Assessing Relationships between Stream Water Quality and Watershed Attributes Using the SWAT2000 Integrated Watershed Model

A Case Study in the Rönne River Basin, Sweden

Assefa Kumssa Afeta
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by

Assefa Kumssa Afeta

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Thesis Assessment Board

Chairman:	Prof. Dr. A.K. Skidmore (NRS, ITC)
External Examiner:	Dr. M. S. Krol (WEM, University of Twente)
Supervisor:	Dr. Ir. C.M.M. Mannaerts (WRS, ITC)
Course Coordinator:	M.sc A. Kooiman (ITC)

International Institute for Geo-information Science and Earth Observation, Enschede, The Netherlands

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Abstract

An integrated spatial modelling approach was used to assess the relationships between watershed attributes and diffuse nitrate loading within the Rönne River basin in Southern Sweden. The soil and water assessment tool (SWAT2000) interfaced with Arc View GIS was used to extract and process input parameters from various GIS data layers including digital elevation model, land cover and soil maps for the SWAT2000 model. The model was calibrated and then validated for stream flow at two locations and for nitrate concentration in stream water at one location within the basin. Several input parameters of the SWAT model were adjusted during calibration of which runoff curve number (CN), soil available water capacity (SOL_AWC), soil evaporation coefficient (ESCO) and nitrate percolation coefficient (NPERCO) were found to be most sensitive. The Nash-Sutcliffe prediction efficiency (E_{NS}), coefficient of determination (r^2) and graphical depictions were used to evaluate the performance of the model. During the yearly time step, $\mathbf{E_{NS}}$ values for the simulations of flow range between 0.65 and 0.78 for both calibration and validation, while values ranging between 0.70 and 0.79 were achieved for $\mathbf{r^2}$, suggesting that the model was able to adequately predict stream flows. However, the result obtained for nitrate calibration on the monthly time step was rather poor with negative $\mathbf{E_{NS}}$ value, may be due to insufficiency of information on nutrient related processes within the calibration sub-basin. The calibrated model for hydrological conditions was then used to assess nitrate loading in relation to various watershed attributes. The result revealed that nitrate loading by surface water is strongly positively correlated to the percentage area occupied by agricultural land (r=0.9), suggesting 79% of variation in diffuse nitrate emission can be explained by variation in agricultural land cover. On the contrary, nitrate loading is found to be negatively correlated with the area occupied by forest (r=-0.7). Finally, this study showed that SWAT model is a powerful tool that could be used for hydrological analysis of Rönne River basin, especially when dealing with water balance and quantity aspects. However, further nutrient data verification is essential for a reliable estimation of nitrate load values, related to water quality problems, to help effective management planning and decision-making processes within the watershed.

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Acronyms

AGRL	Agricultural Land Generic
ALPHA-BF	Base flow alpha factor
AMC	Antecedent Moisture Conditions
AnnGNPS	Annualised Agricultural Non-Point Source model
ANSWERS	Areal Non-point Source Watershed Environment Response Simulation
AOI	Area of Interest
BD	Bulk Density
BMP	Best Management Practice
CBN	Organic carbon content
CN	Curve Number
CORINE	Coordination of Information on the Environment
DEM	Digital Elevation Model
DEWA	Division of Early Warning and Assessment
EEA	European Environment Agency
E_{NS}	Nash-Sutcliffe prediction efficiency
ESCO	Soil evaporation compensation factor
ETM+	Enhanced Thematic Mapper Plus
EU	European Union
Euroharp	towards European Harmonised Procedures for Quantification of Nutrients Losses from Diffuse Sources
FRST	Forest - Mixed
g/l	gram per litre
GIS	Geographic Information System
GUI	Graphical User Interface
GW-REVAP	Groundwater "revap" coefficient
ha	hectare
HRU	Hydrologic Response Unit
K_s	Hydraulic conductivity
Mg	Million grams
mg	milligram
mg/l	milligram per litre
MIKE SHE	European Hydrological System model
mm	millimeter
Mt	Million ton
N	Nitrogen
NO_3	Nitrate
NPERCO	Nitrogen percolation factor
NRCS	Natural Resource Conservation Service
NRS	Department of Natural Resources
NSURQ	Nitrate in surface runoff (kg N/ha)

P	Phosphorous
PAST	Pasture
RIZA	Inst. for Inland Water Mgt. & Waste Water Treatment
RNGB	Range - brush
SCS-CN	Soil Conservation Service – Curve Number
SGU	Geological Survey of Sweden
SMHI	Swedish Meteorological and Hydrological Institute
SOL-AWC	Soil Water Available Capacity
STOWA	Foundation for Applied Water Research
SWAT	Soil and Water Assessment Tool
UNEP	United Nations Environment Programme
URML	Residential – Med/low density
USDA-ARS	United States Department of Agriculture – Agriculture Research Service
USLE	Universal Soil Loss Equation
VASTRA	Swedish Water Management Research Programme
WATR	Water
WEM	Department of Water Engineering & Management
WETL	Wetlands – Mixed
WRS	Department of Water Resources
WWTP	Waste Water Treatment Plant

1. Introduction

1.1 Background and Problem Statement

Water, an essential component of the Earth's ecosystem, is a precious natural resource necessary to maintain human populations and ecosystems that any threat to the sustainability of this resource certainly deserves focused attention. However, freshwaters have been subjected to increasing pressures and suffered quality degradations in the past in many parts of the world (European Commission, 2003; Santhi et al., 2005) despite the continuous renewal of the resources by natural processes of the hydrologic cycle (UNEP/DEWA-Europe, 2004).

"Although we as humans recognize this fact, we disregard it by polluting our rivers, lakes, and oceans. Subsequently, we are slowly but surely harming our planet to the point where organisms are dying at a very alarming rate. In addition to innocent organisms dying off, our drinking water has become greatly affected as is our ability to use water for recreational purposes. In order to combat water pollution, we must understand the problems and become part of the solution."(Krantz and Kifferstein, 2005)

Freshwaters, depending on the purpose of demand, have a certain limit of chemical concentrations suitable for aquatic life and human uses (Chapman, 1996). However, high contents of nutrients in water such as nitrates and phosphorus are the major issues in terms of water quality, which have received most attention worldwide (McDonald and Kay, 1988). Most importantly, excess nitrate and phosphorus concentrations in surface waters such as lakes and coastal waters cause eutrophication, which is considered as a serious environmental problem. Eutrophication over-stimulates the growth of algae resulting in oxygen deficiency that becomes hazardous to the aquatic life through decomposition. High nitrate content in drinking waters can also cause a significant health risk to humans.

Excessive nutrients in the ecosystem are mainly caused by anthropogenic activities such as runoff from industrial, urban and agricultural lands. Additional sources such as direct atmospheric deposition (De Wit, 1999), landscape morphology, hydrological conditions, biogeochemical processes in soil, sediment and geological characteristics have also contributions but less significantly. There has been, however, a general understanding that agricultural sources

due to the increased use of manures and manufactured inorganic fertilizers in global agriculture are the single greatest causes of pollution degrading the quality of surface waters as described by several researchers of the field (Bhuyan et al., 2003; Chapman, 1996; De Wit, 1999; Jonsson et al., 2002; Matejicek et al., 2003; Santhi et al., 2005; Van Herpe and De Troch, 2000). The rate of use of nitrogen fertilizer, for example, has increased steadily from the 1950s until the late 1980s coinciding with the end of World War II and the collapse of the former Soviet Union respectively. The disintegration of the Soviet Union led to great disruptions in agriculture and fertilizer use for a short period of time (Matson et al., 1997), but has shown a rapid growing trend since 1995 again with much of the demand driven by an increased use in China. The annual global nitrogen fertilizer consumption has increased from 10Mt in 1960s to nearly 90Mt in late 1990s (Matson et al., 1997).

Such human activity with increased reliance on manufactured inorganic fertilizers and the rate of change in the pattern of use has caused significant effect (Galloway, 1998; Vitousek et al., 1997) on the global cycling of nitrogen, especially on the movement of nutrients to estuaries and other coastal waters. When inorganic fertilizer is applied to a field, it can move through a variety of flow paths to downstream aquatic ecosystems. Some of the fertilizer leaches directly to groundwater and surface waters, with the range varying from 3 percent to 80 percent of the fertilizer applied, depending upon soil characteristics, climate, and crop type (Howarth, 1996). The amount of global phosphorus flux, for example, carried in eroded materials and wastewater from the land to the seas was estimated at 22 Mt P yr^{-1} (Howarth, 1996), which to have been estimated at about 8 Mt P yr^{-1} prior to increased human agricultural and industrial activity.

The trend in Europe is not significantly different per se from the global trend in terms of fertilizer use and hence water quality problems associated with such agricultural activities. Although point sources such as sewage discharges may contribute significantly to nutrient enrichment in some regions of the continent, diffuse sources, particularly agriculture, are still the major contributors (UNEP/DEWA-Europe, 2004). The greater intensification of agriculture and higher productivity during the past 50 years has resulted in a significant increase in fertilizer use, particularly organic nitrogen use (European Commission, 2002; Wolf et al., 2003). According to European Commission report, mineral nitrogen consumption in EU which was less than 1Mt in 1945 has considerably risen to a peak of over 11Mt in 1985 (European Commission, 2002). Similarly, intensive animal

husbandry increased during the same period, contributing to a greater overall nitrogen load through manure. Nitrogen "pressure" from animal husbandry such as cows, pigs, poultry, and sheep on agricultural soils is approximately 8Mt per year based on 1997 data (European Commission, 2002). The greater density of livestock population and manure applications has resulted not only in the direct deposition of nutrients into the ecosystem but also produced a strong volatilization of ammonia to the atmosphere and deposition back on soils and waters with values up to 50-60 kg N/ha/yr. Categorically, about 50% of the nearly 20Mt annual N input to EU agricultural soils comes from mineral fertilizer while the remaining 50% is attributed to air deposition, biological fixation and livestock manure spreading.

According to the European Commission report (2002), the intensified agricultural activity has also resulted in the reduction of permanent grassland and disappearance of wetland areas, which could serve as buffer and sink zones for nutrients. This has subsequently caused increased erosion, high rate of run-off and more rapid flow of nutrients to the aquatic ecosystem and groundwater (European Commission, 2002), with the most important consequences of surface water eutrophication and possible health effects.

The increasing concentration of nitrate in drinking water and eutrophication of surface waters with adverse environmental and health effects triggered growing public concern, which has prompted the European Union for action to improve water quality since the 1970s. During the last two decades, new agricultural policies and environmental regulations have been enforced in many European countries with the objective of reducing agricultural diffuse source pollution, improve water quality and protect the aquatic habitat from eutrophication (Grizzetti et al., 2003). Limits have been established to agricultural fertilizer inputs in order to control water pollution caused by nitrates from agricultural sources through the nitrate directive issued in 1991 (European Community, 1991). A similar directive was issued concerning urban waste water treatment in the same year to tackle the problem of nitrate pollution. The same nitrate problem has also been addressed in the *water framework directives* of 2000, with the motto of "getting Europe's water cleaner", that has to get polluted waters clean again, and ensure clean waters are kept clean. This includes nitrate under the surveillance monitoring list, mainly aimed at controlling and reducing water pollution resulting from discharge of livestock effluents and the excessive use of fertilizers on agricultural land (European Commission, 2000). The directive, generally, emphasizes on agricultural pollution sources in a watershed and provides a basis for taking actions needed to restore a

water body thus the respective member countries are required to act and implement accordingly.

The efforts made so far have shown significant progress in controlling point sources from sewage and industrial wastes resulting in lower levels of most pollutants such as phosphorus (EEA, 1998a). However, despite the EU Nitrate Directives and the increased use of agri-environmental measures, the agricultural sector remains the main source of diffuse pollutants without as much progress. Nitrogen surpluses from agriculture are rather constant and hence levels in rivers are still as high as they were in the early 1990s (EEA, 2001; EEA, 2003; European Commission, 2003) as indicated in figure 1.

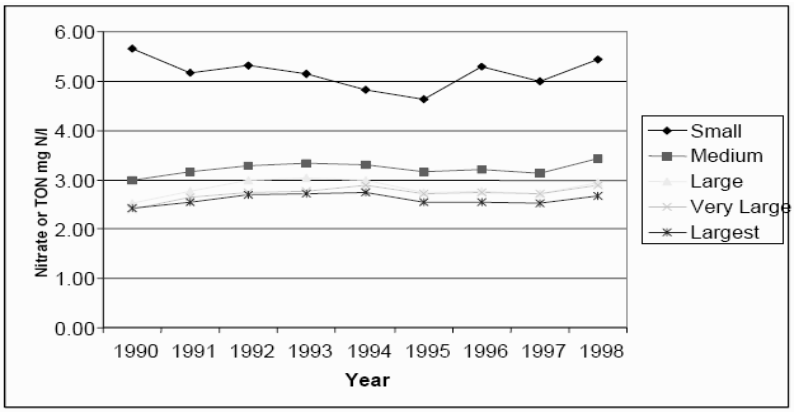

Figure 1: Annual average nitrate concentrations (µg N/l) in different sized European rivers

In Sweden, similar to most EU countries, the load of nutrients on streams, lakes and coastal waters has increased dramatically after World War II mainly due to the intensification of agricultural activities (Arheimer et al., 2004; Ekologgruppen, 2000). This has caused the reduction of wetland areas by 90% thereby stimulating the growth of eutrophication in surface waters and reduction of biological diversity. Nitrogen input to agricultural soils from livestock manure was estimated between 40 to 50 kg N/ha/yr in 1997, while the total nitrogen pressure including mineral fertilizer use, livestock manure spreading, atmospheric deposition and biological fixation was between 100-125 kg N/ha/yr (European Commission, 2002). Average nitrogen atmospheric deposition in Southern Sweden, including the study area, has been estimated between 5 and 10kg N/ha/yr.

In view of the EU water policy, the Swedish Government has put in place policy measures and has been implementing several conservation practices aimed at reducing anthropogenic nitrogen

emissions, mainly due to uncontrolled municipal sewage systems and agricultural practices (Ekologgruppen, 2000) by regulating urban waste disposal systems, the use of excessive fertilizers and handling of manures. In addition to policy enforcements to improve urban sewage systems and changing agricultural practices, complimentary measures to reduce nutrient transport to inland and coastal waters are also being undertaken to improve water quality. These include the creation of wetlands, ponds and buffer-zones in the intensively cultivated areas, to increase nutrient sink areas. But the achievement is rather small (Arheimer et al., 2004) to attain the desired target and eutrophication is still a subject of major concern in many Swedish inland and coastal waters, particularly in the intensively farmed plain lands, which are more common in central and southern part of the country. Although there have been improvements, the level of phosphorus content in several lakes is still high that they can be considered eutrophic. Sources indicate that the concentration of nitrate, especially in groundwater, is even worse showing rather an increasing trend (European Commission, 2002), and "water quality problems still remain due to diffuse leaching from arable land and from sediment loading"(Arheimer et al., 2004).

The proposed research area, the Rönne River basin, is the second largest catchment in Skåne, southern Sweden, known to have been suffering from such elevated nutrient loadings and subsequent eutrophication of surface waters within the watershed (Arheimer et al., 2004). The problem has been persistent that raised a growing concern, due to: firstly, its socioeconomic and ecological importance in that it contains lakes such as Ringsjön, which is frequently affected by algal blooms. In addition to being a habitat for aquatic ecosystem, Lake Ringsjön is currently used as a spare drinking water source for people inhabiting the area. Secondly, the catchment drains through the Rönne River to the coastal water of the North Sea. Coastal waters, in general, are ecologically sensitive from inland pollution sources and in particular the coastal water of the North Sea is a subject of major concern in the EU that led the current study area to have been designated as "nitrate vulnerable zone". As a result, the Rönne river basin has been used as a pilot catchment area for eutrophication control (Arheimer et al., 2004) using an integrated river basin management approach.

Effective decision-making processes for best management options such as the implementation of EU water framework directives require relatively reliable, not expensive and timely information. Conventional monitoring methods like field stream water quality data collection and analysis as a decision-making tool is expensive, time consuming

(Santhi et al., 2005) and even it becomes more complex and difficult especially when dealing with large watersheds such as Rönne river basin characterised by mixed land use, soil types, topography and geological conditions. It is believed that a watershed based modelling approach, with spatial or geographic information system capability, allows for the consideration of such attribute variations, and quantification of the impacts at different spatial and temporal scales. The application of watershed models as decision support tools in reducing nutrient leaching from different land use areas in a watershed and the resulting adverse environmental effects in general and water quality in particular has been discussed in several researched works (Arheimer et al., 2004; Arnold et al., 1998; Bhuyan et al., 2003; Borah and Bera, 2003; Grizzetti et al., 2003; Jayakrishnan et al., 2005; Jha et al., 2004; Jonsson et al., 2002; Santhi et al., 2005; Tolson and Shoemaker, 2004; Tripathi et al., 2003; White et al., 1992; Wolf et al., 2003)

Hence, this study is intending to apply an integrated modelling approach that uses remote sensing data, geographic information system and a physically based simulation model in order to relate nutrient loadings with watershed attributes. Previous scientific effort made, in this regard, in the study area is that of the VASTRA project - The Swedish Water Management Research Programme - which has been focusing on the development and demonstration of models for more efficient eutrophication control. It is believed that this study will contribute towards such efforts, in particular, and the implementation of the EU water framework directive, in general, by identifying the main source and sink areas of nutrients responsible for adverse environmental impacts such as eutrophication and health risks within the Rönne River basin and the North Sea coastal waters.

1.2 Objectives

In light of the above background and problem statement, the proposed research project has the following general and specific objectives:

General Objective:

- To analyze and explain the spatial relationships between nutrient fluxes (NO_3-N loadings) of the Rönne river basin and (sub-) watershed area source/sink attributes using remote sensing data, spatial GIS analyses and the Soil and Water Assessment Tool (SWAT2000 model).

Specific objectives:

- To simulate the hydrologic flow regime and nutrient (NO_3) loading at selected sub-basin outlets from the different (sub-)watershed area attributes;
- To assess the spatial relationship between watershed attributes and NO_3 as a source/sink effect, and hence the contribution of agricultural land as compared to others;
- To test the accuracy of the modelling process against measured/observed stream flow and nutrient loading data;

1.3 Research Questions

The research will try to address the following questions:

- Is there any spatial relationship between stream water quality (NO_3 loading) and watershed characteristics of the Rönne River basin? If so, which of the watershed attributes, for example land use, soil types, topographic features, etc, are the most explanatory variables or predictors?
- Which sources, i.e. agricultural land, forest, etc, contribute most to NO_3 loads in stream water?
- Does change in land use/cover account for changes in stream water quality?
- Can SWAT accurately predict stream flow and nutrient loadings and therefore can probably be used to model similar watersheds?

1.4 Hypotheses

- Agricultural lands are the most contributors of nitrates in stream waters of the Rönne River basin compared to other sources;
- Spatially, the concentration of nitrate increases longitudinally downstream along the Rönne River due to the cumulative effect of the upstream sub-basin characteristics.
- Change in land use scenario would result change in nitrate loading trend and hence stream water quality.
- SWAT predicts about 70% of observed watershed processes, e.g. stream water flow and nitrate loading, within the watershed.

1.5 Research Approach

A simplified research conceptual framework is indicated in the following flowchart, figure 2.

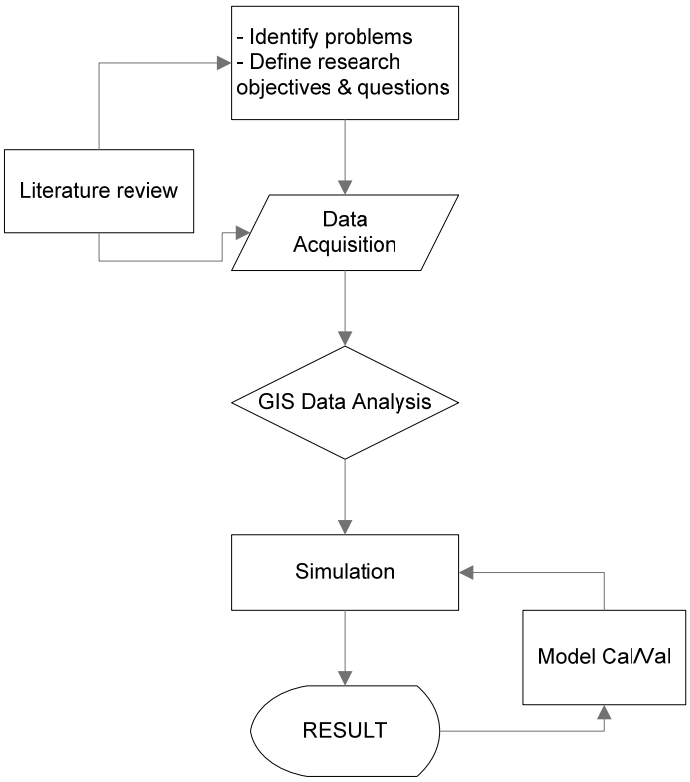

Figure 2: Conceptual framework of the research work

2 An Integrated Spatial Approach for Watershed Management: Overview of Tools

2.1 General

The causes of water pollution are generally attributed to diffuse sources originated mainly from agricultural runoff and point sources originated from domestic and industrial effluents. The sources of such pollutants and their routing mechanism are controlled by several natural and anthropogenic factors such as climatic conditions, hydrologic regimes, soil types, land uses, geological and topographical features, etc. Such natural and human-induced complexities make the protection of pollution and hence water resource management too difficult. Therefore, in order to characterize pollutants and their mechanism of movement within the environment, extensive knowledge and information is required not only about the pollutants in concern but also the controlling factors mentioned within the environment at varying spatial and temporal scales.

The underlying challenge is, however, difficulties that may exist in understanding and evaluating such natural processes leading to impairments in a watershed (Borah and Bera, 2003). Description of the appropriate spatial and temporal variability of processes based on conventional measurements or observations, for example field sampling and analysis of stream water quality, at selected locations to represent and explain the entire watershed is ineffective, expensive and time consuming increasing the intricacy on effective decision-making process. An integrated spatial approach that combines a watershed scale hydrologic and water quality models, remote sensing and geographic information system (GIS) led to tremendous progress during the last few decades (Jayakrishnan et al., 2005) in alleviating such problems and demonstrated to be useful tool in studying, understanding and quantifying the impact of major land use changes and other related watershed characteristics on both water quality and quantity.

2.2 Hydrologic and Water Quality Models

Hydrologic models are becoming increasingly essential in water management (STOWA/RIZA, 1999) as they are powerful tools in

hydrologic system investigation by representing known or assumed functions mathematically that explains the various components of a hydrologic cycle. Basically, two broad categories of hydrologic models are possible – deterministic and stochastic (STOWA/RIZA, 1999). Stochastic models are fully data or field measurement oriented while deterministic models are based on known physical processes and laws.

Deterministic models are the most widely used models and can be further subdivided into whether they are based on a simple empirical relation with spatially lumped description of the watershed or whether they are a physically based and spatially distributed description of the catchment area involving equations with computationally intensive numerical solutions (Borah and Bera, 2003). Empirical models are based on functions relating basin average input data to outputs that may produce reasonable results. However, since they do not have any physical basis, such models are not expected to accurately represent the spatial and temporal details of the watershed properties as desired.

Spatially distributed models, on the other hand, are based on human knowledge of the physical process, which are representative of observed hydrologic phenomena that control the response of the watershed with known geographic locations and attributes. The spatially distributed nature of such models allow a multi-purpose assessment of the effect of spatially and temporally variable processes and watershed properties on the hydrological responses (Romanowicz et al., 2005) making these kind of models more attractive for studying land and water management options. However, despite the growing demands, several shortcomings are associated with the use of spatially distributed hydrologic and water quality models in watershed management decision making process due to: firstly, they involve intensive data demand that cannot be easily met in many practical aspects because of unavailability or do not just satisfy the desired quality standards; secondly, they require tedious computational time and advanced computing machines increasing the cost associated with the modelling process; thirdly, they require an in-depth scientific skill and understanding, for example, related to model parameterization schemes (Romanowicz et al., 2005). Therefore, it will be imperative to decide upon an appropriate model for a given watershed application through compromising between cheap but simple models and detailed but computationally intensive and expensive models (Borah and Bera, 2003).

Models can also vary based on their capability of simulating the temporal scale of processes, categorised as short-term single event based and long-term continuous simulation models. Event-based models are designed for analyzing single-event storms and assessing watershed management practices, especially structural practices. Agricultural Non-Point Source Pollution model (AGNPS), Areal Non-point Source Watershed Environment Response Simulation (ANSWERS), Dynamic Watershed Simulation model (DWSM), and KINmatic runoff and EROSion model (KINEROS) are examples of single rainfall event models. Continuous simulation models are, on the other hand, designed for analyzing long-term effects of hydrological changes and watershed management, especially agricultural, practices. Annualized Agricultural Non-Point Source model (AnnAGNPS), ANSWERS-continuous, Hydrological Simulation Program–Fortran (HSPF), and Soil and Water Assessment Tool (SWAT) are examples of continuous models. While detailed descriptions and references for some selected models in both categories were given in Borah and Bera (2003), a summary of four selected models of continuous types, extracted from the same source, are given in table 1 for comparison purposes.

The extensive review of the various non-point source pollution models and their applications (Borah and Bera, 2003) indicated that SWAT (Arnold et al., 1998) is suitable for long-term continuous predictions in agricultural watersheds. A more detailed description of SWAT model is given in section 3.2

Table 1: Summary of watershed-scale continuous hydrologic and non-point source pollution models

Description/ Criteria	AnnAGNPS	ANSWERS-Continuous	MIKE SHE	SWAT
Model compon-ents/ capabili-ties	Hydrology, transport of sediment, nutrients, and pesticides resulting from snowmelt, precipitation and irrigation, source accounting capability, and user interactive programs including TOPAGNPS generating cells and stream	Daily water balance, infiltration, runoff and surface water routing, drainage, river routing, ET, sediment detachment, sediment transport, nitrogen and phosphorous transformations, nutrient losses through uptake, runoff, and	Interception - ET, overland and channel flow, unsaturated zone, saturated zone, snowmelt, exchange between aquifer and rivers, advection and dispersion of solutes, geochemical processes, crop growth and nitrogen processes in the	Hydrology, weather, sedimentation, soil temperature, crop growth, nutrients, pesticides, agricultural management, channel and reservoir routing, water transfer, and part of the USEPA BASINS modeling system with user interface and ArcViewGIS platform.

	network from DEM.	sediment.	root zone, soil erosion, dual porosity, irrigation, and user interface with pre- and post-processing, GIS, and UNIRAS for graphical presentation	
Tempor-al scale	Long term; daily or sub-daily steps.	Long term; dual time steps: daily for dry days and 30 seconds for days with precipitation.	Long term and storm event; variable steps depending numerical stability.	Long term; daily steps.
Waters-hed represe-ntation	Homogeneous land areas (cells), reaches, and impoundments.	Square grids with uniform hydrologic characteristics, some having companion channel elements; 1-D simulations.	2-D rectangular/square overland grids, 1-D channels, 1-D unsaturated and 3-D saturated flow layers.	Sub-basins grouped based on climate, hydrologic response units (lumped areas with same cover, soil, and management), ponds, groundwater, and main channel.
Rainfall excess on overlan-d/ water balance	Water balance for constant sub-daily time steps and two soil layers (8-in. tillage depth and user supplied second layer).	Daily water balance, rainfall excess using interception, Green- Ampt infiltration equation, and surface storage coefficients.	Interception and ET loss and vertical flow solving Richards's equation using implicit numerical method.	Daily water budget; precipitation, runoff, ET, percolation, and return flow from subsurface and groundwater flow.
Runoff on overlan-d	Runoff curve number generating daily runoff following SWRRB and EPIC procedures and SCS TR-55 method for peak flow.	Manning and continuity equations (temporarily variable and spatially uniform) solved by explicit numerical scheme.	2-D diffusive wave equations solved by an implicit finite difference scheme.	Runoff volume using curve number and flow peak using modified Rational formula or SCS TR-55 method.
Runoff in channel	Assuming trapezoidal and compound cross-sections, Manning's equation is numerically solved for hydraulic parameters and	Manning and continuity equations (temporarily variable and spatially uniform) solved by explicit numerical scheme.	1-D diffusive wave equations solved by an implicit finite difference scheme.	Routing based on variable storage coefficient method and flow using Manning's equation adjusted for transmission losses, evaporation, diversions, and

	TR-55 for peak flow.			return flow.
Overlan-d sedime-nt	Uses RUSLE to generate sheet and rill erosion daily or user-defined runoff event, HUSLE for delivery ratio, and sediment deposition based on size distribution and particle fall velocity.	Raindrop detachment using rainfall intensity and USLE factors, flow erosion using unit-width flow and USLE factors, and transport and deposition of sediment sizes using modified Yalin's equation.	No information.	Sediment yield based on Modified Universal Soil Loss Equation (MUSLE) expressed in terms of runoff volume, peak flow, and USLE factors.
Chemic-al simulat-ion	Soil moisture, nutrients, and pesticides in each cell are tracked using NRCS soil databases and crop information, and reach routing includes fate and transport of nitrogen, phosphorous, and individual pesticides, and organic carbon.	Nitrogen and phosphorous transport and transformations through mineralization, ammonification, nitrification, and denitrification, and losses through uptake, runoff, and sediment.	Dissolved conservative solutes in surface, soil, and ground waters by solving numerically the advection dispersion equation for the respective regimes.	Nitrate-N based on water volume and average concentration, runoff P based on partitioning factor, daily organic N and sediment adsorbed P losses using loading functions, crop N and P use from supply and demand, and pesticides based on plant leaf-area -index, application efficiency, wash off fraction, organic carbon adsorption coefficient, and exponential decay according to half lives.
BMP evaluat-ion	Agricultural management.	Impact of watershed management practices on runoff and sediment losses.	No information.	Agricultural management: tillage, irrigation, fertilization, pesticide applications, and grazing.

Source: (Borah and Bera, 2003)

2.3 The Role of Remote Sensing Data

Models for watershed management require spatial information, concerning the watershed area attributes, as an essential input such as land cover/use, soil types, topography, geology, climatic conditions, population, etc, in order to represent the biophysical

processes of the watershed so that the outputs are technically and scientifically effective. While collection of such datasets using conventional methods, i.e. ground based sampling is proved to be costly and time consuming, the use of the state-of-art tools such as remote sensing techniques offer a cheaper and quick means (Engman, 2002; Kerle et al., 2004; Matejicek et al., 2003; Muller et al., 1993) of measuring watershed parameters both qualitatively and quantitatively. Satellite remote sensing methods can also offer the opportunity to cover an extensive area of a watershed being investigated at a time and, more importantly, the electronic data handling format enables the readily incorporation of the data into computer based hydrologic models. Moreover, the repetitive coverage of the satellite images enables to monitor the changes taking place within the watershed and their long-term effect on watershed management practices such as expansion of agricultural lands, deforestation, urbanisation, etc, by comparing multi-temporal images of the same spatial distribution. In conclusion, remote sensing methods provide reliable and essential input data (Engman, 2002; Kerle et al., 2004; Matejicek et al., 2003; Muller et al., 1993; Rao and Kumar, 2004), especially land cover maps, for watershed models in a time and cost-effective manner.

2.4 The Role of Geographic Information System

Pertaining to their integrative capabilities, geographic information systems are also powerful and indispensable tools for watershed-scale hydrologic analysis and modelling (Khairy et al., 2000). The information extracted from satellite data, topographical maps and other data sources representing soils, land use/cover, weather, topography, population, etc could be stored in GIS as a database in tabular, vector and raster formats. GIS, then, allows the effective and efficient integration of such spatial and non-spatial data for model inputs as well as the spatial visualization of outputs.

Most of the widely used watershed models were developed in the 1970s and 1980s, whereas since the 1990s the main focus of modelling research was targeted at developing user friendly graphical user interfaces (GUI) and linking with geographic information system (GIS) and remote sensing data (Borah and Bera, 2003). Several water quality models have been integrated and coupled with GIS software so far giving the opportunity of improved and effective data management techniques for watershed modelling. The integration of SWAT2000 with ArcView 3.3 (DiLuzio et al., 2002) is one of such examples.

3 Methods and Materials

The modelling process involves a step by step implementation of methods and tools. Basically three main categories can be identified namely: spatial data collection, preparation and analysis. The specific methods and techniques used in this study are schematically represented by the flow chart in figure 3. Most of the spatial GIS data layers in digital format were obtained from various existing sources. GIS Softwares such as Edas & ArcView available at ITC coupled with SWAT2000 were used for data preparation, integration, analysis and presentation. Statistical analyses of the results were performed using Microsoft office excel 2003.

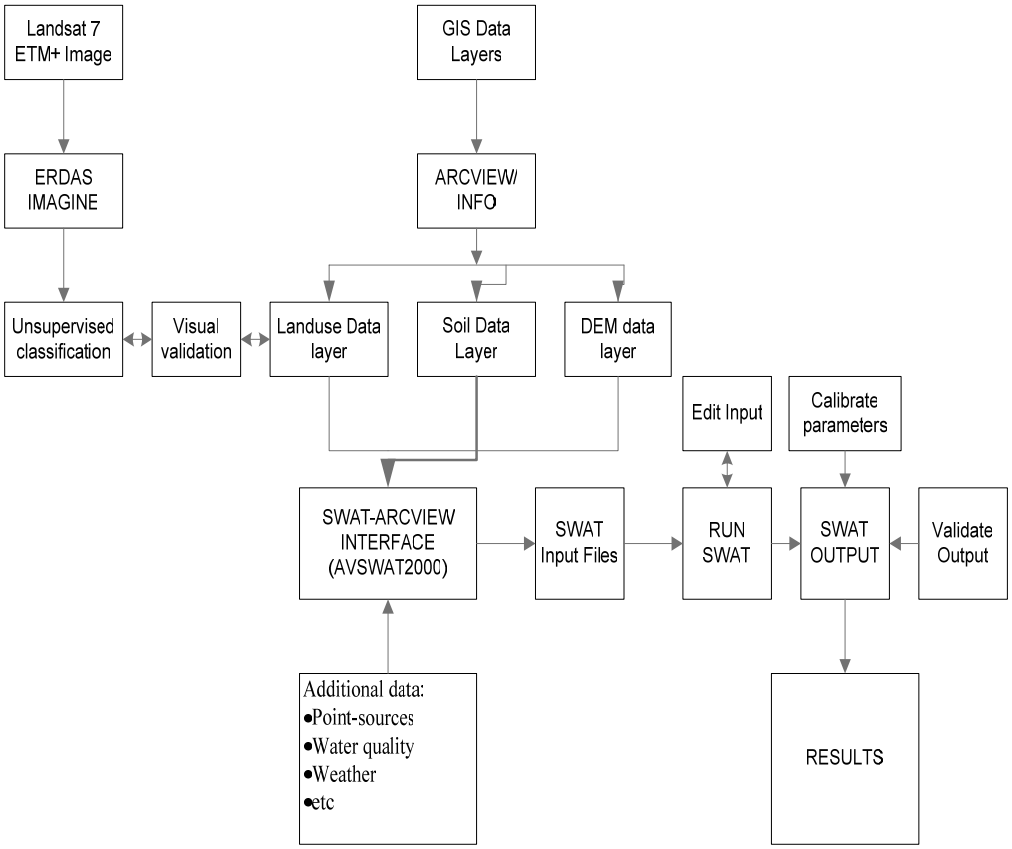

Figure 3: Schematic Diagram of the Integrated Modelling Approach

3.1 Description of the Study Area

The study area, Rönne River basin, is the second largest catchment located in the county of Skåne, the most southern region of Sweden. The watershed is elongated in the southeast-northwest direction from 55.8ºLat/13.2ºLon at the southern east end to 56.4ºLat/12.0ºLon at the northern west end. Location map of the study area is shown in figure 4.

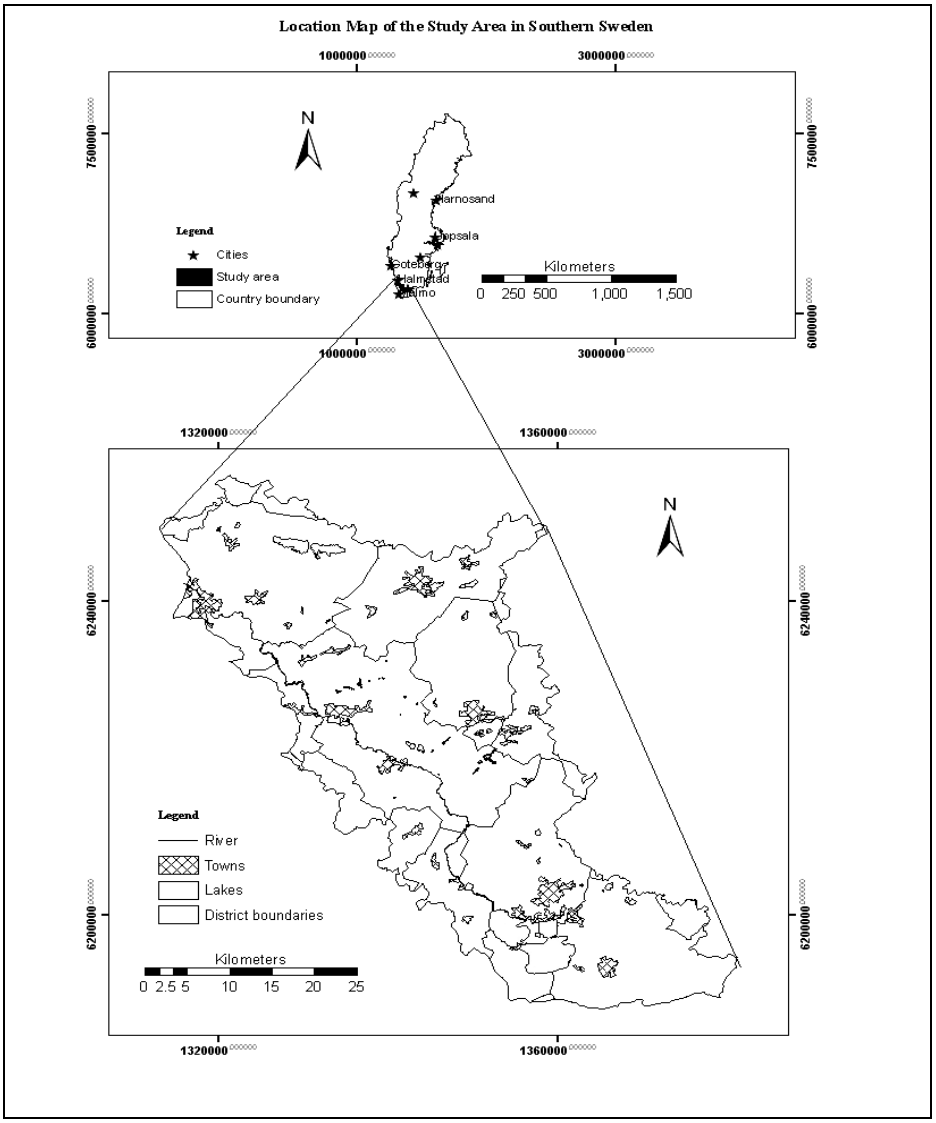

Figure 4: Location of Rönne River basin in Southern Sweden

The basin covers a catchment area of 1900km^2 being drained by the Rönne River in the northwest direction where it finally enters the North Sea near the city of Ängelhom. Figure 5 shows a panoramatic view of the Rönne River as it crosses agricultural fields (left) and enters the North Sea (right).

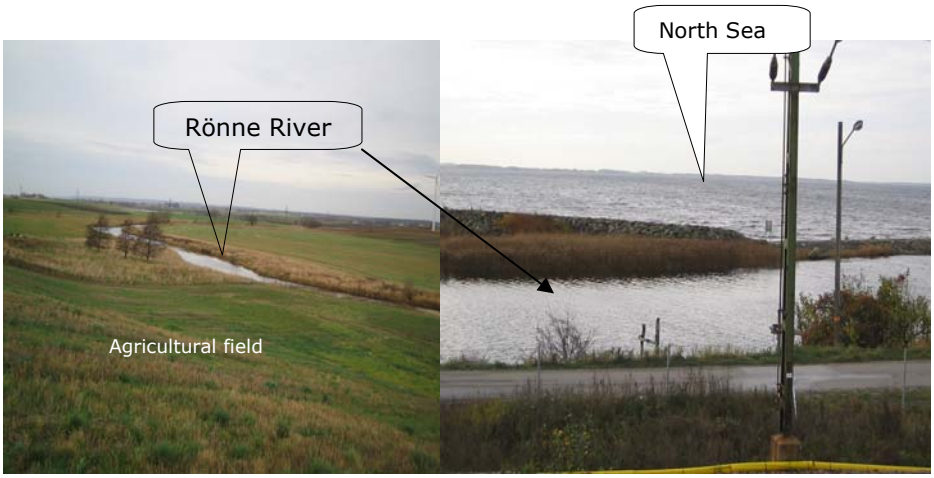

Figure 5: A Panoramatic view of Rönne River with respect to agricultural lands and The North Sea

Elevation within the watershed ranges between 0 m near the North Sea and 220 m further in the upstream area. The watershed and its surroundings receive an average annual precipitation of 700mm and the mean annual temperature is 7.4ºC. January and February are the coldest months of the year with average temperatures of -0.7ºC while the maximum temperature occurs in July and August having an average temperature record of 16.3ºC. The annual evaporation and surface runoff have been estimated between 450-550 mm and 200-300mm respectively (Gustafsson, 1992) while The annual mean flow at the outlet of the Rönne River is about 25 m^3/s (VASTRA, 2005).

Geologically, the watershed area is characterized by the oldest rocks of Precambrian gneisses and granites surrounded by a cover of sedimentary deposits of Early Palaeozoic and Mesozoic ages (Norling and Wikman, 1990). The gneisses and granites are, characterised by deep weathering and intensive fractures, frequently exposed on horsts (figure 6), which are tectonic structures known to occur alternating with grabens along NW-SE trending fracture zones (Daniel, 1978). The lower Palaeozoic is represented by Cambrian quartzitic sandstones and Ordovician-Silurian silty shales while the Mesozoic rocks comprise continental and brackish-marine origin of the Upper Triassic sandstones, clays, and shales succeeded by coal-

seams, which have been mined for some 200 years (Daniel, 1978; Norling and Wikman, 1990). Sand, silty and clayey deposits form the top of the pre-Quaternary sedimentary succession of the Jurassic age. The Quaternary deposits of the area are mostly characterized by younger soils of glacial/postglacial clays, silts and sands. The Quaternary period marks the time when Sweden and the surrounding seas were completely glaciated. The most common soil is till or moraine, composed of mainly clayey and fine-grained sediments, and covers extensive area at the surface as well as below other deposits (Gustafsson, 1992). The major soil types of the area are shown on a generalised soil map in figure 14.

Figure 6: Granite/gneiss rock outcrops on a relatively raised morphology or horst structures

Due to variations in geology and geomorphology, widely differing landscape elements can be found, ranging from acid and poor forest brooks to eutrophied agricultural ditches (VASTRA, 2005). Forest dominates most part of the watershed to the north (about 47%) whereas agricultural land dominates the southern part (about 27%). The area is generally considered one of the most favourable for agriculture in Sweden due to the highly fertile soils relative to other parts of the country. The remaining land use types are characterized by urban areas, water bodies and small proportions of wetlands. Cities with over 10,000 inhabitants are (VASTRA, 2005): Ängelholm (35570), Klippan (15539), Höör (14113) and Hörby (13761). Water bodies occupy about 3.5% of the watershed, Lake Ringsjön being the biggest located in the south-eastern part of the catchment. Figure 7 shows the position of Lake Ringsjön relative to the surrounding cultivated land, while the generalised land use map of the area is shown in fig 13.

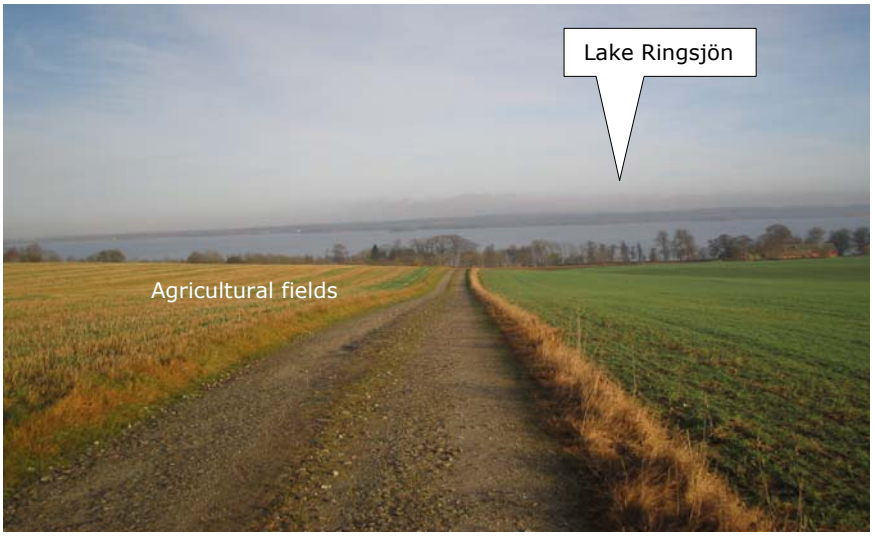

Figure 7: Lake Ringsjön and the surrounding cultivated land

The area is relatively densely populated as compared to other parts of Sweden, a total of nearly 100000 people living in the catchment area out of which about 70000 or 70% are residents of urban areas (VASTRA, 2005). The region has a history of elevated nutrient content due to mainly intensified agricultural activity resulting in nutrient leakage. Lake Ringsjön, which is currently serving as a spare drinking water source, has been suffering from such high nutrient loadings and frequently affected by eutrophication. Consequently, construction and creation of wetlands, ponds and buffer zones have been underway to complement improving water quality, reduce nutrient transport and increase biodiversity in intensively cultivated farmland.

Previous scientific efforts made in implementing the EU water framework directive, in Sweden in general and in the study area in particular, are that of the VASTRA project (The Swedish Water Management Research Programme) which has been focusing on the development and demonstration of models for more efficient eutrophication control. Models for nitrogen flow were already developed during Phase I of the project, and similar models for phosphorus have been under construction (VASTRA, 2005). The scenarios modelled so far in VASTRA phase I, indicated that changing agricultural practices, among others, are the most effective and least expensive way of reducing nitrogen transport from land to lakes and finally the sea.

3.2 Description of the SWAT2000 Model

The watershed scale model, Soil and Water Assessment Tool (SWAT2000) (Arnold et al., 1998), developed by the United States Department of Agriculture–Agriculture Research Service (USDA-ARS), was used in this research. SWAT2000 was selected in this study because of: (1) its semi-distributed nature that simulates spatial details of processes within a watershed, which is the underlying objective of this research work, (2) its availability free of charge that could be downloaded by all interested users from the internet at http://www.brc.tamus.edu/swat with full of its documentation, (3) it provides an easy-to-use graphical-user interface for model set-up and use that has been integrated into ArcView GIS (DiLuzio et al., 2002) so that the user can easily learn and use within few days of practice, and (4) it is one of the nine contemporary methodologies currently used for quantifying diffuse losses of N and P by European research institutes to inform policy makers at national and international levels (Euroharp, 2005).

Soil and Water Assessment Tool (SWAT model) is mostly physically based distributed river basin scale model developed with the objective of simulating the long-time effect of land use practices on water quality, sediment and agricultural chemical yields (Arnold et al., 1998; Neitsch et al., 2002) in a watershed with varying soil types, land uses and climatic conditions. An extensive review of various non-point source pollution models and their applications by Borah and Bera (2003) indicated the suitability of SWAT for long-term continuous simulations in agricultural watersheds. The model has the capability of analysing large watersheds and river basins by subdividing the area into homogeneous sub-basins or sub-watersheds (Santhi et al., 2005). Each sub-basin is further discretized into several hydrologic response units (HRUs) that have distinctive land use and soil combinations. The model requires specific information about the topography, land use, soil type and weather conditions of a watershed. Most of the dataset are commonly readily available from government offices thus reducing the cost and time of field data acquisition.

The main components of the model comprise hydrology, weather, erosion/sedimentation, soil temperature, crop growth, nutrients, pesticides, and agricultural/land management (Borah and Bera, 2003). A complete description of each component is given in Arnold et al. (1998) and Neitsch et al. (2002). Nevertheless, brief descriptions of the more relevant components to this study are given here.

3.2.1 Hydrology

SWAT is an integrated hydrological model that simulates the hydrological balances of a watershed, which is the driving force behind everything that happens in the watershed (Neitsch et al., 2002) such as precipitation, stream flow, groundwater flow, evapotranspiration, and infiltration. Simulation of the hydrologic cycle by SWAT is divided in to two parts: (1) the land phase of the hydrologic cycle that controls the amount of water, sediment, nutrient and pesticide loadings to the main channel for each sub-basin and (2) the water/routing phase of the hydrologic cycle that controls the movement of water, sediment, nutrients, and pesticides through the channel network of the watershed to the outlet (Neitsch et al., 2002). Figure 8 shows the hydrologic cycle simulated by SWAT2000.

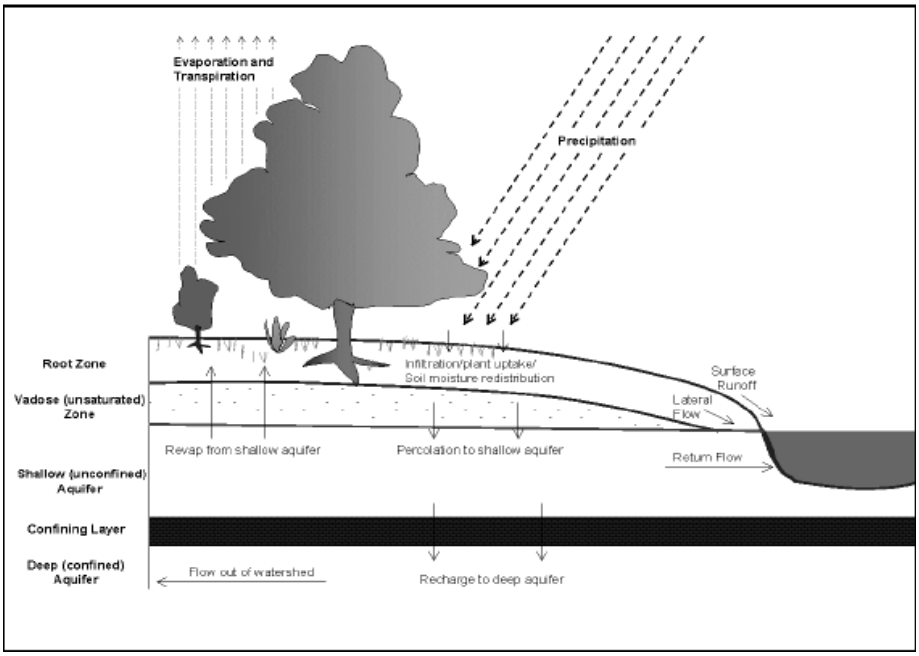

Figure 8: Schematic representation of the hydrologic cycle (after Neitsch et al., 2002)

The land phase of the hydrologic cycle as simulated by SWAT is based on the water balance equation:

$$SW_t = SW_0 + \sum_{i=1}^{t} (R_{day} - Q_{surf} - E_a - w_{seep} - Q_{gw}) \qquad (1)$$

Where SW_t is the final soil water content (mm H_2O), SW_0 is the initial soil water content on day *i* (mm H_2O), *t* is the time (days), R_{day} is the amount of precipitation on day *i* (mm H_2O), Q_{surf} is the amount of surface runoff on day *i* (mm H_2O), E_a is the amount of evapotranspiration on day *i* (mm H_2O), w_{seep} is the amount of water entering the vadose zone from the soil profile on day *i* (mm H_2O), and Q_{gw} is the amount of return flow on day *i* (mm H_2O).

When precipitation exceeds infiltration rate, it becomes surface runoff or overland flow that occurs along a sloping surface. SWAT simulates surface runoff separately for each hydrologic response unit (HRU) and routes to obtain the total runoff for the watershed. Two methods are provided in SWAT to estimate surface runoff: the modified SCS curve number method and the Green and Ampt infiltration method as described in Neitsch (2002). The SCS-CN method is the most widely used method for computing surface runoff for rainfall event, which involves the use of simple empirical formula, and readily available tables and curves. The accumulated runoff depth or rainfall excess is estimated using the SCS curve number equation as:

$$Q_{surf} = \frac{(R_{day} - I_a)^2}{(R_{day} - I_a + S)} \qquad (2)$$

Where Q_{surf} is the accumulated runoff or rainfall excess (mm H_2O), R_{day} is the rainfall for the day (mm H_2O), I_a is the initial abstractions which includes surface storage, interception and infiltration prior to runoff (mm H_2O), and S is a maximum soil water retention parameter (mm H_2O) that varies spatially due to spatial differences of soils, land use, management and slope; and temporally because of changes in soil water content (Chaplot, 2005). The retention parameter, S, is represented by the relationship:

$$S = \left(\frac{1000}{CN}\right) - 10 \qquad (3)$$

Where CN is known as the curve number and relates the runoff potential to the combinations of land use and soil types. Higher curve numbers lead to increased runoff and are a function of hydrological soil group, land cover/land use, cultivation practice, and antecedent moisture conditions (Tetra Tech Inc., 2004). Table 2 shows the four hydrologic soil groups that have been classified by the NRCS from more than 4000 soils based on their minimum infiltration rate.

Table 2: Characteristics of hydrologic soil groups

Soil Group	Characteristics	Minimum Infiltration Capacity (in./hr)
A	Sandy, deep, well drained soils; deep loess; aggregated silty soils,	0.30-0.45
B	Sandy loams, shallow loess, moderately deep and moderately well drained soils	0.15-0.30
C	Clay loam soils, shallow sandy loams with a low permeability horizon impeding drainage (soils with a high clay content), soils low in organic content	0.05-0.15
D	Heavy clay soils with swelling potential (heavy plastic clays), water logged soils, certain saline soils, or shallow soils over an impermeable layer	0.00-0.05

Source: NRCS, 1972 as described in (Tetra Tech Inc., 2004)

The amount of moisture contained in the soil affects the volume and rate of runoff and it is evident from table 2 that runoff is much higher in soils of group D (clay soils) with low infiltration capacity as compared to the soils in group A (sandy soils) with higher infiltration capacity. Curve numbers can be adjusted from antecedent soil moisture conditions developed by NRCS as shown in table 3. Condition I reflects dry soils, Condition II represents the average condition and Condition III represents saturated soils under heavy rainfall, light rainfall and low temperatures.

Table 3: Curve Number adjustments from Antecedent Moisture Conditions I, II, III

CN for AMC II	CN for AMC I	CN for AMC III
100	100	100
95	87	99
90	78	98
85	70	97
80	63	94
75	57	91
70	51	87
65	45	83
60	40	79
55	35	75
50	31	70
45	27	65
40	23	60
35	19	55
30	15	50
25	12	45
20	9	39
15	7	33
10	4	26
5	2	17
0	0	0

Source: NRCS, 1972 as described in (Tetra Tech Inc., 2004)

Unlike the SCS curve number method, the Green and Ampt infiltration method assumes that the soil profile is uniform and hence antecedent moisture is uniformly distributed in the profile.

3.2.2 Erosion / sedimentation

Erosion and sediment yield caused by rainfall and runoff are predicted for each sub-basin with the Modified Universal Soil Loss Equation (MUSLE). Average annual gross erosion is predicted by USLE as a function of rainfall energy. The modified universal soil loss equation developed by Williams (1995) and described in Chaplot (2005) and Neitsch et al., (2002) is given by the following relationship:

$$Y = 11.8 \times (Q_{surf} \times q_{peak} \times Area_{hru})^{0.56} K_{USLE} \times C_{USLE} \times P_{USLE} \times LS_{USLE} \times R \qquad (4)$$

Where Y is the sediment yield on a given day (metric tons), Q_{surf} is volume of surface runoff (mm H_2O)/ha), q_{peak} is the peak runoff rate (m^3s^{-1}), $Area_{hru}$ is the area of HRU (ha), K_{USLE} is the USLE soil erodibility factor, C_{USLE} is the crop management factor, P_{USLE} is the erosion control practice factor, LS_{USLE} is the topographic factor, and R is the coarse fragment factor.

3.2.3 Nutrients

Nutrients, like nitrogen and phosphorus, occur naturally in water, soil and air in various forms. Both nitrogen and phosphorus exist in soil as organic and inorganic forms and the transformation of one form of the nutrient, i.e. N and P, into another is governed by the processes called the nitrogen and the phosphorus cycle respectively. SWAT tracks the movement and transformation of several forms of nitrogen and phosphorus in the watershed. A brief description of both nitrogen and phosphorus and their movement within the ecosystem is given as follows.

Nitrogen

Nitrogen occurs in soils as organic and mineral (inorganic) forms. The mineral component of nitrogen is either held in soil colloids or in solution while the organic component is associated with humus soil. Through nitrogen fixation, nitrogen gas is converted into inorganic nitrogen compounds to be used by plants, animals, and people. This conversion is accomplished through the nitrogen cycle, in which nitrogen as a gas is carried to the earth's surface in precipitation.

Nitrogen may also be added to the soil by bacteria fixation, fertilizer and animal manure application. Nitrogen is removed from the soil by denitrification as gaseous loss to the air, plant uptake, soil leaching and surface runoff. The major components of nitrogen cycles are shown in figure 9.

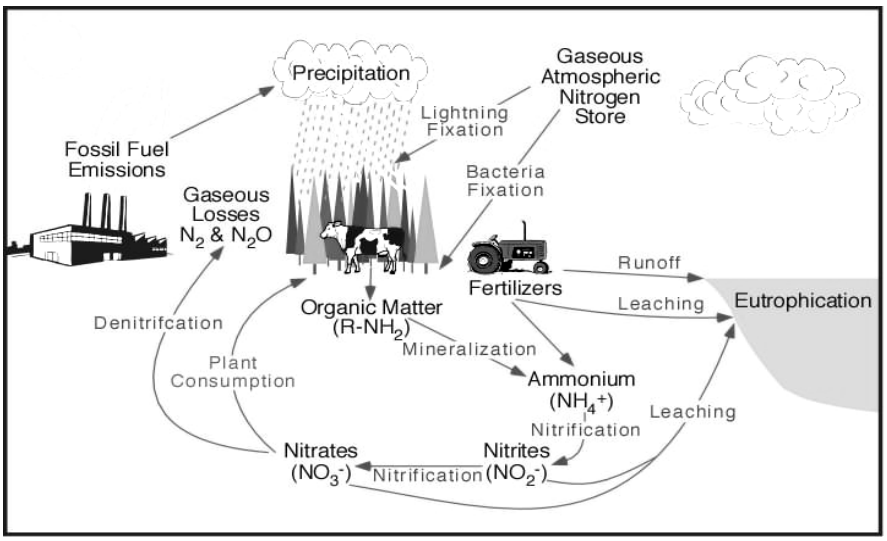

Figure 9: The Nitrogen Cycle (Pidwirny, 2005)

Pertaining to its ability of existing in several chemical valence states, nitrogen is highly reactive and mobile that prediction of its movement between different pools in soil is essential for management with environmental context (Neitsch et al., 2002). SWAT tracks five different pools of nitrogen in the soil: two pools are inorganic compounds (NH_4^+ and NO_3^-), while the other three pools are organic forms of nitrogen as indicated in figure 10.

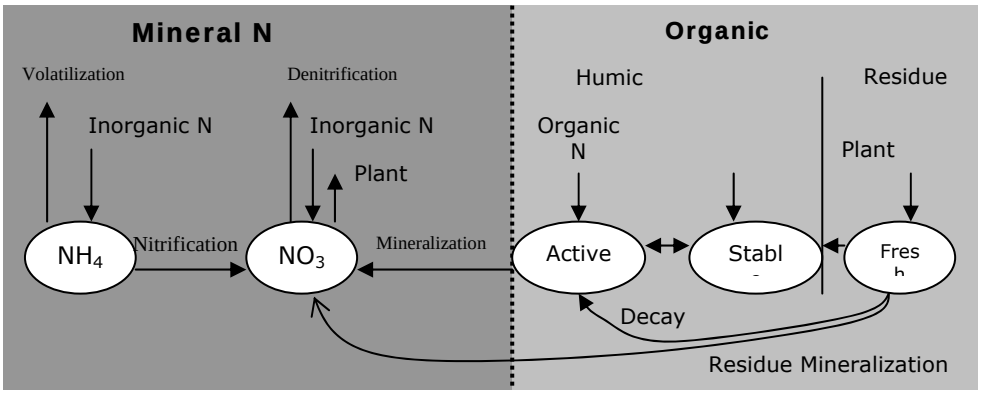

Figure 10: SWAT soil nitrogen pools and processes (after Neitsch et al., 2002)

Nitrate from the mineral N pool (inorganic form) may be transported from soil by surface runoff with water, lateral leaching or percolation to the underlying layer. Howarth (1996) discussed that some of the fertilizer applied to agricultural land leaches directly to groundwater and surface waters depending upon soil characteristics, climate, and crop type and the leaching rate varies from 3 percent to f80 percent of the applied fertilizer. In SWAT, the concentration of nitrate in the mobile water fraction is calculated as:

$$Conc_{NO3,mobile} = \frac{NO_{3ly} * \exp \frac{-w_{mobile}}{(1-\theta_e) * SAT_{ly}}}{w_{mobile}} \quad (5)$$

Where $conc_{NO3,\ mobile}$ is the concentration of nitrate in the mobile water for a given layer (kg N/mm H_2O), $NO3_{ly}$ is the amount of nitrate in the layer (kg N/ha), w_{mobile} is the amount of mobile water in the layer (mm H_2O), θ_ε is the fraction of porosity from which anions are excluded, and SAT_{ly} is the saturated water content of the soil layer (mm H_2O). Anions are excluded due to repulsion from particle surfaces or negative adsorption and replaced by cations (Neitsch et al., 2002). The amount of mobile water in the layer is the amount lost by surface runoff, lateral flow or percolation, given by:

$$w_{mobile} = Q_{surf} + Q_{lat,ly} + w_{perc,ly}, \quad \text{for top 10mm} \quad (6)$$

$$w_{mobile} = Q_{lat,ly} + w_{perc,ly}, \quad \text{for lower soil layers} \quad (7)$$

Where w_{mobile} is the amount of mobile water in the layer (mm H_2O), Q_{surf} is the surface runoff generated on a given day (mm H_2O), $Q_{lat,ly}$ is the water discharged from the layer by lateral flow (mm H_2O), and $w_{perc,ly}$ is the amount of water percolating to the underlying soil layer on a given day (mm H_2O). In SWAT the top 10mm soil layer is allowed to interact with the surface runoff and transport nutrients from this layer, which is calculated as:

$$NO_{3surf} = \beta_{NO_3} * conc_{NO_3,mobile} * Q_{surf} \quad (8)$$

Where NO_{3surf} is the nitrate removed by surface runoff (kg N/ha), β_{NO3} is the nitrate percolation coefficient, $conc_{NO3,mobile}$ is the concentration of nitrate in the mobile water for the top 10 mm of soil (mm H_2O), and Q_{surf} is the surface runoff generated on a given day (mm H_2O).

Organic form of nitrogen may also be transported by surface runoff attached to soil particles. Organic N is associated with the sediment loading from HRU and any change in sediment loading will be reflected in the loading of this form of nitrogen. SWAT estimates the amount of organic nitrogen loading in surface runoff using a loading function developed by McElroy et al (1976) and modified by Williams and Hann (1978), both cited in Neitsch et al. (2002):

$$orgN_{surf} = 0.001 * conc_{orgN} * \frac{sed}{area_{hru}} * \in_{N:sed} \quad (9)$$

Where $orgN_{surf}$ is the amount of organic nitrogen transported to the main channel in surface runoff (kg N/ha), $conc_{orgN}$ is the concentration of organic nitrogen in the top 10mm (g N/metric ton soil), sed is the sediment yield on a given day (metric tons), $area_{hru}$ is the HRU area (ha), and $\varepsilon_{N:sed}$ is the nitrogen enrichment ratio. A larger proportion of clay sized particles in a sediment load to the main channel leads to the occurrence of a greater proportion or concentration of organic N resulting in the enrichment of organic N. The enrichment ratio is defined as the ratio of concentration of organic nitrogen transported with the sediment to the concentration in the soil surface layer (Neitsch et al., 2002). The concentration of organic nitrogen in the soil surface layer and the nitrogen enrichment ratio are, respectively, calculated in SWAT as:

$$conc_{orgN} = 100 . \frac{(orgN_{frsh,surf} + orgN_{sta,surf} + orgN_{act,surf})}{\rho_b . depth_{surf}} \quad (10)$$

$$\in_{Nsed} = 0.78 * (conc_{sed,surq})^{-0.2468} \quad (11)$$

Where $orgN_{frsh,surf}$ is nitrogen in the fresh organic pool in the top 10mm (kg N/ha), $orgN_{sta,surf}$ is nitrogen in the stable organic pool (kg N/ha), $orgN_{act,surf}$ is nitrogen in the active organic pool in the top 10 mm (kg N/ha), ρ_b is the bulk density of the first soil layer (Mg/m^3), $depth_{surf}$ is the depth of the soil surface layer (10 mm), and $conc_{sed,surq}$ is the concentration of the sediment in surface runoff (Mg sed/m^3 H_2O). The concentration of sediment in surface runoff is calculated as:

$$Conc_{sed,surq} = \frac{sed}{10 * area_{hru} * Q_{surf}} \tag{12}$$

Where sed is the sediment yield on a given day (metric ton), $area_{hru}$ is the HRU area (ha), and Q_{surf} is the amount of surface runoff on a given day (mm H_2O).

Phosphorus

Phosphorus is another key nutrient in soils that occurs in organic form associated with humus, insoluble mineral form and plant available phosphorus in solution. Phosphorus may be added to the soil through the application of fertilizer, manure or residue while removal from the soil occurs by plant uptake and erosion. Phosphorus movement in landscapes is closely associated with soil erosion because P is attached to solid soil materials. Phosphorus reacts readily with positively charged iron (Fe), aluminum (Al), and calcium (Ca) ions to form relatively insoluble substances that precipitate out of solution facilitating the build up of phosphorus near the soil surface that is readily available for transport in runoff (Neitsch et al., 2002). SWAT monitors six pools of phosphorus of which three are organic forms and the remaining three are inorganic forms as shown in figure 11.

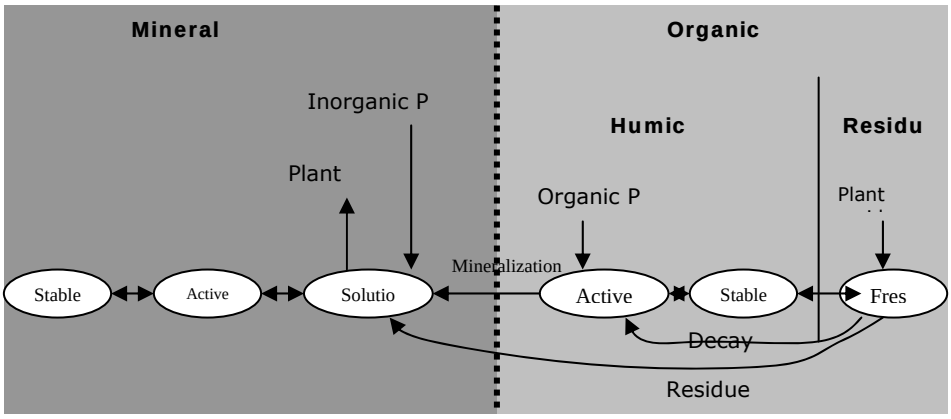

Figure 11: SWAT soil phosphorus pools and processes (after Neitsch et al., 2002)

Phosphorus movement in the soil is primarily by diffusion. Surface runoff only partially interacts with the solution P stored in the top 10 mm of soil due to its low mobility. The amount of solution P transported in surface runoff as calculated by SWAT is:

$$P_{surf} = \frac{P_{solution,surf} * Q_{surf}}{\rho_b * depth_{surf} * k_{d,surf}}, \quad (13)$$

Where P_{surf} is the amount of soluble phosphorus lost in surface runoff (kg P/ha), $P_{solution,surf}$ is the amount of phosphorus in solution in the top 10mm (kg P/ha), Q_{surf} is the amount of surface runoff on a given day (mm H2O), ρ_b is the bulk density of the top 10 mm (Mg/m^3), $depth_{surf}$ is the depth of the surface layer (10 mm), and $k_{d,surf}$ is the phosphorus soil partitioning coefficient (m^3/Mg).

The amount of organic and mineral P that is transported attached to the sediment particles is calculated by SWAT using the loading function developed by McElroy et al (1976) and modified by Williams and Hann (1978), cited in Neitsch et al. (2002):

$$sedP_{surf} = 0.001 * conc_{sedP} * \frac{sed}{area_{hru}} * \in_{p:sed}, \quad (14)$$

Where $sedP_{surf}$ is the amount of phosphorus transported with sediment to the main channel in surface runoff (kg P/ha), $conc_{sedP}$ is the concentration of phosphorus attached to sediment in the top 10 mm (g P/metric ton soil) calculated using equation 12, sed is the sediment yield on a given day (metric tons), $area_{hru}$ is the HRU area (ha), and $\varepsilon_{p:sed}$ is the phosphorus enrichment ratio calculated using the same equation (11) for N above.

The concentration of phosphorus attached to sediment in the soil surface layer, $conc_{sedP}$, is computed as:

$$conc_{sedP} = 100 * \frac{(\min P_{act,surf} + \min P_{sta,surf} + orgP_{hum,surf} + orgP_{frsh,surf})}{\rho_b * depth_{surf}}, \quad (15)$$

Where $minP_{act,surf}$ is the amount of phosphorus in the active mineral pool in the top 10 mm (kg P/ha), $minP_{sta,surf}$ is the amount of phosphorus in the stable mineral pool in the top 10 mm (kg P/ha), $orgP_{hum,surf}$ is the amount of phosphorus in the humic organic pool in the top 10 mm (kg P/ha), $orgP_{frsh,surf}$ is the amount of phosphorus in the fresh organic pool in the top 10 mm (kg P/ha), ρ_b is the bulk density of the first soil layer (Mg/m^3), and $depth_{surf}$ is the depth of the soil surface layer (10 mm).

3.3 Model Inputs

SWAT requires specific information about watershed characteristics such as topography, land use/cover, soil types, weather, and management practices. The model uses a two-level discretization scheme; first basin and sub-basin delineation is performed based on topographic information, followed by further discretization into HRUs using land use and soil type considerations in order to represent heterogeneous watershed properties. Climate inputs are required since they control water balance that drives all the processes simulated in the watershed. Management practice of a watershed is needed because it greatly influences the nutrient emission to surface and subsurface waters.

3.3.1 Watershed Delineation

The subdivision of a watershed into discrete sub-watershed areas enables the modelling process to represent, as much as possible, the heterogeneity of the watershed. SWAT works on a sub-basin basis and the interface delineates the watershed into such sub-basins or sub-watersheds based on topographic information. The delineation of Rönne watershed was performed based on topographic information of the watershed area obtained from the Swedish Land Survey Department in the form of digital elevation model (DEM) with X-Y resolution of 50 metres using the procedure described in ArcView interface for SWAT2000 user's guide (DiLuzio et al., 2002). The total watershed area of Rönne basin is about 190000 ha (VASTRA, 2005) while that of total delineated area by the algorithm was 180500 ha. The discrepancy between the actual and the delineated area could be due to the fact that some modifications were made to the original DEM data to allow complete delineation of the entire watershed area which otherwise disallows the delineation of the most downstream or the main outlet area. The reason was that some cells had zero values near the outlet which persisted even after the pre-processing of the DEM to remove sinks by the interface. The application of one time 3x3 low pass filter in ArcGIS to the DEM was able to solve the problem by averaging the cells having 0 values with surrounding pixels.

Since the size of a sub-basin is believed to affect the assumption of homogeneity or spatial variability of conditions (Jha et al., 2004), definition of the watershed, sub-basin boundaries and streams was made by selecting a threshold area or the minimum draining area to define streams. Configuration of larger number of sub-basins causes

an increased amount of input data preparation and subsequent analysis though detailed spatial variations with in the watershed could be simulated. On the other hand, too small number of sub-watersheds could affect the simulation result by ignoring spatial variability that lumps watershed conditions together. Therefore, a threshold level and hence the number of sub-basins for a given watershed area needs to be selected carefully, and (Jha et al., 2004) suggested a maximum of 35 and 11 sub-basins for predicting nitrate and mineral P losses respectively for watersheds approximately comparable in size to the present study area.

For the present sub-watershed configuration, a threshold area of 4500 ha was used to generate a reasonable number of sub-basins with additional sub-basins delineated to obtain output at 8 water quality monitoring stations for nutrient, especially nitrate, loading simulation. Although the interface could automatically generate sub-basin outlets, manual edition was made in order to include additional outlets at key locations of water quality monitoring sites and remove some of the automatically generated ones so that the desired size of sub-basins would be delineated. Subsequently, selection of the main watershed outlet was made to delineate the main watershed area and 21 sub-basins. Before the task of delineating the main watershed area and sub-basins has been completed, locations of water quality monitoring sites were added into the configuration for simulation and comparison at a later stage of the modelling process. The digital elevation model and the subsequently delineated watershed, sub-basins, streams, selected sub-basin outlet locations and SWAT weather generator locations are shown in figure 12. The statistical summary of topographic data (elevation report) for the delineated watershed and sub-basins is given in appendix 1.

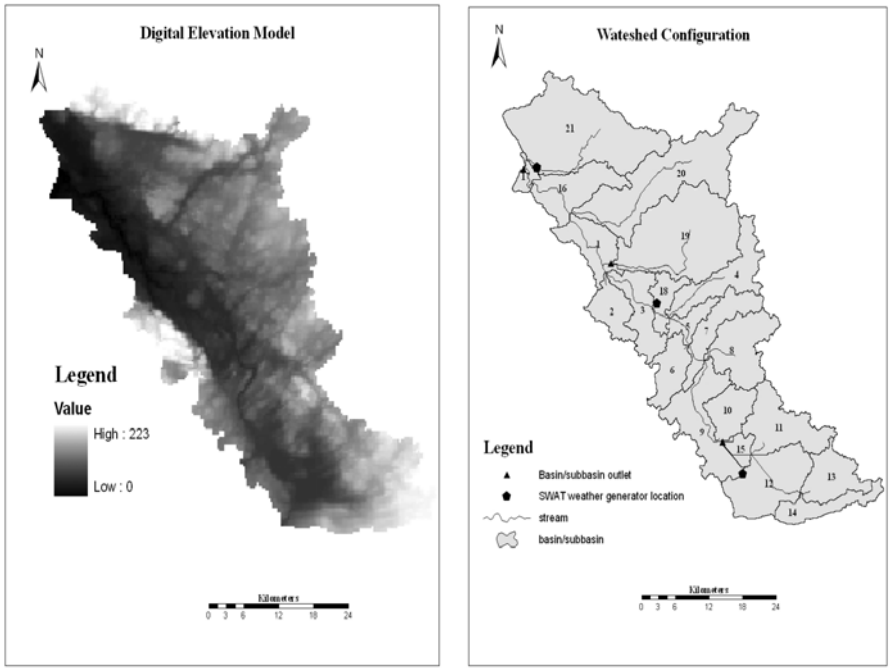

Figure 12: DEM (left) and the delineated watershed/sub-basins (right).

3.3.2 Land Use and Soil Characterization

Land cover and soil, among others, greatly influence the hydrological properties of a watershed that they are two of the five main input datasets required by SWAT to help describe a sub-basin or HRU. Once watershed topographic parameters have been computed for each sub-basin, the interface uses land cover and soils data to generate multiple hydrologic response units (HRUs) within each sub-basin by GIS overlay process to assign soil parameters and SCS curve numbers. HRUs are lumped land areas within the sub-watershed that are comprised of distinctive land cover, soil, and management combinations (Neitsch et al., 2002). Such subdivision of sub-basins into HRUs enables the SWAT model to reflect the spatial variations of the hydrologic conditions for different land cover and soil distributions within the sub-basins

Land Cover Data

Land cover information for the study area was downloaded from The European Topic Centre on Terrestrial Environment (CORINE land cover database) owned by European Environment Agency (EEA,

2000). CORINE is a European Commission programme established with the aim of compiling and coordinating information on the state of the environment and ensuring that the information are consistent and data are compatible within the Member States of the Community as well as at international level. The land cover project is part of the CORINE programme intended to provide consistent geographical information on the land cover of the Member States of the European Community, which is believed to be essential for the management of the environment and natural resources. This data was chosen to be used as model input because:

- It is freely available for the Member States of the Community that downloading from Internet was easily possible through the GIS Centre of Lund University.
- It provides the user with an option in his/her best interest of spatial details to be used by the model.
- It is compatible with most GIS softwares so that data extraction and preparation for the SWAT model input is easier.
- It has been compiled with the aim of providing reliable land cover information to help decision-making process for environmental and natural resources management.

All features in CORINE land cover map were extracted from an interpretation of satellite images, namely Landsat and Spot images, aided by topographic maps obtained from different countries of the Member States and the map has been compiled with various scales and different level of details (3 levels), the surface area of the smallest unit mapped being 25 hectares, to enable decision-makers to identify, analyse and monitor land use management systems. Level 1 is highly generalised information indicating the major categories of land cover while level 2 with 15 land use classes has been compiled for use on scales of 1:500000 and 1:1000000. The third level, with 44 land use classes, is a larger scale map describing more detailed features that can be used for projects on a scale of 1:100000. The different level of details and nomenclature for CORINE land cover is shown in table 4.

A simplified form of the level 2 CORINE land use nomenclature was established for this study. The modification was important because firstly the SWAT input land use data should, more or less, match with that of the SWAT land cover database used by the model, secondly since the data was extracted from a large area covering southern part of Sweden, some of the classes were absent or are not common in the study area, and thirdly minor land use classes would be eliminated at a later stage during definition of the land use/soil distribution or HRUs depending on the selected threshold.

Table 4: CORINE land cover nomenclature

Level 1	Level 2	Level 3
1. Artificial surfaces	1.1. Urban fabric	1.1.1. Continuous urban fabric 1.1.2. Discontinuous urban fabric
	1.2. Industrial, commercial and transport units	1.2.1. Industrial or commercial units 1.2.2. Road and rail networks and associated land 1.2.3. Port areas 1.2.4. Airports
	1.3. Mine, dump and construction sites	1.3.1. Mineral extraction sites 1.3.2. Dump sites 1.3.3. Construction sites
	1.4. Artificial non-agricultural vegetated areas	1.4.1. Green urban areas 1.4.2. Sport and leisure facilities
2. Agricultural areas	2.1. Arable land	2.1.1. Non-irrigated arable land 2.1.2. Permanently irrigated land 2.1.3. Rice fields
	2.2. Permanent crops	2.2.1. Vineyards 2.2.2. Fruit trees and berry plantations 2.2.3. Olive groves
	2.3. Pastures	2.3.1. Pastures
	2.4. Heterogeneous agricultural areas	2.4.1. Annual crops associated with permanent crops 2.4.2. Complex cultivation 2.4.3. Land principally occupied by agriculture, with significant areas of natural vegetation 2.4.4. Agro-forestry areas
3. Forests and semi-natural areas	3.1. Forests	3.1.1. Broad-leaved forest 3.1.2. Coniferous forest 3.1.3. Mixed forest
	3.2. Shrub and/or herbaceous vegetation association	3.2.1. Natural grassland 3.2.2. Moors and heathlands 3.2.3. Sclerophyllous vegetation 3.2.4. Transitional woodland shrub
	3.3. Open space with little or no vegetation	3.3.1. Beaches, dunes and sand plains 3.3.2. Bare rock 3.3.3. Sparsely vegetated areas 3.3.4. Burnt areas 3.3.5. Glaciers and perpetual snow
4. Wetlands	4.1. Inland wetlands	4.1.1. Inland marshes 4.1.2. Peatbogs
	4.2 Coastal wetlands	4.2.1. Salt marshes 4.2.2. Salines 4.2.3. Intertidal flats
5. Water bodies	5.1. Inland waters	5.1.1. Water courses 5.1.2. Water bodies
	5.2. Marine waters	5.2.1. Coastal lagoons 5.2.2. Estuaries 5.2.3. Sea and ocean

Source: (EEA, 2000)

Therefore, ArcView GIS was used to reclassify and prepare the data input as required, considering 7 main classes: urban, agriculture, pasture, shrub/range, forest, wetland and water. All the artificial surfaces with built-up areas were summarised as and designated by urban residential-medium to low density; arable land and permanent crops as agricultural land-generic; pastures as pastures; forests as forest-mixed; shrub and/or herbaceous vegetation association and open space with little or no vegetation as range-brush; inland wetlands as wetlands-mixed and inland waters as water. Some of the classes not mentioned here from the level 2 of the CORINE land cover nomenclature are missing in the study area.

In the subsequent discussion, reference will be made to the generalised land use map of the study area with 7 classes, characterised mainly by forest (46.7%) and arable land (26.6%), both comprising about 73% of the total watershed area. The rest of land use types occupy a very less proportion: pasture (8.5%), brush and rangeland (8.6%), urban areas (4.5%), water bodies (3.5%) and wetlands (only 1.6%), as quantified by SWAT in ArcView interface. Figure 13 shows the reclassified land use map while the proportion of each land use and their description is given in table 5.

Table 5: Proportion and description of the newly classified land use types from CORINE

No.	Area (%)	Swat Land Class	Description
1	4.53	URML	Residential-Med/Low Density
2	26.57	AGRL	Agricultural Land-Generic
3	8.49	PAST	Pasture
4	46.74	FRST	Forest-Mixed
5	8.57	RNGB	Range-Brush
6	1.62	WETL	Wetlands-Mixed
7	3.48	WATR	Water
Total	100.00		

After preparing the land use data to be used in the model, a semi-supervised satellite image classification of the study area, based on few GPS points (only about 26), was performed to have a quick overview and compare with CORINE land cover map. This was to show if semi-supervised maps, with less cost and time, could be used as model input in the absence of carefully prepared maps. The size of ground truth sample was limited to only few points because of time constraint to carryout intensive fieldwork. Landsat 7 ETM+ image from the year 1999 was downloaded from the University of Maryland database, which is freely available for the geo-information community (University of Maryland, 2005). Since it has already been geo-referenced and rectified, pre-processing was not required that only

the area of interest (AOI) was clipped and the coordinate system was re-projected to fit that of the Swedish National Grid. ERDAS software was used for the classification procedure. The resulting classified land use map was somewhat similar to that of the CORINE data whereby most classes were noticeably identified. The satellite image and the resulting classified land use map are attached in appendix 2.

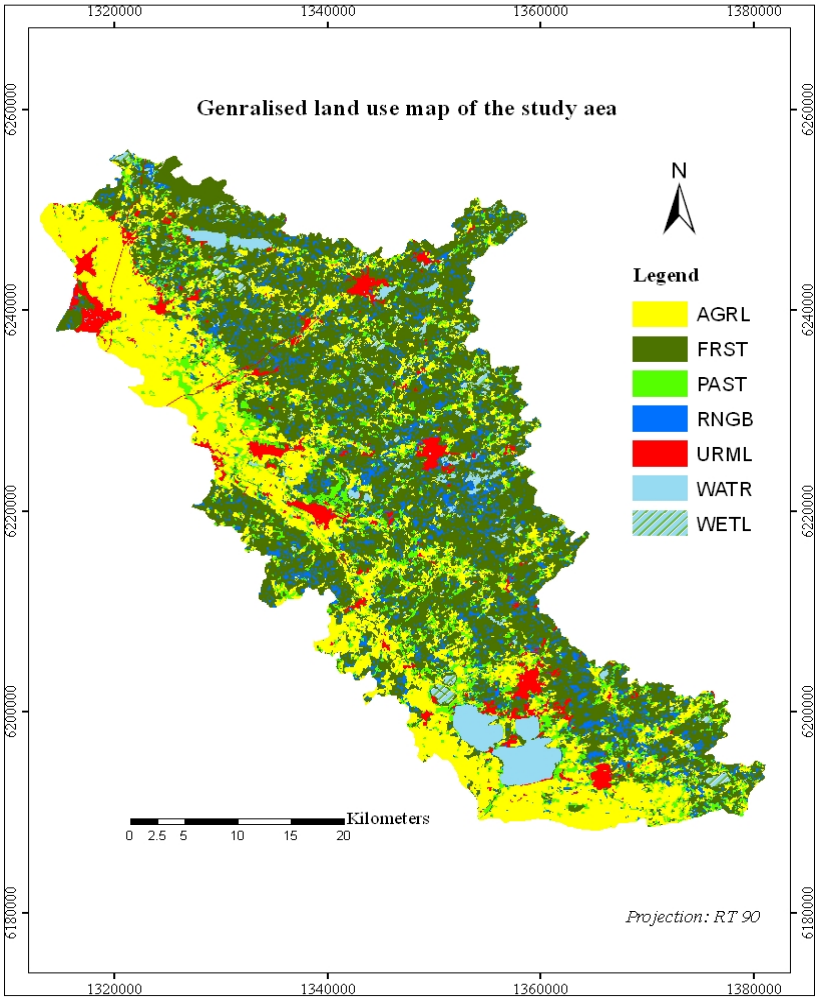

Figure 13: Generalised land use map used as model input

Soil Data

Soil data was obtained from the Geological Survey of Sweden (Sveriges Geologiska Undersökning-SGU) through the GIS Centre of the Lund University. The data was in digital format having different level of soil classes that enables modification according to the interest

of users and the SWAT input data format. The more generalised soil class comprises 14 soil types including water covered areas.

The soils are geologically young and belong to the Quaternary period, formed during the ice age (glacial soils) and subsequent processes (post glacial soils). The glacial soils contain tills/moraines and glacial sediments. Tills are the most common soil types and are mainly clayey (Gustafsson, 1992), but some times also occur as sandy or sandy-silty in some places (Norling and Wikman, 1990) that led to classify them as clay in this study. The glacial sediments consist of coarse grained sediments- sand, gravel, and cobbles, and of fine grained sediments-clay and silt, which were classified as sand for the purpose of this study. The postglacial soils can be divided into re-worked and redeposited types and organic soils. After the glaciation period, there was a strong increase in the production of organic material mixed into the fine grained material to form organic fine grained soils, e.g. organic clay and peat bogs sporadically distributed within the watershed area. A typical soil profile, in the study area, from top to bottom is characterised by organic soil; postglacial and glacial clay; stones, gravel and sand; till and finally bed rock. The most common soil types of the study area, based on the 14 classes of SGU, are indicated in table 6.

Table 6: Common soil types of the Rönne watershed

No.	Soil type	Description	Generalised as
1	Organic Soil	Organic clay, peat bogs, swamps	Organic Soil
2	Clay	Glacial and postglacial clay	Clay
3	Silt	Glacial and postglacial silt	Silt
4	Sand	Glacial sand	Sand
5	Gravel	Glacial gravel	Sand
6	Stone block	Postglacial stone blocks	Rock
7	Fluvioglacial sediment	Mostly glacial sand with gravels	Sand
8	Moraine clay	Moraine; clay >15%	Clay
9	Moraine	Moraine; range from fine grained clays to coarse grained gravel tills.	Clay
10	Thin earth layer	Missing in the study area	-
11	Rock outcrop	bedrock	Rock
12	Landfills	Land fill, natural or artificial	Sand
13	Others	Unknown under water	Water
12	Water	Water bodies	Water

A simplified classification of soils is important in order to avoid further complications and extensive data input demand by the model during soil parameterization. Furthermore, some of the soil types, listed in table 7, are insignificant in terms of proportion in the study area while some of them are closely similar to others in terms of soil physical

properties, such as texture, which is the most important soil parameter in SWAT. Therefore, the original SGU soil data was reclassified into 6 soil types, mainly based on texture. Water was considered as one class in this study because of its unique effect on nutrient movement as the different types of soils do. Figure 14 shows the reclassified soil types while the proportion of each soil type and their description is given in table 7.

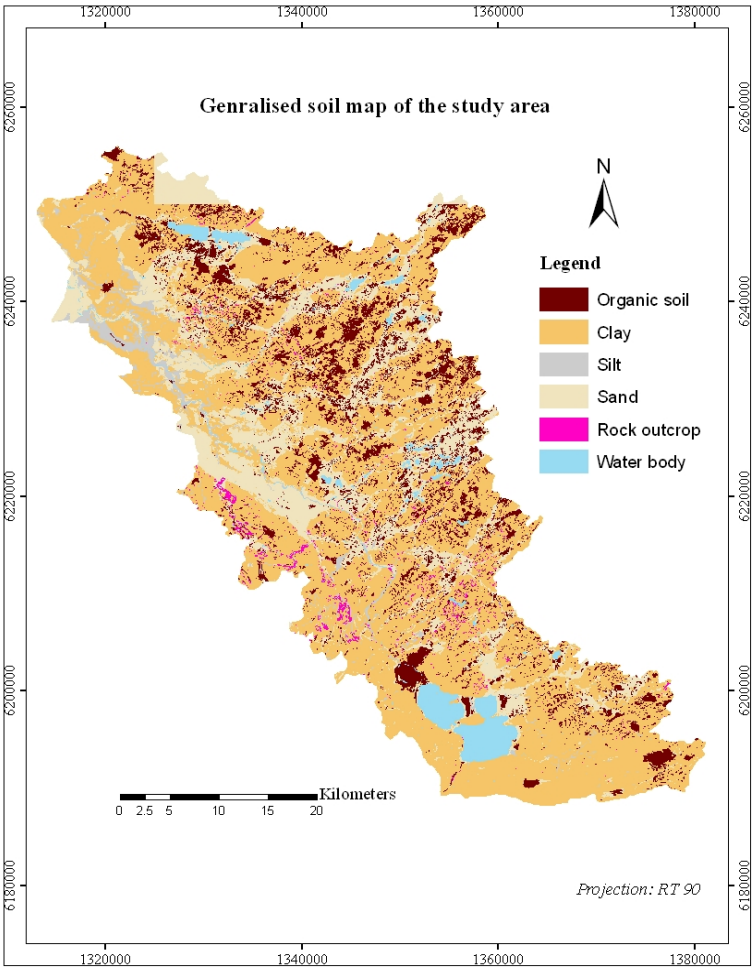

Figure 14: A generalised map of soil types used as model input

According to the newly classified soil types, most of the watershed area is occupied by clayey soils of mainly moraine type (60%) while sandy soils occupy about 20% of the watershed. Organic soils cover about 12%, sporadically distributed over the whole area and a less proportion of the area is made up of the rest minor classes.

Table 7: Proportion and description of the newly classified soil types used as model input

No.	Area (%)	Swat Soil Class	Description
1	11.96	Organic soil	Peat bogs, swamps
2	59.93	Clay	Mainly moraine clay
3	2.65	Silt	Glacial and fluvial origin
4	20.60	Sand	Glacial sand, moraine, sediments and gravels
5	1.11	Rock	Rock outcrops and boulders
7	3.75	Water	Water bodies and soils under water
Total	100.00		

Various physical properties such as bulk density, available water capacity, saturated hydraulic conductivity, k-factor and organic carbon content are required by SWAT in order to carry out the modelling process. The characterisation of the different soil parameters was made by making use of a graphic computer programme known as soil water characteristics (Saxton, 2003). The programme estimates moist bulk density, available water capacity and saturated hydraulic conductivity using the soil texture and organic matter.

The soil types were used to derive soil textures while the soil organic matter content for Sweden, given in literature, was used. The median value of organic matter content for most Swedish arable land, including organic soils, is 4.1%; and for soils with less organic matter, the median value is 3.9% (Eriksson et al., 1997). USLE soil erodibility factor or k-factor was collected from various sources for different soil textures. Table 8 shows the different soil parameters used for the current study. Finally, since there were no available data on different soil layers, only a single soil layer was assumed over the entire watershed for the modelling purpose. In fact, there is no well developed soil layer in the study area since the soils are of much younger age associated to the glacial and post-glacial geological activities.

Table 8: Estimated soil parameters for main soil textures

Soil Type	Soil group	BD g/cm^3	AWC mm/mm	K_s mm/hr	CBN %	Clay %	Silt %	Sand %	k-factor
Organic Soil	C	1.1	0.15	33.3	4.1	40	30	30	0.15
Clay	C	1.10	0.14	26.50	4.10	50	25	25	0.15
Silt	B	1.20	0.24	298.50	3.90	15	15	70	0.42
Sand	A	1.33	0.15	145.30	3.90	10	10	80	0.10
Rock	D	2.30	0.04	0	0.10	7	3	90	0.01
Water	D	1.10	0.15	13	-	-	-	-	0.05

Note: some parameters were changed during calibration

GIS overlay of the land cover and soil information

Once the land cover and soil datasets have been prepared in a format that can be used by the SWAT2000 model, overlay of the two GIS layers was made by the interface to generate detailed land use and soil distribution for the entire watershed area and the 21 sub-basins defined during the watershed delineation stage. This was followed by definition of multiple HRUs within sub-basins. Since the user is allowed to decide whether or not to use HRUs in the modelling application, in this study it was preferred to proceed with the definition of HRUs so that spatial differences of hydrological processes within sub-basins would be reflected. If multiple HRUs are ignored, the interface will use the dominant land use and soil characteristics for the entire watershed which ignores the spatial variability by lumping together the sub-basin processes. A threshold level can also be determined by the user to model multiple HRUs by eliminating minor land uses in each sub-basin. Subsequently, land uses that cover a percentage of the sub-basin area less than the threshold level are eliminated and the area of the land use is reapportioned so that 100 percent of the land area in the sub-basin is included in the simulation. It has been suggested (DiLuzio et al., 2002) that a 20 percent land use threshold and a 10 percent soil threshold are adequate for most modelling applications. However, a 10 percent land use and a 10 percent soil threshold were used in this simulation so that more detailed discretization of sub-basins would be possible to represent more heterogeneous watershed attributes, which resulted in 127 unique land use/soil combinations or HRUs.

3.3.3 Climate Data

Weather data is another key set of input data required by SWAT2000 model. The daily climate inputs include precipitation, maximum and minimum temperature, solar radiation, wind speed and relative humidity. They can be supplied to the model in two general ways: (1) from daily observations of historical data recorded at specific stations near or within the watershed as a user-specified file or (2) monthly statistical data on local weather conditions after which SWAT weather generator simulates the time series data. A combined approach was used in this study where daily precipitation data for the period 1981-2000, obtained from the Swedish Meteorological and Hydrological Institute, was supplied to the model while the rest were left to be generated internally within SWAT for the 40 year period after providing monthly weather statistical data at three locations within the watershed. Most of the statistical datasets were obtained from internet sources, such as WORLDCLIM (Hijmans et al., 2004) for

average minimum and maximum monthly air temperatures (interpolated climate data from local stations) and the European Solar Radiation Atlas (PVCoolBuild, 2003; Scharmer and Greif, 2000) for average daily solar radiation in month.

3.3.4 Management Practice

SWAT2000 predicts the effect of different management practices on a watershed based on the user specified type of management practice input data within the watershed. Some of such input management data include amount and timing of fertilizer, beginning and ending of plant growing season, irrigation, tillage and grazing operations. Since observed data were not available, several assumptions were made on management practices and atmospheric nitrogen deposition based on available literature on the study area and other similar watersheds.

Grizzetti et al. (2003) suggested a concentration of nitrogen in precipitation of 1 ppm for a watershed located in Southern Finland, more or less, similar to the climatic condition in Southern Sweden. The mean annual precipitation for 10 years record at Malmö airport, Southern Sweden, was about 700 mm corresponding to an average annual nitrogen deposition of 700 kg N per km^2 or 7.0 kg N/ha. This value is in agreement with data given in literature, for example an average atmospheric deposition of 5-10 kg/ha/year was estimated for this specific region in the European Communities report (European Commission, 2002). However, the great proportion of nitrogen input in agricultural soils of Sweden comes from inorganic and organic fertilizers while other sources such as atmospheric deposition and biological fixation constitute a lesser amount. The total N "pressure" on agricultural soils of Southern Sweden originating from various sources, given in the European Communities report (European Commission, 2002), is summarized in table 9.

Table 9: Total nitrogen pressure from agriculture, air deposition and biological fixation for South Sweden

Source	N input- kg/ha/yr, range for Sweden	% contribution
Mineral fertilizer	49-61	48.9
Livestock manure	40-50	39.5
Atmospheric deposition	5-10	9.5
Biological fixation	2-3	2.2
Total	100 - 125	100

Source: European Communities report (European Commission, 2002).

Since detailed information was not available on arable crop land distribution, a generic agricultural land (AGRL- equivalent to 4 character SWAT2000 land cover code) and pasture grass (PAST) were chosen to represent agricultural lands. The crops were assumed to grow in the middle of May after fertilizer operation. A fertilizer application was considered for generic agricultural land use and pasture grasses. The maximum fertilizer input rate (125 kg N/ha/yr) was used as model input because the study area is a relatively densely populated and intensely cultivated area due to the presence of favourable fertile soil for agriculture compared to other parts of Sweden. The timing of tillage operation, crop growth, fertilizer application, and harvesting is given in table 10, based on assumption in Southern Finland as mentioned by Grizzetti et al. (2003) and Southern Sweden (European Commission, 2002; VASTRA, 2005).

Table 10: Schedule of tillage, planting, fertilizer application and harvesting for model runs

Year	Date	Operation
1	May 3	Chisel Plow
	May 14	Fertilization (125 kg/ha)
	May 17	Plant Spring Barley
	August 26	Harvest Barley
	September 01	Tillage operation

3.4 Model Calibration and Validation

Once the main SWAT input data are ready to successfully run the model, a number of output files are generated after simulation. The main SWAT output files generated during simulation are indicated in appendix 3. Model calibration and validation was performed based on these output files. Model calibration is a procedure to adjust or fine-tune model parameters so as to represent observed conditions in as much as possible, while validation is testing of the calibrated model results with independent data set without any further adjustment (Neitsch et al., 2002) at different spatial and temporal scales. In water quality modelling, a three step calibration procedure has been suggested by Neitsch et al. (2002), first starting with water balance and stream flow followed by sediment and nutrient consecutively. However, the availability of observed data set determines whether to perform the calibration on all or part of the procedures suggested.

After the initial configuration of SWAT, model calibration and validation were performed on selected locations within the watershed. Stream flow was calibrated first, then nutrient concentration was considered, while sediment was excluded since measured time series

data was not available for calibration purposes. Stream flow was calibrated on annual and monthly average time steps at two locations while NO_3 was calibrated at one location on monthly basis. The calibration and validation locations are indicated in figure 15.

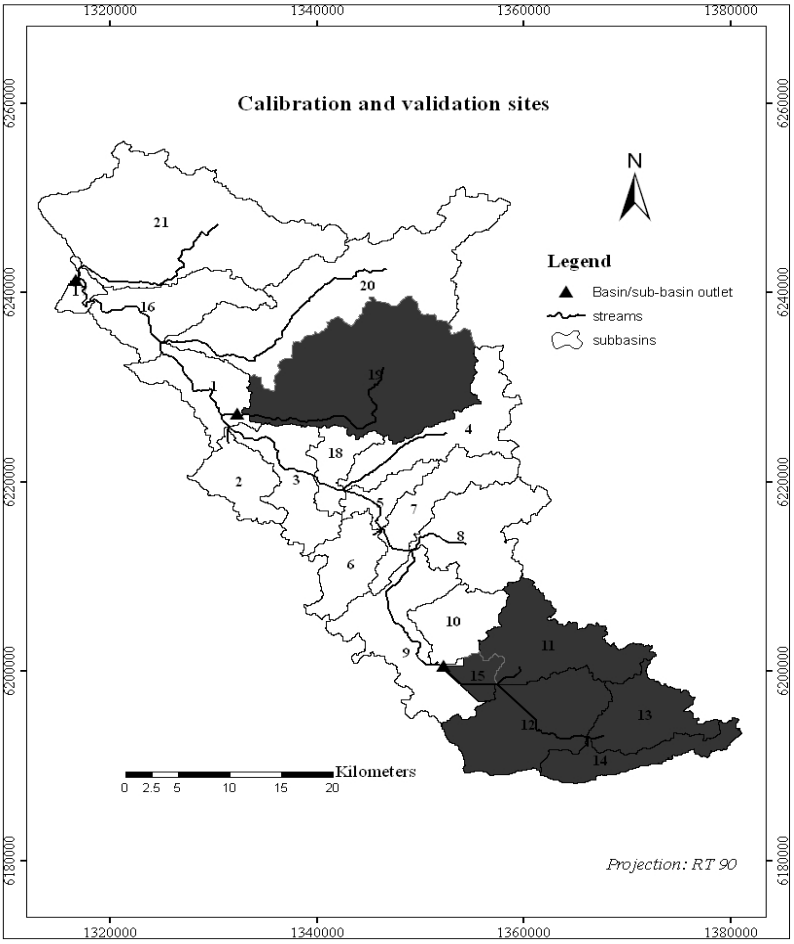

Figure 15: Model calibration and validation sites (the numbers represent designated sub-basins)

The model was first calibrated for flow at the outlet of sub-basin 15 (here after referred as Ringsjön) representing five upstream sub-basins, then extended to the outlet of sub-basin 19 (here after referred as Klippan basin) located further downstream of Ringsjön. The already calibrated parameters, on upstream site, were kept unchanged during calibration of the downstream site. The two calibration sites were selected primarily due to the availability of measured daily stream flow data for a period of 20 years (Jan 1981–

Dec 2000) which was acquired from the VASTRA project of the Swedish Meteorological and Hydrological Institute. Regarding temporal scale, long-term average annual stream flow was calibrated first and after ensuring the satisfactory performance of the model for the yearly time step, monthly stream flow was calibrated for each location again without changing the values already adjusted during yearly time step. The period from 1981-1990 was used for calibration while validation was performed during the period 1991-2000.

Key hydrologic parameters such as SCS runoff curve number for moisture condition II, available water capacity of the soil layer, ground water revaporation coefficient and the threshold depth of the shallow aquifer before evaporation can occur were considered for flow calibration. The parameters were chosen based on previous researched works on SWAT calibration (Jha et al., 2004; Santhi et al., 2005) and suggestions given by Neitsch et al. (2002). The values or ranges of selected sensitive parameters before and after calibration are given in table 11. These parameters were believed to govern the partitioning of precipitation between surface and ground water flows (Tetra Tech Inc., 2004) and hence are sensitive components.

Table 11: Initial and final values of SWAT calibration parameters for stream flow

Variable name	Description	Recommended Range	Initial value	Calibrated value
CN2	SCS runoff curve number for antecedent moisture condition II	±8	-	-8
SOL_AWC	Soil available water capacity	±0.05	-	0.05
ESCO	Soil evaporation compensation factor	0-1	0	0.1
GW_REVAP	Ground water revaporation coefficient	0.02-0.2	0.02	0.1
CH_K2	Effective hydraulic conductivity in main channel alluvium	-0.01-150	0	50
ALPHA_BF	Baseflow alpha factor	0-1	0.048	1
SMFMX	Maximum melt rate for snow during the year	0-10	4.5	10
SMFMN	Minimum melt rate for snow during the year	0-10	4.5	10
SLOPE	Average slope steepness	0-0.6	0.025	0.2
SLSUBBSN	Average slope length	10-150	91.5	50

After confirming that the stream flow calibration results were reasonably convincing and adequate, attempt was made to calibrate nutrient (NO_3-N) concentration in stream water at the outlet of Klippan sub-basin. The calibration process involved comparing water quality time series simulation output to available water quality (NO_3 concentration) observation data, obtained from VASTRA. The available measured water quality data was not a continuous daily record of monitoring data for nutrients. Only about 12 sample data per year (grab samples) were available from the monitoring station for the period Jan 1997-Dec 2000. Additional data on point sources pollution were not also available, whereas it was learned that two such sources from municipal waste water treatment plants exist within the sub-basin. Since point source emissions are believed to greatly influence the concentration of nutrients in streams, comparison of observed and simulated water quality without considering point sources would not be as such meaningful. However, since point sources are expected to be controlled emissions, i.e. constant daily discharges, significant temporal variation is not expected on the model output as a result of the missing data.

Nevertheless, in order to best reproduce the observed data by simulation, fine tuning of pollutant loading and in-stream water quality parameters within recommended range were performed. The SWAT parameters related to nutrient loading such as nitrogen percolation factors (NPERCO), initial organic nutrient concentration in surface soil layer (SOL_ORGN), initial mineral nutrient concentration in the soil layer (SOL_NO_3), biological mixing efficiency (BIOMIX) and fertilizer application rates were adjusted. Although the model was found to be very sensitive to NPERCO and slightly to fertilizer application rates, the remaining parameters were not responsive and much improvement could not be achieved in reproducing the observed scenarios. Validation was not performed for nitrate because of the unavailability of sufficient data as well as due to the poor results obtained on calibration.

Finally, calibration and validation results were evaluated by comparing time series model outputs to measured data. Graphical comparison and statistical analysis methods were used to test for goodness-of-fit. It has been suggested that graphical comparison is enormously useful for judging the results of model calibration as it provides a good visual control over time-variable plots of measured against simulated flow and the model's capability to reproduce storm hydrographs, base flow recession, time lags, and other related factors, which are often overlooked by statistical comparisons (Tetra Tech Inc., 2004). Several statistical methods including the Nash-

Sutcliffe prediction efficiency (E_{NS}), coefficient of determination (r^2), mean error, root mean square error, root mean square, etc, are available for evaluating the model predictions versus the observed values.

The first two statistical approaches, E_{NS} and r^2, were used here since they are the most commonly used methods in several other similar studies. The Nash-Sutcliffe prediction efficiency (Nash and Sutcliffe, 1970) indicates how well the plot of observed against simulated value fits the 1:1 line, which is computed as:

$$E_{NS} = 1 - \frac{\sum_{i=1}^{n} (Q_i - Q'_i)^2}{\sum_{i=1}^{n} (Q_i - Q)^2}, \qquad (16)$$

Where $\boldsymbol{Q}$ represents the mean measured values, $\boldsymbol{Q_i}$ and $\boldsymbol{Q'_i}$ are the measured and simulated values respectively and $\boldsymbol{n}$ is the number of values computed. A value for E_{NS} equal to 0 or less indicates a poor performance of the model while a value of 1 indicates a perfect fit between the measured and model predicted values.

The second statistical goodness-of-fit used was the coefficient of correlation (r^2), which explains how much the measured values are explained by the simulated values indicating the strength of correlation between the two variables. If the value for r^2 is zero or close to zero, the model's prediction capability is poor and unacceptable. If the values are close or equal to one, the model prediction is considered perfect.

4 Results

4.1 Calibration

The calibration of SWAT2000 was performed at two locations for stream flow and at one location for nitrate within the Rönne River basin, which were selected based on the availability of measured time-series data. The period 1981-1990 and 1997-2000 were used as calibration years for stream flow and NO_3-N respectively, and the period from 1991-2000 was used for validation of stream flow. The analysis of the results involved comparison of model calculated annual and monthly average flow with measured flows and comparison of simulated NO_3 concentration against observed stream water quality on a monthly basis.

4.1.1 Stream Flow

Since nitrate fluxes are strongly influenced by water fluxes (Pohlert et al., 2005) parameters controlling water balances were calibrated first. The daily measured stream flow data was averaged to annual daily average for the period 1981 to 1990 and the results were compared to the model calculated annual values after running the model for the same time period of 10 years. In computing the efficiency of the model, the first year of the model run was used as model priming and hence the results of the first year were excluded from all computations so that the influence of the initial conditions such as soil water content would be minimised (Grizzetti et al., 2003). The annual calibration results on both locations are graphically shown in figure 16 and 17 and the statistical summaries are given in table 12.

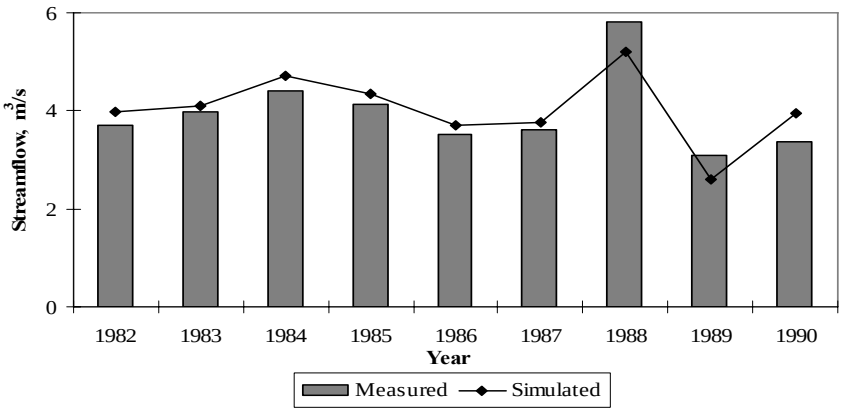

Figure 16: Comparison between measured and simulated annual average stream flow at Ringsjön for model calibration

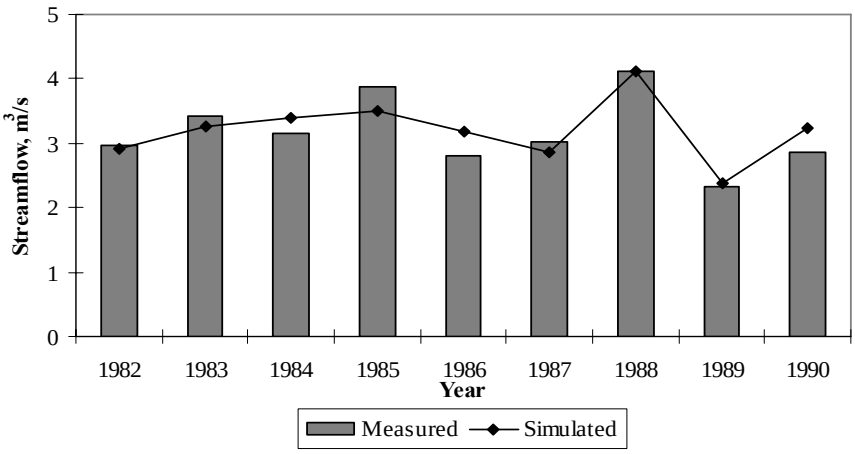

Figure 17: Comparison between measured and simulated annual average stream flow at Klippan for model calibration

Figures 16 and 17 show the time-series comparison of model simulated and observed annual stream flows for the Ringsjön and Klippan locations respectively over the 10 year (1981-1990) calibration period. Generally, the model performed satisfactorily by closely tracking the observed annual flows. There was a good match between the observed and simulated flow values both in magnitude and temporal variations at both locations although some peak flows tended to be under predicted and some of the low flows were over predicted.

Table 12: Summary of calibration results for annual stream flows (m^3/s) at Ringsjön and Klippan

Location	Year	1982	1983	1984	1985	1986	1987	1988	1989	1990	Performance E_{NS}	Performance R^2
Ringsjön	Meas.	3.70	3.97	4.41	4.13	3.51	3.61	5.80	3.10	3.37	0.75	0.77
	Sim.	3.98	4.10	4.72	4.34	3.70	3.77	5.20	2.60	3.96		
Klippan	Meas.	2.98	3.42	3.16	3.87	2.82	3.01	4.11	2.33	2.86	0.78	0.79
	Sim.	2.91	3.28	3.38	3.49	3.19	2.85	4.12	2.39	3.23		

The satisfactory performance of the model was further confirmed by the high statistical Nash-Sutcliffe prediction efficiency (E_{NS}) values of 0.75 and 0.78 for Ringsjön and Klippan, respectively, showing a good agreement between the measured and model calculated values. Regression analysis between the two variables also resulted in high values of r^2 (0.77 for Ringsjön and 0.79 for Klippan), indicating again the strong correlation between measured and simulated annual flows at both locations. The mean of the measured and simulated flows, averaged over the entire period of the model run, were also closely related, with values of 3.96 against 4.04 at Ringsjön and 3.17 against 3.20 at Klippan. They were not significantly different at 95% level of confidence at both locations. The t-calculated values (0.67 and 0.37) were less than the t-critical value (2.31) at Ringsjön and Klippan, respectively (p= 0.26 and 0.36, 1-tail). Overall, the model tended to slightly over predict the measured annual flows at both locations.

Since the results obtained for the calibration of flows during the annual time step were found to be satisfactory as confirmed graphically and statistically, the next step was completed by calibrating for the monthly time step on both locations. The results are summarised in table 14 and graphically represented by figures 18 through 21. The detailed monthly measured and the corresponding SWAT predicted data are given in appendix A-4.1.

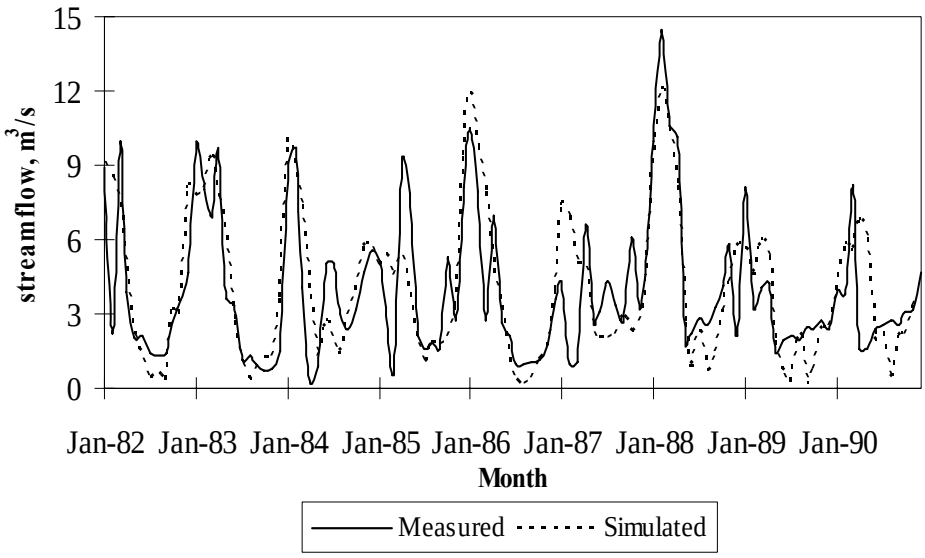

Figure 18: Measured against simulated monthly average stream flow at Ringsjön for model calibration

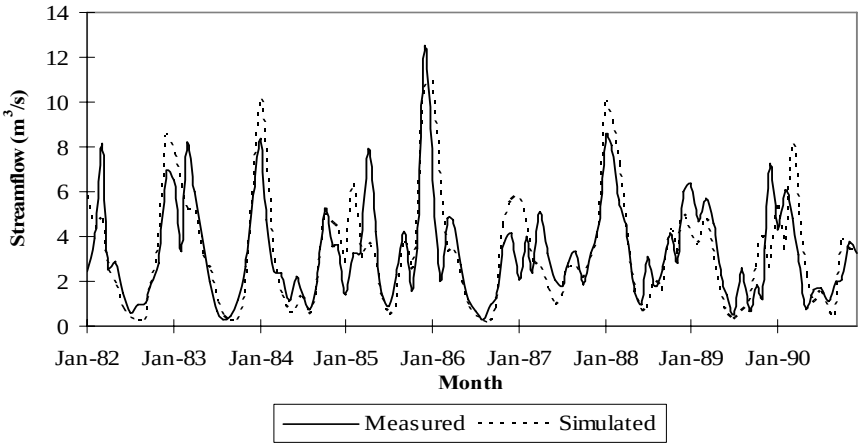

Figure 19: Measured against simulated monthly average stream flow at Klippan for model calibration

The predicted result was able to satisfactorily reproduce and closely follow the corresponding observed monthly flows. The peak flows and temporal variations were well tracked by the model, with some over- and under-predictions, at both locations, the comparisons of which are shown in figure 18 and 19. The monthly predicted values were also regressed and plotted against the measured flows and figure 20 and 21 provide their distribution along the 1:1 line for the two calibration sites. The results showed that the measured and calculated flows were uniformly distributed along the 1:1 line without any significant outlier, indicating a good level of agreement between the observed and simulated values.

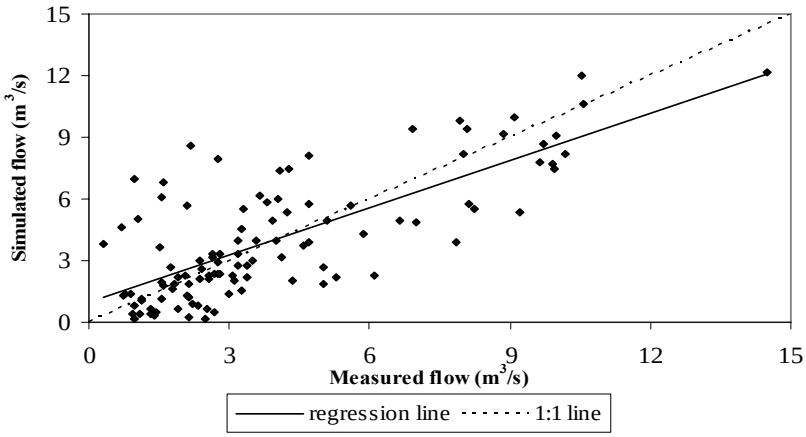

Figure 20: Regression and 1:1 line fit plot of observed versus simulated monthly stream flow at Ringsjön, Jan 1981 to Dec 1990

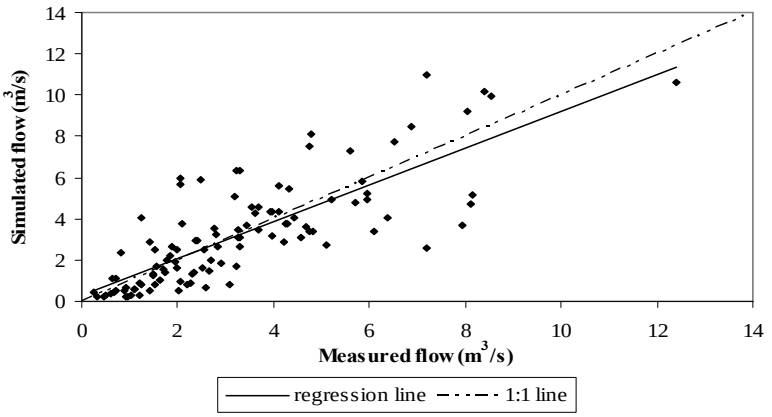

Figure 21: Regression and 1:1 line fit plot of observed versus simulated monthly stream flow at Klippan, Jan 1981 to Dec 1990

Further statistical test for goodness-of-fit, provided in table 13, showed that the Nash-Sutcliffe values of 0.35 for Ringsjön and 0.53 for Klippan were lower as compared to the annual performance of the model, but still considered satisfactory and acceptable. The results of regression analysis, with r^2 values of 0.52 and 0.63 for Ringsjön and Klippan respectively, also showed a good correlation between predicted and measured values for the indicated time step. The mean of the measured and simulated flows, averaged over the entire simulation period, were also closely related with values of 3.96 against 4.05 at Ringsjön and 3.28 against 3.21 at Klippan. They were not significantly different at 95% level of confidence (α=0.05) at both locations since the t-calculated values of 0.40 and 0.44 were less than the t-critical values of 1.66 at both Ringsjön and Klippan, respectively (p= 0.34 and 0.33, 1-tail test). Overall, the model tended to slightly under predict high flows.

Table 13: Summary of calibration results for monthly stream flows at Ringsjön and Klippan

Location	No. of observations	Mean		1-tail t-test, α =0.05			Performance	
		Meas.	Sim.	t-calc	t-crit	p-value	E_{NS}	R^2
Ringsjön	108	3.96	4.05	0.40	1.66	0.34	0.35*	0.52*
Klippan	108	3.28	3.21	0.44	1.66	0.33	0.53	0.63

**The initial calibration results (E_{NS}=0.50 and r^2=0.57) were affected during calibration on downstream sub-basin at Klippan.*

4.1.2 Nutrient (NO_3-N)

Nitrate was considered for nutrient calibration based on the availability of observed stream water quality data. Since the measured nitrate data was concentration in g/l, the SWAT simulated NO_3 load (kg N) was converted to concentration by dividing with the simulated flow out of the reach at Klippan using a concentration–load relationship as:

$$C = \frac{F}{Q} \qquad (17)$$

Where, C is nitrate concentration in stream water (g/l), F is nitrate transported with water out of reach during the simulation time step (e.g. kg N/month) and Q is stream flow out of reach during the time step (e.g. m^3/month). The predicted results were compared with the observed data on a monthly basis for the period of Jan 1997-Dec 2000. The result showed a poor agreement between the observed and measured values by consistently under predicting the observed data, as expected, regardless of efforts made to verify all the key processes related to nutrients. Nutrient parameter adjusted was only the nitrogen percolation factor (NPERCO) which was found to be the only sensitive variable while others such as SOL-NO_3, BIOMIX, RSDCO and fertilizer application rates remained unresponsive or very slightly if any at all. However, the simulated concentration tends to reflect the seasonality of the measured data. Figure 22 shows comparison of observed and simulated monthly NO_3 concentration at the outlet of Klippan sub-basin. The detailed monthly measured and the corresponding SWAT predicted data are given in appendix A-4.3.

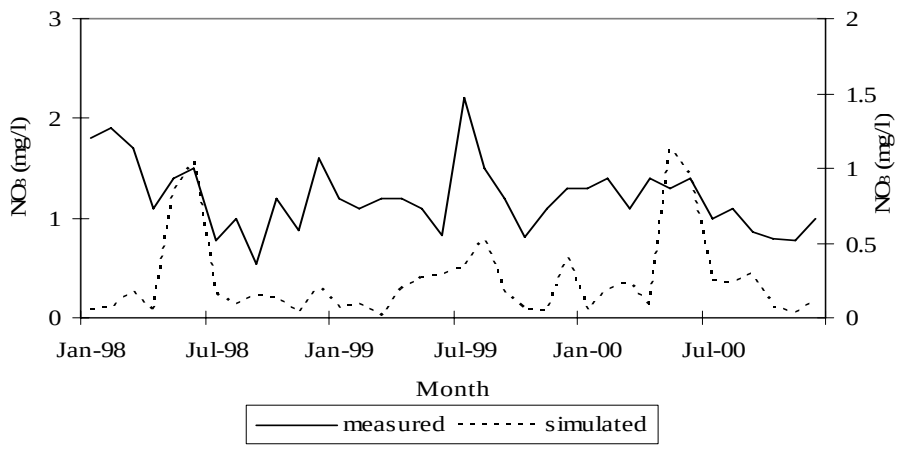

Figure 22: Observed against simulated NO_3-N concentration in stream water at Klippan

The poor agreement between the observed and simulated values was an expected result due to uncertainties associated to the data used for model input as already mentioned in section 3.4. First, the measured data were not continuous daily records of monitoring data (only grab samples) and the chemical analysis procedures were not verified. Second, information on point sources and other possible nitrogen sources in the immediate vicinity of the sampling point that would significantly influence stream water quality were not verified and considered. Lastly, the information on management practice used in the model was purely based on literatures covering wide region and hence might not represent watershed-specific scenarios. All these factors would definitely hinder a thorough calibration process.

Uncertainties that could be associated with NO_3 concentration were further examined by comparing the time-series measured concentration data against measured flow to observe the seasonal behaviour if variation in NO_3 concentration can be explained by seasonal stream flow variation as indicated in figure 23. The result showed a negative correlation with very low coefficient of determination (r^2 = 0.02) suggesting that variation in NO_3 concentration could not be explained by the variation in discharge.

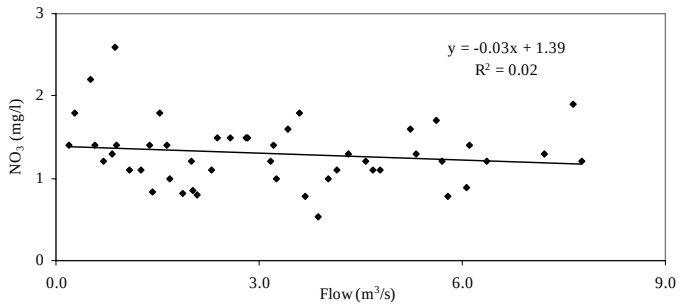

Figure 23: Measured monthly average flow (m^3/s) vs. NO_3 concentration (mg/l) at the outlet of Klippan sub-basin, Jan 1997-Dec2000.

4.2 Validation

Model validation was performed for the annual and monthly stream flows on the same sub-basins used for calibration. The sites were selected based on the availability of measured flow data as well as to avoid the influence that might be resulted due to differences in watershed attributes, such as land cover and soil properties, when a different location is used. In the validation process, the model run was made with input parameters set during the calibration process without any change and the results were compared with observed

data from different time period (1991-2000) used for calibration. The validation result for the annual flow at Ringsjön is graphically shown in figure 24, while figure 25 represents the result for Klippan. Both results showed that SWAT was able to accurately predict the annual stream flow variations, matching well with measured data.

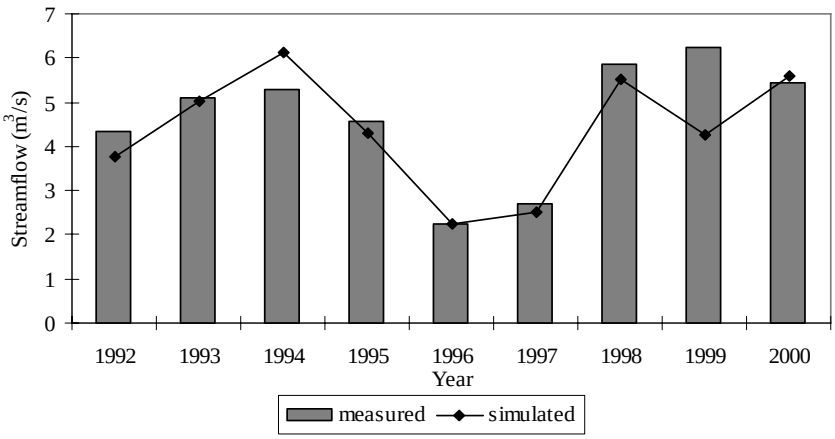

Figure 24: Simulated versus measured annual stream flow at Ringsjön for model validation

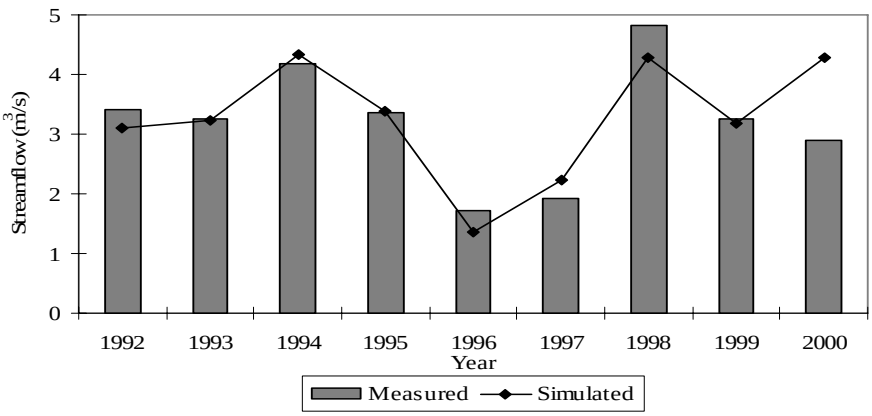

Figure 25: Simulated versus measured annual stream flow at Klippan for model validation

The goodness-of-fit was further evaluated statistically as summarised in table 14. The relatively high Nash-Sutcliffe (0.65 & 0.66) and r^2 (0.75 & 0.70) values confirmed a further agreement and strong correlation between the observed and simulated flows that the model prediction capability of annual flow was adequate, although these results were slightly weaker compared to the calibration results.

Table 14: Summary of validation results for annual flows (m^3/s)

Location	Year	1992	1993	1994	1995	1996	1997	1998	1999	1000	Performance E_{NS}	Performance R^2
Ringsjön	Meas.	4.32	5.11	5.28	4.57	2.25	2.71	5.84	6.24	5.44	0.65	0.75
	Sim.	3.77	5.02	6.13	4.30	2.26	2.53	5.53	4.26	5.59		
Klippan	Meas.	3.41	3.25	4.17	3.36	1.73	1.93	4.81	3.27	2.89	0.66	0.70
	Sim.	3.11	3.23	4.34	3.37	1.37	2.24	4.29	3.17	4.28		

The mean of the measured and simulated flows, averaged over the entire period of simulation, were also closely related (4.64 against 4.38 at Ringsjön and 3.20 against 3.27 at Klippan). They were not significantly different at 95% level of confidence at both locations (Klippan t-calculated=0.34, t-critical=1.86, p=0.37; and Ringsjön: t-calculated=1.06, t-critical=1.86, p=0.16). However, the model had the tendency to slightly under predict during high flow years as observed in 1999 at Ringsjön and in 1998 at Klippan.

The time series observed and simulated monthly flows of the Ringsjön and Klippan validation sites for the validation period of Jan 1991-Dec 2000 were compared and graphically represented as shown in figures 26 and 27. The times of peak flows and seasonal variations matched consistently throughout the simulation period although the model slightly under predicted at Ringsjön and over predicted at Klippan some peak events of stream flow.

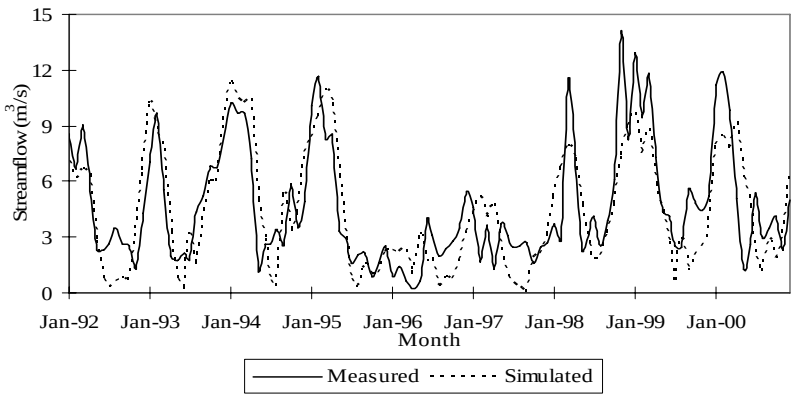

Figure 26: Simulated against measured monthly flow at Ringsjön for model validation

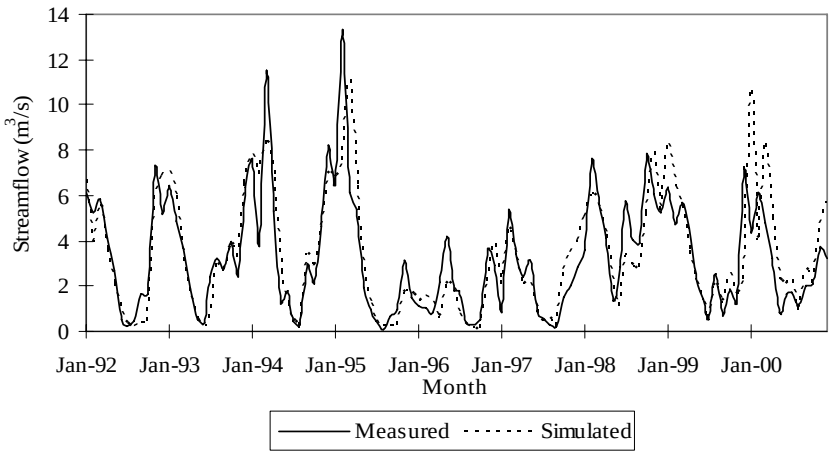

Figure 27: Simulated against measured monthly flow at Klippan for model validation

The distribution of predicted monthly stream flows with respect to the measured values were regressed and the 1:1 line fit were plotted as indicated in figures 28 and 29. The results showed that the values were distributed uniformly about the 1:1 line. However, few outliers were observed for the high flows attributed to the under prediction and over prediction of the model at Ringsjön and Klippan, respectively, during the high flow years.

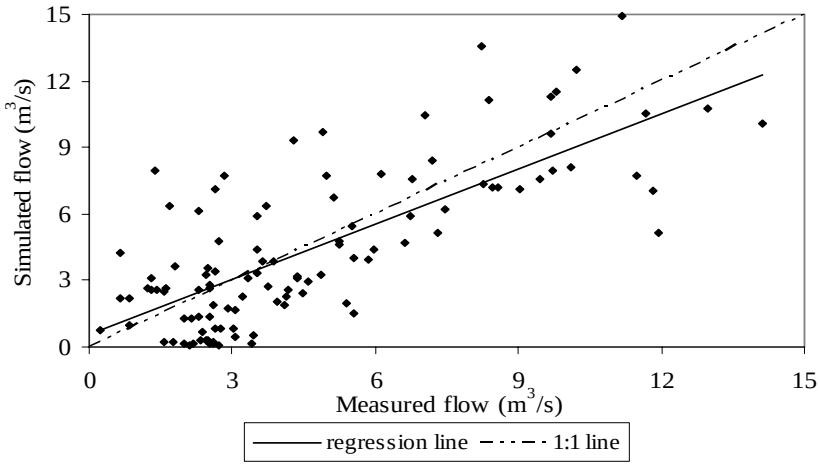

Figure 28: Regression and 1:1 line fit plot of observed versus simulated monthly flow for model validation at Ringsjön

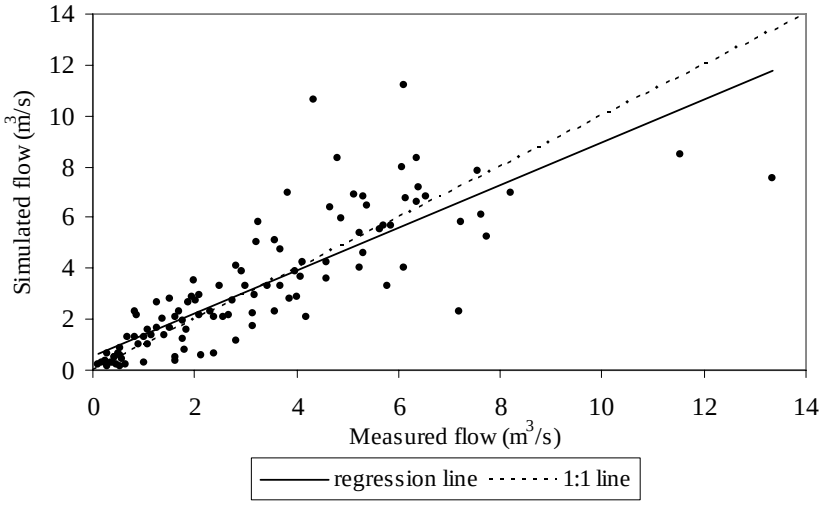

Figure 29: Regression and 1:1 line fit plot of observed versus simulated monthly flow for model validation at Klippan

Although the monthly statistical validation result for Ringsjön had lower E_{NS} and r^2 values (0.43 and 0.56), it was still considered satisfactory and the prediction at Klippan was even higher (E_{NS}=0.62 and r^2=0.66). The mean of the measured and calculated values at both locations were not significantly different at 95% confidence level since the p-values (0.13 and 0.34) were greater than the α-value of 0.05. Table 15 summarises the statistical results of simulated against measured values on both validation sites. The detailed monthly measured and the corresponding SWAT predicted data are given in appendix A-4.2.

Table 15: Summary of validation results for monthly flows

Location	Number of observations	Mean		1-tail t-test, α =0.05			Performance	
		Meas.	Sim.	t-calc	t-crit	p-value	E_{NS}	R^2
Ringsjön	108	4.65	4.39	1.14	1.66	0.13	0.43	0.56
Klippan	108	3.21	3.27	0.41	1.66	0.34	0.62	0.66

4.3 Modelling Nutrient (NO_3) Loading of the Rönne Basin

The calibration and validation results for annual and monthly stream flows performed on two sub-basins of the Rönne River Basin were found to be satisfactory indicating that SWAT can be used to model the hydrologic conditions of the entire basin. But the results obtained for nitrate simulation were not so good probably due to the limitations and uncertainties associated with nutrient data, which otherwise would have significantly influenced the model output. Given such prevailing circumstances, therefore, it would be unwise conclusion to associate the poor results due to less performance of SWAT in modelling nitrate within the study area. In fact, the model was successfully used by Grizzetti et al. (2003) to model diffuse emission of nitrogen from the Vantaanjoki watershed in Southern Finland, which has similar watershed properties to the current study area.

Therefore, SWAT was applied to assess and show the relative importance of different watershed properties, particularly land uses, in terms of nitrate emission and loading to stream water within the study area based on the calibrated hydrological conditions, but without undermining the possible systematic errors associated in estimating nitrate concentrations in stream water due to the already mentioned uncertainties. Temporal variations due to missing point sources data, for example, are insignificant since they are controlled emissions although the estimated outputs are expected to be far under predicted. Finally, it should be noted that in the subsequent nutrient modelling approach, while the estimated outputs could serve the purpose of analysing relationships between nitrate diffuse emission and different watershed characteristics in relative terms, a further verification of nutrient data is still required in order to use it for management planning in its strictest sense.

4.3.1 NO_3 Diffuse Emission Predicting Variables

It was believed that NO_3 diffuse emission by surface and subsurface runoff is influenced by watershed properties such as land use/cover types and hence by the area occupied by such land uses. In this research, it was further hypothesised that agricultural land would contribute most to nitrate emission. In order to relate the most predictors of NO_3 loading by surface water, a comparison involving the proportion of different land covers and the respective nutrient loading in different sub-basins was performed. The correlation

involved the percentage areas of different land uses in 20 out of the 21 sub-basins and the respective nitrate loading by surface runoff. Sub-basin 17 (the main outlet) was considered as spatial outlier and hence excluded from the analysis because the sub-basin area was too small while it contributed high amount of nitrate load compared to the whole area due to high proportion of urban land within the sub-basin. Table 16 presents the correlation matrix that relates the NO_3 loading (NSURQ) and the proportion of land use types from which NO_3 was transported by surface runoff during a 10-year simulation period (1991-2000). The cover proportions of different land uses in different sub-basins and the amount of NO_3 transported from the respective sub-basins in surface runoff is given in appendix 5.

Table 16: Correlation coefficient between NO_3 loading (NSURQ - kg N/ha) and %age field

Landuse	AGRL	FRST	PAST	RANGB	WATR	URBN	**NSURQ**
AGRL	1						
FRST		1					
PAST			1				
RANGB				1			
WATR					1		
URBN						1	
NSURQ	**0.9**	**-0.7**	**0.3**	**-0.5**	**-0.3**	**-0.1**	**1**

Note: the comparison involved only nitrate transported in surface runoff (NSURQ).

Nitrate loading was found to be strongly positively correlated to the area occupied by agricultural land (r=0.9) and negatively correlated to the area occupied by forest (r=-0.7). The remaining land use types were less correlated to nitrate leaching compared to agricultural land and forest areas.

Nitrate loading was further compared by regressing against the proportion of agricultural and forest covers as shown in figures 30 and 30. The result revealed that agricultural land was strong predictor of nitrate emission (r^2 = 0.79) compared to other land use areas suggesting 79% of variation in nitrate loading can be explained by variation in agricultural land proportion within the sub-basin. The positive slope (figure 29) indicates that nitrate loading increases with the increase of agricultural land area and if the watershed was completely free from any agricultural activity, the average nitrate loading from the watershed would be limited to a background value of 0.4 kg N/ha.

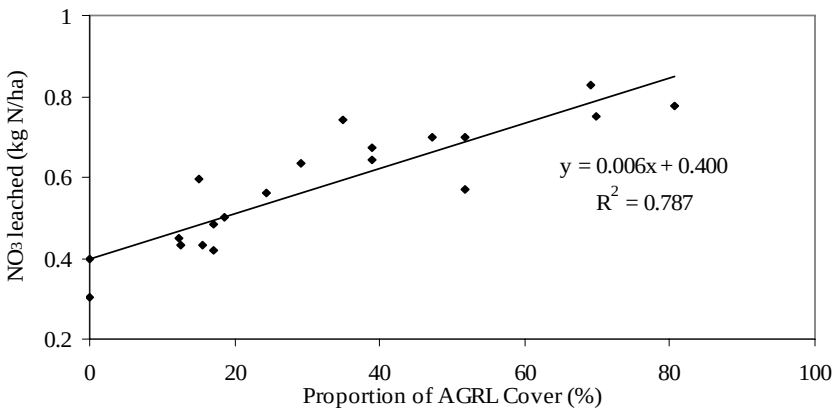

Figure 30: Comparison of proportion of agricultural land cover (%) against annual average NO_3 loading in surface water (kg N/ha), 1991-2000.

Variation in proportion of forest cover also explains variation in nitrate leaching by 43% (r^2 = 0.43) with negative slope as indicated in figure 31. The negative slope indicates the decrease in nitrate leaching with increasing forest area. If the watershed is devoid of any forest, the average nitrate loading from the watershed would increase by nearly 100% from the background value of 0.4 kg N/ha to 0.8 kg N/ha.

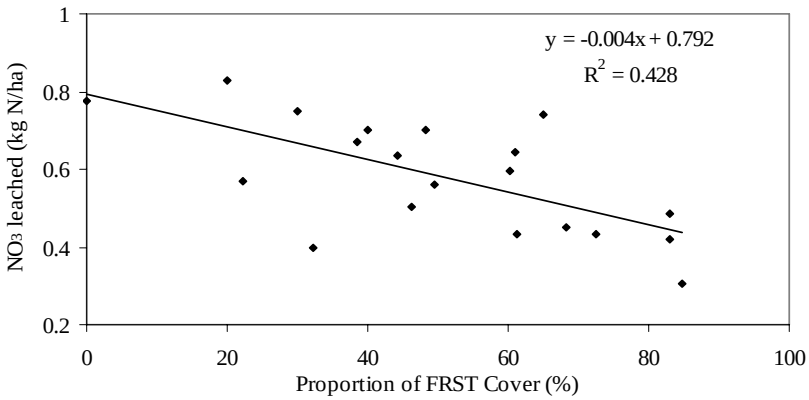

Figure 31: Comparison of proportion of forest cover (%) against annual average NO_3 loading in surface water (kg N/ha), 1991-2000.

4.3.2 NO_3 Concentration in Stream Water: Spatial Variation

The concentration of NO_3 in stream water was also simulated at different sub-basin outlets along the Rönne River to examine the longitudinal spatial trend of stream water quality. The results were plotted from the most upstream sub-basin reach to the most downstream including the main outlet. The simulated results included average annual values for the period 1991-2000 and 1998 wet year as shown in table 17 and figure 32. Longitudinal plot along the main channel at the outlet of the various sub-basins indicated NO_3 concentrations increase downstream. The difference between the 10-year average and the 1998 concentrations were high in the downstream areas while there was no significant difference near the upstream sub-basins.

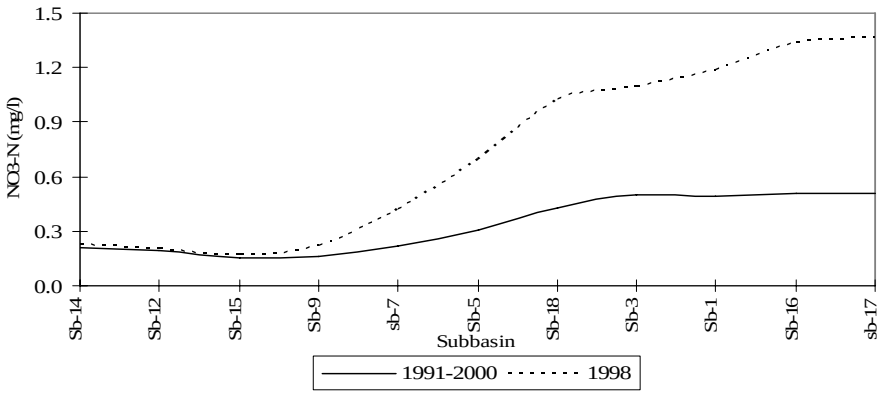

Figure 32: NO3-N concentration in stream water at different locations along the Rönne River

Table 17: Longitudinal NO_3 concentration (mg/l) in stream water along the main channel of Rönne River

Sub-basin		14	12	15	9	7	5	18	3	1	16	17
NO_3	91-00	0.21	0.19	0.15	0.16	0.22	0.31	0.42	0.50	0.49	0.51	0.51
	98	0.23	0.20	0.17	0.22	0.42	0.69	1.02	1.09	1.18	1.34	1.37

4.3.3 NO_3 Loadings under two Management Scenarios

The model was made to run for the period 1991-2000, under two management scenarios to be able to examine the effect of agricultural fertilizer inputs on nitrate loading and stream water quality. The first scenario (scenario 1) was without the application of fertilizer while the second scenario (scenario 2) employed 125kg/ha of nitrogen fertilizer input to agricultural and pasture land. Table 18 summarises and compares the annual basin values averaged over the entire simulation period (1991-2000) for both scenarios, while more details of the model output are given in appendix 6.

Table 18: Summary of average annual basin values (1991-2000) under two management scenarios

Description	Scenario 1: Unfertilized	Scenario 2: Fertilized	Difference
N fertilize applied (kg N/ha)	0.00	125	+125
Surface runoff (mm)	52.63	52.49	-0.3%
Total water yield (mm)	436.73	447.39	+2.4%
NO_3 yield in surface runoff (kg/ha)	0.567	0.589	+3.9%
NO_3 yield in subsurface flow (kg/ha)	0.162	0.494	+205%
NO_3 leached to shallow aquifer (kg/ha)	1.464	2.338	+60%

The result indicated that the application of fertilizer would result in a significant increase of nitrate loading by base flow (205%) while loading by surface runoff increased only by 3.9%. Leaching to groundwater (shallow aquifer) was also increased by 60%. All the three mechanisms of leaching finally contribute to the amount of nitrate concentration in stream water.

The visualisations shown in figures 33 and 34 were the amount of average NO_3 transported by surface runoff (kg N/ha) for unfertilized and fertilized scenarios respectively during the simulation period (1991-2000) from the different sub-basins characterised by different watershed properties such as land use and soil types.

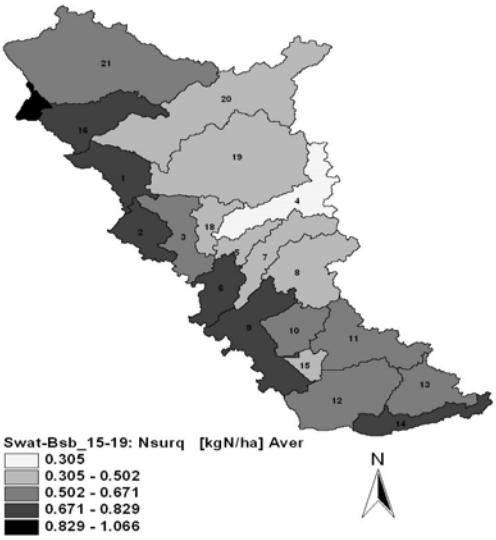

Figure 33: NO_3 loading by surface runoff from different sub-basins, 10-years average (1991-2000) for scenario 1

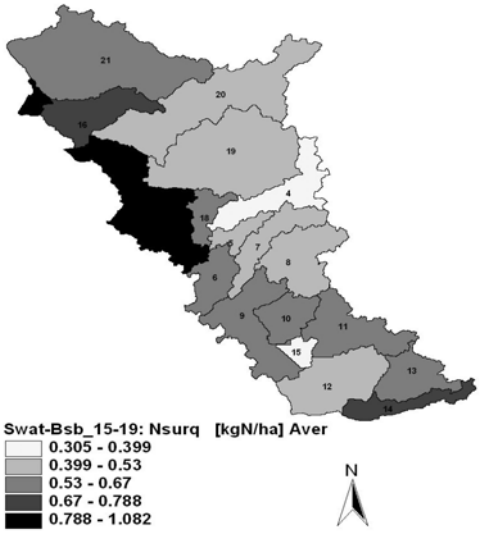

Figure 34: NO_3 loading by surface runoff from different sub-basins, 10-years average (1991-2000) for scenario 2

The difference in loading due to fertilization was not as such significant on the visualizations since the significant amount of leaching occurred by subsurface water that can not be visually observed on surfaces. Nevertheless, both maps show that higher density of loading generally occurred from areas dominated by agricultural land, whereas areas to the north mostly occupied by forest showed less loading density. But, few exceptions to such general distributions were observed, for example in sub-basins 2 and 17. Sub-basin 2 is dominated by forest (65%) and sub-basin 17 is occupied by significant proportion of urban land (47%) and forest (40%) whereas, agricultural land occupies only 13%. The least loading rate occurred from sub-basin 4 under both scenarios.

5 Discussion

5.1 Stream Water Flow Modelling

Several hydrologic parameters were adjusted for annual and monthly stream flow calibrations at two locations within the study area. Most of the calibration parameters were based on suggestions of previous SWAT studies (Arnold et al., 2000; Santhi et al., 2005). Since hydrologic conditions are likely to vary considerably between different watersheds, sensitivity analysis is essential in order to identify the most responsive hydrologic conditions for Rönne River basin. Hence, three of the ten input parameters (table 11) used in this study were evaluated for sensitivity that involve the curve number, soil available water capacity, soil evaporation coefficient, groundwater revaporation factor and base flow alpha factor. The result showed that the runoff curve number (CN) was found to be the most sensitive parameter for both surface and base flow, while SOL-AWC and ESCO also influenced both flows though not as significant as CN. These parameters were also found to be sensitive in other previous SWAT studies performed on several watersheds in the U.S.A., for example by (Arnold et al., 2000; Reungsang et al., 2005; Santhi et al., 2005; White and Chaybey, 2005). Similar studies performed in Europe, for example by Grizzetti et al (2003), on the Vantaanjoki watershed in Southern Finland showed that CN was most sensitive in calibrating stream flow.

CN is a parameter that adjusts the proportioning between surface runoff and infiltration based on watershed properties such as soil hydrologic group, land use, and antecedent moisture conditions. The sensitivity calculated for flow as a function of average annual values, by varying CN2 from -0.1 to -8 decreases the surface runoff by 38% and 48%, and increases the base flow by 12% and 9% for the calibration sub-basins at Ringsjön and Klippan, respectively. This indicates that surface runoff significantly increases as the curve number increases, while base flow is inversely related to curve number since infiltration rate decreases with increasing surface runoff. The sensitivity also varies between different sub-basins, as observed in this study, due to likely variations in combinations of land use and soil types in different sub-basins. The land use and soil combinations in Klippan (>80% forest and 23 % sand soil) favour more infiltration compared to the upstream sub-basins of Ringsjön characterised by significant proportion of land use and soil types that favour more runoff such as water bodies, agricultural land, urban areas and clay soils with less forest cover.

ESCO was the second parameter evaluated for sensitivity because it determines the depth distribution for evaporation from the soil (Neitsch et al., 2002) to account for the effect of capillary action, crusting and cracking (Jha et al., 2004). The sensitivity evaluation performed showed that the variation of ESCO value from 1 (no compensation) to 0.1 decreases the surface runoff by 5.4% and 4%, and base flow by 9.5% and 9% for Ringsjön and Klippan sub-basins respectively. This indicates that reduced ESCO value increases the evaporative demand from soil layer making less water available for surface runoff and base flow. The change in base flow at both locations is closely similar may be because only a single soil layer was assumed for the entire basin and hence spatial variation does not account for layer depth variation. SOL-AWC decreases, in both surface and base flow, between 4-5% for an increase of parameter value from 0.001 to 0.05. Increasing the SOL-AWC increases the water holding capacity of the soil that would be more available for plant uptake by reducing the net outflow of water from a sub-basin. On the contrary, decreasing soil water capacity results in less pore space between soil particles and hence less water holding capacity favouring higher runoff and percolation.

The calibration and validation of SWAT2000 performed on two locations within the Rönne River Basin produced a satisfactory prediction of stream water flow, for both annual and monthly time steps with slight under predictions of high flows. The mean of the measured and calculated values are not significantly different at 95% confidence level ($p<0.05$) for all the calibration and validation results. The Nash-Sutcliffe simulation efficiency, $\mathbf{E_{NS}}$, ranges from 0.65 to 0.78 for the annual simulations indicating that the model sufficiently tracked the average annual flows. A coefficient of determination, r^2, ranging between 0.70 – 0.79 for annual simulation further confirms the suitability of SWAT to model, especially, the hydrological conditions of the current study area. However, the model performed less efficiently on a monthly basis with computed E_{NS} and r^2 values ranging from 0.35 to 0.62 and 0.52 to 0.66 respectively indicating that SWAT is more suitable to model long-term annual effects than short-term monthly variations of basin hydrologic conditions. Although the results obtained can be considered adequate as evaluated graphically and statistically, they showed generally less performance compared to results obtained in previous SWAT studies on similar watersheds. Grizzetti et al. (2003) for example, obtained $\mathbf{E_{NS}}$ value of 0.57-0.81 and 0.75-0.81 for daily and monthly time steps respectively during the calibration and validation of SWAT on the Vantaanjoki watershed in Southern Finland. This may indicate

that the present calibration results could be further improved by verifying the model input data such as land use, soil and climate since these dataset might had been greatly generalised somewhat overlooking the spatial variability of basin characteristics.

5.2 Nutrient (NO_3) Load Modelling

Although the simulated time-series plot tends to follow the temporal variations of the observed data (figure 22), the calibration result for the selected nutrient (NO_3-N) was poor with negative Nash-Sutcliffe prediction efficiency. All parameters suggested in various literatures for NO_3 calibration (Neitsch et al., 2002; Santhi et al., 2005; White and Chaybey, 2005), such as SOL-NO_3, BIOMIX, NPERCO, RSDCO and fertilizer application rates were rigorously adjusted to effectively simulate the measured data, but the result remained far under predicted without significant improvement. The nitrate percolation coefficient (NPERCO) was found to be the only sensitive parameter that significantly changes NO_3 concentration (>300%) with a parameter variation from 0 to 1. It adjusts the proportion of concentration of nitrate between surface runoff and percolating water (Neitsch et al., 2002) in such a way that higher value of NPERCO increases NO_3 transport in surface water and hence concentration in stream water.

The consistent under prediction of the model in calibrating NO_3 is not surprising since nutrient related model data input was not exhaustively verified due to lack of information. One of such data input involves point sources from waste water treatment plants (WWTPs) located within the basin that is believed to greatly influence the model output. Several point sources from WWTPs are found in the study area and two of them are located within the sub-basin selected for calibration for which point discharge data could not be available to include in this particular study. A SWAT study in Germany (Pohlert et al., 2005) demonstrated that the contribution of point source pollution, especially during low flow periods in summer can reach up to 90% of the total nitrate load at the outlet. This implies the significance of point sources in altering the under predicted nitrate concentration in stream in this study although the point source contribution proportion may differ from that of Germany depending on local conditions.

Management data, such as timing, type and amount of fertilizer applications, crop types and growing seasons are also closely related to nutrient loading. The management data used in this study were

solely based on available literature sources for the whole Sweden, regional in nature, which are not watershed-specific to exceptionally represent the study area that may lead to additional discrepancies. However, the discrepancy due to such management data may not be significant as discussed by Grizzetti et al. (2003) since other watershed characteristics such as soil properties are more important than crop and management parameters in model predictions.

Uncertainties associated with NO_3 concentration were further examined by comparing the time-series measured concentration data against measured flow to observe the seasonal behaviour if variation in NO_3 concentration can be explained by seasonal stream flow variation as indicated in figure 23. The result showed a negative correlation with very low coefficient of determination (r^2 = 0.02) suggesting that variation in NO_3 concentration could not be explained by the variation in discharge. This may be partially true since other factors such as soil types and land uses also control NO_3 exports from sub-basins.

However, this is in disagreement with previous works performed in other watersheds, for example, in Belgium (Van Herpe and De Troch, 2000) and several other researched works cited there-in. Van Herpe and De Troch (2000) reported positive correlations, and on average 32% (r^2=0.32) of the variation in NO_3 concentration was explained by the variation in discharge volume with out any significant negative correlations between discharge and concentration. The low value of r^2 in this study may, therefore, be attributed to the likely error associated to the sampling and analysis procedures for NO_3 concentration. This can be further explained by the fact that the measured data were not continuous daily discharge data (only grab samples) so that the possibility of low precision and less representativeness of the data may not be ruled out. Santhi et al. (2005) underscored the necessity of continuous collection of monitoring data for adequate calibration and validation of SWAT model.

Uncertainties are also likely due to the model itself since some nutrient related processes are not considered to fully account for nitrogen balances. The contribution of Nitrous oxide (N_2O), for example, as a possible pathway for N-losses is not included into the model. Therefore, further verification of the N-balances for the sub-basin, by attempting to estimate the overall N-budgets, i.e. inputs, exports and transformations, may help minimize such uncertainties. Groundwater quality data in terms of N-levels, for example, could be

used to verify the N- leaching to groundwater and the contributions expected back from the groundwater store.

5.3 Scenario Analysis

Obviously, the uncertainty associated with nitrate modelling due to the limitations discussed above is a drawback for comprehensive scenario analysis that would enable the estimation and study of nitrate loading from different sub-basins and concentration in stream water since the resulting model outputs are greatly influenced by the quality of input data. However, different scenarios could be examined in relative terms so that the spatial and temporal variations of nutrient in relation to different watershed attributes such as land use, soil types, topographic feature, and management scenarios would be investigated.

The first scenario analysis was introduced to scrutinize the relationship between different land use types and NO_3 loading by surface water that finally makes its way to stream waters. NO_3 (kg N) transported by surface water from the various land covers was statistically compared to the field percentages and the result showed the amount of nitrate leached positively correlated to the percentage area occupied by agricultural land ($r= 0.9$, $r^2= 0.79$). A similar strong correlation was obtained (Grizzetti et al., 2003) in the Vantaanjoki basin of Southern Finland by considering total nitrogen loads and agricultural surface of the drained catchments. This analysis also indicated the positive correlation between in-stream NO_3 concentration (mg/l) and proportion of arable land although the correlation coefficient obtained ($r= 0.37$) is weaker than that of the NO_3 load ($r=0.90$). The lower correlation coefficient for nitrate concentration compared to loading may be attributed to additional factors that influence nutrient concentrations such as in-stream nutrient processes that undergo different transformations. Nonetheless, both nitrate transport in surface water (kg N) and concentration in stream water (mg/l) are well explained by agricultural land, which could be attributed to the application of fertilizers. Similarly, Grizzetti et al. (2003) discussed the well predictability of nutrient load by agricultural surface even without specifying crop type. Previous scenarios modelled in the study area (Arheimer et al., 2004) also suggested that changing agricultural practices is the most effective way to reduce nitrogen transport from land to lakes and the sea indicating that nutrient emission is well predicted by agricultural land.

This study also showed diffuse emissions are lower for other land use types and particularly negatively correlated with percentage of forest cover (r = -0.7). A statistical analysis performed on a watershed in Belgium (Van Herpe and De Troch, 2000) showed analogous results whereby higher percentages of forested area resulted in lower NO_3 concentrations. This indicates that NO_3 emission to surface and subsurface waters decreases with increasing forest surface mainly due to the increased demand of plant nitrogen uptake as indicated by the nitrogen cycle (figure 9) and SWAT nitrogen pool (figure 10). The CN associated to forest cover is also believed to influence, at least, nitrate transport in surface runoff by increasing infiltration.

The second scenario studied was the spatial variation of NO_3 concentration in stream water at the outlet of various sub-basins along the Rönne River. The longitudinal pattern along the main stream, as observed in figure 32 and table 17, shows that the concentration of NO_3 increases downstream. This is not an unexpected trend, since the volume of water that carries nutrients from sub-basins increases in the downstream direction. This can be further explained by comparing the ten year average and the 1998 wet year NO_3 concentration as indicated in figure 32. The difference between the two in the first few upstream sub-basins is practically insignificant since the volume of water and sub-basin area contributing to NO_3 loading are small. But it becomes significant in the downstream areas as the volume of water for 1998 rises above the ten year average due to large draining area to carry more nutrients. The downstream increase of nitrate concentration may be explained not only by the enhanced volume of water (hydrologic regime) responsible for transporting but also by the spatial distribution of basin characteristics such as topography, soil type and land use. In this study, it was hypothesised that arable land is the major source of nutrient concentrations in stream water. This can be justified by comparing the increasing nitrate concentration (figure 32 and table 17) corresponding to the downstream intensification of agricultural activity along the main channel (figure 13 and appendix 2). Similar phenomenon was observed and discussed (Van Herpe and De Troch, 2000) in Belgium where the influence of anthropological land use activities (increasing arable land) was reflected in the longitudinal NO_3 concentration pattern along the main stream with positive correlation.

Finally, two different management scenarios for fertilizer application were considered in order to evaluate the effects on NO_3 loading. Scenario 1 was without the application of fertilizer while scenario 2 considers 125kg/ha of nitrogen fertilizer input to agricultural and

pasture lands represented by spring barley. The simulation result averaged over 10 years showed an annual increase of NO_3 loading by 4%, 267% and 63% to surface water, subsurface water and shallow aquifer, respectively under fertilized scenario (table 18). The spatial distribution of NO_3 leaching exhibits similar densities for both scenarios except slight increase observed after fertilization (cf. figures 33 and 34). Generally, most of the leaching is related to agricultural land as observed from the simulation visualised in figures 33 and 34) and this has been statistically confirmed as explained above. The least rate of leaching occurred in sub-basin 4 under both scenarios. Sub-basin 4 is covered by forest and totally devoid of agricultural land.

However, exceptions are also observed in this simulation whereby high density of loading is related to other areas different from agricultural land. The high rate of loading from sub-basin 2, for example may not be explained by agricultural land since the proportion of forest cover is much higher (65%) than agricultural land surface (35%). This indicates that other watershed characteristics may control the NO_3 loading within the basin. One of such factors could be the effect of topographic feature that controls the rate of surface runoff and hence the rate of nutrient leaching. This has been discussed by Van Herpe and De Troch (2000) that steep slopes generate more surface runoff and less infiltration compared with gentle or flat slopes. Although such hydrologic process has a diluting effect on stream water NO_3 concentrations due to greater volume of water, on the other hand it carries a considerable amount of nutrients from land surface causing huge loss of NO_3 from sub-basins. This is a possible circumstance in sub-basin 2 mainly covered by forest but generating high nitrate export. The steepest slope is in fact found in sub-basin 2 with mean standard deviation of elevation being the highest (59) compared to the rest of sub-basins, only closely followed by sub-basin 21 with standard deviation of 55 (appendix 1) for which surface leaching is also high. This indicates that the sub-basins generate high rate of runoff and hence high nitrate export in surface water influenced by the topographic feature as observed in sub-basins 2 and 21.

6 Conclusion

The hydrologic conditions and nitrate loading from land to stream water within the Rönne River basin were modelled using SWAT hydrologic and water quality model. The model calibration and validation procedures for stream flow confirmed that the model adequately simulates the hydrologic regime of the study area. The simulated stream flow was found to be sensitive to variations in curve number, the soil evaporation coefficient and soil available water capacity that the selection of values of the parameters can significantly change the predicted stream flow results. The calibration result for nitrate concentration in stream water was rather poor probably attributed to the scantiness of information and input data used for the model.

Nevertheless, the calibrated model for hydrologic conditions was used to assess the relationships between different watershed attributes and nitrate losses within the watershed. The simulation result over the period of 10-years (1991-2000) showed that nitrate loading by surface water is strongly positively correlated to the percentage area occupied by agricultural land and negatively correlated to the area occupied by forest. Nitrate concentration in stream water is also correlated and spatially varied with intensively cultivated agricultural land longitudinally along the main channel of the Rönne River. There is also an indication that nutrient loading could be influenced by topographic features, steeper slopes being responsible for enhanced transport of nitrate by surface runoff.

Generally, agricultural land is a strong predictor of nitrate emission from land that 79% of variation in nitrate loading can be explained by variation in agricultural land cover. The loading in surface water is also influenced by change in forested area with r^2 value of 0.43. Therefore, changing agricultural practice and increasing forest area will effectively reduce nitrate leaching to surface waters within the Rönne basin and the nearby coastal North Sea.

Finally, this study showed that SWAT model is a powerful tool that can be used for hydrological analysis of Rönne River basin, especially when dealing with water balance and quantity aspects. However, further nutrient data verification is essential for a reliable estimation of nitrate load values, related to water quality problems, to help effective management planning and decision-making processes within the watershed.

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Appendices

Appendix 1: Topographic statistical summary

A-1.1 Elevation report for the watershed

Min. Elevation: 0
Max. Elevation: 220.333
Mean. Elevation: 89.87
Std. Deviation: 41.1029

A-1.2 Elevation report for each sub-basin

Sub-basin # 1
Min. Elevation: 0.366667
Max. Elevation: 77.6444
Mean. Elevation: 27.1339
Std. Deviation: 14.4413

Sub-basin # 2
Min. Elevation: 4.43333
Max. Elevation: 206.489
Mean. Elevation: 115.026
Std. Deviation: 59.0218

Sub-basin # 3
Min. Elevation: 5.26667
Max. Elevation: 204.089
Mean. Elevation: 68.7082
Std. Deviation: 47.1641

Sub-basin # 4
Min. Elevation: 38.7778
Max. Elevation: 150.617
Mean. Elevation: 95.448
Std. Deviation: 22.3323

Sub-basin # 5
Min. Elevation: 37.6444
Max. Elevation: 117.467
Mean. Elevation: 66.207

Sub-basin # 12
Min. Elevation: 53.5333
Max. Elevation: 159.567
Mean. Elevation: 82.6792
Std. Deviation: 27.0774

Sub-basin # 13
Min. Elevation: 77.1222
Max. Elevation: 186.8
Mean. Elevation: 146.521
Std. Deviation: 20.3408

Sub-basin # 14
Min. Elevation: 77.3667
Max. Elevation: 187.9
Mean. Elevation: 147.044
Std. Deviation: 22.7102

Sub-basin # 15
Min. Elevation: 53.5889
Max. Elevation: 113.856
Mean. Elevation: 65.0945
Std. Deviation: 14.7067

Sub-basin # 16
Min. Elevation: 0
Max. Elevation: 127.089
Mean. Elevation: 41.61

Std. Deviation: 18.9847

Sub-basin # 6
Min. Elevation: 42.6889
Max. Elevation: 168.8
Mean. Elevation: 84.5067
Std. Deviation: 31.373

Sub-basin # 7
Min. Elevation: 42
Max. Elevation: 167.078
Mean. Elevation: 90.5339
Std. Deviation: 27.9726
– – ---
Sub-basin # 8
Min. Elevation: 44.8889
Max. Elevation: 172.611
Mean. Elevation: 108.967
Std. Deviation: 23.5944

Sub-basin # 9
Min. Elevation: 42.8889
Max. Elevation: 127
Mean. Elevation: 79.5981
Std. Deviation: 17.6689

Sub-basin # 10
Min. Elevation: 53.3889
Max. Elevation: 169.256
Mean. Elevation: 83.7107
Std. Deviation: 23.6541

Sub-basin # 11
Min. Elevation: 53.4889
Max. Elevation: 175.48
Mean. Elevation: 109.173
Std. Deviation: 27.6952

Std. Deviation: 29.6787

Sub-basin # 17
Min. Elevation: 0.0333333
Max. Elevation: 37.9556
Mean. Elevation: 11.0494
Std. Deviation: 9.22288

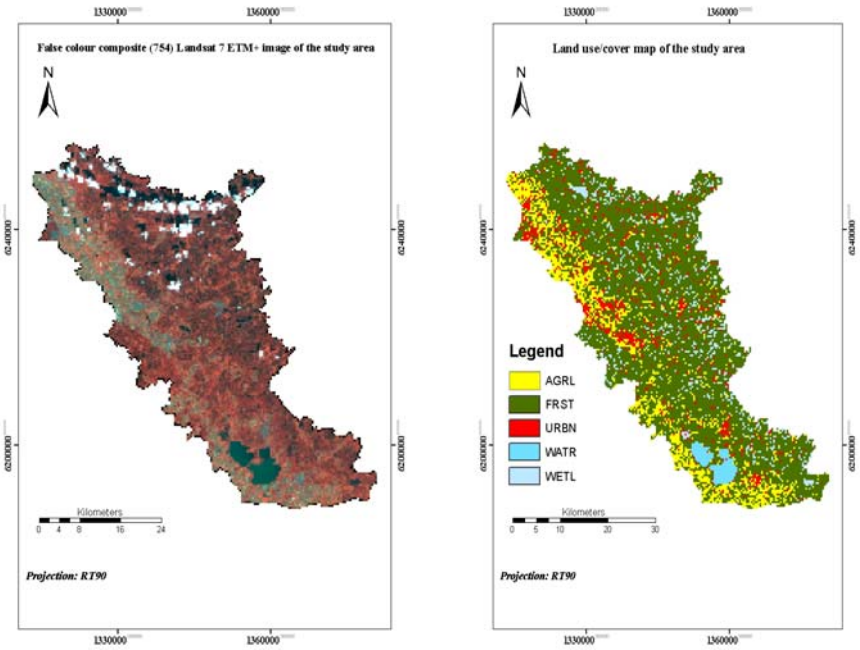
Sub-basin # 18
Min. Elevation: 34.3889
Max. Elevation: 93.7556
Mean. Elevation: 54.6465
Std. Deviation: 12.2109

Sub-basin # 19
Min. Elevation: 11.5444
Max. Elevation: 149.722
Mean. Elevation: 93.8542
Std. Deviation: 24.4071

Sub-basin # 20
Min. Elevation: 2.04444
Max. Elevation: 175.525
Mean. Elevation: 96.9643
Std. Deviation: 28.6877

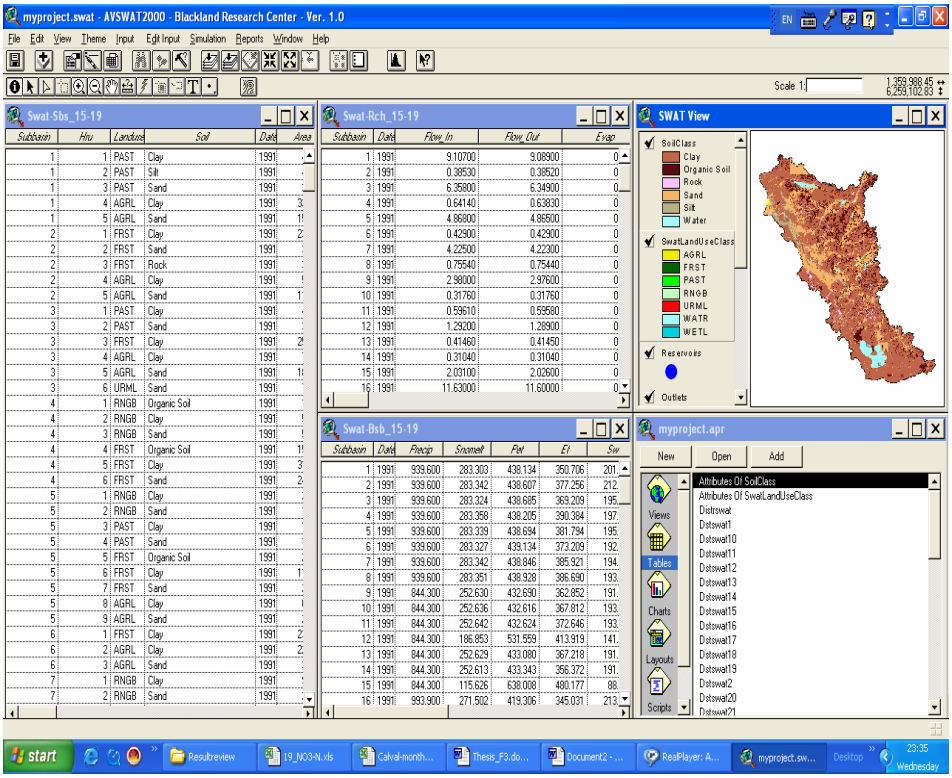
Sub-basin # 21
Min. Elevation: 6.94444
Max. Elevation: 220.333
Mean. Elevation: 93.3582
Std. Deviation: 54.5715

Appendix 2: Landsat image and classified land use map



Landsat ETM+ 7 image of the study area, acquisition date: 03 Aug 1999, with display colour bands: R=7, G=5, and B=4. The resulting classified land use map was based on semi-supervised image classification method.

Appendix 3: SWAT main output files



Three main SWAT output files: the HRU file (.sbs), the main channel or reach out put file (.rch) and the sub-basin file (.bsb). In this study, (.rch) and (.bsb) were widely used for data analysis.

Appendix 4: Calibration and validation data

A-4.1: Flow calibration data

Table showing measured stream flow (monthly daily average) and the corresponding SWAT predicted values for two locations within the watershed. The calibration was performed for a period of 10-years (Jan 81-Dec 90).

Date	Ringsjön		Klippan	
	Meas. (m^3/s)	Sim. (m^3/s)	Meas. (m^3/s)	Sim. (m^3/s)
Jan-81	15.85	0.05	6.22	0.00
Feb-81	15.70	0.09	8.08	0.26

Mar-81	12.21	1.89	8.60	1.60
Apr-81	5.79	1.74	2.40	2.15
May-81	1.32	0.66	1.30	0.88
Jun-81	2.22	1.40	1.66	0.41
Jul-81	2.23	1.14	1.60	0.52
Aug-81	1.71	0.79	0.82	0.49
Sep-81	1.73	0.90	0.56	0.32
Oct-81	1.73	6.51	5.07	4.43
Nov-81	6.78	11.86	10.19	9.23
Dec-81	10.44	10.40	4.14	8.48
Jan-82	8.88	8.41	2.47	5.87
Feb-82	2.18	5.65	3.95	4.35
Mar-82	9.94	6.68	8.12	4.74
Apr-82	3.92	3.16	2.56	2.53
May-82	2.07	1.51	2.91	1.83
Jun-82	2.14	0.94	1.51	0.82
Jul-82	1.45	0.16	0.61	0.34
Aug-82	1.31	0.67	0.93	0.21
Sep-82	1.41	0.29	1.04	0.31
Oct-82	2.65	5.21	1.94	1.90
Nov-82	3.51	4.98	2.78	3.53
Dec-82	4.69	10.10	6.87	8.46
Jan-83	9.91	9.28	6.53	7.74
Feb-83	8.02	7.38	3.30	6.33
Mar-83	6.93	7.33	8.14	5.19
Apr-83	9.63	7.20	5.97	5.20
May-83	3.65	4.26	3.99	3.17
Jun-83	3.39	1.84	2.00	2.50
Jul-83	1.14	0.51	0.63	1.11
Aug-83	1.32	0.15	0.26	0.41
Sep-83	0.98	0.75	0.46	0.24
Oct-83	0.75	1.31	1.22	0.30
Nov-83	0.79	2.15	2.65	1.46
Dec-83	1.51	7.15	5.85	5.83
Jan-84	9.08	13.77	8.38	10.16
Feb-84	9.72	9.01	4.75	7.50
Mar-84	4.09	3.23	2.43	2.95
Apr-84	0.30	1.31	2.34	1.42
May-84	0.88	0.50	1.10	0.62
Jun-84	5.00	3.79	2.19	0.80
Jul-84	5.01	2.26	1.47	1.33
Aug-84	2.99	1.10	0.72	0.52
Sep-84	2.36	4.26	2.00	1.61
Oct-84	3.19	6.54	5.23	4.92
Nov-84	4.70	5.72	3.55	4.61

Dec-84	5.58	5.31	3.62	4.26
Jan-85	5.08	4.03	1.43	2.90
Feb-85	3.29	7.49	3.24	6.37
Mar-85	0.69	3.95	3.25	3.08
Apr-85	9.21	4.59	7.93	3.69
May-85	7.85	2.52	4.23	2.85
Jun-85	2.14	1.48	1.49	1.25
Jul-85	1.55	0.73	0.90	0.53
Aug-85	1.83	1.95	2.28	0.87
Sep-85	1.59	3.41	4.25	3.73
Oct-85	5.28	1.77	1.54	2.54
Nov-85	2.80	4.67	3.45	3.72
Dec-85	8.09	15.62	12.41	10.63
Jan-86	10.54	13.49	7.19	10.96
Feb-86	7.95	6.58	2.06	5.98
Mar-86	2.78	4.03	4.83	3.37
Apr-86	7.00	4.16	4.75	3.37
May-86	2.64	2.52	2.71	1.97
Jun-86	2.09	0.81	1.22	0.92
Jul-86	0.94	0.18	0.69	0.44
Aug-86	0.97	0.09	0.31	0.21
Sep-86	1.07	0.20	0.95	0.21
Oct-86	1.11	1.48	1.23	0.80
Nov-86	1.78	4.02	3.68	4.54
Dec-86	3.59	6.95	4.13	5.62
Jan-87	4.26	7.69	2.05	5.68
Feb-87	0.99	5.41	3.96	4.34
Mar-87	1.04	4.08	2.38	2.92
Apr-87	6.66	3.43	5.11	2.69
May-87	2.68	1.84	3.23	1.69
Jun-87	3.11	1.99	2.05	0.95
Jul-87	4.36	2.70	1.74	1.41
Aug-87	3.37	2.63	2.85	2.62
Sep-87	2.77	3.54	3.31	2.65
Oct-87	6.10	2.16	1.85	2.18
Nov-87	3.17	4.30	3.28	3.07
Dec-87	4.59	5.61	4.44	4.08
Jan-88	10.00	13.32	8.55	9.95
Feb-88	14.50	11.99	8.05	9.20
Mar-88	10.58	9.01	5.60	7.30
Apr-88	10.17	4.57	4.29	3.78
May-88	1.75	1.12	1.55	1.68
Jun-88	2.23	0.33	0.93	0.68
Jul-88	2.80	2.39	3.09	0.80
Aug-88	2.55	0.63	1.76	1.98

Sep-88	3.27	2.13	2.53	1.60
Oct-88	4.13	5.96	4.12	4.32
Nov-88	5.87	4.48	2.80	3.23
Dec-88	2.11	6.62	5.97	4.96
Jan-89	8.12	4.72	6.37	4.06
Feb-89	3.26	4.23	4.68	3.61
Mar-89	4.05	6.18	5.71	4.82
Apr-89	4.24	3.87	4.58	3.08
May-89	1.54	1.05	2.31	1.35
Jun-89	1.91	0.31	1.43	0.54
Jul-89	2.12	0.14	0.50	0.28
Aug-89	1.92	2.23	2.58	0.67
Sep-89	2.49	0.08	0.70	1.12
Oct-89	2.35	1.58	1.88	2.64
Nov-89	2.76	3.69	1.25	4.05
Dec-89	2.42	3.28	7.21	2.57
Jan-90	4.01	6.89	4.33	5.44
Feb-90	3.81	4.51	6.10	3.41
Mar-90	8.24	10.12	4.79	8.12
Apr-90	1.61	5.19	3.21	5.09
May-90	1.57	2.36	0.83	2.33
Jun-90	2.38	0.88	1.64	1.04
Jul-90	2.58	2.09	1.69	1.53
Aug-90	2.70	0.22	1.09	0.62
Sep-90	2.56	2.12	2.03	0.49
Oct-90	3.06	3.85	2.09	3.79
Nov-90	3.18	4.68	3.68	3.44
Dec-90	4.69	4.76	3.25	3.48

A-4.2: Flow validation data

Table showing measured stream flow (monthly daily average) and the corresponding SWAT predicted values for two locations within the watershed. The validation was performed for a period of 10-years (Jan 1991-Dec 2000).

Date	Ringsjön		Klippan	
	Meas. (m^3/s)	Sim. (m^3/s)	Meas. (m^3/s)	Sim. (m^3/s)
Jan-91	3.95	0.06	5.35	0.00
Feb-91	6.55	0.03	3.05	0.00
Mar-91	3.53	0.02	3.40	0.01
Apr-91	2.82	0.03	2.25	0.01

May-91	4.95	0.27	2.21	0.30
Jun-91	5.08	1.98	4.93	1.97
Jul-91	6.43	1.75	2.56	2.48
Aug-91	2.85	1.09	0.80	1.08
Sep-91	2.24	1.27	1.35	0.86
Oct-91	3.53	2.12	2.55	2.18
Nov-91	2.40	7.08	5.53	5.95
Dec-91	4.00	7.88	6.39	6.39
Jan-92	8.57	7.18	6.16	6.75
Feb-92	6.63	4.73	5.23	4.00
Mar-92	9.04	7.12	5.86	5.64
Apr-92	5.98	4.37	4.09	3.65
May-92	2.28	1.35	2.67	2.19
Jun-92	2.32	0.27	0.53	0.87
Jul-92	2.61	0.14	0.24	0.38
Aug-92	3.45	0.49	0.48	0.23
Sep-92	2.65	0.80	1.64	0.38
Oct-92	2.54	1.39	1.62	0.48
Nov-92	1.37	7.99	7.23	5.82
Dec-92	4.29	9.35	5.15	6.87
Jan-93	7.05	10.49	6.41	7.15
Feb-93	9.72	7.93	4.89	5.95
Mar-93	5.55	4.02	3.67	3.28
Apr-93	1.99	1.30	1.84	1.59
May-93	1.75	0.26	0.54	0.61
Jun-93	2.17	0.13	0.39	0.26
Jul-93	1.82	3.62	2.39	0.63
Aug-93	4.36	3.07	3.20	2.93
Sep-93	5.24	4.81	2.74	2.72
Oct-93	6.79	7.55	3.99	3.89
Nov-93	6.74	5.91	2.49	3.28
Dec-93	8.40	11.15	6.38	6.60
Jan-94	10.23	12.47	7.55	7.85
Feb-94	9.69	9.61	3.84	6.96
Mar-94	9.69	11.32	11.55	8.49
Apr-94	7.19	8.39	5.30	6.81
May-94	1.29	3.13	1.28	2.69
Jun-94	2.60	1.86	1.79	1.26
Jul-94	2.61	0.21	0.44	0.49
Aug-94	3.39	0.15	0.29	0.23
Sep-94	2.65	7.15	3.01	3.34
Oct-94	5.85	3.95	2.10	2.92
Nov-94	3.52	5.89	4.61	4.26
Dec-94	4.89	9.69	8.21	6.97
Jan-95	10.09	8.14	6.55	6.79

Feb-95	11.66	10.56	13.35	7.53
Mar-95	8.24	13.53	6.13	11.17
Apr-95	8.47	7.21	5.38	6.47
May-95	3.34	3.12	1.95	2.87
Jun-95	2.91	1.71	1.01	1.31
Jul-95	1.56	0.24	0.43	0.53
Aug-95	2.09	0.10	0.10	0.21
Sep-95	2.14	1.32	0.66	0.21
Oct-95	0.84	0.97	1.00	0.29
Nov-95	1.57	2.50	3.16	1.69
Dec-95	2.54	2.67	1.52	1.69
Jan-96	0.85	2.17	1.15	1.38
Feb-96	1.41	2.56	1.08	1.54
Mar-96	0.66	2.19	0.82	1.32
Apr-96	0.24	0.79	2.14	0.58
May-96	0.64	4.21	4.18	2.07
Jun-96	3.95	2.03	1.78	1.95
Jul-96	3.06	1.65	1.82	0.82
Aug-96	2.00	0.18	0.36	0.31
Sep-96	2.39	0.65	0.30	0.17
Oct-96	2.74	0.86	0.53	0.12
Nov-96	3.51	4.38	3.58	2.26
Dec-96	5.50	5.49	2.94	3.91
Jan-97	4.59	2.99	0.87	2.13
Feb-97	1.69	6.33	5.32	4.58
Mar-97	3.63	3.88	3.42	3.29
Apr-97	1.24	2.66	2.38	2.06
May-97	3.76	2.71	3.16	2.21
Jun-97	3.06	0.45	0.90	1.01
Jul-97	2.44	0.32	0.58	0.43
Aug-97	2.52	0.16	0.27	0.63
Sep-97	2.71	0.06	0.19	0.26
Oct-97	1.59	2.68	1.52	2.79
Nov-97	2.50	3.58	2.00	3.55
Dec-97	2.71	4.76	2.81	4.06
Jan-98	3.72	6.34	3.60	5.10
Feb-98	2.85	7.72	7.63	6.09
Mar-98	11.47	7.74	5.63	5.50
Apr-98	7.46	6.23	4.14	4.24
May-98	2.31	2.55	1.38	2.00
Jun-98	3.21	2.26	2.83	1.13
Jul-98	4.10	1.88	5.78	3.30
Aug-98	2.54	2.81	4.02	2.86
Sep-98	3.85	3.87	3.88	2.83
Oct-98	6.12	7.78	7.76	5.24

Nov-98	14.11	10.04	6.07	7.94
Dec-98	8.26	7.34	5.24	5.39
Jan-99	12.97	10.74	6.37	8.33
Feb-99	9.46	7.61	4.68	6.42
Mar-99	11.84	7.05	5.71	5.64
Apr-99	7.29	5.13	4.58	3.56
May-99	4.35	3.18	2.31	2.27
Jun-99	4.13	2.26	1.43	1.38
Jul-99	2.50	0.28	0.50	0.63
Aug-99	2.43	3.22	2.58	2.06
Sep-99	5.53	1.48	0.70	1.30
Oct-99	4.87	3.28	1.88	2.66
Nov-99	4.48	2.41	1.25	1.66
Dec-99	5.24	4.66	7.21	2.30
Jan-00	11.17	14.96	4.33	10.61
Feb-00	11.93	5.19	6.10	4.04
Mar-00	9.81	11.52	4.79	8.30
Apr-00	5.11	6.75	3.21	5.06
May-00	1.29	2.57	0.83	2.31
Jun-00	2.63	3.41	1.64	2.11
Jul-00	5.39	1.96	1.69	2.33
Aug-00	3.02	0.83	1.09	0.99
Sep-00	3.50	3.37	2.03	2.74
Oct-00	4.16	2.59	2.09	2.19
Nov-00	2.28	6.14	3.68	4.76
Dec-00	4.98	7.73	3.25	5.80

A-4.3: NO_3 Calibration data

Table showing measured NO3 concentration in stream water (monthly average) and the corresponding SWAT predicted values at Klippan. The calibration was performed for a period of 4-years (Jan 1997-Dec 2000)

	Klippan	
Month	Meas. (mg/l)	Sim. (mg/l)
Jan-97	2.60	0.00
Feb-97	1.30	0.00
Mar-97	1.60	1.65
Apr-97	1.50	2.77
May-97	1.20	18.03
Jun-97	1.40	14.12
Jul-97	1.40	6.66

Aug-97	1.80	1.61
Sep-97	1.40	1.01
Oct-97	1.80	0.13
Nov-97	1.20	0.05
Dec-97	1.50	0.20
Jan-98	1.80	0.06
Feb-98	1.90	0.07
Mar-98	1.70	0.18
Apr-98	1.10	0.03
May-98	1.40	0.61
Jun-98	1.50	0.83
Jul-98	0.78	0.14
Aug-98	1.00	0.09
Sep-98	0.54	0.15
Oct-98	1.20	0.14
Nov-98	0.88	0.04
Dec-98	1.60	0.20
Jan-99	1.20	0.07
Feb-99	1.10	0.09
Mar-99	1.20	0.02
Apr-99	1.20	0.20
May-99	1.10	0.21
Jun-99	0.83	0.25
Jul-99	2.20	0.36
Aug-99	1.50	0.55
Sep-99	1.20	0.18
Oct-99	0.81	0.07
Nov-99	1.10	0.05
Dec-99	1.30	0.41
Jan-00	1.30	0.06
Feb-00	1.40	0.19
Mar-00	1.10	0.23
Apr-00	1.40	0.06
May-00	1.30	0.82
Jun-00	1.40	0.81
Jul-00	0.99	0.21
Aug-00	1.10	0.21
Sep-00	0.86	0.29
Oct-00	0.80	0.07
Nov-00	0.78	0.04
Dec-00	1.00	0.12

Appendix 5: Different land use proportions vs. Nsurq

Sub-basin	AGRL	FRST	PAST	RANGB	WATR	URBN	NSURQ (kg N/ha)
1	80.69	0	19.31	0	0	0	0.778
2	35.01	64.99	0	0	0	0	0.742
3	38.85	38.44	11.37	0	0	11.34	0.671
4	0	84.67	0	15.33	0	0	0.305
5	12.55	61.34	10.43	15.68	0	0	0.431
6	51.82	48.18	0	0	0	0	0.701
7	12.21	68.33	0	19.46	0	0	0.450
8	16.99	83.01	0	0	0	0	0.486
9	47.14	40.03	12.83	0	0	0	0.701
10	24.3	49.47	15.69	10.54	0	0	0.560
11	15.15	60.33	0	11.4	0	13.12	0.595
12	51.75	22.22	0	0	26.03	0	0.568
13	29.21	44.36	12.41	14.02	0	0	0.633
14	69.08	19.93	10.99	0	0	0	0.829
15	0	32.37	0	0	54.23	13.4	0.400
16	69.95	30.05	0	0	0	0	0.749
17	13.39	39.55	0	0	0	47.06	1.066
18	18.63	46.36	18.38	16.63	0	0	0.502
19	15.54	72.62	0	11.84	0	0	0.432
20	17.12	82.88	0	0	0	0	0.418
21	39	61	0	0	0	0	0.642

Appendix 6: Annual Basin Values: Water balance and nutrient

SWAT Feb.'01 VERSION2000

Thu Feb 02 00:19:26 2006 AVSW

AVE ANNUAL BASIN VALUES

PRECIP = 899.9 MM
SNOW FALL = 267.53 MM
SNOW MELT = 266.36 MM
SUBLIMATION = 1.17 MM
SURFACE RUNOFF Q = 52.63 MM
LATERAL SOIL Q = 18.59 MM
TILE Q = 0.00 MM
GROUNDWATER (SHAL AQ) Q = 366.21 MM
REVAP (SHAL AQ => SOIL/PLANTS) = 12.50 MM
DEEP AQ RECHARGE = 15.38 MM
TOTAL AQ RECHARGE = 393.34 MM
TOTAL WATER YLD = 436.73 MM
PERCOLATION OUT OF SOIL = 398.61 MM
ET = 405.2 MM
PET = 461.9MM
TRANSMISSION LOSSES = 0.70 MM
TOTAL SEDIMENT LOADING = 37.528 T/HA

SWAT Feb.'01 VERSION2000

Thu Feb 02 00:19:26 2006 AVSW

AVE ANNUAL BASIN VALUES

SWAT Feb.'01 VERSION2000

Wed Feb 01 23:42:26 2006 AVSW

AVE ANNUAL BASIN VALUES

PRECIP = 899.9 MM
SNOW FALL = 267.53 MM
SNOW MELT = 266.23 MM
SUBLIMATION = 1.29 MM
SURFACE RUNOFF Q = 52.49 MM
LATERAL SOIL Q = 18.77 MM
TILE Q = 0.00 MM
GROUNDWATER (SHAL AQ) Q = 376.81 MM
REVAP (SHAL AQ => SOIL/PLANTS) = 12.82 MM
DEEP AQ RECHARGE = 6.04 MM
TOTAL AQ RECHARGE = 404.94 MM
TOTAL WATER YLD = 447.39 MM
PERCOLATION OUT OF SOIL = 410.27 MM
ET = 393.5 MM
PET = 461.9MM
TRANSMISSION LOSSES = 0.68 MM
TOTAL SEDIMENT LOADING = 9.435 T/HA

SWAT Feb.'01 VERSION2000

Wed Feb 01 23:42:26 2006 AVSW

AVE ANNUAL BASIN VALUES

NUTRIENTS- UNFERTILIZED	NUTRIENTS- FERTLIZED
ORGANIC N = 34.793 (KG/HA)	ORGANIC N = 24.678 (KG/HA)
ORGANIC P = 4.261 (KG/HA)	ORGANIC P = 3.129 (KG/HA)
NO3 YIELD (SQ) =0.567 (KG/HA)	NO3 YIELD (SQ) =0.589 (KG/HA)
NO3 YIELD (SSQ)=0.162(KG/HA)	NO3 YIELD(SSQ) =0.494(KG/HA)
SOL P YIELD = 0.008 (KG/HA)	SOL P YIELD = 0.027 (KG/HA)
NO3 LEACHED = 1.464 (KG/HA)	NO3 LEACHED = 2.338 (KG/HA)
P LEACHED = 0.216 (KG/HA)	P LEACHED = 0.528 (KG/HA)
N UPTAKE = 26.845 (KG/HA)	N UPTAKE = 36.859 (KG/HA)
P UPTAKE = 5.588 (KG/HA)	P UPTAKE = 4.511 (KG/HA)
ACTIVE TO SOLUTION P FLOW = 10.486 (KG/HA)	ACTIVE TO SOLUTION P FLOW = 20.462 (KG/HA)
ACTIVE TO STABLE P FLOW = 7.863 (KG/HA)	ACTIVE TO STABLE P FLOW = 15.068 (KG/HA)
N FERTILIZER APPLIED = 0.000 (KG/HA)	N FERTILIZER APPLIED = 124.993 (KG/HA)
P FERTILIZER APPLIED = 0.000 (KG/HA)	P FERTILIZER APPLIED = 12.200 (KG/HA)
N FIXATION = 0.000 (KG/HA)	N FIXATION = 0.000 (KG/HA)
DENITRIFICATION = 87.329 (KG/HA)	DENITRIFICATION = 189.611 (KG/HA)
HUMUS MIN ON ACTIVE ORG N = 100.156 (KG/HA)	HUMUS MIN ON ACTIVE ORG N = 109.319 (KG/HA)
ACTIVE TO STABLE ORG N = -57.216 (KG/HA)	ACTIVE TO STABLE ORG N = -52.803 (KG/HA)
HUMUS MIN ON ACTIVE ORG P = 17.066 (KG/HA)	HUMUS MIN ON ACTIVE ORG P = 18.643 (KG/HA)
MIN FROM FRESH ORG N = 5.527 (KG/HA)	MIN FROM FRESH ORG N = 17.882 (KG/HA)
MIN FROM FRESH ORG P = 0.826 (KG/HA)	MIN FROM FRESH ORG P = 3.528 (KG/HA)
NO3 IN RAINFALL=8.802(KG/HA)	NO3 IN RAINFALL=8.801(KG/HA)
INITIAL NO3 IN SOIL = 29.976 (KG/HA)	INITIAL NO3 IN SOIL = 29.976 (KG/HA)
FINAL NO3 IN SOIL = 0.936 (KG/HA)	FINAL NO3 IN SOIL = 1.034 (KG/HA)
INITIAL ORG N IN SOIL = 32490.656 (KG/HA)	INITIAL ORG N IN SOIL = 32490.656 (KG/HA)
FINAL ORG N IN SOIL = 31161.080 (KG/HA)	FINAL ORG N IN SOIL = 31339.012 (KG/HA)
INITIAL MIN P IN SOIL = 490.597 (KG/HA)	INITIAL MIN P IN SOIL = 490.597 (KG/HA)
FINAL MIN P IN SOIL = 606.869 (KG/HA)	FINAL MIN P IN SOIL = 724.198 (KG/HA)

INITIAL ORG P IN SOIL = 3980.106 (KG/HA)	INITIAL ORG P IN SOIL = 3980.106 (KG/HA)
FINAL ORG P IN SOIL = 3769.930 (KG/HA)	FINAL ORG P IN SOIL = 3798.609 (KG/HA)
NO3 IN FERT = 0.000 (KG/HA)	NO3 IN FERT = 81.573 (KG/HA)
AMMONIA IN FERT = 0.000 (KG/HA)	AMMONIA IN FERT = 14.838 (KG/HA)
ORG N IN FERT = 0.000 (KG/HA)	ORG N IN FERT = 28.582 (KG/HA)
MINERAL P IN FERT = 0.000 (KG/HA)	MINERAL P IN FERT = 6.971 (KG/HA)
ORG P IN FERT = 0.000 (KG/HA)	ORG P IN FERT = 5.228 (KG/HA)
N REMOVED IN YIELD = 0.000 (KG/HA)	N REMOVED IN YIELD = 27.433 (KG/HA)
P REMOVED IN YIELD = 0.000 (KG/HA)	P REMOVED IN YIELD = 2.521 (KG/HA)
AMMONIA VOLATILIZATION = 0.000 (KG/HA)	AMMONIA VOLATILIZATION = 1.848 (KG/HA)
AMMONIA NITRIFICATION = 0.000 (KG/HA)	AMMONIA NITRIFICATION = 12.989 (KG/HA)
NO3 EVAP-LAYER 2 TO 1 = 0.509	NO3 EVAP-LAYER 2 TO 1 = 2.368