Transport, Retention, Källfördelning. Belastning på havet. Swedish Environmental Protection Agency, Report No. 5247.

Brandt, M., Bergström, S., Gardelin, M. (1988) Modelling the effects of clearcutting on runoff - Examples from Central Sweden. Ambio, 17, 5: 307-313.

Breeuwsma, A. and O.F. Schoumans, 1987. Forecasting the impact of environmental legislation on leaching of phosphate and nitrate from agricultural soils. In: Aspects of the impact of agriculture on the environment. Proc. EC-Workshop Copenhagen, October 1986.

Breeuwsma, A., and Schoumans, O.F. 1987. Forecasting phosphate leaching from soils on a regional scale. In: W. van Duijvenbooden et al. (ed.), Vulnerability of soil and groundwater to pollutants; TNO Comm. on Hydrological Res. Proc. and Information no. 38, The Hague, The Netherlands, pp. 973-981.

Bruen, M. (1999, ed.) Forecasting floods in urban areas downstream of steep catchments, Final report from the TELFLOOD project, Centre for water resources research, University College Dublin, Dept of Civil Engineering.

Campling, P., S. van de Walle, J. van Orshoven, Feyen, J., 2001. Final Report. Calculation of Agricultural Nitrogen Quantity for EU River Basins. Spatial Application Division K.U. LEUVEN Research & Development. Version 20/12/2001

Caraco, N.F. & Cole, J.J. 1999. Human impact on nitrate export. an analysis using major world rivers. Ambio 28: 167-170.

Chambers, B. J., Lord, E. I., Nicholson, F. A. and Smith, K. A. 1999. Predicting nitrogen availability and loses following applications of manures to arable land: MANNER. Soil Use and Management, 15, 137-143.

Chardon, W.J. & O.F. Schoumans, Review of concepts and process description on chemical mechanisms. Meeting WG2, COST Action 832, Quantifying the agricultural contribution to eutrophication. Cordoba, Spain, 13/14/15-5-1999.

De Vries, W. 1994. Soil response to acid deposition at different regional scales. Ph.D. Thesis. DLO Winand Staring Centre for Integrated Land, Soil and Water Research, Wageningen, The Netherlands

De Willigen, P. 1991. Nitrogen turnover in the soil-crop system; comparison of fourteen simulation models. Fert. Res. 27:141-149.

De Witt, M. J. M. (2001) Nutrient fluxes at the river basin scale. I: the PolFlow model. Hydrological Processes, 15, 743-759.

Department of the environment and local government (1997). Managing Ireland's Rivers and Lakes: A catchment-based Strategy against Eutrophication.

Diekkruger, B., Söndgrath, D., Kersebaum, K.C. and McVoy, C.W. 1995. Validity of agroecosystems models. A comparison of results of different models applied to the same dataset. Ecological modelling 81:3-29.

Dümenil, L. and Todini, E., 1992. A rainfall-runoff scheme for use in the Hamburg climate model. Advances in theoretical hydrology. J. P. o´Kane, (Editor), Elsevier, 129-157.

Eckersten, H. & Jansson, P.-E. 1991. Modelling water flow, nitrogen uptake and production for wheat. Fert. Res. 27, 313-329.

Edwards, A.C. and Withers, P.J.A. (1998) Soil phosphorus management and water quality: a UK perspective. Soil Use and Management, 14, 124-130.

Eklund, A., Gardelin, M. & Lindroth, A. (2000): Vinteravdunstning i HBV-modellen - jämförelse med mätdata (Winter evaporation in the HBV model - comparison with observations) (in Swedish). SMHI Reports Hydrology No. 83, Norrköping.

European Environment Agency (1995) Europe's Environment - The Dobris Assessment. D. Stanners & P. Bourdeau (Eds.), Copenhagen.

European Environment Agency/IFEN (2000). Technical report N°51. Calculation of nutriment surplus from agricultural sources. Statistics spatialisation by means of CORINE Land Cover. Application to the case of Nitrogen.

Fogelberg, S. (2003): Comparison of nitrogen retention modelling using the HBV-N and the MONERIS models. Master thesis, Uppsala Technical University, Report (in press).

Friend, A.D. 1998. Parameterisation of a global daily weather generator for terrestrial ecosystem modelling. Ecological modelling, 109, 121-140.

Gardelin, M., Bergström, S., Carlsson, B., Graham, L.P. & Lindström, G. (2001) Climate change and water resources in Sweden - Analysis of uncertainties. In: M.Beniston Ed.), Climatic Change: Implications for the Hydrological Cycle and for Water Management. Advances in Global Change Research. Kluwer Academic Publishers, Dordrecht

Gleeson, T. (1992). Runoff Risk Categories for Soils. Teagasc.

Granlund, K., Rekolainen, S., Grönroos, J., Nikander, A. & Laine, Y. 2000. Estimation of the impacts of fertilisation rates on nitrate leaching in Finland using a mathematical simulation model. Agric. Ecosystems & Environment 80:1-13.

Grant, R., Laubel, A., Kronvang, B., Andersen, H.E., Svendsen, L.M. and Fuglsang, A. (1996) Loss of dissolved and particulate phosphorus forms in drainage water from four arable catchments on structured soils in Denmark. Water Research, Vol. 30, No. 11, 2633-2642.

Grimvall, A. and Stålnacke, P. 1996. Statistical methods for source apportionment of riverine loads of pollutants. Envirometrics 7: 201-213.

Groenendijk, P. and J.G. Kroes, 1999. Modelling the nitrogen and phosphorus leaching to groundwater and surface water with ANIMO 3.5. Report 144. Winand Staring Centre, Wageningen

Gustafson, A. 1988. Simulation of nitrate leaching from arable land in southern Sweden. Acta Agric. Scand. 38, 13-23.

Harlin, J. & Kung, C.S. (1992) Parameter uncertainty and simulation of design floods in Sweden. Journal of Hydrology, Vol. 137, pp. 209-230.

Harlin, J. (1992) Modelling the hydrological response of extreme floods in Sweden. Nordic Hydrology, Vol. 23, pp. 227-244.

Haygarth, P.M., Chapman, P.J., Jarvis, S.C. and Smith, R.V. (1998) Phosphorus budgets for two contrasting grassland farming systems in UK. Soil Use and Management, 14, 160-167.

Hoffmann, M. & Johnsson, H. 1999. A method for assessing generalised nitrogen leaching estimates for agricultural land. Environmental Modelling and Assessment, 4:35-44.

Hoffmann, M. & Johnsson, H. 2000. Nitrogen leaching from agricultural land in Sweden. Model calculated effects of measures to reduce leaching loads. Ambio 29:67-73.

Hoffmann, M. & Johnsson, H., 2003. Test of a modelling system for estimating nitrogen leaching - A pilot study in a small agricultural catchment. Environmental Modelling & assessment 8:15-23.

Hoffmann, M., Johnsson, H., Gustafson, A. and Grimvall, A. 2000. Leaching of nitrogen in Swedish agriculture - a historical perspective. Agric. Ecosystems & Environ. 80:277-290.

Hough, M., Palmer, S., Weir, A., Lee, M and Barrie, I. (1996). The Meteorological Office rainfall and evaporation calculation systems: MORECS Version 2.0 (1995). An update to hydrological memorandum 45.

Iritz, L., Johansson, B. & Lundin, L. (1994) Impacts of forest drainage on floods. Hydrological Sciences Journal, Vol. 39, No. 6, 637-659.

Iversen, T.M., Kjeldsen, K., Kristensen, P., Haan, B., Oirschot, M., Parr, W. and Lack, T. (1997) Integrated Environmental Assessment on Eutrophication. Technical Report No. 207, Ministry of Environment, National Environmental Research Institute, Silkeborg, Denmark, 101 p.

Jansson, P.-E. & Andersson, R. 1988. Simulation of runoff and nitrate leaching from an agricultural district in Sweden. J. Hydrol. 99, 33-47.

Jansson, P.-E. Singh Antil, R. & Borg, G.Ch. 1989. Simulation of nitrate leaching from arable soils treated with manure. In Nitrogen in Organic Wastes Applied to Soils (ed. J.A. Hansen & K. Henriksen), 150-166. International Solid Waste Professional Library, London: Academic Press.

Jansson, P.-E., 1991. Simulation model for soil water and heat conditions. Description of the SOIL model. Report 165, Department of Soil Sciences, Division of Biogeophysics, SLU, P.O. Box 7014, SE-75007, Uppsala, Sweden. 72 pp.

Jansson, P.-E., Borg, G.C., Lundin, L.-C. & Lindén, B. 1987. Simulation of Soil Nitrogen Storage and Leaching. Applications to Different Swedish Agricultural Systems. The Swedish National Environmental Protection Board, Solna. Report 3356. 63 pp.

Jansson, P.-E., Halldin, S., 1979. Model for annual water and energy flow in a layered soil. In: Halldin, S. (Ed.), Comparison of Forest Water and Energy Exchange Models, International Society for Ecological Modelling, Copenhagen. pp. 145-163.

Jeppesen, E. (1998) The Ecology of Shallow Lakes - Trophic interactions in the pelagial. Doctor´s dissertation (DSc) NERI Technical Report No. 247, Silkeborg, Denmark, 358 p.

Johansson, B. & Seuna, P. (1994) Modelling the effects of wetland drainage on high flows. Aqua Fennica, Vol. 24, No. 1, 59-67.

Johansson, B. (2002) Estimation of areal precipitation for hydrological modelling in Sweden. Ph.D.thesis. Earth Science Centre, Göteborg University, Report A76 2002.

Johnsson, H. & Hoffman, M. 1998. Nitrogen leaching from agricultural land in Sweden - Standard rates and Gross Loads in 1985 and 1994. Ambio 27: 481-488.

Johnsson, H. & Mårtensson, K. 2002. Kväveläckage från svensk åkermark. Beräkningar av normalutlakning för 1995 och 1999. Naturvårdsverket rapport nr 5248. 89 pp.

Johnsson, H. 1991. Simulation of nitrogen losses using the SOILN model. NPO-forskning fra Miljöstyrelsen, A20. Ministry of Environment, National Agency of Environmental Protection, Copenhagen.

Johnsson, H., Bergström, L., Jansson, P.-E. & Paustian, K. 1987. Simulated nitrogen dynamics and losses in a layered agricultural soil. Agric. Ecosystems Environ. 18, 333-356.

Johnsson, H., Hoffmann, M., 1996. Nitrate leaching simulations. In Rekolainen, S., Leek, R. (Eds.), Regionalisation of Erosion and Nitrogen Losses from Agricultural Land in Nordic Countries. Tema Nord 1996:615, Copenhagen, Denmark, pp. 18-24.

Johnsson, H., Klemedtsson, L., Nilsson, Å. & Svensson, B. H. 1991. Simulation of field scale denitrification losses from soils under grass ley and barley. Plant and Soil, 138, 281-302.

Johnsson, H., Larsson, M., Mårtensson, K. & Hoffmann, M. SOILNDB: a decision support tool for assessing nitrogen leaching losses from arable land. Environmental Modelling & software 17: 505-517.

Johnsson, H., Larsson, M., Mårtensson, K., Hoffmann, M. 2002. SOILNDB: A decision support tool for assessing nitrogen leaching losses from arable land. Environmental Modelling & Software 17:505-517.

Kallio, K., Rekolainen, S., Ekholm, P., Granlund, K., Laine, Y., Johnsson, H. and Hoffmann, M. 1997. Impacts of climatic change on agricultural nutrient losses in Finland. Boreal Environ. Res. 2: 35-52.

Kätterer, T., Eckersten, H., Heidmann, T., Borg, G., Sippola, J., Breland, T.A., Vold, A., Haugen, L.E., Thomsen, A., Blombäck, K. & Bakken, L. 1999. Application of the nitrogen model to agricultural land. In Jansson, P-E., Persson, T. & Kätterer, T. (eds.) Nitrogen Processes in Arable and Forest Soils in the Nordic Countries. Field-scale modelling and experiments. TemaNord 1999:560. Nordic Council of Ministers, Copenhagen.

Kirkby, M. J. (1976) Hydrological slope models: the influence of climate. In: Derbyshire, E (Ed.) *Geomorpholgy and Climate*, Wiley, London, 247-267.

Kirk McClure Morton (1999). Low Level Trial - Implementation of Draft OSPAR Guidelines for Harmonised Quantification & Reporting Procedures for Nutrients" - September 1999 - Contribution by Ireland.

Kirk McClure Morton (2001). The Lough Derg and Lough Ree Catchment Monitoring and Management System. Final Report.

Kristensen, P. and Hansen, O.H. (1994) European Rivers and Lakes - Assessment of their environmental state. European Environment Agency, EEA Environmental Monographs 1, Copenhagen, 122 p.

Kroes, J.G. and J. Roelsma, 1997. ANIMO 3.5. User's guide for the ANIMO version 3.5 nutrient leaching. model Technical Document 46. Winand Staring Centre, Wageningen.

Kronvang, B., Græsbøll, P., Larsen, S.E., Svendsen, L.M. and Andersen, H.E. (1996) Diffuse nutrient losses in Denmark. Wat. Sci. Tech., 33, 81-88.

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

Kronvang, B., Hoffmann, C.C., Svendsen, L.M., Windolf, J., Jensen, J.P. and Dørge, J. (1999) Retention of nutrients in river basins. Aquatic Ecology.

Kronvang, B., Laubel, A. and Grant, R. (1997) Suspended sediment and particulate phosphorus transport and delivery pathways in an arable catchment, Gelbæk stream, Denmark. Hydrological Processes, 11, 627-642.

Kronvang, B., Svendsen, L.M., Jensen, J.P. and Dørge, J. (1999) Scenario analysis of nutrient management at the river basin scale. Hydrobiologia.

Krysanova, F., Wechsung, A. Becker, W., Poschenrieder and Gräfe, J. (1999) Mesoscale ecohydrological modelling to analyse regional effects of climate change, Environmental Modelling and Assessment (in press).

Larocque, M. & Banton, O. 1994. Determining Parameter Precision for Modelling Nitrate Leaching: Inorganic Fertilisation in Nordic Climates. Soil Sci. Soc. Am. J. 58, 396-400.

Larsson, M. & Johnsson, 2003. Simulation of nitrate leaching using a modelling system with automatic parameterisation routines. Soil, Use and Management (In press).

Leonard, R.A., W.G. Knisel, Still, D.A. 1987. GLEAMS: Groundwater loading effects on agricultural management systems. Trans. ASAE 30(5): 1403-1428.

Lewan, E. 1993. Evaporation and discharge from arable land with cropped or bare soils during winter. Measurements and simulations. Agricultural and Forest Meteorology, 64, 131-159.

Lewan, E. 1994. Effects of a catch crop on leaching of nitrogen from a sandy soil: Simulations and measurements. Plant and Soil, 166, 137-152.

Lidén, R. and Harlin, J. (2000) Analysis of conceptual rainfall-runoff modelling performance in different climates, Journal of Hydrology, Vol. 238, pp. 231-240.

Lidén, R., Vasilyev, A., Loigu, E., Stålnacke, P., Grimvall, A. and Wittgren, H. B., (1999). Nitrogen source apportionment - a comparison between a dynamic and a statistical model. Ecological Modelling 114:235-250.

Liere, L. van, J. Janse, M. Jeuken, P. van Puijenbroek, O.[F.] Schoumans, R.[F.A.] Hendriks, J. Roelsma & D. Jonkers: Effect of nutrient loading on surface waters in polder Bergambacht, the Netherlands. In: J.[H.A.M.] Steenvoorden, F. Claessen & J. Willems (eds.), Agricultural effects on ground and surface waters: research at the edge of science and society. Wallingford (UK), IAHS, 2002. IAHS Publ. 273, pp. 213-218.

Lindström, G. & Harlin, J. (1992) Spillway design floods in Sweden. II: Applications and sensitivity analysis. Hydrological Sciences Journal, Vol. 37, No. 5, pp. 521-539.

Lindström, G. & Rodhe, A. (1986): Modelling water exchange and transit times in till basins using oxygen-18. Nordic Hydrol. 17, 325-334.

Lindström, G. (2000): HBV model simulations of oxygen-18 flow in small forested basins in Sweden. Proceedings XXI Nordic Hydrological Conference in Uppsala, 26-30 June 2000, NHP-Report No. 46, 374-379.

Lindström, G., Bishop, K. & Ottosson-Löfvenius, M. (2002) Soil frost and runoff at Svartberget, northern Sweden - measurements and model analysis. Hydrological Processes, Vol. 16, No. 17, 3379-3392.

Lindström, G., Johansson, B., Persson, M., Gardelin, M. & Bergström, S. (1997): Development and test of the distributed HBV-96 model. J. Hydrol. 201, 272-288.

Lord, E. I. 1992. Modelling of nitrate leaching: Nitrate Sensitive Areas. Aspects of Applied Biology 30, 19-28.

Lord, E. I. and Anthony, S. G., 2000. MAGPIE: A modelling framework for evaluating nitrate losses at national and catchment scales. Soil Use and Management, 16: 167-174.

Magette WL (1998). Factors affecting losses of nutrients from agricultural systems and delivery to water resources. Draft Guidelines for Nutrient Use in Intensive Agricultural Enterprises, Teagasc.

McGechan, M.B. & Wu, L. 2001. A review of carbon and nitrogen processes in European soil nitrogen dynamic models. In: Shaffer, M.J., Ma, L. & Hansen, S. (eds.) Modelling carbon and nitrogen dynamics for soil management. Lewis publishers (CRC Press LLC), Boca Raton/London/New York/Washington, pp 103-171.

Moog, O. (1984) Stream phosphorus exports from prealpine watersheds with respects to geology and land use. Verh. Internat. Verein. Limnol. 22, 1071-1076.

Morton, K. McC (1999). Low Level Trial - Implementation of Draft OSPAR Guidelines for Harmonised Quantification & Reporting Procedures for Nutrients" - September 1999 - Contribution by Ireland.

Morton, K. McC (2001). The Lough Derg and Lough Ree Catchment Monitoring and Management System. Final Report.
Naturvårdsverket (1997) Nitrogen from land to sea. Swedish Environmental Protection Agency report no 4801.
Neill, M. (1989) Nitrate concentrations in river waters in the southeast of Ireland and their relationship with agricultural practices. Wat. Res. 23(11), 1339-1355.
Neitsch S.L., Arnold J.G., Kiniry J.R., Williams J.R., (2001), Soil and Water Assessment Tool - Theoretical Documentation - Version 2000, Blackland Research Center - Agricultural Research Service, Texas - USA
Pettersson, A., Arheimer, B. and Johansson, B., (2001). Nitrogen concentrations simulated with HBV-N: new response function and calibration strategy. Nordic Hydrology 32(3):227-248.
Pionke, H.B., Gburek, W.J., Sharpley, A.N. and Schnabel, R.R. (1996) Flow and nutrient export patterns from agricultural hill-land watershed. Water Resources Research, 32, 1795-1804.
Ragab, R., Cooper, D.M., Harrris, D.M., Catt, J.A. 1996. Simulating nitrate leaching under winter wheat grown on a structured clay soil considering bypass flow. J. Hydrology 182:157-173.
Rekolainen, S., Ekholm, P., Ulén, B. and Gustafson, A. (1997) Phosphorus losses from agriculture to surface waters in the Nordic countries. In: Tunney, H., Carton, O.T., Brookes, P.C. and Johnston, A.E. (Eds) Phosphorus loss from soil to water, CAB International, Wallingford, Oxon, UK, 77-94.
Rekolainen, S., Grönros, J., Bärnlund, I., Nikander, A., and Laine, Y. (1998) Modelling the impact of measurement practices on agricultural phosphorus losses to surface waters of Finland. Proc. 3th International Conference on Diffuse Pollution. Edinburgh, Scotland, 31 August - 4 September 1998, SEPA.
Rijtema, P.E. & J.G. Kroes, 1991: Some results of nitrogen simulations with the model ANIMO. Fertilizer Research 27: 189-198
Rijtema, P.E., P. Groenendijk and J.G. Kroes, 1999. Environmental impact of land use in rural regions. The development, validation and application of model tools for management and policy analysis. Series on environmental science and management, vol. 1. Imperial College Press, London.
Sandén, P. & Warfvinge, P. (Eds.) (1992): Modelling groundwater response to acidification. SMHI Reports RH No. 5, Norrköping.
Scholefield, D., Lockyer, D.R., Tyson, K.C. & Whitehead, D.C. 1991. A model to predict transformations and losses of nitrogen in UK pastures grazed by beef cattle. Plant & Soil 132, 165-177.
Schoumans, O.F. & P. Groenendijk, 2000. Modelling soil phosphorus levels and phosphorus leaching from agricultural land in the Netherlands. J. Environ. Qual. 29 (2000), 1: 111-116.
Schoumans, O.F. 1995. Beschrijving en validatie van de procesformulering van de abiotische fosfaatreacties in kalkloze zandgronden. Staring Centrum Wageningen, Rapport no. 381.
Schoumans, O.F. and P. Groenendijk, 2000. Modelling Soil Phosphorus Levels and Phosphorus Leaching from Agricultural Land in the Netherlands. J. Environ. Qual. 29:111-116.
Schoumans, O.F., 1999. Beschrijving van het gedrag van anorganisch fosfaat in veengronden. Alterra Rapport 522.
Schoumans, O.F., A. Breeuwsma and W. de Vries (1987). Use of soil survey information for assessing the phosphate sorption capacity of heavily manured soils. Proc. Int. Conf. on the Vulnerability of Soil and Groundwater to Pollutants. Noordwijk aan Zee, 1987, The Netherlands: 1079-1088
Schoumans, O.F., B.A. Marsman and A. Breeuwsma (1989). Assessment of representative soil data for phosphate leaching. Proc. Int. Symposium Land qualities in space and time. Eds. J. Bouma and A.K. Bregt, Pudoc, Wageningen, 201-205
Schoumans, O.F., J. Roelsma, H.P., Oosterom, P. Groenendijk, H. Van Zeijts, G.J. van der Born, S. van Tol, H.F.M. van den Berg, H.G. van der Meer, F.K. Ehlert, 2002.

Nutriëntenemissies vanuit landbouwgronden naar het grondwater en oppervlaktewater bij varianten van verliesnormen. Modelberekeningen met STONE 2.0. Clusterraopport 4: deel 1. Alterra, Wageningen.

Schoumans, O.F., J.Mol-Dijkstra, L.M.W. Akkermans and C.W.J. Roest, 2002. SIMPLE: Assessment of non-point phosphorus pollution from agricultural land to surface waters by means of a new methodology. Water Science & Technology. Vol. 45 No9 pp 177-182.

Seibert, J. (1997) Estimation of parameter uncertainty in the HBV model. Nordic Hydrology, Vol. 28, No. 4/5. pp 247-262.

Shaw, E.M. (1988) Hydrology in practice, second edition. Chapman and Hall, London, U.K.

Silgram, M. (1997) Hydrological controls on nitrate leaching: an integrated monitoring and modelling study. PhD thesis, University of London. 357pp.

Silgram, M. and Shepherd, M. (1999) The effects of cultivation on soil nitrogen mineralization. Advances in Agronomy 65, 267-311.

Silgram, M., Waring, R., Anthony, S. and Webb, J. 2001. Intercomparison of national and IPCC methods for estimating N loss from agricultural land. *Nutrient Cycling in Agroecosystems* 60 (1-3), 189-195.

Simmelsgaard, S. E., Kristensen, K., Andersen, H. E., Grant, R., Jørgensen, J. O. and Østergaard, H. S. (2000): An empirical model for calculation of root zone nitrate leaching. DJF rapport Markbrug no. 32, Danmarks JordbrugsForskning, 67 pages (in Danish)

Swedish EPA, 1997. Nitrogen from land to sea. Main report. Swedish Environmental Protection Agency, Report 4801, Nordstedts tryckeri AB, Stockholm.

Teagasc (1998). Nutrient Advice for Phosphorus and Potassium Fertiliser. Teagasc Research Centre. Johnstown Castle, Wexford.

Thirup C. (2000) Simulation of nitrate leaching on the isle of Taasinge. Report made for the County of Funen, Denmark by WaterTech (in Danish)

Todini, E., 1995. The ARNO rainfall-runoff model. J. Hydrol., 175:339-382.

Torstensson, G. & Aronsson, H. 2000. Nitrogen leaching and crop availability in manured catch crop systems. Nutrient Cycling in Agroecosystems 56:139-152.

Torstensson, G. & Johnsson, H. 1996. Simulation of Water and Nitrogen dynamics in a Five Year Leaching Experiment with Varying Fertilisation and Manure Treatments. Reports and Dissertations 29. Dep. of Soil Sciences, Div. of Water Quality Management, Swedish University of Agricultural Sciences. Box 7072, S-750 07 Uppsala, Sweden.

Turpin, O., Ferguson, R. & Johansson, B. (1999): Use of remote sensing to test and update simulated snow cover in hydrological models. Hydrol. Processes 13, 2067-2077.

Tychon, B., Vander Borght, P. & De Backer, L.W. 1998. Water and Nitrogen Transfer Study Through Soils of a Small Agricultural Water Catchment. In Proceedings of 3rd International Conference on Diffuse Pollution. Edinburgh, UK

Ulén, B 1998. Simulation of nitrate leaching before and after conversion to ecological farming. Biological Agriculture & Horticulture, 17:59-75.

Ulén, B, Johansson G. and Kyllmar, K., 2001, Model predictions and long-term trends in phosphorus transport from arable lands in Sweden. Agricultural Water Management 49, 197-210.

Vereecken, H., E.J. Jansen, M.J.D. Hack-ten Broecke, M. Swerts, R. Engelke, S. Fabrewitz & S. Hansen, 1991: Comparison of simulation results of five nitrogen models using different datasets. In: Soil and Groundwater Research Report II, Nitrate in Soils, Final report of contracts EV4V-0098- NL and EV4V-00107-C, Commission of the European Communities.

Vighi, M., Soprani, S., Puzzarini, P. and Menghi, G. (1991) Phosphorus loads from selected watersheds in the drainage area of the Northern Adriatic Sea. J. Environ. Qual. 20, 439-444.

Vries, W. de, O.F. Schoumans, J.F. Kragt and A. Breeuwsma, (1989). Use of models and soil survey information in assessing the regional water quality. In: Groundwater contamination: use of models in decision-making. Eds. G. Jousma et al., Kluwer Academic Publishers, Dordrecht, The Netherlands.

Withers, P.J.A. and Jarvis, S.C. (1998) Mitigation options for diffuse phosphorus loss to water. Soil Use and Management, 14, 186-192.

Witt, de, M. J. M. (2001) Nutrient fluxes at the river basin scale. I: the PolFlow model. Hydrological Processes, 15, 743-759.

Wittgren, H. B., Gippert, L., Jonasson, L., Pettersson, A., Thunvik, R., and Torstensson, G. (2001). An actor game on implementation of environmental quality standards for nitrogen. In: Steenvoorden, J., Claessen, F. and Willems, J. (Eds) Agricultural Effects on Ground and Surface Waters. IAHS Publ. no. 273.

WMO (1986) Intercomparison of models of snowmelt runoff. Operational Hydrology Report No. 23, WMO-No. 646, WMO, Geneva, Switzerland.

WMO (1992) Simulated Real-time Intercomparison of Hydrological Models. WMO Operational Hydrological Report No. 38, Geneva, Switzerland.

Wright, G.G., Edwards, A.C. Morrice, J.G. and Pugh, K. (1991) North east Scotland river catchment nitrate loading in relation to agricultural intensity. Chemistry and Ecology, 5, 263-281.

Wu, L., and McGechan, M.B. 1998. A review of carbon and nitrogen processes in four soil nitrogen models. J. Agric. Eng. Res. 69:279-305.

Wu, L., McGechan, M.B., Lewis, D.R., Hooda, P.S. & Vinten, A.J.A. 1998. Parameter selection and testing the soil nitrogen dynamics model SOILN. Soil Use and Management, 14, 170-181.

Zhang, X. and Lindström, G. (1996) A comparative study of a Swedish and a Chinese hydrological model. Water Resources Bulletin, October 1996, Volume 32, Number 5, pp. 985-994.

ANNEX A Hydrological pathways

I Frost and Snow

NL-CAT (ANIMO)	Simple snow balance simulated; Compaction not considered; Snow melt occurs when air temperature exceeds a temperature threshold; Soil temperature simulated; Frozen soil: soil water flow set to zero below a temperature threshold;
REALTA	-
N-LES CAT	The hydrological submodule treats precipitation as snow when air temperature is below zero. Snow accumulates or melts from a snow reservoir depending on air temperature. Soil temperature and frost effect on water flow in the soil column is not considered.
MONERIS	Only permanently frozen soils
TRK	Nitrogen concentrations in the root zone (SOILNDB) are affected by freezing-thawing, computed by a coupled heat/water flow equation. In the water discharge model (HBV) the snow routine is a degree-day approach. Soil frost can be computed from air temperature and snow conditions.
SWAT	Snowmelt is described, impact of frozen soils on soil waterflow has not been considered.
EVENFLOW	The original version of EVENFLOW does not include a snowmelt function. A new function is being developed to support work in this project.
NOPOLU	-
SOURCE APP.	-

The process oriented models ANIMO and TRK take frost and snow into account. The Swedish model TRK describes the phenomenon in the greatest detail. SWAT comprises a description of snowmelt. MONERIS takes the effect of permanent frozen soils on nutrient losses to surface water systems into account by means of an empirical parameter which should be derived by calibration.

II Point sources and water transfer

NL-CAT (ANIMO)	Discretisation of the area includes the description of main water courses; Water and dissolved nutrient transport through surface water system is considered explicitly; Point sources are considered explicitly in surface water quality module
REALTA	-
N-LES CAT	-
MONERIS	-
TRK	In the water discharge model (HBV), catchments are divided into coupled subbasins. Point sources and water discharge are routed between subbasins.
SWAT	Distributed model comprising the watersheds. The water transfer between the watersheds and the point sources are described explicitly.
EVENFLOW NOPOLU	Point sources are explicitly included in the river routing module.
SOURCE APP.	-

ANNEX A Hydrological pathways (Cont.)

III Canopy interception

NL-CAT (ANIMO)	A fraction of gross precipitation by the Von Hoyningen-Hüne and Braden relation as a function of Leaf Area Index and one crop dependent parameter
REALTA	-
N-LES CAT	Hydrological submodule incorporates interception storage. Evaporation from the interception storage is assumed to take place at a potential rate.
MONERIS	-
TRK	A simple interception storage can be used in HBV. The evaporation from the interception storage is assumed to take place at a potential rate.
SWAT	Canopy storage can be taken into account. A parameter should be defined for the maximum storage of the reservoir.
EVENFLOW	Canopy interception is calculated as a function of Leaf Area Index and evaporation from the canopy is calculated using Penman-Monteith assuming a zero resistance.
NOPOLU	-
SOURCE APP.	-

IV Evapotranspiration

NL-CAT (ANIMO)	Distinction is made between soil evaporation and plant evaporation. Partition between potential soil evaporation and potential plant evaporation is made on the basis of Leaf Area Index and a crop dependent parameter. Potential soil evaporation is reduced for dry conditions on the basis of the empirical evaporation functions of Black (1969) or Boesten and Stroosnijder (1986). Potential plant evaporation is reduced on the basis of the soil moisture suction profile in the root-zone.
REALTA	-
N-LES CAT	Reference evapotranspiration (PET) is an input to the hydrological submodule. It can be calculated by e.g. the Penman or the Makkink equation. Actual water uptake by roots from each layer is calculated according to a time-dependent depth distribution of roots and an empirical reduction function accounting for soil water availability.
MONERIS	Implicitly derived from flow and rainfall
TRK	Calculation of potential evapotranspiration is based on the Penman equation. In SOILNDB, the actual water uptake by roots from each layer is calculated according to a time-dependent depth distribution of roots and an empirical reduction function accounting for soil water availability. In the water discharge model (HBV), potential evaporation is reduced when the soil moisture reaches below a critical threshold.
SWAT	Three methods included: Thornthwaite, Penman-Monteith, and Priestly-Tailor. Choice is dependent on climatological conditions and data availability. Actual water uptake is defined by a root development distribution function with depth and the soil moisture content. Below a certain threshold, the water uptake will be reduced
EVENFLOW	PE is calculated using the Penman-Monteith equation. AE is calculated and takes into account both moisture stress and the potential rate of evapotranspiration. Crop parameters are fixed by calendar month and are assumed for a healthy crop.
NOPOLU	
SOURCE APP.	-

ANNEX A Hydrological pathways (Cont.)

V Overland flow

NL-CAT (ANIMO)	Conceptual model for surface runoff has been implemented. The distributed model NL-CAT considers a number of sub watersheds. The (power function) relation comprises three parameters. For each sub watershed the surface runoff relation should be parameterised, dependent on soil, slope and distance to watercourses.
REALTA	
N-LES CAT	-
MONERIS	
TRK	In the HBV model, overland flow and surface runoff are components in water balance which are described implicitly in the runoff response relation. The overland flow component can be determined from a subroutine based on either groundwater conditions or by the SCS curve number method.
SWAT	Surface runoff occurs whenever the rate of water application to the ground surface exceeds the rate of infiltration. The SCS curve number procedure (SCS, 1972) and the Green & Ampt infiltration method (1911) can be used. Information for parameterisation of the relations is available.
EVENFLOW	EVENFLOW provides for overland flow through saturation excess and infiltration excess. The magnitude of infiltration excess is determined by the Kirkby equation (Kirkby, 1976) and the timing is controlled buy a dynamic version of the USDA-SCS curve number procedure. The magnitude of the saturation excess is determined from the HOST class.
NOPOLU	
SOURCE APP.	-

ANNEX A Hydrological pathways (Cont.)

VI Subsurface drainage

NL-CAT (ANIMO)	Preferential flow in macropores can be described in detail. Richards equation is solved numerically taking into account the distribution with depth of the water retention curve and the hydraulic conductivity. Different types of drainage systems and water courses can be described by means of a piecewise linear relation between groundwater level and drain discharge. Subsurface drains can be represented.
REALTA NLES-CAT MONERIS	-
TRK	In the SOIL model, drainage flows are described explicitly. In the water discharge model (HBV), the percolation from the root zone is transferred to surface water systems by means of a system of linear and non-linear groundwater reservoirs.
SWAT	SWAT does not solve the Richards equation but utilises a conceptual model for the soil water movement
EVENFLOW	Subsurface flows from different depths within the soil profile are calculated using a characterisation of the hydrology of soil types (HOST). Such lateral flows can represent the effect of drains in controlling the level of the groundwater table.
NOPOLU SOURCE APP.	-

The NL-CAT soil moisture module describes the water movement through soil in the greatest detail. The SWAT model and the HBV model use semi-empirical relations to describe the percolation from the root zone to deeper soil layers.

VII Relation to groundwater flow

NL-CAT (ANIMO)	The lower boundary of a field/plot in the distributed model consists of a boundary condition by which the influence of regional deep groundwater flow can be taken into account. For water flow the boundary condition can be either of defined potential, a flux or a mixed condition and for solute transport the boundary condition is a concentration value.
REALTA N-LES CAT MONERIS	-
TRK	The groundwater storage in HBV is described by an upper, non-linear reservoir, and a deeper, linear, reservoir. Deep, large aquifers are not described explicitly.
SWAT	Considers the upper groundwater zone as a model layer. Properties of this model layer determine the response of inputs to the groundwater system. Upward seepage and exchange with vadose zone is considered
EVENFLOW	A groundwater contribution to river flow is estimated but in the current version of the model is considered to have a constant pollutant concentration.
NOPOLU SOURCE APP.	-

ANNEX A Hydrological pathways (Cont.)

VIII River hydrograph

NL-CAT (ANIMO)	Water flow in the main water courses is described explicitly. Levels and flows are results of the simulation.
REALTA	
N-LES CAT	-
MONERIS	Inputs and outputs (annual water balance) are taken into account.
TRK	The shape of the HBV hydrograph is determined by the transfer through the groundwater reservoirs, a transformation function, and routing between subbasins and through lakes.
SWAT	Water routing through channel networks can be described explicitly. A distinction is made between ephemeral, intermittent or perennial streams. Most relevant terms of the channel water balance are simulated.
EVENFLOW	A hydrograph is simulated by a conceptual model, based on the one-dimensional Topmodel equation parameterised from the mean baseflow index associated with the soils present in the catchment. Diffuse pollutant routing is by simple advection.
NOPOLU	-
SOURCE APP.	-

Ranking according to the detail of process description:
SWAT>NL-CAT>TRK>EvenFlow>Moneris>Realta=Nopolu=Source App.

IX Travel time

NL-CAT (ANIMO)	Water discharge to groundwater and surface water is represented by a pseudo-two dimensional flow in a vertical soil column with unit surface. The distribution with depth of the drainage outflow in the one-dimensional soil column introduces implicitly the travel time distribution of exfiltrating groundwater.
REALTA	-
N-LES CAT	-
MONERIS	
TRK	The water residence time, in HBV, is determined by the flow through and volume of the different water bodies, e.g. in the soil, groundwater and lakes. Travel time between subbasins can be computed by a Muskingum routing.
SWAT	Hydraulic residence time coefficient should be defined explicitly.
EVENFLOW	Travel-time is explicitly calculated using reach specific velocity equations to follow the movement of water parcels from the point of entry to the mouth of the river system. Velocity is a function of instantaneous discharge and mean annual discharge for each reach.
NOPOLU	-
SOURCE APP.	-

ANNEX B Short model information

Model name	Nutrient Losses at Catchment scale
Acronym	**NL-CAT**
Ref. Model description	**Animo** Groenendijk, P. and J.G. Kroes, 1999. Modelling the nitrogen and phosphorus leaching to groundwater and surface water with ANIMO 3.5. Report 144. Winand Staring Centre, Wageningen Schoumans, O.F. and P. Groenendijk, 2000. Modeling Soil Phosphorus Levels and Phosphorus Leaching from Agricultural Land in the Netherlands. J. Environ. Qual. 29:111-116. Vereecken, H., E.J. Jansen, M.J.D. Hack-ten Broecke, M. Swerts, R. Engelke, S. Fabrewitz & S. Hansen, 1991: Comparison of simulation results of five nitrogen models using different datasets. In: Soil and Groundwater Research Report II, Nitrate in Soils, Final report of contracts EV4V-0098- NL and EV4V-00107-C, Commission of the European Communities. Rijtema, P.E., P. Groenendijk and J.G. Kroes, 1999. Environmental impact of land use in rural regions. The development, validation and application of model tools for management and policy analysis. Series on environmental science and management, vol. 1. Imperial College Press, London. Rijtema, P.E. & J.G. Kroes, 1991: Some results of nitrogen simulations with the model ANIMO. Fertilizer Research 27: 189-198 Boogaard, H.L. & J.G. Kroes (1998) Leaching of nitrogen and phosphorus from rural areas to surface waters in the Netherlands. Nutrient cycling in Agroecosystems 50, 321-324. Groenendijk, P. (1999) Surface water pollution from diffuse agricultural sources at a regional scale. Paper for IAHS conference in 1999, Birmingham. Hack-ten Broeke, M.J.D (1998) Evaluation of nitrate leaching risk at site and farm level. Nutrient cycling in Agroecosystems 50, 271-276. Hack-ten Broeke, M.J.D (2001) Irrigation management for optimising crop production and nitrate leaching on grassland. Agricultural Water Management 49, 97-114. Hendriks, R.F.A., K. Oostindie & P. Hamminga (1999) Simulation of bromide tracer and nitrogen transport in a cracked clay soil with the FLOCR/ANIMO model combination. Journal of Hydrology 215, 94-115. Vinten, A.J.A. (1999) Predicting nitrate leaching from drained arable soils derived from glacial till. Journal of Environmental Quality 28, 988-996. Wu, L. & M.B. McGechan (1998) A review of carbon and nitrogen processes in four soil nitrogen dynamics models. Journal of Agricultural Engineering Research 69, 279-305.

	Swap For a complete review of recently published papers about agro-en ecohydrological studies with SWAP, see Table 8.1 in: Dam, J.C. van (2000) Field-scale water flow and solute transport. SWAP model concepts, parameter estimation and case studies. Ph.D.-thesis, Wageningen University, Wageningen. Bierkens, M.F.P., P.J.T. van Bakel & J.G. Wesseling (1999) Comparison of two modes of surface water control using a soil water model and surface elevation data. Geoderma 89, 149-175. Clemente, R.S., R. de Jong, H.N. Hayhoe, W.D. Reynolds & M. Hares (1994) Testing and comparisons of three unsaturated soil water flow models. Agricultural Water Management 25, 135-152. Dam, J.C. van (2000) Simulation of field-scale water flow and bromide transport in a cracked clay soil. Hydrological Processes 14, 1101-1117. Droogers, P. (2000) Estimating actual evapotranspiration using a detailed agro-hydrological model. Journal of Hydrology 229, 50-58. Droogers, P., W.G.M. Bastiaanssen, M. Beyazgül, Y. Kayam, G.W. Kite & H. Murray-Rust (2000) Distributed agro-hydrological modelling of an irrigation system in western Turkey. Agricultural Water Management 43, 183-202. Hack-ten Broeke, M.J.D (1998) Evaluation of nitrate leaching risk at site and farm level. Nutrient cycling in Agroecosystems 50, 271-276. Hack-ten Broeke, M.J.D (2001) Irrigation management for optimising crop production and nitrate leaching on grassland. Agricultural Water Management 49, 97-114. Jong, R. de & A. Bootsma (1997) Estimates of water deficits and surpluses during the growing season in Ontario using the SWATRE model. Canadian Journal of Soil Science 77, 285-294. Kabat, P., B.J. van den Broek, & R.A. Feddes (1992) SWACROP: A water management and crop production simulation model. ICID Bulletin 92 Vol. 41 no. 2, 61-84. Kelleners, T.J., J. Beekma & M.R. Chaudhry (1999) Spatially variable soil hydraulic properties for simulation of field-scale solute transport in the unsaturated zone. Geoderma 92, 199-215. Kroes, J.G., J.G. Wesseling & J.C. van Dam (2000) Integrated modelling of the soil-water-atmosphere-plant system using the model SWAP 2.0, an overview of theory and an application. Hydrological processes 14, 1993-2002. Dam, J.C. van, J. Huygen, J.G. Wesseling, R.A. Feddes, P. Kabat, P.E.V. van Walsum, P. Groenendijk & C.A. van Diepen (1997) Theory of SWAP version 2.0. Simulation of water flow, solute transport and plant

growth in the Soil-Water-Atmosphere-Plant environment. Technical Document 45, DLO-Winand Staring Centre & Report 71, Department Water Resources, Wageningen Agricultural University, Wageningen.

Hack-ten Broeke, M.J.D (2000) Nitrate leaching from dairy farming on sandy soils. Case studies for experimental farm De Marke. Ph.D. thesis, Wageningen University, Wageningen.

Huygen, J., J.C. van Dam & J.G. Kroes (2000) SWAP graphical user interface.

Kroes, J.G., J. Roelsma & J. Huygen (2000) Toetsing van modellen SWAP en ANIMO. Bijlage 1a in: Interregproject Watermanagement in het Benelux-Middengebied. Projectonderdeel B: Watermanagement op bedrijfsniveau; Integratie van het beregeningsadviessysteem met het peil- en nutrientenbeheer.

Kruijne, R., J.G. Wesseling & O.F. Schoumans (1996). Onderzoek naar maatregelen ter vermindering van de fosfaatuitspoeling uit landbouwgronden. Ontwikkeling en toepassing van een- en tweedimensionale modellen. Rapport 374.4, DLO-Staring Centrum, Wageningen.

Wesseling, J.G., J.G. Kroes & K. Metselaar (1998) Global sensitivity analysis of the Soil-Water-Atmosphere-Plant (SWAP) model. Report 160, DLO-Winand Staring Centre, Wageningen.

SWQN

Rijtema P.E., M.F.R. Smit, D. Boels, S.T. Abdel Gawad, and D.E. El Quosy, 1991. Formulation of the Water Distribution Model WATDIS. Reuse of Drainage Water Project Report 23. Drainage Research Institute, Cairo, Egypt and The Winand Staring Centre, Wageningen, The Netherlands.

Abdel Gawad, S.T., M.A. Abdel Khalek, D. Boels, D.E. El Quosy, C.W.J. Roest, P.E. Rijtema, M.F.R. Smit, 1991. Analysis of Water Management in the Eastern Nile Delta. Reuse of Drainage Water Project Report 30. Drainage Research Institute, Qanater, Cairo, Egypt and The Winand Staring Centre, Wageningen, The Netherlands.

SWQL (or NUSWALite)

Kolk, J.W.H. van der & J. Drent (1996). NUSWA – a mathematical model to predict the fate of nutrients in surface water systems. Internal Report 402, DLO-Winand Staring Centre, Wageningen.

Hendriks, R.F.A., J.W.H. van der Kolk en H.P. Oosterom, 1994. Effecten van beheersmaatregelen op de nutriëntenconcentraties in het oppervlaktewater van

	peilgebied Bergambacht. Een modelstudie. DLO-Staring Centrum. Rapport 272. (in Dutch) Kolk, J. W. H. van der & R. F. A. Hendriks (1995) Prediction of effects of measures to reduce eutrophication in surface water in rural areas - a case study. Water Science and Technology Vol 31 No 8 pp 155–158 © IWA Publishing 1995. Liere, L. van, Janse, J.H., Jeuken, M., Schoumans, O.F., Hendriks, R.F.A.& Roelsma, J (2002). Effect of nutrient loading on surface waters in polder Bergambacht, The Netherlands. In: Agricultural effects on ground and surface waters: research at the edge of science and society : proceedings. Ed. Steenvoorden, J.H.A.M. IAHS, Wallingford. Pagina's: p.213-218. Kolk, J.W.H. van der & Hendriks, R.F.A. (1994). Prediction of effects of measures to reduce eutrophication in surface water in rural areas : a case study. In: Living with water : conference on integrated water resources management. NVA, Bergen. p.529-531. Drent, J., Hendriks, R.F.A. & Kolk, J.W.H. van der (1997). Maatregelen ter verbetering van de kwaliteit van het oppervlaktewater in Bergambacht. H twee O, 30(1997)3, p.70-73,90. (in Dutch)
Ref. Users guide	**ANIMO** Kroes, J.G. and J. Roelsma, 1997. ANIMO 3.5. *User's guide for the ANIMO version 3.5 nutrient leaching. model Technical Document* 46, Winand Staring Centre, Wageningen. Boogaard**,** H.L. & J.G. Kroes (1997) GONAT. Geographical Orientated National simulations with ANIMO 3.5 of nutrients. Technical Document 41, DLO-Winand Staring Centre, Wageningen. **SWAP** Kroes, J.G., J.C. van Dam, J. Huygen & R.W. Vervoort (1999) User's Guide of SWAP version 2.0, Simulation of water flow, solute transport and plant growth in the Soil-Water-Atmosphere-Plant environment. Technical Document 53, DLO-Winand Staring Centre & Report 81, Department Water Resources, Wageningen Agricultural University, Wageningen. **SWQN** Not yet available **SWQL (or NUSWALite)** Not yet available

Main contact	**ANIMO** Ir. P. Groenendijk P.O. Box 47 NL-6700 AA Wageningen + 31 317 474425 p.groenendijk@alterra.wag.nl **SWAP** J.G. Kroes P.O. Box 47 NL-6700 AA Wageningen + 31 317 474372 j.g..kroes@alterra.wag.nl **SWQN** Ir. M.H.J.L. Jeuken P.O.Box 47 NL-6700 AA Wageningen + 31 317 474717 m.h.j.l.jeuken@alterra.wag.nl **SWQL (or NUSWALite)** Ir. M.H.J.L. Jeuken P.O.Box 47 NL-6700 AA Wageningen + 31 317 474717 m.h.j.l.jeuken@alterra.wag.nl
Alternative contact	**ANIMO** Drs. ing. J. Roelsma P.O. Box 47 NL-6700 AA Wageningen + 31 317 474368 j.roelsma@alterra.wag.nl **SWAP** Ir. M. Groenendijk P.O. Box 47 NL-6700 AA Wageningen + 31 317 474785 j.g..kroes@alterra.wag.nl **SWQN** Ir. A.A.M.F.R. Smit P.O. Box 47 NL-6700 AA Wageningen + 31 317 474369 a.a.m.f.r.smit@alterra.wag.nl **SWQL (or NUSWALite)** Ir. P. Groenendijk P.O.Box 47 NL-6700 AA Wageningen + 31 317 474425 p.groenendijk@alterra.wag.nl

Model name	**The Irish Phosphorus Model**
Acronym	**Réalta**
Ref. Model description	Kirk McClure Morton (2001). The Lough Derg and Lough Ree Catchment Monitoring and Management System. Final Report. Magette WL (1998). Factors affecting losses of nutrients from agricultural systems and delivery to water resources. Draft Guidelines for Nutrient Use in Intensive Agricultural Enterprises, Teagasc.
Ref. Users guide	Not available
Main contact	Dr Alan Barr Kirk McClure Morton 74 Boucher Road Belfast BT12 6RZ N Ireland Phone +44 2890 667914 e-mail: alan.barr@kmm.co.uk
Alternative contact	Ms Alison Murdock Kirk McClure Morton 74 Boucher Road Belfast BT12 6RZ N Ireland Phone +44 2890 667914 e-mail: alison.murdock@kmm.co.uk

Model name	**Nitrate Leaching Estimator**
Acronym	**N-LES CAT**
Ref. Model description	Simmelsgaard, S. E., Kristensen, K., Andersen, H. E., Grant, R., Jørgensen, J. O. and Østergaard, H. S. (2000): An empirical model for calculation of root zone nitrate leaching. DJF rapport Markbrug no. 32, Danmarks JordbrugsForskning, 67 pages (in Danish)
Ref. Users guide	Not published
Main contact	Hans Estrup Andersen Nat. Env. Research Inst. Vejlsøvej 25, p.o.box 314 DK-8600 Silkeborg, Denmark phone +45 89 20 14 00 e-mail: hea@dmu.dk
Alternative contact	Brian Kronvang Nat. Env. Research Inst. Vejlsøvej 25, p.o.box 314 DK-8600 Silkeborg, Denmark phone +45 89 20 14 00 e-mail: bkr@dmu.dk

Model name	**MO**delling **N**utrient **E**missions in **RI**ver **S**ystems
Acronym	**MONERIS**
Ref. Model description	Behrendt, H., Huber, P., Ley,M., Opitz, D., Schmoll, O., Scholz, G. & Uebe, R. (1999): Nährstoff-bilanzierung der Flußgebiete Deutschlands. UBA-texte, 75/99, 288 S. Behrendt, H., Huber, P., Kornmilch,M., Opitz, D., Schmoll, O., Scholz, G. & Uebe, R. (2002): Estimation of the nutrient inputs into river basins - experiences from German rivers. Regional Environemental Changes, Spec. Issue, (in print; online published). Behrendt, H., Dannowski, R., Deumlich, D., Dolezal, F., Kajewski, Kornmilch, M., Korol, R., Mioduszewski, W., Opitz, D., Steidl, J. & Stronska, M. (2002): Nutrient and heavy metal emissions into the river system of Odra - results and comparison of models. Schriftenreihe des Institutes für Abfallwirtschaft und Altlasten, Technische Universität Dresden, Bd. 28, Vol.2, 213-221.
Ref. Users guide	Not available
Main contact	Horst Behrendt Forchungsverbund Berlin e.V., Müggelseedamm 310 12561 Berlin Germany +493064181683 e-mail: behrendt@igb-berlin.de
Alternative contact	Ulrike Zweynert Forchungsverbund Berlin e.V., Müggelseedamm 310 12561 Berlin Germany +493064181699 e-mail: u.zweynert@igb-berlin.de

Model name	**TRK-The Swedish system**
Acronym	**TRK**
Ref. Model description	The TRK system: Swedish EPA, 1997. Nitrogen from land to sea. Main report. Swedish Environmental Protection Agency, Report 4801, Nordstedts tryckeri AB, Stockholm. Brandt M., and Ejhed H., 2002. TRK, Transport-Retention-Källfördelning, Belastning på havet. Swedish Environmental Protection Agency, Report 5247, Lindblom &Co, Stockholm. HBV (catchment water-balance): Bergström, S., 1995. The HBV model. In Singh, V. P. (ed.) *Computer Models of Watershed Hydrology*, Water Resources Publications, Littleton, Colorado, pp. 443-476. Lindström, G., Johansson, B., Persson, M., Gardelin, M., and Bergström, S., 1997. Development and test of the distributed HBV-96 hydrological model, *J. Hydrol., Vol. 201*, pp. 272-288. SOILNDB & SOILN (arable N leaching): Johnsson, H., Larsson, M., Mårtensson, K., Hoffmann, M. 2002. SOILNDB: A decision support tool for assessing nitrogen leaching losses from arable land. *Environmental Modelling & Software* 17: 505-517. Johnsson, H., Bergström, L., Jansson, P.-E. & Paustian, K. 1987. Simulated nitrogen dynamics and losses in a layered agricultural soil. *Agric. Ecosystems Environ. 18*, 333-356. *Method N leaching estimates:* Hoffmann, M. & Johnsson, H. 1999. A method for assessing generalised nitrogen leaching estimates for agricultural land. *Environmental Modelling and Assessment*, 4:35-44. Johnsson, H. & Hoffmann, M. 1998. Nitrogen leaching from agricultural land in Sweden – Standard rates and Gross loads in 1985 and 1994. *Ambio 27*:481-488 *Revised Method for N leaching estimates:* Johnsson, H. & Mårtensson, K. 2002. Kväveläckage från svensk åkermark – beräkningar av normalutlakning för 1995 och 1999. Swedish Environmental Protection Agency, Report 5248, Lindblom & Co, Stockholm (in Swedish). Phosphorus model for arable land: Ulén, B, Johansson G. and Kyllmar, K., 2001, Model predictions and long-term trends in phosphorus transport from arable lands in Sweden. *Agricultural Water Management* 49, 197-210.

	HBV-N (catchment N-transport and retention): Pettersson, A., Arheimer, B. and Johansson, B., 2001. Nitrogen concentrations simulated with HBV-N: new response function and calibration strategy. *Nordic Hydrology 32(3):227-248.* Arheimer, B and Brandt, M., 2000. Watershed modelling of non-point nitrogen pollution from arable land to the Swedish coast in 1985 and 1994. *Ecological Engineering 14:389-404.* Arheimer, B. and Brandt, M., 1998. Modelling nitrogen transport and retention in the catchments of Southern Sweden. *Ambio 27*(6):471-480.
Ref. Users guide	Not available
Main contact	Helene Ejhed, IVL Swedish Environmental Research Institute, Box 210 60, 100 31 Stockholm, Sweden
Alternative contact	Berit Arheimer, SMHI, 601 76 Norrköping, Sweden
Alternative contact	Holger Johnsson, Dep. of Soil Sciences, SLU, Box 7072, 750 07 Uppsala, Sweden

Model name	**Soil and Water Assessment Tool**
Acronym	**SWAT**
Ref. Model description	Neitsch S.L., Arnold J.G., Kiniry J.R., Williams J.R., (2001), Soil and Water Assessment Tool – Theoretical Documentation - Version 2000, Blackland Research Center – Agricultural Research Service, Texas - USA
Ref. Users guide	Neitsch S.L., Arnold J.G., Kiniry J.R., Williams J.R., (2001), Soil and Water Assessment Tool – User Manual Version 2000, Blackland Research Center – Agricultural Research Service, Texas - USA
Main contact	Antonio Lo Porto IRSA-CNR, Via De Blasio 5, 70123 Bari, Italy Tel +39 0805820532, Fax +39 080 5313365, e-mail: loporto@area.ba.cnr.it
Alternative contact	Faycal Bouraoui JRC-UE, TP460 Via Fermi, 21020 ISPRA (VA), Italy Tel +390332785173, Fax +390332789328, e-mail: Faycal.bouraoui@jrc.it
Alternative contact	Ekaterini Varanou NTUA, Iroon Polytechnioy 5, Zografoy 157 80, Athens, Greece Tel +30107722885 Fax +30107722879 e-mail: Evaran@chi.civil.ntua.gr

Model Name	**EveNFlow**
Ref. Model description	Anthony, S. G., Quinn, P., and Lord, E. I. 1996. Catchment scale modelling of nitrate. *Aspects of Applied Biology* 46, 23-32. Lord, E. I. 1992. Modelling of nitrate leaching: Nitrate Sensitive Areas. *Aspects of Applied Biology* 30, 19-28. Lord, E. I. and Anthony, S. G., 2000. MAGPIE: A modelling framework for evaluating nitrate losses at national and catchment scales. *Soil Use and Management*, 16: 167-174. Scholefield, D., Lockyer, D.R., Tyson, K.C. & Whitehead, D.C. 1991. A model to predict transformations and losses of nitrogen in UK pastures grazed by beef cattle. *Plant & Soil* 132, 165-177. Addiscott, T.M. & Whitmore, A.P. 1991.Simulation of solute leaching in soils of differing permeabilities. *Soil Use & Management* 7(2), 94-102. Chambers, B. J., Lord, E. I., Nicholson, F. A. and Smith, K. A. 1999. Predicting nitrogen availability and loses following applications of manures to arable land: MANNER. Soil Use and Management, 15, 137-143. Beven K, Lamb R, Quinn P, Romanowicz R, Freer J. 1994. TOPMODEL. In *Computer Models of Watershed Hydrology*. Singh V, (ed). Water Resource Publications; 1-43. Boorman, D., Hollis, J. and Lilly, A. 1995 Hydrology of soil types: a hydrologically based classification of the soils of the United Kingdom. Institute of Hydrology Report No. 126, Wallingford, Oxfordshire. A. D. Friend. 1998. Parameterisation of a global daily weather generator for terrestrial ecosystem modelling. *Ecological modelling*, 109, 121-140. De Witt, M. J. M. (2001) Nutrient fluxes at the river basin scale. I: the PolFlow model. Hydrological Processes, 15, 743-759. U.S. Army Corps of Engineers, 1998. 'Engineering and Design Runoff from Snowmelt Engineer Manual. Washington, DC 20314-1000 , Engineer Manual 1110-2-1406.
Ref. Users guide	Not yet published
Main contact	Dr Steven G. Anthony ADAS Woodthorne, Wergs Road, Wolverhampton, WV6 8TQ., UK +44 (0)1902 693192 e-mail: Steve.Anthony@adas.co.uk
Alternative contact	Dr Martyn Silgram ADAS Woodthorne, Wergs Road, Wolverhampton, WV6 8TQ., UK +44 (0)1902 693354 e-mail: Martyn.Silgram@adas.co.uk

Model Name	**NOPOLU system 2®**
Acronym	**NOPOLU**
Ref. Model description	European Environment Agency/IFEN (2000). Technical report N°51. Calculation of nutriment surplus from agricultural sources. Statistics spatialisation by means of CORINE Land Cover. Application to the case of Nitrogen. Spatial Application Division K.U. LEUVEN Research & Development. Version 20/12/2001. Dr. P. CAMPLING, Lic. S. VANDE WALLE, Dr. Ir. J. VAN ORSHOVEN, Prof. Dr. Ir. J.FEYEN. Final Report. Calculation of Agricultural Nitrogen Quantity for EU River Basins.
Ref. Users guide	Not yet published
Main contact	Guillaume LE GALL BETURE CEREC 2 rue Stephenson F-78181 Saint Quentin en Yvelines France Tél. : +33 (0)130 129 105 Fax.: +33 (0)139 449 187 e-mail : guillaume.legall@beture-cerec.com
Alternative contatct	Hervé REISSER BETURE CEREC 2 rue Stephenson F-78181 Saint Quentin en Yvelines France Tél. : +33 (0)130 129 106 Fax.: +33 (0)139 449 187 e-mail : herve.reisser@beture-cerec.com

Model name	**Source Apportionment**
Acronym	**SA**
Ref. Model description	Guideline 8: Principles for Source Apportionment for Quantifying Nitrogen and Phosphorus Discharges and Losses *(Reference Number: 2000-12).* (Source: OSPAR 00/9/2 Add.8 and OSPAR 00/20/1, § 9.5a)
Ref. Users guide	See above
Main contact	Brian Kronvang Nat. Env. Research Inst. Vejlsøvej 25, p.o.box 314 DK-8600 Silkeborg, Denmark phone +45 89 20 14 00 e-mail: bkr@dmu.dk
Alternative contact	Søren E. Larsen Nat. Env. Research Inst. Vejlsøvej 25, p.o.box 314 DK-8600 Silkeborg, Denmark phone +45 89 20 14 00 e-mail: sel@dmu.dk