Water quality assessment and management of lowland river catchments

Dissertation

Zur Erlangung des Doktorgrades
der Mathematisch-Naturwissenschaftlichen Fakultät
der Christian-Albrechts-Universität zu Kiel
vorgelegt von

MSc. Lam Quang Dung

Department of Hydrology and Water Resources Management
Institute for the Conservation of Natural Resources
Kiel University, Kiel, Germany

April 2011

Referentin:	Prof. Dr. Nicola Fohrer
Koreferentin:	Prof. Dr. Natascha Oppelt
Koreferent:	

Tag der mündlichen Prüfung: 07 June 2011

Zum Druck genehmigt:

Der Dekan

Summary

This dissertation describes at first the hydrology and the long-term impact of point and diffuse source pollution on nutrient loads based on the current agricultural practices and sewage disposals in rural lowland catchments. The evaluations of Best Management Practices (BMPs) for water quality improvement was then implemented aiming at controlling and reducing pollution from point and diffuse sources in the entire catchments. The study catchments including Kielstau catchment (50 km²) and its subcatchment (7.6 km²), namely Moorau catchment, are located in the North German lowlands. These catchments are characterized by low hydraulic gradients, shallow groundwater, and flat topography. Sandy, loamy and peat soils are characteristic for these catchments. The water quality is not only influenced by the predominating agricultural land use in the catchments as cropland and pasture, but also by municipal wastewater treatment plants. The major environmental problems consist of nutrient losses from agricultural land resulting in water pollution in these study areas. Water quality models have proven to be a reliable tool for water quality assessment, scenario analysis, and decision-making. Model scenarios can be helpful in finding appropriate measures for assessing the water environmental status while taking into account climate, land use, soils, and water use. In this research a SWAT (Soil and Water Assessment Tool) model is used for the lowland catchments. SWAT is an ecohydrological model with the objective to predict the impact of land management practices on water, sediment and agricultural chemical yields on meso- and macroscale catchments.

The SWAT model was applied and calibrated and validated for daily flow, sediment, and nutrient loads in the Kielstau catchment first, and its performance capabilities were then tested in the Moorau catchment. The modeled results showed that SWAT performed satisfactorily in simulating daily flow, sediment, and nutrient loads at the catchment outlets, achieving the coefficient of determination (R^2) and the Nash-Sutcliffe Efficiency (E_{NS}) in the range of 0.42 to 0.84 for both the validation and calibration period. After set up and calibration, the model was used for scenario analysis in order to evaluate the cost and effectiveness of BMP implementation in reducing pollution load at the catchment outlets. Two approaches to structural and nonstructural BMPs including extensive land use management (ELUM), grazing management practice (GZM), field buffer strip (FBS), and nutrient management plan (NMP), were considered in this study. The results indicate that the implementation of BMPs in these lowland catchments would result in a significant reduction of nutrient loads at the watershed outlets in general, especially for nitrogen loads. This study also reveals that the implementation of a single BMP did not achieve the target value for water quality according to the European Water Framework Directive. The combination of BMPs improved significantly the water quality in the Kielstau catchment, achieving a 53.9 % and a 46.7% load reduction in nitrate and total nitrogen load, respectively, with annual implementation costs of 93,000 Euro. The results of spatial distribution of nutrient loads demonstrate that the SWAT model can be used to indentify crucial pollution areas within a watershed. This approach helps decision makers to improve suitable measures aiming at further controlling more effectively nutrient loads to water bodies.

Zusammenfassung

Diese Dissertation beschreibt zunächst die Hydrologie und die langfristigen Auswirkungen von Punkt- und diffusen Quellen auf die Nährstoffbelastungen, die auf den derzeitigen landwirtschaftlichen Praktiken und Abwassereinleitungen in ländlichen Tiefland-Einzugsgebieten basieren. Bewertungen von Best Management Practices (BMPs) für die Verbesserung der Wasserqualität wurden dann mit dem Ziel der Kontrolle und Verringerung der Umweltverschmutzung aus Punkt- und diffusen Quellen in gesamten Einzugsgebieten durchgeführt. Die Untersuchungsgebiete schließen das Kielstau-Einzugsgebiet (50 km²) und sein Teileinzugsgebiet (7,6 km²), das Moorau-Einzugsgebiet, in der Norddeutschen Tiefebene ein. Diese Einzugsgebiete werden durch geringe hydraulische Gradienten, oberflächennahes Grundwasser und flache Topographie charakterisiert. Sandige, lehmige und Torfböden sind charakteristisch für diese Einzugsgebiete. Die Wasserqualität wird nicht nur durch die vorherrschende landwirtschaftliche Nutzung im Einzugsgebiet wie Acker- und Weideland, sondern auch durch kommunale Kläranlagen beeinflusst. Die wesentlichen Umweltprobleme entstehen durch Nährstoffverluste aus landwirtschaftlichen Flächen und führen zu Wasserverschmutzung in diesen Untersuchungsgebieten. Wasserqualitätsmodelle haben sich als zuverlässiges Werkzeug für die Bewertung der Wasserqualität, Szenario-Analyse und Entscheidungsfindung bewährt. Modell-Szenarien können hilfreich sein bei der Suche nach geeigneten Maßnahmen für die Beurteilung des Umweltzustands des Wassers unter Berücksichtigung von Klima, Landnutzung, Boden und Wassernutzung. In dieser Untersuchung wurde ein SWAT (Soil and Water Assessment Tool)-Modell für die Tiefland-Einzugsgebiete verwendet. SWAT ist ein ökohydrologisches Modell mit dem Ziel, die Auswirkungen der Landbewirtschaftung auf Wasser, Sedimente und landwirtschaftlichen Ertrag in meso- und makroskaligen Einzugsgebieten vorherzusagen.

Das SWAT-Modell wurde zunächst angewandt, kalibriert und validiert für tägliche Abflüsse, Sediment- und Nährstofffrachten im Einzugsgebiet der Kielstau, und seine Leistungsfähigkeit wurde dann in dem Moorau-Einzugsgebiet getestet. Die modellierten Ergebnisse zeigten, dass SWAT zufriedenstellend die täglichen Abflüsse, Sediment- und Nährstofffrachten an den Einzugsgebietsauslässen simuliert, und dabei ein Bestimmtheitsmaß (R^2) und eine Nash-Sutcliffe-Effizienz (ENS) im Bereich von 0,42 bis 0,84 für den Validierungs- und Kalibrierungszeitraum erreicht. Nach Aufbau und Kalibrierung wurde das Modell für Szenario-Analysen verwendet, um die Kosten und die Effektivität von BMPs zur Verringerung der Verschmutzungsbelastung an den Einzugsgebietsauslässen zu untersuchen. Zwei Ansätze zu strukturellen und nicht-strukturellen BMPs einschließlich von extensiven Landnutzungsmanagements (ELUM), Weidewirtschaftsmanagements (GZM), Uferrandstreifen (FBS) und Nährstoff-Management-Plänen (NMP), wurden in dieser Studie berücksichtigt. Die Ergebnisse zeigen, dass die Einführung von BMPs in diesen Tiefland-Einzugsgebieten zu einer erheblichen Reduzierung der Nährstoffbelastung an den Einzugsgebietsauslässen im allgemeinen, aber vor allem für Stickstofffrachten, führen würde. Diese Studie zeigt auch, dass die Einführung einer einzelnen BMP den Zielwert für die Wasserqualität nach der Europäischen Wasserrahmenrichtlinie nicht

erreicht. Die Kombination von BMPs verbessert deutlich die Wasserqualität im Einzugsgebiet der Kielstau und erreicht jeweils eine 53,9% und eine 46,7% Reduzierung in Nitrat- und Gesamt-Stickstoff-Fracht, mit jährlichen Implementierungskosten von 93.000 Euro. Die Ergebnisse der räumlichen Verteilung der Nährstofffrachten zeigen, dass das Modell SWAT verwendet werden kann, um kritische Verschmutzungsbereiche innerhalb eines Einzugsgebiets zu identifizieren. Dieser Ansatz hilft Entscheidungsträgern, geeignete Maßnahmen zur weiteren effizienteren Kontrolle von Nährstoffbelastungen der Gewässer zu verbessern.

Table of Contents

Chapter IV The impact of agricultural Best Management Practices on water quality in a North German lowland catchment

Environmental Monitoring and Assessment
Submitted 20.05.2010, Accepted 08.02.2011, Published 11.03.2011

Chapter V Assessing the spatial and temporal variations of water quality in lowland areas, Northern Germany
Journal of Environmental Management
Submitted 05.04. 2011

List of Figures

List of Tables

Chapter I Introduction

The impairment of water quality due to point and diffuse source pollution is becoming an increasing global concern. Agriculture has been identified as the major contributor of diffuse source pollution of water resources (Humenik et al., 1987; Duda, 1993; Behrendt et al., 1999). Agricultural pollutants such as sediment, fertilizers, pesticides, salts and trace elements resulting from various activities, lead to the degradation of surface and ground water resources through soil erosion, chemical runoff and leaching (Donoso et al., 1999; Zalidis et al., 2002; Thorburn et al., 2003). Land application of manure provides nutrients and organic matter that enhance crop growth and can improve soil physical properties. However, excessive nutrient application can result in the degradation of water quality. Assessment of the environmental impact of diffuse source pollutants at a regional scale is a key component to achieve sustainably agricultural management.

Lowland catchments are ecosystems with low flow velocity, a high groundwater table, and flat topography (Schmalz et al., 2009; Krause et al., 2007; Müller et al., 2004). Hydrological conditions and nutrient dynamics of lowland river systems and the adjacent floodplains are strongly controlled by the interactions between surface water and shallow groundwater (Osman and Bruen, 2002; Sophocleous, 2002; Krause and Bronstert, 2004, 2005; Winter, 1999). The environmental goals for the lowland Kielstau catchment are to reduce the diffuse source pollution from the agriculture to the river so that nutrient concentrations could be met the target value for water quality according to the European Water Framework Directive (WFD) (EC, 2000). The WFD is a relatively new legislation that demands the good status of surface waters and groundwater to be accomplished until 2015 for all European Union member states. The main water management plans implemented by WFD are (a) classification of waters in relation to environmental quality targets, (b) evaluation of status and effects of measures on water quality, and (c) action plans including counter-measures and associated costs for the implementation of these measures.

Models can be helpful tools for assessing environmental ecological status (Grizzetti et al., 2003; Krysanova et al., 2005) and can be used for producing background information to the water management plans, since diffuse nutrient losses from agricultural land cannot be monitored and quantification of the effects of different measures is difficult without models. In recent years, commonly used agricultural watershed models are DRAINMOD (Skaggs, 1980), HSPF (Johanson et al., 1984), AGNPS (Yoon et al., 1993), MIKE SHE (Xevi et al., 1997), ANSWERS2000 (Bouraoui and Dillaha, 1996), SWAT (Arnold et al., 1998), SWIM (Krysanova et al., 1998). These models have gained wide spread acceptance as effective tools to assess the impact of agriculture on the quality of surface and groundwater. In the study catchments, monitoring combined with modeling can be used for further clarification of the relationship between agricultural management and water pollution.

1.1 Statement of the problems

The Kielstau catchment is classified as lowland-flood plain landscape in the North of Germany.

The water quality in the Kielstau catchment is strongly influenced by natural regulation functions regarding water balance, nutrient dynamics and subsequently floodplain ecology. In the past centuries different melioration measures such as river regulation have been implemented in order to render better cultivation conditions for agriculture and faster discharge. These have led to a change in the natural water and nutrient balance. Besides, the installation of tile drainage systems and open ditches (Figure 1.1) aiming at improving aeration conditions, allowing access to the fields with farm maschinery, and preventing frequent flooding in lowland areas resulting in faster and more intensive transport of water and nutrient to the river network.

Figure 1.1: Tile drainage and open ditch in the Kielstau catchment (Photo by Bieger, 2007)

Intensive agriculture within the last years is one of the main reasons resulting in enhancing water pollution. The excessive fertilizer application in the arable lands may cause large amount of nutrients accumulated in the soil and washed off by surface runoff or leached to groundwater. In addition, improper management of grazing activities and high livestock density can negatively affect water quality through nutrients from urine and manure dropped by the animals.

Additional pollution sources leading to high nutrient concentrations in this catchment are wastewater treatment plants (WWTPs). In the Kielstau catchment, there are six WWTPs located along the tributaries and the river Kielstau. Emission from these WWTPs is mostly originating from waste resources of residential areas. Moreover, one poultry farm sited in the near of the Moorau tributary also contributes to increase in nutrient concentration at the watershed outlet.

1.2 Study area

The study area Kielstau catchment is located in the Northern part of Germany as a typical example of lowland - flood plain landscape, region of Schleswig-Holstein (Figure 1.2). It has a size of approximately 50 km². The Kielstau catchment is part of the drainage area of the Treene, which is the most important tributary of the Eider River. The river rises from the eastern part of the catchment and flows through Lake Winderatt towards the gauge Soltfeld, situated at the outlet of the Kiestau catchment (Figure 1.3). The total length of the river Kielstau is about 17 km.

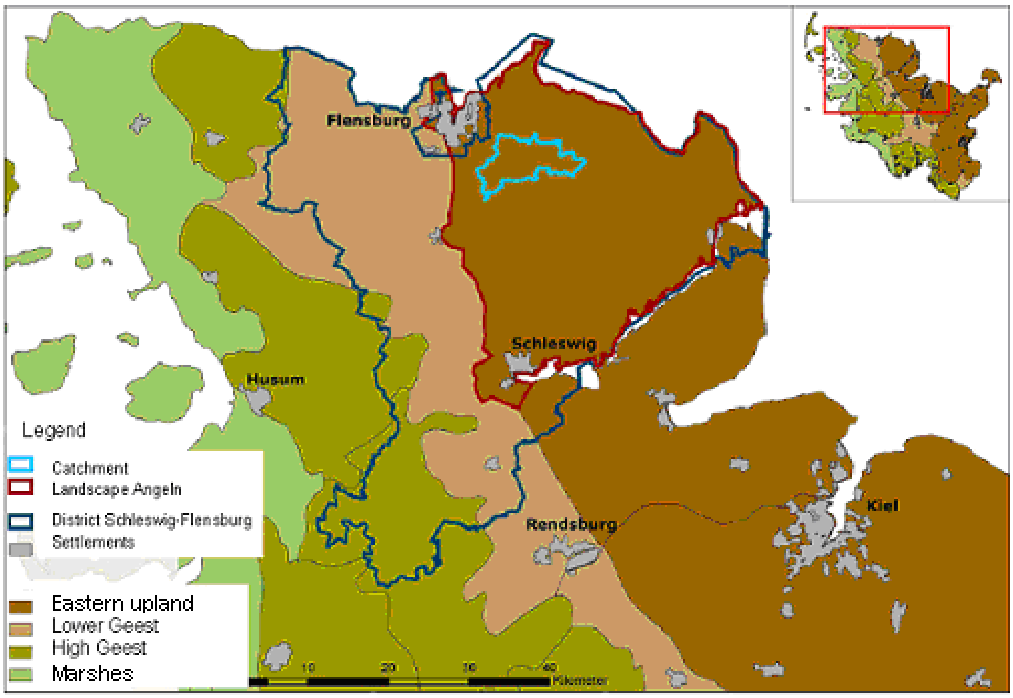


Figure 1.2: Location of study area (Map by Bieger 2007, data source: LVermA, 2005)

The stream network of the Kielstau catchment is formed by six tributaries and many other smaller tile drainage systems and open ditches that are connected directly to the main river or to the tributaries (Figure 1.3). Several tributaries and drainage ditches contribute to the main stream water flow and chemistry. The six WWTPs built within the Kielstau watershed are Husby, Hürup Nord, Hürup Weseby, Hürup Süd, Ausacker, and Freienwill (Figure 1.3). Husby is situated at the beginning of the Moorau tributary with 3000 population equivalents. Hürup Nord, Hürup Weseby, and Hürup Süd are located along the longitudinal Hennebach tributary (461, 447, and 240 population equivalents). Ausacker and Freienwill are located on the river Kielstau (1880 and 350 population equivalents, SÄDBUDL, 2009).

The altitude in the study area ranges from a minimum of 27.3 meters to a maximum of 79.9 meters (Figure 1.4). The mean altitude is 45 meters (extracted from LVermA, 1995). Land use is dominated by arable land and pasture. The arable land area occupies over 55%, and pasture over 26%, of the catchment area. The dominant soils of the Kielstau catchment are Stagnic Luvisols and Haplic Luvisols. The annual average precipitation is 841 mm/a (station Satrup, 1961-1990, DWD, 2009); the mean annual temperature is 8.2 ^{0}C (station Flensburg, 1961-1990, DWD, 2009).

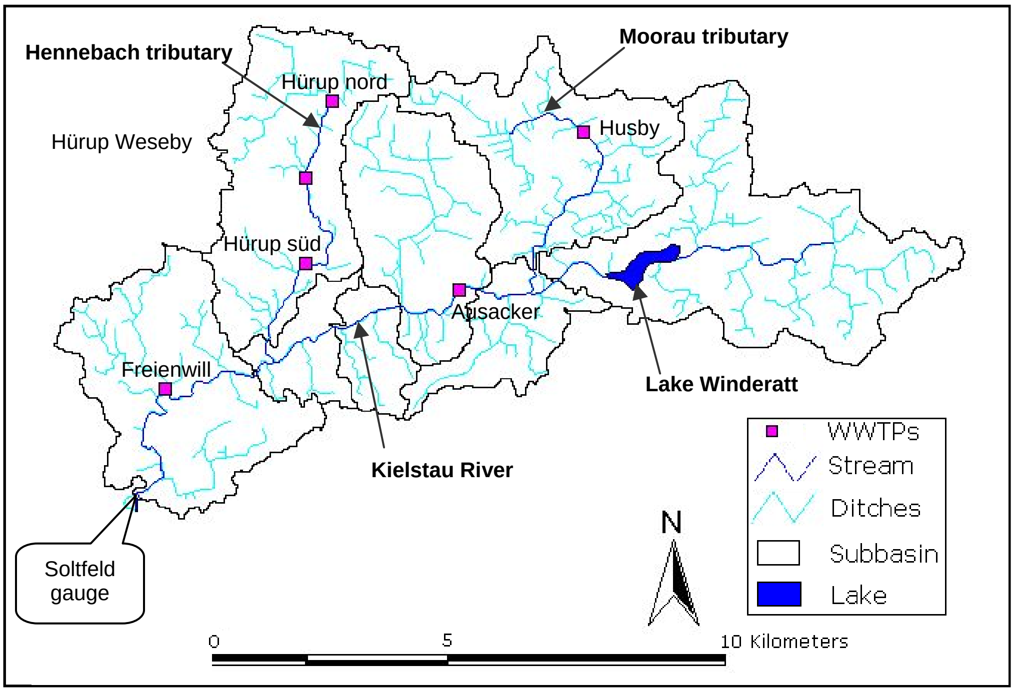


Figure 1.3: Kielstau catchment and its subcatchments, stream network, Soltfeld gauge station, and waste water treatment plants

The discharge of the Kielstau catchment is measured at the Soltfeld gauging station (Figure 1.5). The hourly discharge data were measured from this station by Staatliches Umweltamt Schleswig (2009).

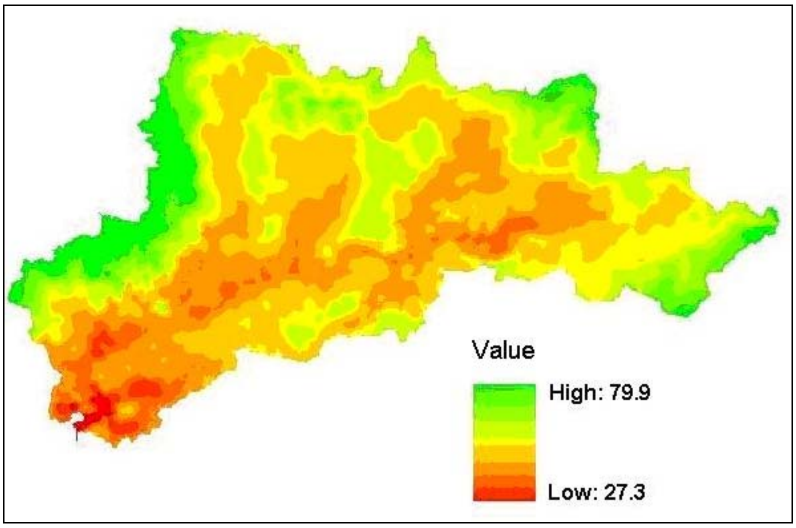


Figure 1.4: Topography of the study area (LVermA, 1995)

Figure 1.5 shows the daily measured data of the discharge for the period from Nov.1999–Nov.2008. As a consequence of the precipitation events and lower evapotranspiration, higher peaks of discharge commonly occur in the winter season. The amount of discharge varies depending on the seasons and the years. As can be seen from Figure 1.5, the lowest discharge in winter season occurs in the years 2000-2001 and 2005-2006, while the highest discharge occurs in the period from January to February 2002. According to the data measured at the Soltfeld gauging station in the time span from 1999 till 2008, discharge had its minimum measured value on 22nd of September 2003, with a value of 0.02 m³/s and its maximum on 28th of January 2002 with a flow of 3.14 m³/s. The average discharge calculated for the period from 1999–2008 is 0.42 m³/s.

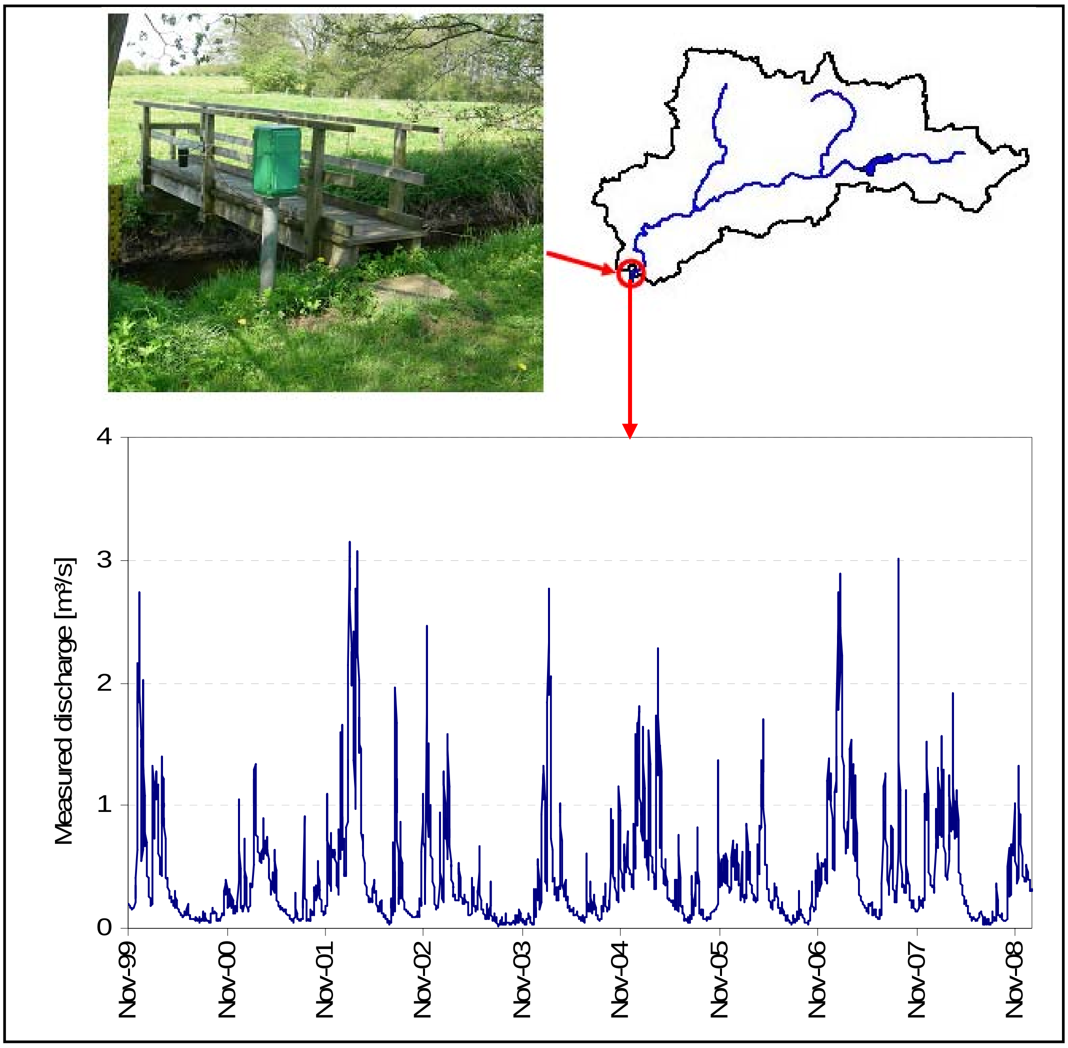


Figure 1.5: The location of Soltfeld gauging station and measured discharge in the Kielstau catchment (Staatliches Umweltamt Schleswig, 2009)

1.3 Objectives and outline

This research aims to evaluate the long-term impact of point and diffuse source pollution on nutrient load in lowland catchments using the ecohydrological SWAT model, and to identify the impacts of different land use management scenarios on diffuse source nutrients for delivery of water quality improvements as well as to select appropriate management scenarios based on the trade-off relationship between the effectiveness in nutrient reduction and the corresponding cost of BMPs implementation in the Kielstau catchment.

The dissertation is cumulatively organized as stand-alone manuscripts that are published or awaiting publication in international peer-reviewed journals. These papers are reproduced here unmodified except for cross-references.

Chapter I introduces the study area, the statement of problems, and the summary of objectives as well as the main contents which have been mentioned from chapters II - VI.

Chapter II focuses on the application of the SWAT model for sensitivity analyses, streamflow, and nitrate load prediction in a measoscale lowland catchment, which has special hydrological characteristics such as flat topography, shallow groundwater, low hydraulic gradients, and high potential for water retention in peatland and lakes in comparison with those of mountainous or urban catchments. The aim of this chapter is to identify the capacities of applying an ecohydrological model for simulating water balance and stream discharge, and to evaluate the impact of point and diffuse source pollution on nitrate load in the lowland Kielstau catchment. The model performance was verified by comparing simulated and measured daily discharge and nitrate loading for the period from 1998–2007 and from June 2005–December 2007, respectively. The results of flow and nitrate load were predicted by the SWAT model based on current agricultural practices and sewage disposals within the catchment.

Chapter III discusses the evaluation of the long-term impact of point and diffuse source pollution on nitrate load, and the influences of different land use cover types on nitrate load at different subbasins. In addition, the contribution of point and diffuse sources to nitrate load was also determined at the outlet of the Kielstau catchment. Additional measured flow and nitrate load for the period of 2008 were expanded and used to increase the validation period in this chapter.

Chapter IV presents the influences of point and diffuse sources pollution on nutrient loads at the catchment outlet, and the development of selection processes of the structural and nonstructural BMPs aiming at finding effective BMPs, which can mitigate the highest load of agricultural diffuse source pollution and be the most consistent with farming condition in this lowland area. The objectives of this chapter are the assessment of the long-term impact of point and diffuse source pollution on sediment and nutrient loads in a lowland catchment using the ecohydrological SWAT model, and the evaluation of cost and effectiveness of BMPs in minimizing the diffuse sources pollution within the watershed.

Chapter V concentrates on the evaluation of the SWAT model performance in simulating nutrient loads in the Kielstau catchment and its subbasin, Moorau catchment, and the assessment of parameter transferability in these lowland catchments as well as in other catchments with similar environmental conditions by comparing data sets resulting from auto-calibration. The main objectives of this chapter are the assessment of the transferability of parameter sets between lowland catchments by using the SWAT model, the evaluation of the temporal and spatial variations of nutrients in the whole catchment before and after implementation of BMPs, and the identification of crucial subbasins which provide great nutrient load compared to other subbasins within the watershed.

Chapter VI consisting of summary and conclusion draws key finding and important results from this study.

Chapter II Ecohydrological modelling of water discharge and nitrate loads in a mesoscale lowland catchment, Germany

Q. D. Lam, B. Schmalz, N. Fohrer

Advances in Geosciences, Volume 21 (2009), Pages 49 – 55.

Submitted 15.01.2009, Accepted 28.04.2009, Published 11.08.2009

Abstract

The aims of this study are to identify the capacities of applying an ecohydrological model for simulating flow and to assess the impact of point and non-point source pollution on nitrate loads in a complex lowland catchment, which has special hydrological characteristics in comparison with those of other catchments. The study area Kielstau catchment has a size of approximately 50 km^2 and is located in the North German lowlands. The water quality is not only influenced by the predominating agricultural land use in the catchment as cropland and pasture, but also by six municipal wastewater treatment plants.

Ecohydrological models like the SWAT model (Soil and Water Assessment Tool) are useful tools for simulating nutrient loads in river catchments. Diffuse entries from the agriculture resulting from fertilizers as well as punctual entries from the wastewater treatment plants are implemented in the model setup.

The results of this study show good agreement between simulated and measured daily discharges with a Nash-Sutcliffe efficiency and a correlation coefficient of 0.76 and 0.88 for the calibration period (November 1998 to October 2004); 0.75 and 0.92 for the validation period (November 2004 to December 2007). The model efficiency for daily nitrate loads is 0.64 and 0.5 for the calibration period (June 2005 to May 2007) and the validation period (June 2007 to December 2007), respectively. The study revealed that SWAT performed satisfactorily in simulating daily flow and nitrate loads at the lowland catchment in Northern Germany.

Keywords: Lowland hydrology, simulated discharge, nitrate load, SWAT

2.1 Introduction

The degradation of water quality due to non-point source and point source pollution is becoming an increasing global concern. In order to improve the quality of polluted water bodies, the European Framework Directive was implemented in the year 2000 to protect the various types of water bodies in question (EC, 2000). One of the main objectives of the European Framework Directive is for water bodies to achieve a good ecological state by 2015.

Lowland catchments are ecosystems with low flow velocity, a high ground water table, and flat topography (Müller et al., 2004; Krause et al., 2007; Schmalz et al., 2008). Agricultural practices such as fertilizer and pesticide use as well as sewages are main reasons causing the pollution of stream water in these catchments in Northern Germany. Furthermore the installation of artificial drainage systems and pumping stations have changed the natural water balance considerably and influenced the in-stream water quality due to an accelerated nutrient transport (Schmalz et al., 2008). Many studies have observed that installation of drainage ditches increases the leaching of nutrients. Adamson et al. (2000) reported small changes in nitrate concentrations by installing drainage ditches in blanket peat. Evans et al. (1995) have suggested the implementation of controlled drainage as management practices to minimize nitrate losses. David et al. (1997) have found high nitrate concentrations with the range of 5 to 49 mg/l in drainage tiles in an gricultural catchment area in Illinois.

For the prediction of hydrological processes and nutrient loads, simulation models that describe the water and nutrient dynamics might be considered as useful tools. A number of ecohydrological models have already been used in lowland catchments: the IWAN model (Krause and Bronstert, 2005) was used for modeling water balance and nutrient dynamics of floodplains. Hattermann et al. (2006) integrated wetlands and riparian zones into SWIM (Krysanova et al., 1998) to determine their influence on water and nutrient fluxes. The SWAT model (Arnold et al., 1998) has been widely used all around the world to predict stream discharge and nutrient loads from various sizes of watersheds (Tripathi et al. 2004). Borah and Bera (2003) found that SWAT was the most useful for long-term simulation in predominantly agricultural watersheds when they compared eleven hydrologic and non-point source pollution models. In addition, the computational efficiency of SWAT is convenient for parametric adjustment and multiple simulations implemented in minimal time (Arnold and Fohrer, 2005).

The objective of this paper is to evaluate the performance of the SWAT model in simulating water balance and stream discharge in a complex lowland catchment which has special hydrological characteristics in comparison with those of other catchments, and to predict the impact of point and nonpoint source pollution on nitrate loads based on current agricultural practices and sewage disposals at the watershed outlet.

2.2 Materials and methods

2.2.1 Study area

The study area Kielstau catchment is located in Northern Germany as part of a lowland area of Schleswig-Holstein (Figure 2.1). The area of the Kielstau catchment is about 50 km^2. Land use is dominated by arable land and pasture. The arable land area occupies over 55%, pasture accounts for 26% of the total area and urban area makes up over 3% (Table 2.1). The soils of the rural catchment are sandy or loamy, and the river valleys are characterized by peat soils.

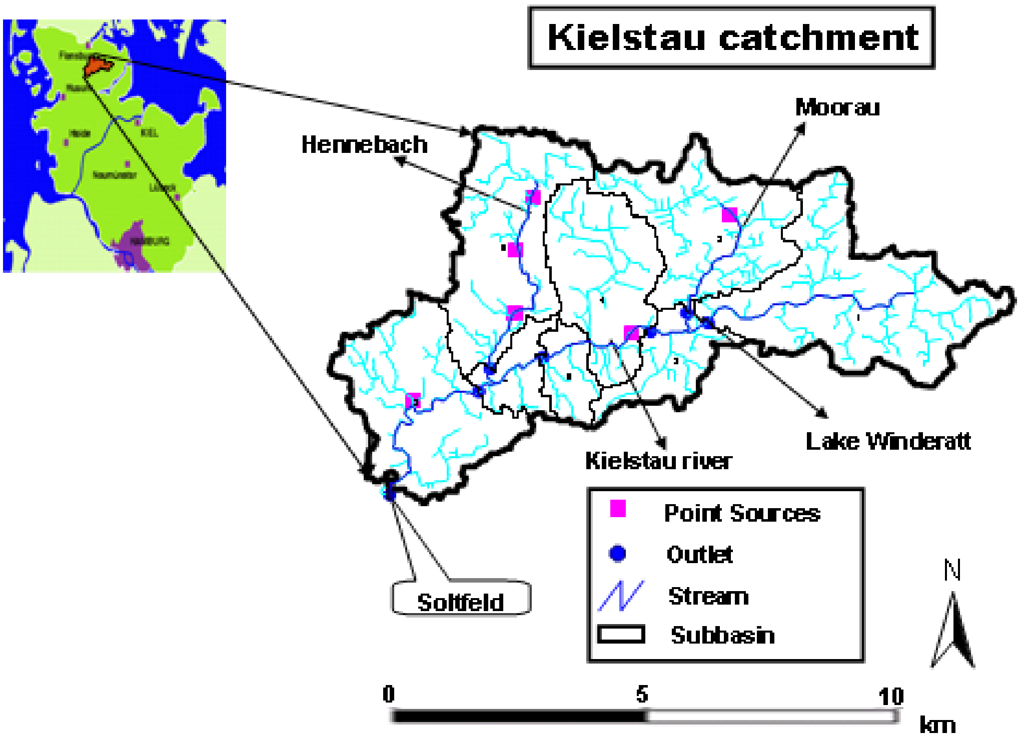


Figure 2.1: Location of the Kielstau catchment and its subbasins in chleswig-Holstein, Northern Germany

The Kielstau River has a total length of 17 km and flows through Lake Winderatt towards the gauge Soltfeld, located at the outlet of the Kielstau catchment (Figure 2.1). There are two important tributaries of the Kielstau River from the north, the Moorau and the Hennebach, and wastewater treatment plants have been built in both (Figure 2.1). Specifically, one wastewater treatment plant has been built in Moorau tributary, three others in Hennebach tributary and two others in the Kielstau River; all of which can have a remarkable influence on the water quality of the Kielstau River downstream (Schmalz et al., 2007). In addition various small tributaries and water from drainage pipes and ditches flow into the Kielstau River. The drainage fraction of agricultural area in the Kielstau catchment is estimated at 38% (Fohrer et al., 2007).

Table 2.1 Characteristics of the Kielstau catchment. Information on topography is derived from the DEM, Climate data are taken from Meierwik station (Deutscher Wetterdienst DWD, 2008), and Land use distribution is taken from Deutsches Zentrum für Luft- und Raumfahrt (DLR, 1995)

Parameter		Value	Parameter		Value
Topography			*Land use class*		5
	Area	50 km²	*Land use, % of area (>2%)*		
	Maximum height difference	49 m		Agriculture	55.82 %
	Mean river slope	1 %		Pasture	26.14 %
	Main river length	17 km		Range Bush	5.64 %

				Forest Deciduous	8.62 %
Climate				Urban	3.13 %
	Annual precipitation	841 mm	*Dominant crops*	Wheat, Rape, Maize	
	Mean annual evapotranspiration	400 mm	*Demography*		
	Mean annual temperature	8.2 ^{0}C		Population	85.955

2.2.2 The SWAT model

The ecohydrological model SWAT (Soil and Water Assessment Tool, Arnold et al., 1998, version 2005) was applied to simulate both the water balance and the nitrate loads in this complex hydrological catchment. SWAT is a semi-distributed, process-oriented hydrological model. It is a continuous time model which simulates water and nutrient cycles with a daily time step. The SWAT model represents the large-scale spatial heterogeneity of the study area by dividing the watershed into subbasins. The subbasins are then further subdivided into hydrologic response units (HRUs) that are assumed to consist of homogeneous land use and soils. Major components of the model include hydrology, weather, and agricultural management. The details of all components can be found in Arnold et al. (1998) and Neitsch et al. (2002).

In the SWAT model, soil water content, surface runoff, nutrient cycles, crop growth and management practices are simulated for each HRU and then aggregated for the subbasin by a weighted average. The model's hydrological components are comprised of surface runoff, percolation, lateral flow, ground water, evapotranspiration and channel transmission loss. Surface runoff volume is estimated using a modification of Soil Conservation Services (SCS) curve number method (Williams and Laseur, 1976).

The soil profile is subdivided into multiple layers including infiltration, evaporation, plant uptake, lateral flow, and percolation to lower layers. SWAT offers various methods to estimate the potential evapotranspiration (PET) such as Hargreaves, Penman-Monteith, and Priestley. The Penman-Monteith method was chosen in this study because the PET evaluation is based on the basic data such as solar radiation, wind speed, air temperature and relative humidity, whereas wind speed is not considered by the Hargreaves and Priestley methods. The model computes evaporation from soils and plants separately. Potential soil water evaporation is predicted as a function of potential evapotranspiration and leaf area efficiency, while actual soil water evaporation is predicted by using exponential functions of water content and soil depth. Plant transpiration is predicted as a linear function of potential evapotranspiration and leaf area efficiency.

SWAT simulates the nitrogen cycle in the soil profile and in the shallow aquifer (Neitsch et al., 2002). In soil and water, nitrogen is extremely reactive and exists in a number of dynamic forms. It may be added to the soil by rain, mineral and organic fertilizers, residue application, and bacteriological fixation. It can be removed from the soil through plant consumption, soil erosion, leaching, volatilization and denitrification. In the SWAT model, there are five different pools of nitrogen in the soil. Two pools (NH_4^+, NO_3^-) are inorganic forms of nitrogen, while the other three pools are organic forms of nitrogen. Nitrate may be transported with surface runoff, lateral flow or percolation. The amount of nitrate moving with the water is calculated by multiplying the concentration of nitrate in the mobile water by the volume of water moving in each pathway.

Since nitrate fluxes strongly depend on water fluxes, parameters controlling water balance were calibrated as the first step, and only then were nitrate loads considered. The application of the model first involved the analysis of parameter sensitivity, which was then used for model auto-calibration following the hierarchy of sensitive model parameters. The sensitivity analysis method (Morris, 1991) was conducted and aims to assess the most sensitive parameters for setting up the model in this catchment (Table 2.2). Model auto-calibration was performed by changing each parameter ten times within the allowable range of values for the specific parameter. Detailed calibration procedures for SWAT model and the definitions of various calibration parameters are described by Neitsch et al. (2002).

Table 2.2 Sensitivity analysis results

Rank	Parameters	Description*	Rank	Parameters	Description*
1	GWQMN	Threshold depth of water in shallow aquifer	8	SOL_Z	Depth from soil surface to bottom of layer
2	RCHRG_DP	Deep aquifer percolation coefficient	9	CH_K_2	Channel effective hydraulic conductivity
3	ALPHA_BF	Base flow recession constant	10	SURLAG	Surface runoff lag coefficient
4	SOL_AWC	Available water capacity	11	SOL_K	Saturated hydraulic conductivity
5	ESCO	Soil evaporation compensation factor	12	CN_2	Curve number
6	GW_REVAP	Groundwater revap coefficient	24	EPCO	Plant uptake compensation factor
7	GW_DELEY	Delay time for aquifer recharge	28	BLAI	Maximum potential leaf area index

* Detailed description is available at http://www.brc.tamus.edu/swat/swatdoc.html (Neitsch et al., 2005).

The auto-calibration was carried out using daily flow data of the hydrological years 1998–2004. The validation was done for the continuous time 2004–2007. For the nitrate loads simulation, the manual calibration was performed for a period of two years (June 2005–May 2007) after which the validation was done for a period of six months (June 2007–December 2007). The performance of the model was evaluated by the Nash-Sutcliffe efficiency (Nash and Sutcliffe, 1970), correlation coefficient, and root mean square error to determine the quality and reliability of the predictions compared to measured values.

The performance of the model was evaluated by the Nash-Sutcliffe efficiency (Nash and Sutcliffe, 1970), correlation coefficient, and root mean square error to determine the quality and reliability of the predictions compared to measured values.

Table 2.3 Model input data sources for the Kielstau watershed

Data type	Source	Data description/properties
Topography	LVermA, 1995	Digital elevation model, a grid size of 25 m×25 m
Soil map	BGR, 1999	Soil physical properties such as texture, saturated conductivity, etc. Scale of soil map (1:200 000)
Land use map	DLR, 1995	Land use classifications, 25 m×25 m resolution
Climate data	DWD, 2008	Temperature, precipitation, wind speed, humidity, (Meierwik station, 1993–2007)
Sewage disposal	Kreis Schleswig-Flensburg, 2007	Sewage disposal (point sources) data of 6 waste water treatment plants (Figure 2.1)
Hourly discharge	Staatliches Umweltamt Schleswig, 2008	Hourly discharge data of Kielstau at gauge Soltfeld, (1993–2007)

2.2.3 Input data

The main input data used for the SWAT model are shown in Table 2.3. A three-year crop rotation (winter wheat – winter wheat – rape) and monocultural maize were simulated on arable land. Fertilizer application and cultivation schedule are in conformance with the conventional cultivation (Table 2.4). Measured nitrate concentrations from June 2005 to December 2007 (weekly data for the year 2005 and daily data for the years from 2006 to 2007) were collected and analyzed by the Department of Hydrology and Water Resources Management-Ecology Centre at Kiel University (Tavares, 2006 and Bieger, 2007). Data of average monthly point source effluents from January 2002 to December 2007 (Kreis Schleswig-Flensburg, 2007) were implemented as point sources in the model.

Table 2.4 Crop types and fertilization for different land use classes of the Kielstau catchment

Crop rotation (3 years): winter wheat - winter wheat – rape Monocultural maize		
Crops	Date of fertilizer application	Total fertilization
Winter wheat	15. March; 15. April; 15. June	220 (kg N/ha); 240(kg manure/ha)
Rape	20. March; 15. April	200 (kg N/ha)
Maize	20. March; 01. May; 10. June; 10. August	180 (kg N/ha); 300(kg manure/ha)
Pasture	15. March; 30. May; 10. July; 25. August	160 (kg N/ha)
Range bush	01. March	80 (kg N/ha)

2.3 Results and discussion

Parameters that significantly affected water balance have been adjusted in their values in order to provide the best fit between the measured and simulated data by the auto-calibration tool of the model. These parameters include the SCS runoff curve number for moisture condition II, available water capacity, soil evaporation compensation factor, groundwater parameters, and channel effective hydraulic conductivity. For instance, soil evaporation compensation factor, the groundwater revap coefficient, and depth-to- subsurface drain have been adjusted from the default values of 1; 0.02; 0 to the simulated values of 0.95; 0.2; 800, respectively. Groundwater revap coefficient is an important parameter controlling the upwelling of groundwater into the unsaturated soil zone. The allowable range of this parameter is between 0.02 and 0.2. As the groundwater revap coefficient approaches 0, movement of water from the shallow aquifer to the root zone is restricted. For the Kielstau catchment, this parameter was changed from its initial value of 0.02 to 0.2 in order to obtain a better fit of the model results to the measured data. The auto-calibration processes were also implemented similarly for other parameters within their allowable range in SWAT.

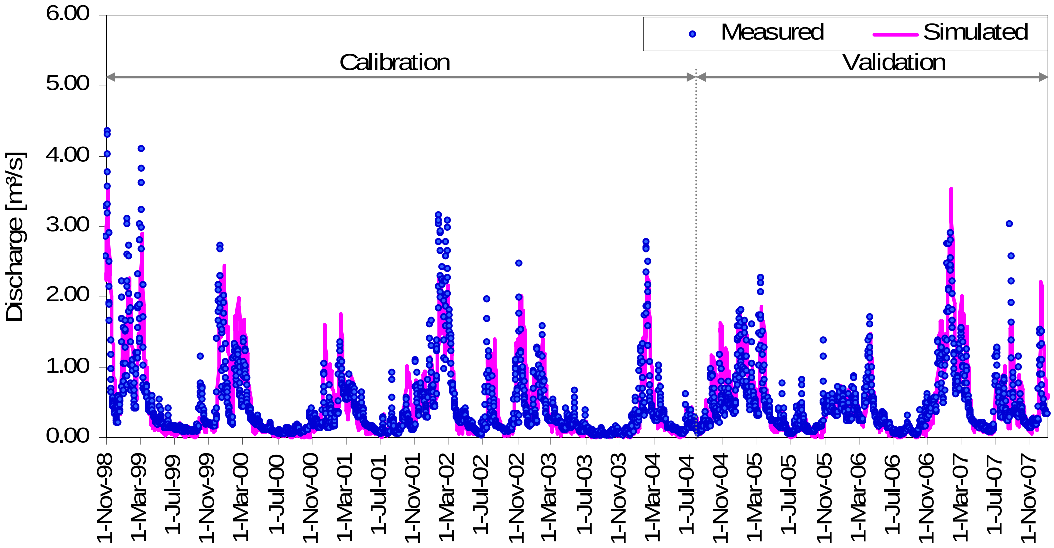


Figure 2.2: Measured and simulated daily discharge at the Kielstau catchment outlet, gauge Soltfeld (Nash-Sutcliffe efficiency and correlation coefficient of 0.76 and 0.88 for the calibration period; 0.75 and 0.92 for the validation period)

Figure 2.2 shows good agreement between simulated and measured daily discharge with a Nash-Sutcliffe efficiency and a correlation coefficient of 0.76 and 0.88 for the calibration period and 0.75 and 0.92 for the validation period at the outlet of Kielstau catchment, respectively. Overall, the model performance was satisfactory in both calibration and validation periods in daily time step.

The model results for daily nitrate loads at the gauge Soltfeld are illustrated in Figure 2.3. Parameters which remarkably impacted the nitrate concentrations such as humus mineralization (CMN), Nitrogen percolation coefficient (NPERCO), and residue mineralization (RSDCO) have been manually adjusted so that the prediction corresponds to the measured values during the calibration period (Table 2.5).

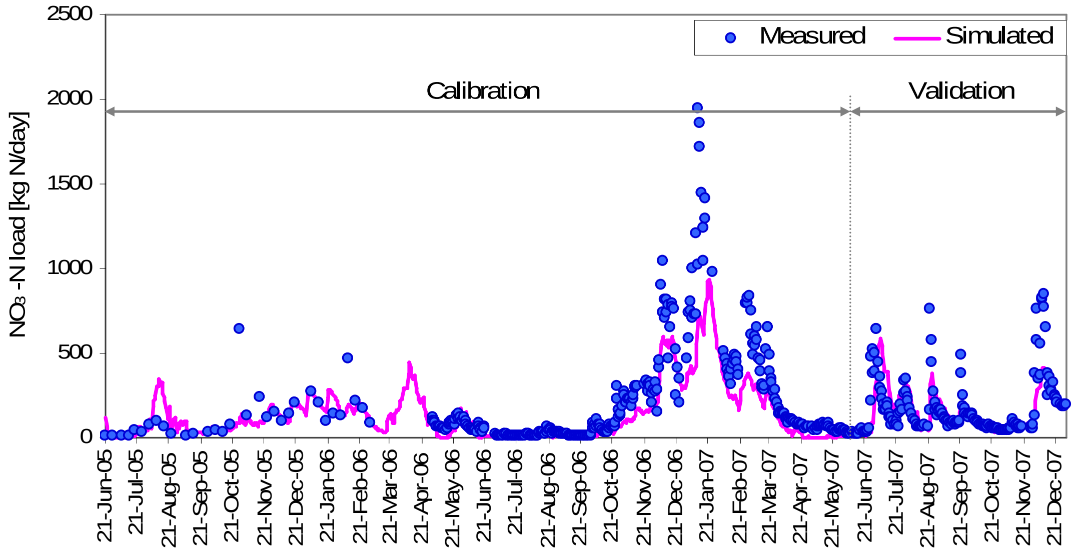


Figure 2.3: Measured and simulated daily nitrate loads at the Kielstau catchment outlet, gauge Soltfeld (Nash-Sutcliffe efficiency, correlation coefficient, and root mean square error of 0.64, 0.86, and 96.9 for the calibration period; 0.5, 0.71, and 67.5 for the validation period)

During the summer periods of 2005, 2006, and 2007 the model simulated well for both the range and the dynamic of the nitrate loads in general. In contrast, the model underpredicted the nitrate loads during the winter periods. This can be attributed to the following main reasons: Firstly, the underestimation of some peak flows led to the underestimation of the corresponding nitrogen peaks. Secondly, the higher nitrogen concentrations in the winter caused by nitrogen mobilization from the catchment are not represented in the model. Thirdly, the lack of plant uptake which causes accumulation of leachable nitrate resulted in increasing nitrogen concentrations in stream flow during the winter period.

For the winter period of 2005, the disagreement between measured and simulated nitrate loads was due to the underestimation of discharge. Furthermore the measured data of nitrate loads were only collected once a week in this period, while the model outputs were daily nitrate loads. The marked difference of weekly and daily resolution may influence the model efficiency in the whole simulation period.

Table 2.5 Calibrated parameters with SWAT model for simulating nitrate loads

Parameter	Process	Initial	Final
CMN	Humus mineralization	0.0003	0.002
NPERCO	Nitrate transport	0.2	0.95
RSDCO	Residue mineralization	0.01	0.05
BIOMIX	Biological mixing efficiency	0.2	0.1

The Nash-Sutcliffe efficiency and the correlation coefficient of the nitrate model are 0.64 and 0.86 for the calibration period and 0.5 and 0.71 for the validation period, respectively. These results are in accordance with previous studies using SWAT on various catchments. Bieger (2007) simulated daily nitrate loads on the same catchment and obtained a Nash-Sutcliffe efficiency and a correlation coefficient of 0.55 and 0.84 for the calibration period of June 2005–October 2006, respectively. However, differing from the present study, point source effluents were used as input data for constant daily loading by Bieger (2007). Grizzetti et al. (2003) obtained a Nash-Sutcliffe efficiency of 0.51 when they used SWAT to model diffuse emissions and retentions of nutrients on daily time step at the Vantaanjoki watershed (1680 km2), which is situated in Southern Finland and classified as a lowland catchment. Behera and Panda (2006) concluded that SWAT simulated nitrate concentration satisfactorily throughout the entire rainy season based on comparisons with daily observed data from an agricultural watershed located in eastern India. They obtained a Nash-Sutcliffe efficiency and a correlation coefficient of 0.92 and 0.93 for the calibration period, respectively.

With the above outlined results, we achieved a comparable simulation efficiency of daily nitrate loads. The simulated results indicated that the SWAT model was applied to simulate seasonable daily nitrate loads in the Kielstau catchment.

2.4 Conclusions

The ecohydrological SWAT model has been used to simulate both water balance and nitrate loads from different point and non-point sources in the mesoscale Kielstau catchment – a typical lowland area in Northern Germany. The basis input data comprises climate, land use, soil, topography, and current agricultural data as well as sewage disposals of six wastewater treatment plants were used in this study to predict the current nitrate loads. The simulated flow and nitrate loads were compared with corresponding in-stream measurements at the Soltfeld station of Kielstau catchment. The results of this study show good agreement between simulated and measured daily discharges with a Nash-Sutcliffe efficiency and a correlation coefficient of 0.76 and 0.88 for the calibration period and 0.75 and 0.92 for the validation period. The statistical coefficients of the nitrate model performance were relatively reasonable (Nash-Sutcliffe efficiency, correlation coefficient of 0.64, 0.86 for the calibration period; 0.5, 0.71 for the validation period, respectively) and demonstrate that SWAT results are reliable at a daily time scale for nitrate loads. Overall, SWAT performed satisfactorily in simulating both daily flow and nitrate loads at the Kielstau catchment.

In our ongoing research, the measured data of nitrate concentration will be continuously expanded and used to increase the validation period of nitrate loads. It is expected that a wider range of data will prove helpful in more clearly understanding the trend of nitrate loads. Furthermore, different management practice scenarios will be considered with the goal of minimizing nitrate loads in the long term.

Chapter III Modelling point and diffuse source pollution of nitrate in a rural lowland catchment using the SWAT model

Q. D. Lam, B. Schmalz, N. Fohrer

Agricultural Water Management

Submitted 15.01.2009, Accepted 06.10.2009, Published 01.02.2010

Abstract

The assessments of potential environmental impacts of point and diffuse source pollution at regional scales are necessary to achieve the sustainable development of natural resources such as land and water. Nutrient related diffuse source pollutant inputs can enhance crop growth and improve soil eutrophication. However, excessive nutrient input can result in the impairment of water quality. The objectives of this study were to evaluate the long-term impact of point and diffuse source pollution on nitrate load in a lowland catchment using the ecohydrological model SWAT (Soil and Water Assessment Tool) and to determine the contribution of point and diffuse sources to nitrate load in the entire catchment.

The study area Kielstau catchment has a size of approximately 50 km^2 and is located in the North German lowlands. The water quality is not only influenced by the predominating agricultural land use in the catchment as cropland and pasture, but also by six municipal wastewater treatment plants. Diffuse entries as well as punctual entries from the wastewater treatment plants are implemented in the model set-up. The model was first calibrated and then validated in a daily time step. The values of the Nash- Sutcliffe efficiency for the simulations of flow and nitrate load range from 0.68 to 0.75 for the calibration period and from 0.76 to 0.78 for the validation period. These statistical results revealed that the SWAT model performed satisfactorily in simulating daily flow and nitrate load in lowland catchment of Northern Germany. The results showed that diffuse sources are the main contributor to nitrate load in the entire catchment accounting for about 95% of the total nitrate load, while only 5% results from point sources. The model results also indicated that agriculture is the dominant contributor of diffuse sources and the percentage of agricultural land area is considerably positively correlated to nitrate load at the different subbasins. The area covered by forest is found to be negatively correlated with nitrate load.

Keywords: Lowland hydrology, point and diffuse sources, nitrate load, SWAT

3.1 Introduction

Lowland catchments are ecosystems with low flow velocity, a high groundwater table, and flat topography (Schmalz et al., 2007; Krause et al., 2007; Müller et al., 2004). In the past centuries different melioration measures such as river regulation and pumping stations have been implemented in order to enlarge areas as well as render better cultivation conditions for agriculture. These have led to a change in the natural water and nutrient balance, which has contributed to eutrophication problems and ecological damage of lakes and river network systems. Besides, tile drainage is also a common agricultural practice aimed at improving aeration conditions and moisture in lowland areas. It reduces the retention time of water in the soil and hence forms an important pathway for nitrate to surface water bodies (Tiemeyer et al., 2006; Kladivko et al., 1999). Many studies have stated that the installation of drainage ditches increases the leaching of nutrients. Adamson et al. (2000) reported small changes in nitrate concentrations by installing drainage ditches in blanket peat. Evans et al. (1995) have suggested the implementation of controlled drainage as a management practice to minimize nitrate losses. David et al. (1997) have found high nitrate concentrations with the range of 5-49 mg/l in drainage tiles in an agricultural catchment area in Illinois. On the other hand, these areas are also affected by additional human interferences such as agricultural practices (e.g. fertilizer, pesticides utilization) and point source emissions, which also have relevant influence on nutrient load and water quality. Some other studies have found that pollutants such as fertilizers, pesticides and sediment, resulting from various agricultural practices, lead to the degradation of surface and ground water (Donoso et al., 1999; Zalidis et al., 2002).

Knowing these problems, environmental regulations and new agricultural policies have been developed in several European countries in order to mitigate the negative impact of diffuse source pollution, to protect the stream habitat from eutrophication in general as well as to attain a better ecological status of lowland catchments in particular. Through the Nitrate Directive, the European Commission was established to protect water pollution caused by nitrates from agricultural sources (EC, 1991; 91/676/ EEC), and in the year 2000 the European Framework Directive was implemented to protect the various types of water bodies in question (EC, 2000; 2000/60/EC). One of the main objectives of the European Framework Directive is for water bodies to achieve a good ecological state by 2015.

The evaluation of new regulations as well as degradation of water bodies caused by point and diffuse source pollution requires modelling studies in order to assess the impact of those new policies as well as water pollution on the surrounding environment. Model scenarios can be helpful in finding reasonable measures for assessing environmental ecological status while taking into account relevant factors such as climate, land, and water use (Krysanova et al., 2005; Højberg et al., 2007). In recent years, a large number of models have been developed to generalize the effect of environmental conditions and agricultural practices on nutrient losses on field and catchment scale. Models like DRAINMOD (Skaggs, 1980), AGNPS (Yoon and Disrud, 1993), ANSWERS2000 (Bouraoui and Dillaha, 1996) have been used to simulate flow and nutrient movement within a watershed.

For the lowland catchments, a number of ecohydrological models have already been used such as: the IWAN model (Krause and Bronstert, 2005), which was used for modelling water balance and nutrient dynamics of floodplains. Hattermann et al. (2006) integrated wetlands and riparian zones into SWIM (Krysanova et al., 1998) to determine their influence on water and nutrient fluxes. The SWAT model (Arnold et al., 1998) has been widely used all around the world to predict stream discharge and nutrient load from watersheds of various sizes (Saleh et al., 2000; Tripathi et al., 2004; Gassman et al., 2006; Lenhart et al., 2003). Borah and Bera (2003)found that SWAT was the most useful for long-term simulation in predominantly agricultural watersheds when they compared eleven hydrologic and diffuse source pollution models. In addition, the computational efficiency of SWAT is convenient for parametric adjustment and multiple simulations implemented in minimal time (Arnold and Fohrer, 2005).

Therefore, the aims of this study were: (1) the evaluation of the long-term impact of point and diffuse source pollution on nitrate load in a lowland catchment which has special hydrological characteristics such as flat topography, upwelling of groundwater, low hydraulic gradients, and high potential for water retention in peatland and lakes in comparison with those of mountainous or urban catchments using the ecohydrological SWAT model, (2) the determination of point and diffuse sources contribution to nitrate load, and (3) the influences of different land use cover types on nitrate load at different subbasins.

3.2 Materials and methods

3.2.1 Study area

The study area Kielstau catchment is located in Northern Germany as part of a lowland area in Schleswig-Holstein (Figure 3.1). The area of the Kielstau catchment is about 50 km^2. The river Kielstau has a total length of 17 km and flows through Lake Winderatt towards the gauge Soltfeld, located at the outlet of Kielstau catchment. There are two important tributaries of the river Kielstau from the north, the Moorau and the Hennebach. The six wastewater treatment plants built within the Kielstau watershed are Husby, Hürup Nord, Hürup Weseby, Hürup Süd, Ausacker, and Freienwill (Figure 3.1). Husby is situated at the beginning of the Moorau tributary with 3000 population equivalents. Hürup nord, Hürup Weseby, and Hürup süd are located along the longitudinal Hennebach tributary (461, 447, and 240 population equivalents). Ausacker and Freienwill are located on the river Kielstau (1880 and 350 population equivalents). In addition, various small tributaries and water from drainage pipes and ditches flow into the river Kielstau. The drainage fraction of agricultural area in the Kielstau catchment is estimated at 38% (Fohrer et al., 2007). The maximum height difference is 49 m. The precipitation is about 841 mm/a (station Satrup, 1961–1990, DWD, 2009a,b); the mean annual temperature is 8.2 8C (station Flens- burg, 1961–1990, DWD, 2009a,b).

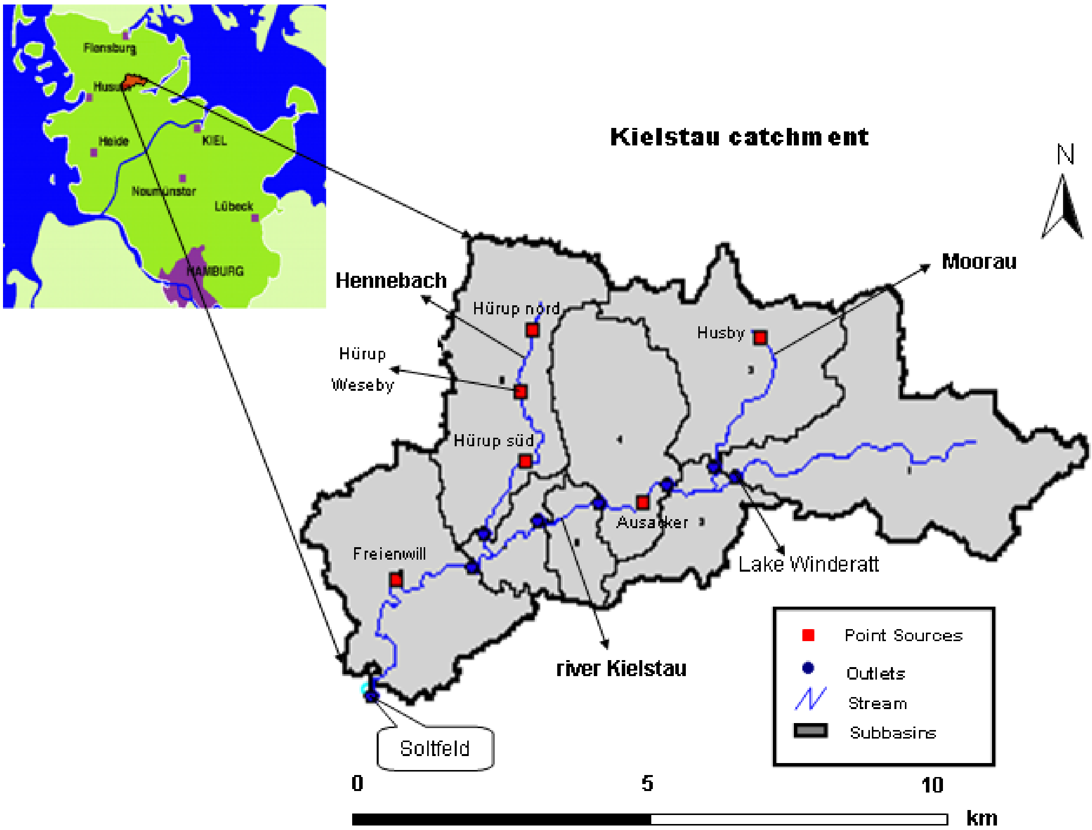


Figure 3.1: Location of the Kielstau catchment and its subbasins in Schleswig-Holstein, Northern Germany

Land use is dominated by arable land and pasture. The arable land area occupies over 55%, and pasture over 26%, of the catchment area. The dominant soils of the Kielstau catchment are Stagnic Luvisols and Haplic Luvisols. Land use and soil map used in this study can be seen in Figure 3.2.

In the Kielstau catchment, diffuse source pollution of nutrients comes mainly from various farms which apply fertilizers or animal husbandry in the vicinity of the river as well as from urban areas. The combination of these diffuse sources and point sources from the six wastewater treatment plants influences in-stream water quality considerably (Schmalz et al., 2007).

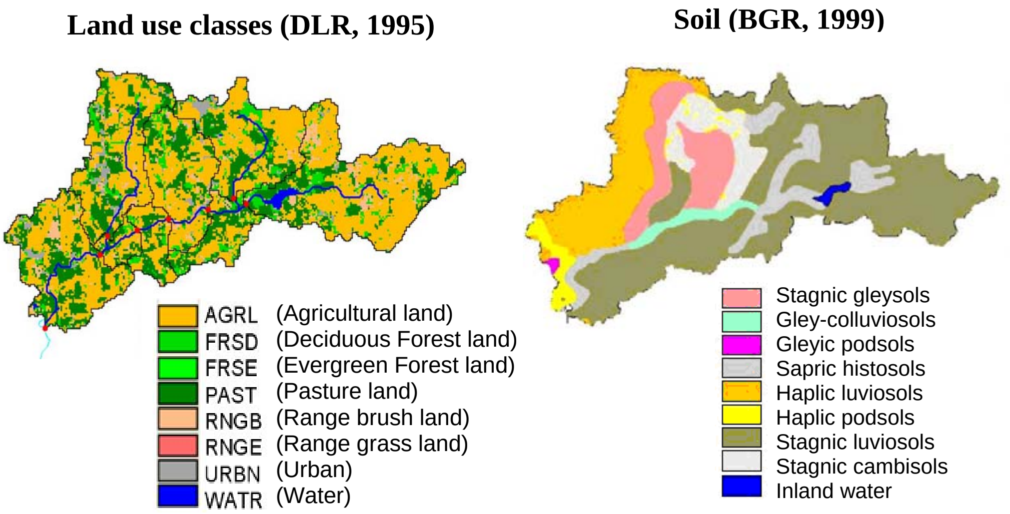


Figure 3.2: Land use and soil classification

3.2.2 Monitoring of the watershed

The Soltfeld gauging measurement station has been installed at the outlet of the Kielstau catchment (Figure 3.1). The daily discharge data (1993–2008) were measured from this station by Staatliches Umweltamt Schleswig. The average daily discharge is listed in Table 3.1.

The collection and analysis of daily water samples took place during the period from May 2006 to November 2008 by the Department of Hydrology and Water Resources Management – Ecology Centre at Kiel University (Tavares, 2006; Bieger, 2007). At the Soltfeld gauging station, water samples were collected (2 x 50 ml) and frozen for further laboratory analysis. The nitrate (NO_3-N) concentrations in the water samples were quantified by photometry and ion chromatography in the laboratory. The average, maximum, and minimum concentrations of NO_3-N are given in Table 3.1.

Table 3.1 Average measured water discharge (Staatliches Umweltamt Schleswig), NO_3-N concentrations (Ecology Centre) at the Soltfeld gauging station

Variable	Unit	Period/day	Value
Average discharge	m^3/s	January 1993 to December 2008	0.43
Average NO3-N	mg/l	May 2006 to October 2008	4.37
Maximum NO3-N	mg/l	12 July 2006	10.05
Minimum NO3-N	mg/l	28 August 2007	1.34

Regarding water quality, the river network of the Kielstau catchment can be clearly differentiated between less polluted in the upper part and more polluted in the lower part of the catchment. Schmalz et al. (2008) showed the measured data of nitrate concentrations at different points along the longitudinal profile of the river Kielstau on two dates: 19 July 2006, representative of the summer season, and 15 November 2006, representative of the autumn season; the findings of which being that, in general, nitrate concentrations of measured points often exceed the target value for a ‘‘good ecological status’’ (Class II, NO_3-N≤ 2.5 mg/l (LAWA, 1998)). The average NO_3-N value of 4.37 mg/l (Table 3.1) also exceeds the allowable limit in comparison with the above target value. These initial assessments of NO_3-N concentrations partly illustrated the status of water quality and can be helpful in gaining a preliminary understanding of the NO_3-N behavior at different times in the Kielstau catchment. The assessment of amounts as well as the trend of nitrate load will later be interpreted by the model using long time series of input and measured data.

3.2.3 The SWAT model

The ecohydrological model SWAT (Soil and Water Assessment Tool, Arnold et al., 1998, version 2005) has been widely used for watershed scale studies dealing with water quantity and quality. SWAT is a semi-distributed, process-oriented hydrological model. It is a continuous time model, which simulates water and nutrient cycles with a daily time step. The SWAT model represents the large-scale spatial heterogeneity of the study area by dividing the watershed into subbasins. The subbasins are then further subdivided into hydrologic response units (HRUs) that are assumed to consist of homogeneous land use and soils. The climatic variables required by SWAT include daily precipitation, maximum/minimum air temperature, solar radiation, wind speed and relative humidity. Major components of the model include hydrology, weather, and agricultural management. The details of all components can be found in Arnold et al. (1998) and Neitsch et al. (2002).

In the SWAT model, soil water content, surface runoff, nutrient cycles, crop growth and management practices are simulated for each HRU and then aggregated for the subbasin by a weighted average. The model’s hydrological components are comprised of surface runoff, percolation, lateral flow, ground water, and evapotranspiration and channel transmission loss. Simulation of the hydrology of a watershed is split into two major divisions. The first division is the land phase of the hydrologic cycle, which controls the amount of water, sediment, nutrient and pesticide loading into the main channel in each subbasin. The second division is the water or routing phase of the hydrologic cycle that can be defined as the movement of water, sediment, nutrient, etc. through the channel network of the watershed to the outlet (Neitsch et al., 2005). The SWAT model simulates surface runoff volumes and peak runoff rate for each HRU using daily rainfall or subdaily rainfall amounts. Surface runoff is calculated using a modification of the Soil Conservation Service (SCS, 1972) curve number method, which is a function of the soil’s permeability, land use and antecedent soil water conditions.

The soil profile is subdivided into multiple layers including infiltration, evaporation, plant uptake, lateral flow, and percolation. SWAT offers various methods to estimate the potential evapotranspiration (PET), such as Hargreaves, Penman-Monteith, and Priestley. The Penman-Monteith method was chosen to be employed in this study because the PET evaluation is based on the basic data such as solar radiation, wind speed, air temperature and relative humidity, while wind speed is not considered by the Hargreaves and Priestley methods. The model computes evaporation from soils and plants separately. Potential soil water evaporation is predicted as a function of potential evapotranspiration and leaf area index, whereas actual soil water evaporation is predicted by using exponential functions of water content and soil depth. Plant transpiration is predicted as a linear function of potential evapotranspiration and leaf area index.

SWAT simulates the nitrogen cycles in the soil profile and in the shallow aquifer (Neitsch et al., 2005). In soil and water, nitrogen is extremely reactive and exists in a number of dynamic forms. It may be added to the soil in the form of fertilizer, manure or residue application, bacteriological fixation, and rain. It can be removed from the soil through plant uptake, soil erosion, leaching, volatilization and denitrification. In the SWAT model, there are five different pools of nitrogen in the soil. Two of the pools are inorganic forms of nitrogen, while the other three pools are organic forms of nitrogen. Nitrate may be transported with surface runoff, lateral flow or percolation. Nitrate entering the shallow aquifer in recharge from the soil profile through the percolation may be remained in the aquifer, moved with groundwater flow into the main channel, or be transported out of the shallow aquifer with water moving into the soil zone in response to water deficiencies, and moved with recharge to the deep aquifer. For the lowland catchment, groundwater component is dominant pathway and plays an important role in transporting nitrate from the shallow aquifer to the main channel or to the soil zone through the upwelling of groundwater processes.

The amount of nitrate moved with the water is calculated by multiplying the nitrate concentration in the mobile water by the volume of water moving in each pathway. The concentration of nitrate in the mobile water fraction is calculated as:

$$Conc_{NO3,\ mobile} = \frac{NO_{3ly} \cdot \left(1 - \exp\left[\frac{-W_{mobile}}{(1-\theta e).SAT_{ly}}\right]\right)}{W_{mobile}}$$

where $Conc_{NO3,\ mobile}$ is the concentration of nitrate in the mobile water for a given layer (kg N/mm H_2O), NO_{3ly} is the amount of nitrate in the layer (kg N/ha), W_{mobile} is the amount of mobile water lost by surface runoff, lateral flow or percolation in the layer (mm H_2O), θe 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).

Organic N transport with sediment is calculated with a loading function developed by McElroy et al. (1976) and modified by William and Hann (1978) for application to separate runoff events. Estimation of the daily organic N runoff loss is based on the concentration of organic N in the topsoil layer, the sediment yield, and the enrichment ratio: that of organic N in sediment to organic N in soil (Neitsch et al., 2005).

The SWAT in-stream water quality algorithms incorporate constituent interactions and relationships used in the QUAL2E model (Brown and Barnwell, 1987), which contains the major interactive factors such as the nutrient cycles, algae production, and benthic oxygen demand.

3.2.4 Model inputs

The ArcSWAT interface for SWAT version 2005 (Winchell et al., 2007) was used to compile the SWAT input files. The basic data sets required to set up the model inputs are: topography, soil, land use and climatic data. These data are described in Table 3.2. The topographic information was used for automatic delineation of the watershed. Land use and soil map were superimposed on the catchment's subbasins. In this study the SWAT model was conducted by dividing the watershed of Kielstau into 8 subbasins and 154 HRUs (Figure 3.1).

Table 3.2 Model input data sources for the Kielstau watershed

Data type	Source	Data description/properties
Topography	LvermA (1995)	Digital elevation model, a grid size of 25 m×25 m
Soil map	BGR (1999)	Soil physical properties such as texture, saturated conductivity, etc. Scale of soil map (1:200,000)
Land use map	DLR (1995)	Land use classifications, 25 m×25 m resolution
Climate data	DWD (2009a,b)	Temperature, precipitation, wind speed, humidity, (Meierwik station, 1993–2008)

The current management practice in this catchment involves a 3-year crop rotation (winter wheat–winter wheat–rape) and monocultural maize, which were simulated in the model. In our simulation, winter wheat is planted from mid-September after a tillage operation and is harvested in the beginning of August in the subsequent year whereupon soil is tilled again. Rape is planted from the end of August and harvested in the beginning of July in the subsequent year. The amounts and date of fertilizer application, which are in conformance with the conventional cultivation, are listed in Table 3.3.

For the calculation of the total nitrogen application, a concentration of nitrogen in precipitation of 1 ppm was assumed. The mean annual precipitation in the Kielstau basin is about 841 mm/a corresponding to an average annual nitrogen deposition of 841 kg N/km^{2}per year or 8.4 kg N/ha/year. This value is in agreement with the European standards (EC, 2002). Data of average monthly nitrogen taken from the six wastewater treatment plants from 2002 to 2008 (Kreis Schleswig-Flensburg, 2008) were implemented as point sources in the model.

Table 3.3 Crop types and fertilization for different land use classes of the Kielstau catchment

Crops	Date of Fertilizer application	Mineral fertilizer (kg/ha/year) (1)	Atmospheric deposition [a] (kg N/ha/year)	Total (kg N/ha/year) (1+2)	Livestock manure[b]

			(2)		
Winter wheat	20.March; 15.April; 01.June	201.6	8.4	210	240
Rape	01.September; 01.March	181.6	8.4	190	120
Maize	01.May; 01.July	170.6	8.4	179	150
Pasture	15. March; 30.May, 10. July; 25.August	153.6	8.4	162	130
Range brush	15. March; 30.May	72.1	8.4	80.5	80

[a] Source: European Communities report (EC, 2002).

[b] Livestock manure application (slurry cow and pig) is split mainly 2 times in spring and autumn seasons during the cultivation period.

3.2.5 Model calibration

The application of the model first involved the analysis of parameter sensitivity, which was then used for model auto- calibration following the hierarchy of sensitive model parameters. The sensitivity analysis method (Morris, 1991) was conducted and aims to assess the most sensitive parameters for setting up the model in this catchment. Model auto-calibration was performed by changing each parameter ten times within the allowable range of values for the specific parameter. Detailed calibration procedures for the SWAT model and the definitions of various calibration parameters are depicted by Neitsch et al. (2002).

The auto-calibration was carried out using flow data from the hydrological years 1998–2004. The validation was done for the continuous time 2004–2008. For the nitrate simulation, the manual calibration was performed for the period of June 2006 to October 2007; the validation was then conducted for the period of November 2007 until October 2008. Simulation of flow and nitrate load was performed for daily time step using measured data from the Soltfeld gauging station at the catchment outlet (Figure 3.1). Surface runoff and base flow were calibrated simultaneously. Parameters adjusted for surface runoff was curve number (CN2) and available water capacity (SOL_AWC). The main parameters adjusted for base flow were soil evaporation compensation factor (ESCO), plant uptake compensation factor (EPCO), ground water revap coefficient GW_REVAP, and threshold depth of water in shallow aquifer (GWQMN). For instance, ESCO, GW_REVAP, and depth-to-subsurface drain (DDRAIN) have been adjusted from the default values of 1; 0.02; 0 to the simulated values of 0.95; 0.2; 800, respectively. The different pathways modeled by the SWAT model are separated into the transport: (i) via surface runoff (0.3%), (ii) via subsurface runoff (14.7%), (iii) via groundwater and transport to the river (57.4%), (iv) via upwelling of groundwater (22.2%), and (v) deep aquifer recharge (4.2%). These results indicated that the surface runoff is very low, while groundwater flow is very high in this lowland area. In addition, the results of the sensitivity analysis showed that groundwater parameters are found to be most sensitive and they turned out to be the most influential factors on simulated water discharge (Lam et al., 2009). Therefore, the groundwater component is very important in lowlands because of the high interaction between surface water and the shallow groundwater. For this lowland catchment, GW_REVAP is an important parameter controlling the upwelling of groundwater and nitrate transport with water from the shallow aquifer to the unsaturated soil zone in response to water deficiencies. SWAT models the movement of water into overlying unsaturated soil layers as a function of water demand for evapotranspiration. This process is significant in watershed where the saturated zone is near below the surface or where deep-rooted plants are growing. Revap is allowed to occur only if the amount of water stored in the shallow aquifer exceeds a threshold water level. The allowable range of this parameter is between 0.02 and 0.2. As GW_REVAP approaches 0, movement of water from the shallow aquifer to the root zone is restricted. This parameter was changed from its initial value of 0.02–0.2 in order to obtain a better fit between the model results and the measured data. The auto-calibration processes were also implemented similarly for other parameters within their allowable range in SWAT. Main parameters used for the sensitivity analysis and the calibration of flow are described in Table 3.4.

Table 3.4 Main variables used for sensitivity analysis and calibration in SWAT

Variable name	Model processes	Description	Allowable range	Actual value used
GW_REVAP	Flow	Ground water revap coefficient	0.02-0.2	0.2
GW_DELAY	Flow	Delay time for aquifer recharge	0-500	14
GWQMN	Flow	Threshold depth of water in shallow aquifer	0-5000	50
ALPHA_BF	Flow	Base flow recession constant	0-1	1
RCHRG_DP	Flow	Deep aquifer percolation coefficient	0-1	0.05
ESCO	Flow	Soil evaporation compensation factor	0.01-1	0.95
EPCO	Flow	Plant uptake compensation factor	0.01-1	1
CN2 (arable land, pasture, range bush, and forest)	Flow	Curve number	35-98	64, 46, 46, 35
SOL_AWC	Flow	Available water capacity	0-1	0.12-0.54
DDRAIN	Flow	Depth to subsurface drain	0-2000	800
TDRAIN	Flow	Time to drain soil to field	0-72	24
GDRAIN	Flow	Drain tile lag time	0-100	8
BIOMIX	Sediment, organic and mineral nutrients	Biological mixing efficiency	0.01-0.4	0.2
NPERCO	Mineral nitrogen	Nitrogen percolation coefficient	0.01-1	0.95
RSDCO	Sediment, organic and mineral nutrients	Residue decomposition coefficient	0.01-0.1	0.05
AL1	Nitrogen in channel	Fraction of algae biomass that is nitrogen	0.07-0.09	0.08

Since nitrate fluxes strongly depend on water fluxes, parameters controlling water balance were calibrated as the first step, and only then were nitrate load considered (Pohlert et al., 2005). The nitrate was calibrated manually by changing parameters so that simulated nitrate results match measured nitrate data well. A constant initial (SOL_NO3) concentration of 5 mg/kg in soil layers was assumed. Main parameters adjusted for nitrate calibration were nitrate transport (NPERCO), residue mineralization (RSDCO), and biological mixing efficiency (BIOMIX). These parameters were verified within their allowable ranges (Table 3.4). Several statistics, including the root mean square error, coefficient of determination, and Nash-Sutcliffe efficiency (Nash and Sutcliffe, 1970) have been used to evaluate the accuracy of model predictions. The coefficient of determination (R^2) is the square of the Pearson's product–moment correlation coefficient, which is an indicator of strength of relationship between measured and simulated values. Nash-Sutcliffe efficiency (E_{NS}) has been widely used to evaluate the performance of hydrologic models. The values of E_{NS} and R^2 range from 0 to 1. If the E_{NS} and R^2 values are less than or close to 0, the model simulation is taken as an indication of poor or unacceptable performance. The closer the values get to 1, the more perfect the model simulation. Santhi et al. (2001) found that a value greater than 0.5 for these variables constitutes an acceptable simulation model.

3.3 Results and discussion

3.3.1 Simulation of flow

The simulated results of flow were compared with the corresponding measured data at the watershed outlet for both the calibration and the validation period.

Figure 3.3 shows good agreement between simulated and measured daily discharge with E_{NS} and R^2 at 0.75 and 0.78 for the calibration period and 0.78 and 0.84 for the validation period at the outlet of Kielstau catchment, respectively. Overall, the model performance was satisfactory in both the calibration and validation periods in daily time step. However, the timing and the height of some single flood peaks, which occurred in the recent winter periods of 2005, 2006, 2007 and 2008, were less accurately predicted. This is presumably due to the model structure of SWAT. Since it is a continuous time model with a daily time step, subscale processes such as single-event flood routing cannot be efficiently and elaborately predicted. In addition, temporally, the daily measured precipitation for 24 h beginning at 6:00 AM may not well match the daily average discharge values, which were measured for 24 h from midnight on.

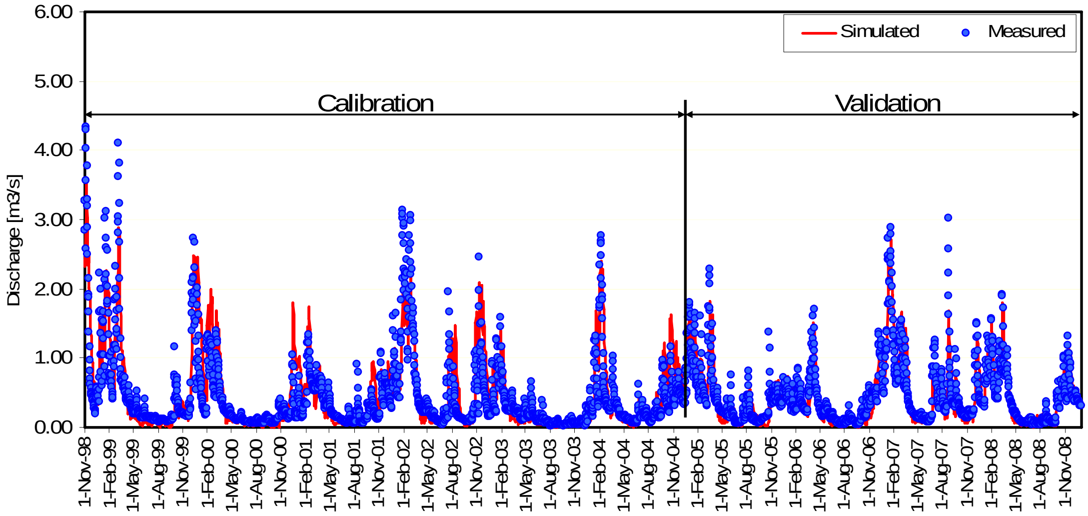


Figure 3.3: Simulated and measured daily discharge at the Soltfeld gauging station (E_{NS} and R^2 of 0.75 and 0.78 for the calibration period; 0.78 and 0.84 for the validation period)

3.3.2 Simulation of nitrate load

Parameters used for manual calibration of nitrate load are given in Table 3.5. The comparison between simulated and measures results of daily nitrate load at the gauge Soltfeld is shown in Figure 3.4. During the summer periods of 2006, 2007, and 2008, the model simulated well for both the range and the dynamic of the nitrate load in general. In contrast, the nitrate load is underpredicted during the winter periods. This can be attributed to the underestimation of some peak flows, which lead to the underestimation of the corresponding nitrogen peaks.

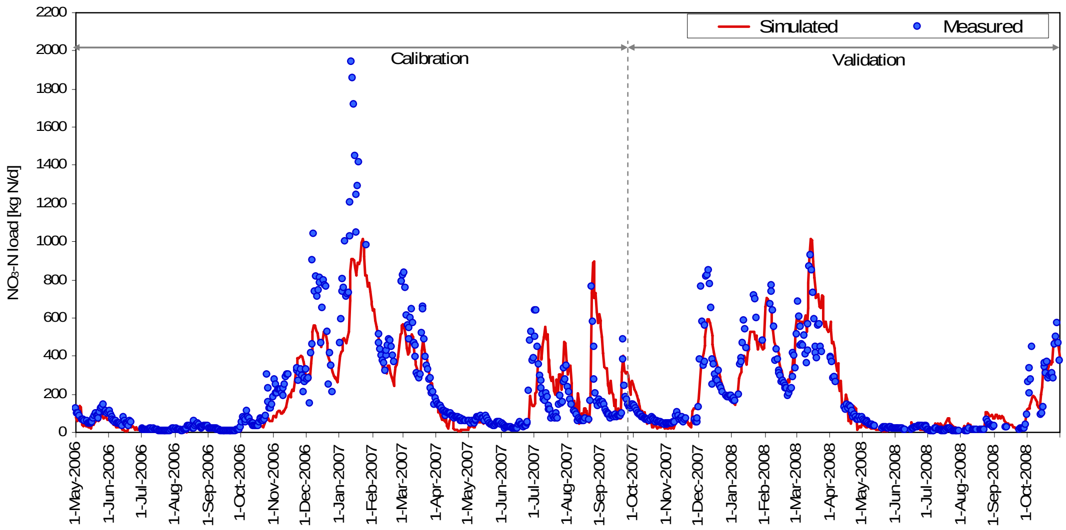


Figure 3.4: Simulated and measured daily nitrate load at the Soltfeld gauging station

As can be seen from Table 3.5, the differences between the mean values of measured and simulated nitrate load were not significant. These differences are estimated at about 1% for the calibration period and 5% for the validation period, respectively. The high values of root mean square error are mainly due to the magnitude of measured and simulated values of nitrate load in kg/day. On the other hand, the underestimation of nitrate load in winter periods is a significant reason for increasing higher values of root mean square error. The model's efficiency E_{NS} of 0.68 and 0.76 for the respective calibration and validation periods showed a close agreement between the measured and simulated nitrate load at the watershed outlet. Moreover, high values of the coefficient of determination R^2 indicated that the model simulates considerable accuracy of nitrate load during both the calibration and the validation period.

Table 3.5 Simulated and measured daily nitrate load for the calibration period from May 2006 to October 2007 and the validation period from November 2007 until October 2008 at gauge Soltfeld

Statistic parameters	Calibration		Validation	
	Measured	Simulated	Measured	Simulated
Mean ($Kg.d^{-1}$)	195.75	194.0	199.01	209.53
Root mean square error		92.7		52.9
R^2		0.7		0.84
E_{NS}		0.68		0.76

The results of nitrate simulation are in accordance with previous studies using SWAT on various catchments. Bieger (2007) simulated daily nitrate load on the same catchment and obtained an E_{NS} and R^2 of 0.55 and 0.84 for the calibration period of June 2005 to October 2006, respectively. However, differing from the present study, point source effluents were used as input data for average annual values by Bieger (2007). These assumptions may not clearly depict the variation of point source emissions at different times and thus the model did not adequately simulate the influences of point source emissions on nitrate concentrations. Grizzetti et al. (2003) obtained an E_{NS} of 0.51 when they used SWAT to model diffuse emissions and retentions of nutrients on daily time step at the Vantaanjoki watershed (1680 km^2), which is situated in Southern Finland and classified as a lowland catchment. Behera and Panda (2006) obtained an E_{NS} ranging from 0.83 to 0.92 for both the calibration and validation period. They concluded that SWAT simulated nitrate concentration satisfactorily throughout the entire rainy season based on comparisons with daily-observed data from an agricultural watershed located in eastern India. These results implied that SWAT can be used to evaluate reliably nitrate load in daily time step on various catchments in general and lowland catchments in particular. As outlined above, we achieved a comparable simulation efficiency of daily nitrate load at the catchment outlet. The values of E_{NS} and R^2obtained are satisfactory in both the calibration and the validation period. The values of statistical parameters also indicated that the SWAT model has been used to assess successfully nitrate load for the entire lowland catchment.

3.3.3 Contribution of point and diffuse sources to nitrate load

Nitrate concentrations in stream have been strongly influenced by emissions of point and diffuse sources within the Kielstau watershed. To evaluate the sources of pollution and quantify the nitrate load entering the entire watershed, nitrate load in-stream water was simulated at different subbasin outlets (Sb) along the longitudinal river Kielstau including Sb1, Sb3, Sb4, Sb5, Sb7, and Sb8. The six wastewater treatment plants were considered (Figure 3.1). Daily simulations were performed over a 7-year period, from 2002 to 2008. This simulation period was chosen according to the availability of point source information. Two scenarios were examined to compare the influence of point and diffuse source pollution on nitrate load at the outlet of subbasins as well as the watershed outlet. These scenarios, respectively, take into account point and diffuse sources (scenario 1), or diffuse sources only (scenario 2). Figure 3.5 shows the simulated average annual values of nitrate load at subbasin outlets along the longitudinal river Kielstau.

The nitrate load results at the different subbasin outlets and the watershed outlet were drawn from the upstream subbasin to the downstream of the river Kiestau. As can be seen from Figure 3.5, the simulated average annual values of nitrate load for the period 2002–2008 at the watershed outlet are 112.7 tons, 107.4 tons for scenario 1, scenario 2, respectively. The nitrate load value of 107.4 tons in scenario 2 is estimated at about 95% in comparison with the corresponding value in scenario 1. This means diffuse sources contribute 95% of the total nitrate load in the whole Kielstau watershed and the remaining 5% is attributed to point sources. Regarding the nitrate load at the subbasin outlets in scenario 1, the nitrate load increased gradually from subbasin 1 to subbasin 7 before finally reaching the highest value of 112.7 tons at the watershed outlet. This may be explained by the enhanced nitrate concentrations or by the increased volume of water that transports nutrients from upper subbasins of the watershed in the downstream direction. The loading of nitrate at the different subbasin outlets is consistent with the increasing trend of measured nitrate concentrations, especially in autumn, at different transects along the longitudinal river Kielstau (Schmalz et al., 2008).

From the results shown in Figure 3.5 it can be concluded that diffuse sources are the dominant factor enhancing nitrate concentrations in the Kielstau catchment. Reduction of diffuse sources may help to minimize the magnitude of the nitrate concentrations within the entire catchment, while reduction of point sources may contribute to improving the water quality in the individual subbasins, where point sources are located.

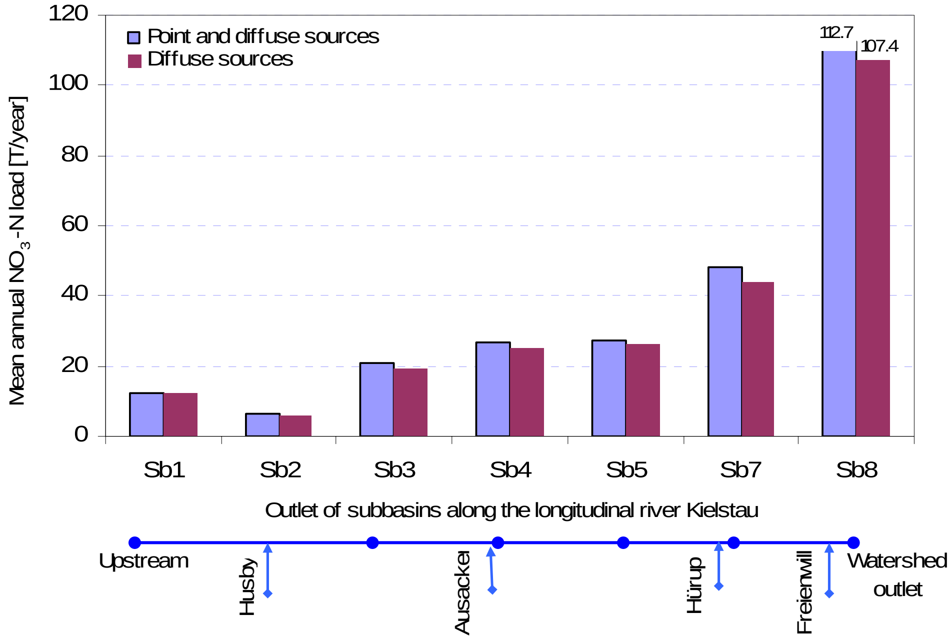


Figure 3.5: Mean annual load of nitrate at subbasin outlets along the longitudinal river Kielstau considering different scenarios

3.3.4 Diffuse source emissions of nitrate

Diffuse source emissions of nitrate by surface runoff, lateral flow and percolation are impacted by watershed properties such as land use/cover types and thus the area occupied by such land uses influences the loading of nitrate. In order to determine land use types which have considerable influences on the nitrate concentrations in stream, a comparison including the percentage of different land covers and their respective nitrate loads in different subbasins was considered. The proportions of different land use cover types in different subbasins and the average annual amount of nitrate load transported from the respective subbasins in stream are given in Table 3.6.

Table 3.6 Different land uses proportion compared with nitrate load at subbasin outlets

Subbasins	AGRL (%)	RNGB (%)	PAST (%)	RNGE (%)	WATR (%)	URBN (%)	FRSD (%)	FRSE (%)	Nitrate load (T/year)
1	46.71	7.97	29.12	0.07	2.18	1.72	10.90	1.33	12.33
2	51.24	3.80	27.14	0.04	0.07	2.78	10.67	4.26	6.63
3	46.01	3.41	27.04	2.30	0.05	3.53	10.96	6.70	20.90
4	59.73	4.59	27.63	0.03	0.00	1.98	6.00	0.04	26.55
5	47.87	2.94	37.52	0.00	0.00	1.09	6.35	4.23	27.54
6	51.78	3.48	26.43	1.14	1.24	5.79	8.08	2.06	7.69
7	54.39	3.64	32.17	0.17	0.00	0.11	9.22	0.30	48.44
8	59.98	4.98	29.61	0.03	0.25	1.44	3.71	0.00	112.70

Notes: AGRL = agricultural land; RNGB = range-brush land; PAST = pasture land; RNGE = range-

Grasses land; WATR = water; URBN = urban; FRSD = forest-deciduous land; FRSE = forest-evergreen land.

Figure 3.6 shows the correlation between the average annual nitrate load and the percentage of land use types from which nitrate was transported by runoff during the simulation period 2002–2008. The values of the correlation coefficients revealed that nitrate load at the subbasin outlets was found to be fairly positively correlated to the area occupied by agricultural land (r = 0.63) and negatively correlated to the area occupied by forest (r = 0.71). The remaining land use types were found to be less correlated to nitrate load in comparison with agricultural land and forest areas. The influences of agricultural and forest cover on the variation of nitrate load was further compared by regression analysis as shown in Figures 3.7 and 3.8.

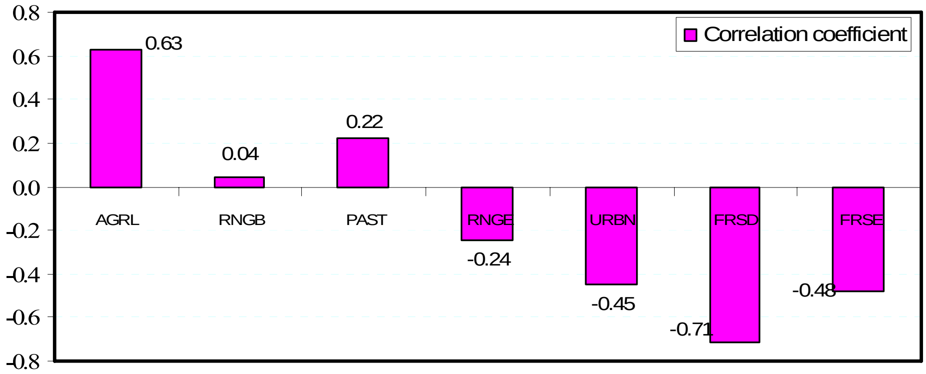


Figure 3.6: Correlation coefficient between the NO_3-N load and the percentage of land use types

Figure 3.7 illustrates that agricultural land significantly affected nitrate load (r^2= 0.4) as compared to other land use areas. The positive slope shows that nitrate load increases with the increase of agricultural land area.

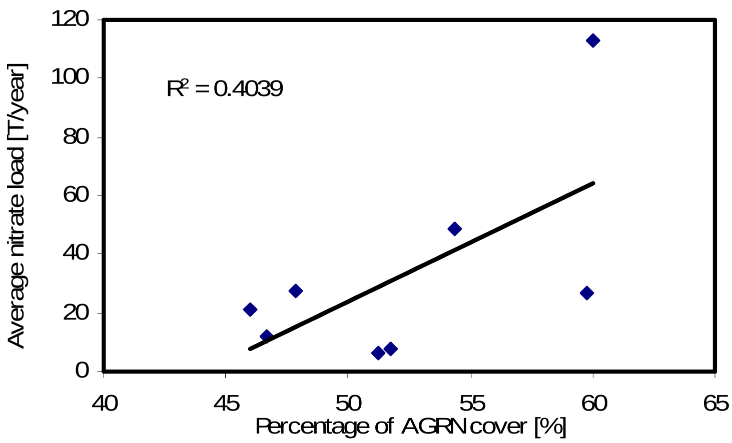


Figure 3.7: Comparison between the percentage of agricultural land cover and average annual

NO_3-N load for the period 2002-2008

In contrast to agricultural land, forest area was negative with respect to nitrate load. It can be seen in Figure 3.8 that nitrate load decreases as forest area increases.

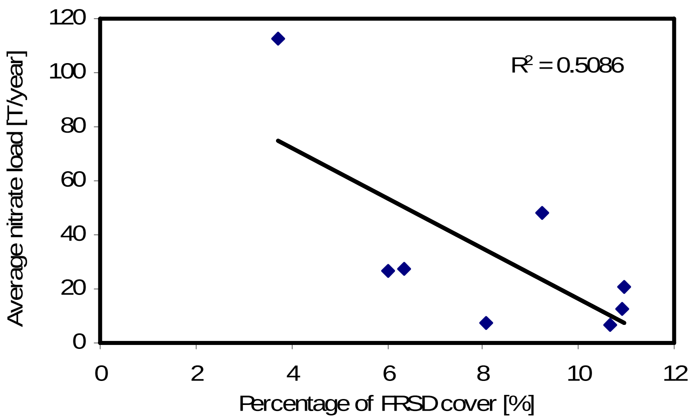


Figure 3.8: Comparison between the percentage of forest cover and average annual NO3-N load for the period 2002-2008

The results outlined above revealed that land use types significantly influenced nitrate load at the different subbasins outlets. Diffuse source emissions of nitrate are higher for agricultural land, which could be attributed to the application of fertilizers, and lower for other land use types, especially for forest cover in the watershed.

3.4 Conclusions

In this study the ecohydrological model SWAT was applied to simulate both water discharge and nitrate load from different point and diffuse sources in the mesoscale Kielstau catchment - a typical rural lowland area in Northern Germany. The basis input data includes climate, topography, soil, land use, and agricultural data; all of which, as well as consideration of sewage disposals of six wastewater treatment plants, was used to predict the current nitrate load. The calibration and validation of the SWAT output were implemented by comparing simulated flow and nitrate load with corresponding in-stream measurements at the watershed outlet. Simulations of flow and nitrate were performed in daily time step. The results of this study showed good agreement between simulated and measured daily discharges with an E_{NS} and R^2 of 0.75 and 0.78 for the calibration period, 0.78 and 0.84 for the validation period. The statistical coefficients of the nitrate model performance were relatively reasonable (E_{NS} and R^2 of 0.68, 0.70 for the calibration period; 0.76, 0.84 for the validation period, respectively). These statistic values revealed that SWAT performed satisfactorily in accurately simulating daily flow and nitrate load at the Kielstau lowland catchment.

The comparison between the contribution of point and diffuse sources to nitrate load in the whole catchment was performed by comparing two scenarios: simulation of point and diffuse sources (scenario 1); simulation of only diffuse sources (scenario 2). The results indicated that diffuse sources are the dominant source of nitrate in the entire catchment. The contribution of point sources and diffuse sources to nitrate load at the watershed outlet were about 5% and 95%, respectively.

An evaluative analysis of the influences of different land use cover types on nitrate load at different subbasins was carried out. Agriculture is found to be the dominant contributor of diffuse sources, while forest areas have the lowest influence on nitrate load in comparison with those of other land use types.

The accurate prediction of nitrate load by the SWAT model in this study is an important contribution to the assessment of total nitrogen as well as water quality of the whole watershed. In this study we mentioned the current agricultural practice in order to represent the current status of the Kielstau catchment. These results also serve as a baseline for evaluating the impact of the best management practice scenarios on long-term in-stream nutrient pollution, which may well assist in meeting the objective of the European Water Framework Directive of putting sustainable river basin management into effect by 2015.

Chapter IV The impact of agricultural Best Management Practices on water quality in a North German lowland catchment

Q. D. Lam, B. Schmalz, N. Fohrer

Environmental Mornitoring and Assessment
Submitted 20.05.2010, Accepted 08.02.2011, Published 10.03.2011

Abstract

Research on water quality degradation caused by point and diffuse source pollution plays an important role in protecting the environment sustainably. Implementation of Best Management Practices (BMPs) is a conventional approach for controlling and mitigating pollution from diffuse sources. The objectives of this study were to assess the long-term impact of point and diffuse source pollution on sediment and nutrient load in a lowland catchment using the ecohydrological model Soil and Water Assessment Tool (SWAT) and to evaluate the cost and effectiveness of BMPs for water quality improvement in the entire catchment.

The study area, Kielstau catchment, is located in the North German lowlands. The water quality is not only influenced by the predominating agricultural land use in the catchment as cropland and pasture, but also by six municipal wastewater treatment plants. Diffuse entries as well as punctual entries from the wastewatertreatment plants are implemented in the model set-up. Results from model simulations indicated that the SWAT model performed satisfactorily in simulating flow, sediment, and nutrient load in a daily time step. Two approaches to structural and nonstructural BMPs have been recommended in relation to cost and effectiveness of BMPs in this study. These BMPs include extensive land use management, grazing management practice, field buffer strip, and nutrient management plan. The results showed that BMPs would reduce fairly the average annual load for nitrate and total nitrogen by 8.6% to 20.5%. However, the implementation of BMPs does not have much impact on reduction in the average annual load of sediment and total phosphorus at the main catchment outlet. The results obtained by implementing those BMPs ranged from 0.8% to 4.9% and from 1.1% to 5.3% for sediment and total phosphorus load reduction, respectively. This study also reveals that reduction only in one type of BMP did not achieve the target value for water quality according to the European Water Framework Directive. The combination of BMPs improved considerably water quality in the Kielstau catchment, achieving a 53.9% and a 46.7% load reduction in nitrate and total nitrogen load, respectively, with annual implementation cost of 93,000 Euro.

Keywords: Best management prctice, environment, SWAT

4.1 Introduction

Lowland catchments are ecosystems with low flow velocity, a high groundwater table, and flat topography (Müller et al., 2004; Krause et al., 2007; Schmalz et al., 2009). In the past centuries, different melioration measures such as river regulation and pumping stations have been implemented in order to enlarge areas as well as render better cultivation conditions for agriculture. These have led to a change in the natural water and nutrient balance, which has contributed to eutrophication problems and ecological damage of lakes and river network systems. Besides, tile drainage is also a common agricultural practice aimed at improving aeration conditions and moisture in lowland areas. It reduces the retention time of water in the soil and hence forms an important pathway for nitrate to surface water bodies (Tiemeyer et al., 2006; Kladivko et al., 1999). On the other hand, these areas are also affected by additional human interferences such as agricultural practices (e.g., fertilizer, pesticides utilization) and point source emissions, which also have relevant influence on nutrient load and water quality. Some other studies have found that pollutants such as fertilizers, pesticides and sediment, resulting from various agricultural practices, lead to the degradation of surface and groundwater (Donoso et al., 1999; Zalidis et al., 2002).

Point and diffuse source pollution are become a serious problem causing the impairment of water quality in many European countries. Diffuse source pollution generally results from surface and subsurface runoff, drainage, atmospheric deposition, and precipitation. As runoff from precipitation moves, it picks up and transports pollutants resulting from nature and human activity, ultimately depositing them into rivers, lakes, wetlands, and groundwater. Agriculture has been identified as the major contributor of diffuse source pollution of water resources (Humenik et al., 1987; Duda, 1993; Behrendt et al., 1999; Lam et al., 2010). Therefore, application of Best Management Practices (BMPs) is a useful method to eliminate or minimize diffuse source pollution resulting from agricultural activities in order to achieve good ecological and chemical conditions of water quality standard regulated by the European Framework Directive (EC, 2000). Many types of agricultural BMPs can be used for controlling diffuse source pollution such as conservation tillage, nutrient management plans, animal waste management, stream protection, grazing land management, forest riparian buffers, etc. However, water quality problems cannot usually be solved with one type of BMP because single practices do not typically provide the full range and extent of control needed at a site. Different practices are therefore combined to treat pollutants more effectively from different source pollution within a watershed. Implementation of BMPs is challenged by integration of environmental, economic, and institutional criteria. Assessment of environmental issues in watersheds relates to social benefits such as achieving the goal of maximum productions, minimum yield reduction, and unchanged farming habits. Establishment cost and environmental effectiveness of BMPs are often crucial factors in selecting and adopting BMPs (Arabi et al., 2004). Identifying optimal combination of BMPs requires systematic approaches that allow decision makers to quickly assess trade-off among environmental and economic criteria.

Watershed models are useful tools and have been used widely on the globe to predict the long-term impact of BMPs application on water quality. Model scenarios can be helpful in finding reasonable measures for assessing environmental ecological status while taking into account relevant factors such as climate, land, and water use (Krysanova et al., 2005; Højberg et al., 2007).

In recent years, a large number of diffuse source pollution models, ANSWERS (Beasley and Hugins, 1982), AGNPS (Yoon and Disrud, 1993), HSPF (Johanson et al., 1984), EPIC (William et al., 1984), have been developed to generalize the effect of environmental conditions and agricultural practices on nutrient losses on field and catchment scale.

For the lowland catchments, a number of ecohydrological models have already been used: The IWAN model (Krause and Bronstert, 2005) was used for modeling water balance and nutrient dynamics of floodplains. Hattermann et al. (2006) integrated wetlands and riparian zones into SWIM (Krysanova et al., 1998) to determine their influence on water and nutrient fluxes. The Soil and Water Assessment Tool (SWAT) model (Arnold et al., 1998) has been widely used all around the world to predict flow, sediment and nutrient load from watersheds of various sizes (Saleh et al., 2000; Tripathi et al., 2004; Gassman et al., 2006; Lenhart et al., 2003; Schmalz et al., 2008b). Gassman et al. (2007) have indicated that a key strength of SWAT is a flexible framework allowing the simulation of a wide variety of structural and nonstructural BMPs such as conservation tillage, cover crops, application rate and timing of fertilizers, nutrient management, buffer strips, flood prevention structures, grass waterway, and parallel terraces. Many studies have used SWAT (Tripathi et al., 2005; Arabi et al., 2007; Behera and Panda, 2006; Bracmort et al., 2006; Rode et al., 2008) to evaluate the impact of BMPs on water quality at different watershed scales. However, the application of reasonable BMPs to a specific watershed is often not similar due to the different watershed characteristics, land use, soil resolution, and groundwater level. Because model outputs are affected by those factors, the evaluation of performance and effectiveness of BMPs based on model predictions will be influenced as well. For lowland areas located in Northern Germany, no previous study has been made in using the SWAT model to develop an appropriate management strategy for quantifying and controlling of sediment and diffuse source pollution as well as evaluating the cost and effectiveness of BMPs at the watershed scale.

As a result, the aims of this study were: (1) the assessment of the long-term impact of point and diffuse source pollution on sediment and nutrient load in a lowland catchment which has special hydrological characteristics such as flat topography, shallow groundwater, low hydraulic gradients, and high potential for water retention in peatland and lakes in comparison with those of mountainous or urban catchments using the ecohydrological SWAT model, (2) the evaluation of cost and effectiveness of BMPs in minimizing the diffuse sources pollution within the watershed.

4.2 Materials and methods

4.2.1 Study area

The study area Kielstau catchment is located in Northern Germany as part of a lowland area in Schleswig–Holstein (Figure 4.1). The area of the Kielstau catchment is about 50 km2. The river Kielstau has a total length of 17 km and flows through Lake Winderatt towards the gauge Soltfeld, located at the outlet of Kielstau catchment. There are two important tributaries of the river Kielstau from the north, the Moorau and the Hennebach. The six wastewater treatment plants built within the Kielstau watershed are Husby, Hürup Nord, Hürup Weseby, Hürup Süd, Ausacker, and Freienwill (Figure 4.1). Husby is situated at the beginning of the Moorau tributary with 3000 population equivalents. Hürup Nord, Hürup Weseby, and Hürup Süd are located along the longitudinal Hennebach tributary (461, 447, and 240 population equivalents). Ausacker and Freienwill are located on the river Kielstau (1880 and 350 population equivalents). In addition, various small tributaries and water from drainage pipes and ditches flow into the river Kielstau. The drainage fraction of agricultural area in the Kielstau catchment is estimated at 38% (Fohrer et al., 2007). The maximum height difference is 49 m (Schmalz et al., 2008a). The precipitation is about 841 mm/a (station Satrup 1961–1990, DWD, 2009b); the mean annual temperature is 8.2◦C (station Flensburg, 1961–1990, DWD, 2009a).

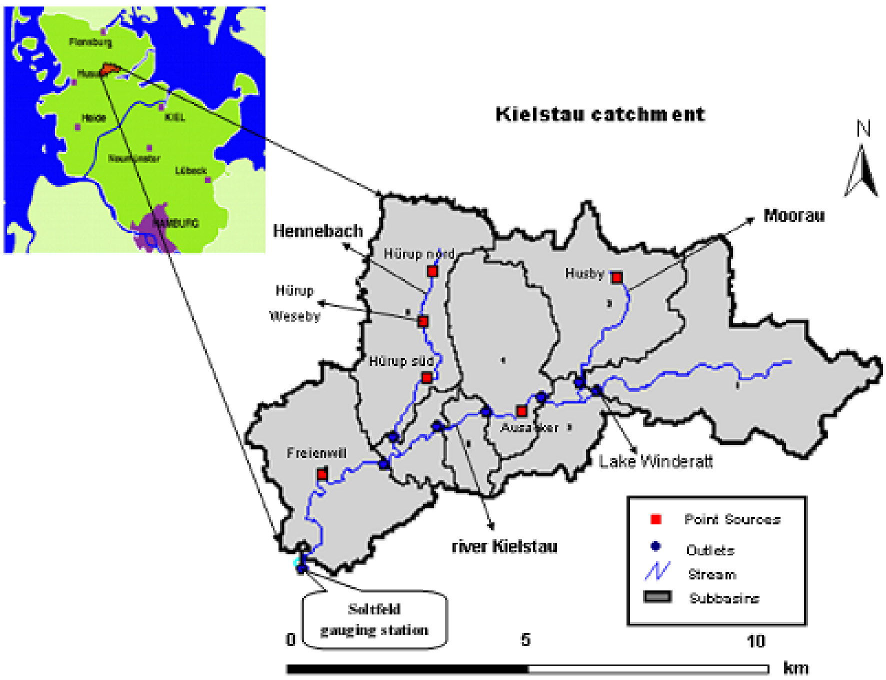


Figure 4.1: Location of the Kielstau catchment and its subbasins in Schleswig-Holstein, Northern Germany

Land use is dominated by arable land and pasture. The arable land area occupies over 55%, and pasture over 26%, of the catchment area (Tavares, 2006). The dominant soils of the Kielstau catchment are Stagnic Luvisols and Haplic Luvisols. Land use and soil maps used in this study can be seen in Figure 4.2.

In the Kielstau catchment, diffuse source pollution of nutrients results mainly from various farms which apply fertilizers or animal husbandry in the vicinity of the river as well as from urban areas. The combination of these diffuse sources and point sources influences instream water quality considerably (Schmalz et al., 2007).

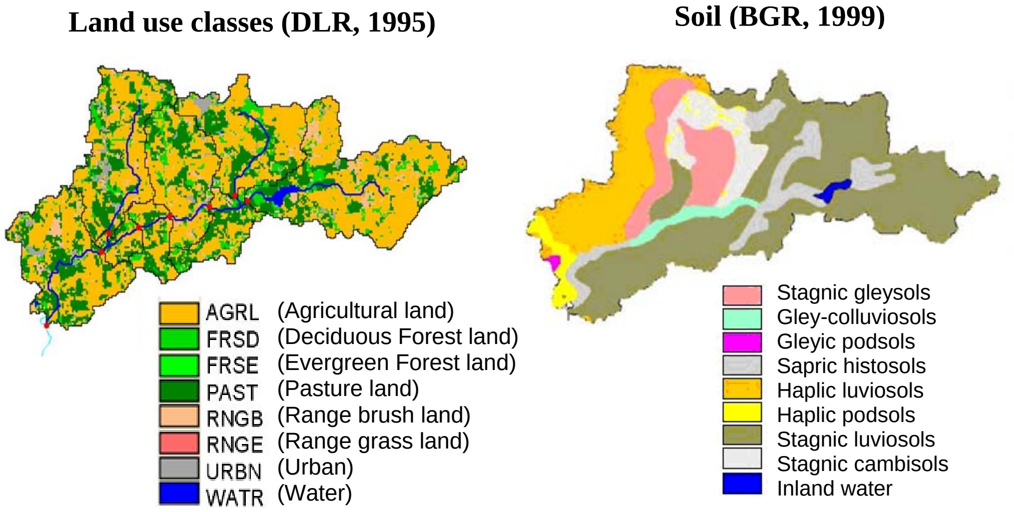


Figure 4.2: Land use and soil classification (Lam et al., 2010)

4.2.2 Monitoring of the watershed

The Soltfeld gauging measurement station has been installed at the outlet of the Kielstau catchment (Figure 4.1). The hourly discharge data (1993–2008) were measured from this station by Staatliches Umweltamt Schleswig (2009). The average daily discharge is listed in Table 4.1.

Table 4.1 Mean measured water discharge (Staatliches Umweltamt Schleswig, 2009) and nutrient concentrations (Ecology Centre) at the Soltfeld gauging station

Variable	Unit	Period	Value	LAWA (1998), class II
Average discharge	m^3/s	January 1993 to December 2008	0.43	-
Average NO_3-N	mg/l	May 2006 to December 2008	4.48	2.5
Average NH4-N	mg/l	May 2006 to December 2008	0.15	0.3
Average TN	mg/l	May 2006 to December 2008	5.81	3.0
Average PO4-P	mg/l	May 2006 to December 2008	0.18	0.1
Average TP	mg/l	May 2006 to December 2008	0.23	0.15

The collection and analysis of daily water samples took place during the period from May 2006 to December 2008 by the Department of Hydrology and Water Resources Management – Ecology Centre at Kiel University (Schmalz and Fohrer 2010). At the Soltfeld gauging station, water samples were collected (2 x 50 ml) and frozen for further laboratory analysis. Nutrient concentrations in the water samples were quantified by photometry and ion chromatography in the laboratory. The average concentrations of nutrient are given in Table 4.1.

Regarding water quality, the river network of the Kielstau catchment can be differentiated between less and more polluted parts. Schmalz et al. (2008a) showed measured nitrate concentrations at different points along the longitudinal profile of the river Kielstau on two dates: 19 July 2006, representative of the summer season, and 15 November 2006, representative of the autumn season;

the findings of which being that, in general, nitrate concentrations of measured points often exceed the target value for a "good water quality" (Class II, NO_3-N ≤ 2.5 mg/l (LAWA, 1998)). The target stipulations are based on the LAWA procedure for the chemical classification of water bodies. Quality class II (moderately polluted) of LAWA standard (Table 4.1) represents the target value for water quality until the year 2015 according to the European Water Framework Directive. In other words, nutrient concentration needs to be lower or equal that value to satisfy with LAWA quality standards. The measured average concentration values of all parameters (Table 4.1) exceed the allowable limit of class II as a whole. These initial assessments of nutrient concentrations can be helpful in gaining a preliminary understanding of the nutrient behavior at different times in particular and partly illustrated the status of water quality in the whole Kielstau catchment. The assessment of amounts as well as the trend of nutrient load will later be interpreted by the model using long time series of input and measured data.

4.2.3 The SWAT model

The ecohydrological model SWAT (Soil and Water Assessment Tool, Arnold et al., 1998, version 2005) has been widely used for watershed scale studies dealing with water quantity and quality. SWAT is a semi-distributed, process-oriented hydrological model. It is a continuous time model, which simulates water and nutrient cycles with a daily time step. The SWAT model represents the large-scale spatial heterogeneity of the study area by dividing the watershed into subbasins. The subbasins are then further subdivided into hydrologic response units (HRUs) that are assumed to consist of homogeneous land use and soils. The climatic variables required by SWAT include daily precipitation, maximum/minimum air temperature, solar radiation, wind speed and relative humidity. Major components of the model include hydrology, weather, and agricultural management. The details of all components can be found in Arnold et al. (1998) and Neitsch et al. (2002).

In the SWAT model, soil water content, surface runoff, nutrient cycles, crop growth and management practices are simulated for each HRU and then aggregated for the subbasin by a weighted average. The model's hydrological components are comprised of surface runoff, percolation, lateral flow, groundwater, and evapotranspiration and channel transmission loss. Simulation of the hydrology of a watershed is split into two major divisions. The first division is the land phase of the hydrologic cycle, which controls the amount of water, sediment, nutrient and pesticide loading into the main channel in each subbasin. The second division is the water or routing phase of the hydrologic cycle that can be defined as the movement of water, sediment, nutrient, etc. through the channel network of the watershed to the outlet (Neitsch et al., 2005). The SWAT model simulates surface runoff volumes and peak runoff rate for each HRU using daily rainfall or subdaily rainfall amounts. Surface runoff is calculated using a modification of the Soil Conservation Service (SCS, 1972) curve number method, which is a function of the soil's permeability, land use and antecedent soil water conditions.

The soil profile is subdivided into multiple layers including infiltration, evaporation, plant uptake, lateral flow, and percolation. SWAT offers various methods to estimate the potential evapotranspiration (PET), such as Hargreaves (Hargreaves and Samani, 1985), Penman-Monteith (Monteith, 1965), and Priestley-Taylor (Priestley and Taylor, 1972). The Penman-Monteith method was chosen to be employed in this study because the PET evaluation is based on the basic data such as solar radiation, wind speed, air temperature and relative humidity, while wind speed is not considered by the Hargreaves and Priestley-Taylor methods. The model computes evaporation from soils and transpiration from plants separately. Potential soil water evaporation is predicted as a function of potential evapotranspiration and leaf area index, whereas actual soil water evaporation is predicted by using exponential functions of water content and soil depth. Plant transpiration is predicted as a linear function of potential evapotranspiration and leaf area index.

Erosion and sediment yield are estimated for each HRU with the Modified Universal Soil Loss Equation (Williams, 1975). The sediment transport in the channel is controlled by the simultaneous operation of deposition and degradation processes. The channel sediment routing equation uses a modification of Bagnold (1977) that defined stream power as the production of water density, flow velocity and water surface slope. The maximum amount of sediment that can be transported from a reach segment is a function of the peak channel velocity. The SWAT model either deposits excess sediment or re-entrains sediment via channel erosion depending on the sediment load entering the channel.

SWAT simulates the nitrogen cycles in the soil profile and in the shallow aquifer (Neitsch et al., 2005). In soil and water, nitrogen is extremely reactive and exists in a number of dynamic forms. It may be added to the soil in the form of fertilizer, manure or residue application, bacteriological fixation, and rain. It can be removed from the soil through plant uptake, soil erosion, leaching, volatilization and denitrification. Plant use of nitrogen is estimated using the supplying and demand approach (Williams et al., 1984). In the SWAT model, there are five different pools of nitrogen in the soil. Two of the pools are inorganic forms of nitrogen, while the other three pools are organic forms of nitrogen.

Nitrate may be transported with surface runoff, lateral flow or percolation. Nitrate entering the shallow aquifer in recharge from the soil profile through the percolation may be remained in the aquifer, moved with groundwater flow into the main channel, or be transported out of the shallow aquifer with water moving into the soil zone in response to water deficiencies, and moved with recharge to the deep aquifer. For the lowland catchment, groundwater component is dominant pathway and plays an important role in transporting nitrate from the shallow aquifer to the main channel or to the soil zone through the upwelling of groundwater processes. The amount of nitrate moved with the water is calculated by multiplying the nitrate concentration in the mobile water by the volume of water moving in each pathway. Organic N transport with sediment is calculated with a loading function developed by McElroy et al. (1976) and modified by William and Hann (1978) for application to separate runoff events. Estimation of the daily organic N runoff loss is based on the concentration of organic N in the topsoil layer, the sediment yield, and the enrichment ratio: that of organic N in sediment to organic N in soil (Neitsch et al., 2005).

The different phosphorus processes modeled by SWAT in the HRUs and the various pools of phosphorus in the soil are described in Neitsch et al. (2005). Plant use of phosphorus is estimated using the supply and demand approach similar to nitrogen. Three major forms of phosphorus in mineral soils are organic P associated with humus, insoluble forms of mineral P, and plant-available P in soil solution. Due to the low mobility of phosphorus, surface runoff will only partially interact with the solution P stored in the top 10 mm of soil. Sediment transport of organic and mineral P to the stream is calculated with a loading function as described in organic N transport.

The SWAT in-stream water quality algorithms incorporate constituent interactions and relationships used in the QUAL2E model (Brown and Barnwell, 1987), which contains the major interactive factors such as the nutrient cycles, algae production, and benthic oxygen demand.

4.2.4 Model inputs

The ArcSWAT interface for SWAT version 2005 (Winchell et al., 2007) was used to compile the SWAT input files. The basic data sets required to set up the model inputs are: topography, soil, land use and climatic data. These data are described in Table 4.2. The topographic information was used for automatic delineation of the watershed. Land use and soil map were superimposed on the catchment's subbasins. In this study the SWAT model was conducted by dividing the watershed of Kielstau into 8 subbasins (Figure 4.1) and 154 HRUs.

The current management practice in this catchment involves a 3-year crop rotation (winter wheat – winter wheat – rape) and monocultural maize, which were simulated in the model. In our simulation, winter wheat is planted from mid-September after a tillage operation and is harvested in the beginning of August in the subsequent year whereupon soil is tilled again. Rape is planted from the end of August and harvested in the beginning of July in the subsequent year. The amount and date of fertilizer application are in conformance with the conventional cultivation (Lksh, 2006).

Table 4.2 Model input data sources for the Kielstau watershed

Data type	Source	Data description/properties
Topography	LVermA (1995)	Digital elevation model, a grid size of 25 m×25 m
Soil map	BGR (1999)	Soil physical properties such as texture, saturated conductivity, etc. Scale of soil map (1: 200, 000)
Land use map	DLR (1995)	Land use classifications, 25 m × 25 m resolution
Climate data	DWD (2009a, b)	Temperature, precipitation, wind speed, humidity (Meierwik station, 1993–2008)

The nitrogen deposition was estimated to be 8.4 kg N/ha/year (Lam et al., 2010). Rates of N-fertilizer including nitrogen deposition and P-fertilizer application (in kg ha-1) were set, respectively, to 210 and 50 for winter wheat, 190 and 60 for rape, 179 and 72 for maize, and 160 and 30 for pasture. The manure application for arable land ranged from 80 kg ha-1 to 240 kg ha-1. In the pasture lands, cow is dominant animal that has been grazing mainly in summer season. Average livestock (LU) density obtained from local people consultation and other literatures is estimated to be 2 LU ha-1 (Kreins et al., 2003; Gömann et al., 2005). Data of average monthly nutrient taken from the six wastewater treatment plants from 2002 to 2008 (Kreis Schleswig-Flensburg, 2009) were implemented as point sources in the model. All the default values mentioned above were used for baseline simulation.

4.2.5 Model calibration

The application of the model first involved the analysis of parameter sensitivity, which was then used for model auto-calibration following the hierarchy of sensitive model parameters. The sensitivity analysis method (Morris, 1991) was conducted and aims to assess the most sensitive parameters for setting up the model in this catchment. Model auto-calibration was performed by changing each parameter ten times within the allowable range of values for the specific parameter. Detailed calibration procedures for the SWAT model and the definitions of various calibration parameters are depicted by Neitsch et al. (2002).

The auto-calibration was carried out using flow data from the hydrological years 1998–2004. The validation was done for the continuous time 2004–2008. The manual sediment simulation was conducted from October 2006 to December 2008 due to the availability of measured data. For the nutrient simulation, the manual calibration was performed for the period of May 2006 to October 2007; the validation was then conducted for the period of November 2007 until December 2008. Simulation of flow, sediment and nutrient load was performed for daily time step using measured data from the Soltfeld gauging station at the catchment outlet (Figure 4.1). Measured daily nutrient and sediment loads are obtained by multiplying their daily average concentration with daily average discharge at the watershed outlet. The parameters used for model calibration are presented in Table 4.3.

4.2.5. 1 Flow

Surface runoff and base flow were calibrated simultaneously. Parameters adjusted for surface runoff were curve number (CN2) and available water capacity (SOL_AWC). The main parameters adjusted for base flow were soil evaporation compensation factor (ESCO), plant uptake compensation factor (EPCO), groundwater revap coefficient (GW_REVAP), and threshold depth of water in shallow aquifer (GWQMN). For instance, ESCO, GW_REVAP, and depth-to-subsurface drain (DDRAIN) have been adjusted from the default values of 1; 0.02; 0 to the simulated values of 0.95; 0.2; 800, respectively. The different pathways modeled by the SWAT model are separated into the transport through surface and subsurface runoff, upwelling of groundwater, groundwater and transport to the river, and deep aquifer recharge. The groundwater flow (57.4%) and upwelling of groundwater (22.2%) are dominant components and very important in lowlands because of the high interaction between surface water and the shallow groundwater

(Schmalz et al., 2009), while the surface runoff is very low (0.3 %) due to the flat topography - a special characteristic of lowland area (Lam et al., 2010). For this lowland catchment, GW_REVAP is an important parameter controlling the upwelling of groundwater and nitrate transport with water from the shallow aquifer to the unsaturated soil zone in response to water deficiencies. SWAT models the movement of water into overlying unsaturated soil layers as a function of water demand for evapotranspiration. This process is significant in watershed where the saturated zone is near below the surface or where deep-rooted plants are growing. Revap is allowed to occur only if the amount of water stored in the shallow aquifer exceeds a threshold water level. The allowable range of this parameter is between 0.02 and 0.2. As GW_REVAP approaches 0, movement of water from the shallow aquifer to the root zone is restricted. This parameter was changed from its initial value of 0.02 to 0.2 in order to obtain a better fit between the model results and the measured data. The auto-calibration processes were also implemented similarly for other parameters within their allowable range in SWAT. Main parameters used for the sensitivity analysis and the calibration of flow are described in Table 4.3.

Table 4.3 Main variables used for sensitivity analysis and calibration in SWAT

Variable name	Model processes	Description	Allowable range	Actual value used
GW_REVAP	Flow	Ground water revap coefficient	0.02-0.2	0.2
GW_DELAY	Flow	Delay time for aquifer recharge	0-500	14
GWQMN	Flow	Threshold depth of water in shallow aquifer	0-5000	50
ALPHA_BF	Flow	Base flow recession constant	0-1	1
RCHRG_DP	Flow	Deep aquifer percolation coefficient	0-1	0.05
ESCO	Flow	Soil evaporation compensation factor	0.01-1	0.95
EPCO	Flow	Plant uptake compensation factor	0.01-1	1
CN_2 (arable land, pasture, range brush, and forest)	Flow	Curve number	35-98	64, 46, 46, 35
SOL_AWC	Flow	Available water capacity	0-1	0.12-0.54
DDRAIN	Flow	Depth to subsurface drain	0-2000	800
TDRAIN	Flow	Time to drain soil to field capacity	0-72	24
GDRAIN	Flow	Drain tile lag time	0-100	8
C FACTOR	Sediment	Cover or management factor	0.003-0.2	AGRN:0.2, PAST: 0.003
USLE_P	Sediment	Support practice factor	0-1	0.5
SPEXP	Sediment	Exponential factor for channel sediment routing	1-1.5	1
SPCON	Sediment	Linear factor for channel sediment routing	0.0001-0.01	0.0001
CH_EROD	Sediment	Channel erodibility factor	0-1	0.25
CH_COV	Sediment	Channel cover factor	0-1	0.2

BIOMIX	Sediment, organic and mineral nutrient	Biological mixing efficiency	0.01-0.4	0.2
RSDCO	Sediment, organic and mineral nutrient	Residue decomposition coefficient	0.01-0.1	0.05
SOL_NO_3	Nitrogen	Initial NO_3 concentration in the soil layer	0-5	5
SOL_ORGN	Nitrogen	Initial organic nitrogen concentration in the soil layer	0-10000	100
NPERCO	Nitrogen	Nitrogen percolation coefficient	0.01-1	0.95
AL1	Nitrogen	Fraction of algae biomass that is nitrogen	0.07-0.09	0.08
BC2	Nitrogen	Rate constant for biological oxidation of NO_2 to NO_3	0.2-2	1.1
SOL_SOLP	Phosphorus	Initial soluble phosphorus concentration in the soil layer	0-100	5
SOL_ORGP	Phosphorus	Initial organic phosphorus concentration in the soil layer	0-4000	100
PPERCO	Phosphorus	Phosphorus percolation coefficient	10-17.5	10
PHOSKD	Phosphorus	Phosphorus soil partitioning coefficient	100-175	100
AL2	Phosphorus	Fraction of algae that is phosphorus	0.01-0.02	0.02
RS5	Phosphorus	Organic phosphorus setting rate in the reach	0.001-0.1	0.1
BC4	Phosphorus	Rate constant for mineralization of organic phosphorus to dissolved phosphorus	0.01-0.7	0.1

4.2.5. 2 Sediment and nutrients

Sediment calibration in the SWAT model was implemented by changing parameters in relation to the loading from subbasins and channel degradation/ deposition. While surface runoff is the primary factor controlling sediment load to the stream, some important parameters affecting sediment movement from subbasins into the stream were changed. These include cover or management factor (C FACTOR) using for agricultural crops and pasture, support practice factor (USLE_P) for contour farming terraced field. Variables affecting channel degradation/ deposition have also been changed subsequently to represent the cohesive nature of the channels in the watershed. These variables involve exponential factor for channel sediment routing SPEXP, channel erodibility factor CH_EROD, and channel cover factor CH_COV. Sediment deposition and channel degradation are the two dominant channel processes, which affect sediment yield at the watershed outlet.

Nutrient calibration was simulated into two steps, calibration of nutrient load and calibration of in-stream water quality processes. Before implementing simulation of nutrient, initial concentration of organic and mineral nitrogen and phosphorus in the upper soil layer (SOL_ORGN, SOL_NO_3, SOL_ORGP, and SOL_SOL P) were assumed. Main parameters adjusted for nutrient calibration were nitrogen and phosphorus percolation (NPERCO, PPERCO), phosphorus soil partitioning coefficient (PHOSKD), residue mineralization (RSDCO), and biological mixing efficiency (BIOMIX). These parameters were verified within their allowable ranges. In addition, some parameters affecting in-stream water quality processes such as fraction of algae biomass that is nitrogen and phosphorus (AL1, AL2), rate constant for biological oxidation of NO_2 to NO_3 (BC2), and rate constant for mineralization of organic phosphorus to dissolved phosphorus (BC4) were also considered.

Several statistics, including mean, coefficient of determination, and Nash-Sutcliffe efficiency (Nash and Sutcliffe, 1970) have been used to evaluate the accuracy of model predictions. The coefficient of determination (R^2) is the square of the Pearson's product-moment correlation coefficient, which is an indicator of strength of relationship between measured and simulated values. Nash-Sutcliffe efficiency (E_{NS}) has been widely used to evaluate the performance of hydrologic models. The values of E_{NS} and R^2 range from 0 to 1. If the E_{NS} and R^2 values are less than or close to 0, the model simulation is taken as an indication of poor or unacceptable performance. The closer the values get to 1, the more perfect the model simulation. Santhi et al. (2001) found that a value greater than 0.5 for these variables constitutes an acceptable simulation model.

4.2.6 Best management practices scenarios

Best Management Practices are used broadly as field measures that reduce the negative impact of an activity on the environment. In relation to agriculture and water quality, a BMP could be a change in farm or land management to improve the water quality in agricultural fields. In this study, the calibrated and validated model was used to assess the impacts of various structural and nonstructural BMPs on water quality in this watershed. Changes in sediment and nutrient load between these scenarios compared to a baseline scenario, which based on the current practices provided the percentage of reduction in pollution in the watershed. Because diffuse sources were found to be the dominant sources of nitrogen in the entire catchment (Lam et al., 2010), BMPs implemented in the fields in order to reduce diffuse source pollution were considered.

Structural and nonstructural management approaches have been considered and selected in accordance with realistic condition (topography, agricultural cultivation, and farming technique) of the lowland catchment. Structural BMPs can be used to control the volume of pollutants through crop rotation, physical containment and flow restrictions of pollutants. Such structural BMPs recommended in this study include extensive land use management (ELUM), grazing management practice (GZM), and field buffer strips (FBS). On the other hand, nonstructural BMPs do not require permanent structures but typically require modifying farming practices or farmer behaviors (Lambert et al., 2007; Taylor and Wong, 2002). A nonstructural BMP recommended in this study is nutrient management plan (NMP). In addition, a combination of feasible scenarios (combined scenarios, CBN) was also considered to assess generally the impact of BMPs on reduction in diffuse source pollution at the main watershed outlet. The recommended BMPs can be found in Table 4.4.

Table 4.4 Description of BMPs simulated for the Kielstau catchment

Measure	Description	Code
Extensive land use management	Combination of different crop rotations and tillage	ELUM
Nutrient management plan	Reducing nutrient application (both mineral fertilizer and manure) in arable land by 20%	NMP
Grazing management practice	Reduction of livestock density from 2 LU ha^{-1} to 1.1 LU ha^{-1} and no fertilizer application on pasture land	GZM
Field buffer strip	Application of 10 m field buffer strips on arable and pasture land. Field buffer strips are installed along the edge of main channel	FBS
Combination scenarios	Combination of the most efficient scenarios	CBN

4.2.6.1 Extensive land use management

To assess the impacts of ELUM on the behaviors of sediment and nutrient load at the watershed outlet, ELUM scenarios were created relating to various crop rotations and different tillage treatments. Several scenarios were performed considering 21 combinations of three types of tillage and seven crop rotations. The three tillage systems employed were conventional tillage (baseline), conservation tillage, and no till (Table 4.5). Conventional tillage used for current agricultural practices in the Kielstau catchment comprised a moldboards plow operation after harvesting. The depth for this operation was set at 150 mm and the residue mixing efficiency at 95%. Conservation tillage or mulch tillage used the chisel plow, which causes less residue disturbance than moldboard plows. No till is one kind of conservation tillage involving minimum disturbance of the soil to maintain the residue level after harvest time.

Table 4.5 Tillage treatments and their mixing efficiency

Tillage treatment	Mixing efficiency (%)	Tillage depth (mm)
Conventional (baseline)	0.95	150
Conservation	0.25	100
No till	0.05	25

Seven different crop rotations were created on arable land (Table 4.6). These rotations have been chosen basing on realistic cultivation condition at the locality. The timing and magnitude of fertilization for each crop was not varied across scenarios. The amount of N-fertilizer and P-fertilizer application ranged from 140–210 kg/ha/year and from 40–72 kg/ha/year, respectively (Lksh, 2006). Detailed crop rotations and fertilizer application rate are given in Table 4.6.

Table 4.6 Crop rotations and fertilizer application for the Kielstau catchment

Crop rotations	Mineral fertilizer		Livestock manure[a]	Code
	N (kg/ha/year)	P (kg/ha/year)	(kg/ha/year)	
winter wheat–winter wheat–rape (Baseline scenario)	210–210–190	50–50–60	240–240–120	WWR

rape–winter wheat–winter barley	190–210–180	60–50–40	120–240–120	RWB
sugar beet–winter wheat–winter barley	140–210–180	45–50–40	135–240–120	SbWB
maize–maize–maize	179–179–179	72–72–72	150–150–150	MMM
winter wheat–winter rye–winter rye	210–150–150	50–40–40	240–190–190	WRyRy
winter rye–winter rye–winter rye	150–150–150	40–40–40	190–190–190	RyRyRy
winter wheat–maize–winter rye	210–179–150	50–72–40	240–150–190	WMRy

[a] Livestock manure application (slurry cow and pig) is split mainly two times in spring and autumn seasons during the cultivation period.

4.2.6.2 Nutrient management plan

Assessment of efficient NMP helps to enhance the economic and environmental sustainability of the agricultural system. NMP typically involves soil nutrient testing, equipment calibration, timing of fertilizer application, and record keeping (Ribaudo and Johansson, 2007). Reduction in fertilizer application for arable land has been applied in various watershed scales (Jha et al. 2006; Hesse et al., 2008). The appropriate rate of reduction in fertilizer application suggested in those studies ranged from 10 to 20%. In this paper, decrease in nutrient application for both mineral fertilizer and manure in arable land assumed for this scenario is 20% aiming at evaluating the ability of pollution reduction in nutrient at the outlet of the lowland Kielstau catchment.

4.2.6.3 Grazing management practice

Implementation of efficient grazing management in a field will result in: (a) reduction of chemical fertilizers by utilizing manure fertilizer, (b) reduction of rill erosion by preventing animals from trampling vegetative cover, disturbing soil surfaces, and destroying stream banks and riparian areas, (c) and reduction of water resource pollution by increasing nitrogen availability to plants, controlling livestock density. The pasture areas occupied more than 26% of the study area, the reduction of pollution resulting from these pasture areas therefore contributed importantly in minimizing water pollution at downstream of the lowland catchment. SWAT provides grazing option to simulate plant biomass removal and manure deposition over a specified time period on pasture and range grazed by animal. Information required in the grazing operation includes the length of grazing period (GRZ_DAYS), dry weight of biomass consumed daily (BIO_EAT), the amount of manure deposited daily (MANURE_KG) and minimum plant biomass for grazing (BIO_MIN, kg/ha).

In this scenario, no fertilizer application has been used for pasture land. The rate of livestock density was reduced from 2 to 1.1 LU ha^{-1} in accordance with the target of German policy (BLE, 2008). Grazing was rotated three times per year, on May 1, July 1, and September 15. Rotational grazing areas are carried out within pasture areas as well. Number of grazing days was obtained from various literature sources: FAPRI (2006) suggested grazing period of 30 days for grass land using the SWAT model, Frame (1992) suggested 10 (20) to 30 days depending on stocking rate and on the time in the year. For this study, length of grazing period of 28 days was assumed for cow. The manure deposited was estimated to be 4.1 kg dry matter/ha/day according to the recommendations proposed by Frame (1992). Minimum plant biomass (BIO_MIN) for grazing was set to 200. This variable was created so that the plant cover in an HRU would not be reduced to zero when grazing has been implemented in pature land. If the plant biomass falls below 200, the model will not graze, trample, or apply manure in the HRU on that day.

4.2.6.4 Field buffer strip

Vegetative buffer strips are installed along the edge of main channel segment in the lowland catchment to reduce sediment and nutrients in surface and subsurface runoff. Permanent vegetation planted in buffer strips is Bermuda grass or alfalfa. A primary mechanism of vegetative buffer strips is to reduce flow velocities because the vegetation provides greater resistance to water flow. The reduction in surface runoff velocity then causes deposition of suspended particles and increased infiltration, which minimizes pollution. Furthermore, buffer strips can facilitate biological transformations which diminish the nutrient load of subsurface flows such as plant uptake, microbial immobilization, nitrification, and denitrification (Lowrance et al., 1984).

The pollutants loads resulting from areas within the watershed which drain into the channel segment are trapped in the vegetative strip. SWAT provides a specific method to incorporate edge-of-field filter strips through the FILTERW parameter that reflects the width of the strip. The trapping efficiency for sediment and nutrient ($trap_{ef}$) is calculated by the equation:

$$Trap_{ef} = 0.367.\ FILTERW^{0.2967} \qquad (1)$$

Filter strips can also reduce loads of constituents in subsurface flow that pass through the strip (Neitsch et al., 2005). The trapping efficiency for subsurface flow constituents ($Trap_{ef,\ sub}$) is calculated by the equation:

$$Trap_{ef,\ sub} = \frac{(2.1661.FILTERW - 5.1302)}{100} \qquad (2)$$

Although there are many factors affecting sediment and nutrient trapping efficiency, such as runoff volume, soil properties, and vegetative properties, the SWAT model algorithm to simulate FBS effects on sediment and nutrient reduction is set only as a function of width. These equations were taken from US empirical data on buffer strip efficiency. Numerous studies have used these equations to evaluate the efficiencies of buffer strips on reduction in sediment and nutrient loads in various region scales (Bracmort et al., 2006; Bärlund et al., 2007; Arabi et al., 2007).

For this scenario field buffer strips were applied only for arable land and pasture land along the main channel. The width of edge-of field filter strips (FILTERW) was assumed 10 m in the model.

4.2.7 Cost estimation of BMPs

Cost estimates for establishing BMPs tend to vary from farm to farm, and depend on factors such as geographic area, topography, accessibility of equipment, and government regulations, etc. For this study, a method for calculating BMP costs is estimated item-by-item using information from a regional database or other relevant cost estimation resources. The important aspects of BMPs costs have been considered including costs of investigation, design, installation, operation, and maintenance. In addition, opportunity cost and interest rate were also considered. For both the structural and nonstructural BMPs, the cost analysis was based on data obtained from farmer records and document involving in BMP design, construction, and implementation. Cost estimates were based on current prices in the year of 2008/2009 in Germany (KTBL, 2008; SBD, 2009).

Compared to structural BMPs, the cost estimates for nonstructural BMPs are difficult to determine, because of their indirect and highly variable implementation levels (US EPA, 2004). In addition, data on such BMP performance tends to be limited. The main costs estimated for three structural BMPs include (a) the costs of materials for construction (b) costs of machinery, labor, and equipment for installation and operation, while main costs of nonstructural BMP comprise costs of soil test, experiment, and field investigation. Costs of materials employed in BMP construction and establishment were estimated by multiplying the input levels used by 2008/2009 prices obtained from a local database. Implementation of tillage operation systems was considered as equipment rental at current retail rates and their costs were derived from KTBL (2008) in which costs of machinery, tractors, labor, interest, insurances, taxes, technical inspections, and repair as well as maintenance have been taken into account. Labor costs for construction and other aspects varied depending on the task. The actual amounts were computed by multiplying hourly rates by the number of hours of hired labor. The opportunity cost of land used for constructing FBS was also included. It was estimated as the loss of revenue from crop production when the land is switched into 10 m wide strip of land.

The expected lifetime of BMPs was considered in this study. Total BMP costs were reduced to annual values using the equation (Degarmo et al., 1997; Gitau et al., 2004)

$$A_{BMP} = \frac{Z\left(\frac{r}{100}\right)}{1-\left(1+\frac{r}{100}\right)^{-n}} \tag{3}$$

where A_{BMP} is the annualized cost for a BMP (€), Z is the capital cost of a BMP (€), r represents the time value of money (%) and n is the expected lifetime of the BMP (years). Interest rate assumed for the time value of money was 6%. The expected lifetimes of BMPs implementation are given in Table 4.9.

4.3 Results and discussion

4.3.1 Simulation of flow

The simulated results of flow were obtained from the previous study (Lam et al., 2010). Detailed analyses of the flow can be found also in Lam et al. (2010). The modeled results shows good agreement between simulated and measured daily discharge with E_{NS} and R^2 at 0.75 and 0.78 for the calibration period and 0.78 and 0.84 for the validation period at the outlet of the Kielstau catchment, respectively (Figure 4.3).

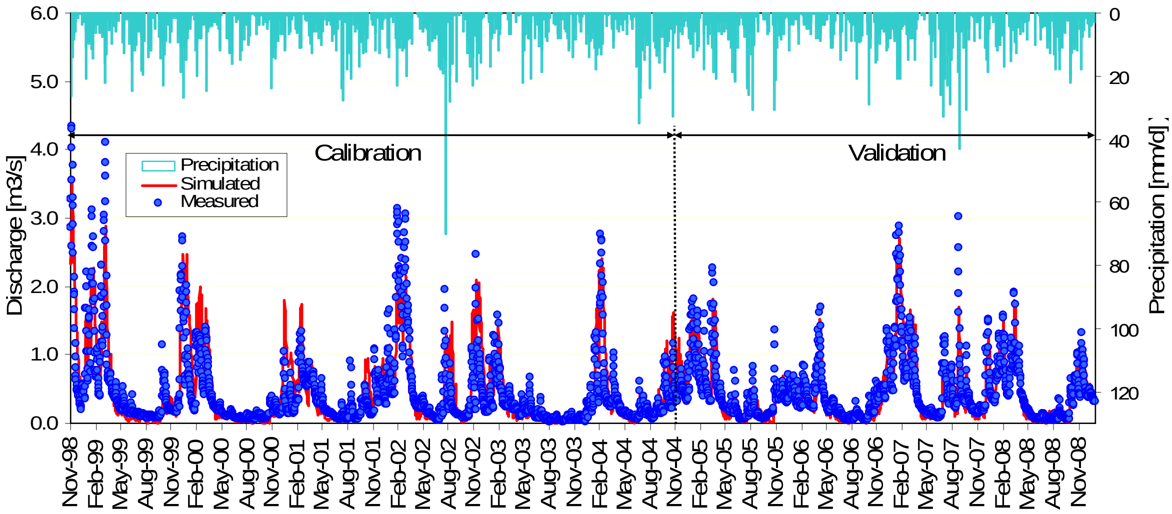


Figure 4.3: Simulated and measured daily discharge at the Soltfeld gauging station

The results of different runoff components obtained from the previous study (Lam et al., 2010) are also illustrated in Figure 4.4. These results indicate that groundwater flow is dominant pathway in this lowland catchment, while surface runoff is very low compared to other components. This is one of the special characteristics of lowland areas in comparison with mountainous areas where surface runoff is considered to be dominant pathway (Lenhart et al., 2002; Van Griensven at al., 2006).

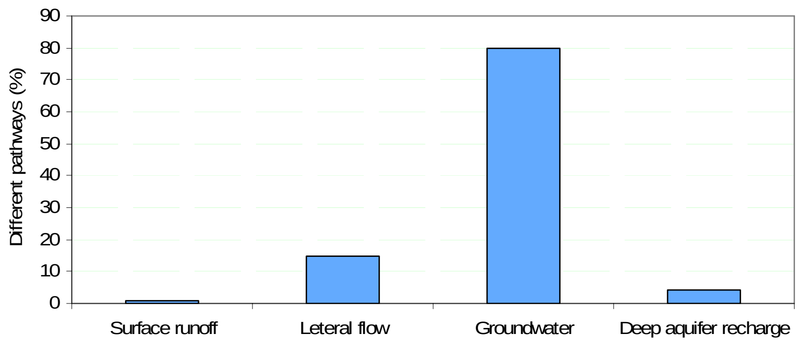


Figure 4.4: Comparison of runoff components in the Kielstau lowland catchment

4.3.2 Simulation of sediment load

The temporal variation of sediment load at the main watershed outlet is represented in Figure 4.5. The trend in sediment load was generally in reasonable agreement with measured data. This is confirmed by the model's efficiency E_{NS} of 0.57 for the calibration period, which increased to 0.58 for the validation period (Table 4.7). The coefficient of determination R^2 increased also from 0.63 to 0.65. However, considerably underestimated sediment load occurred during the winter periods of 2007 and 2008. This underestimation is mainly caused by the underestimated flow in these periods. The sediment load at the watershed outlet is contributed by the loading from subbasins and channel systems within the watershed. The main variables adjusted in the calibration model are given in Table 4.3. The results of the model showed that average annual sediment input from channel to the river was estimated to be 72% of the total average annual sediment load at the main watershed outlet, while only 28% of that resulting from the fields and drainage systems within the catchment. The reasons for low sediment load from the fields are due to the characteristic of flat area and low surface runoff in this lowland catchment (Figure 4.4). Thus it can be stated that sediment load from channel caused by bank erosion is dominant, while the loading of sediment from the fields is negligible. These results are also similar to the studied results by Kiesel et al. (2009) when they used the SEPAL approach to quantify sediment pathways in the same catchment and found that the bank erosion is predominant with 71%, followed by the drains with 15% and fields with 14%.

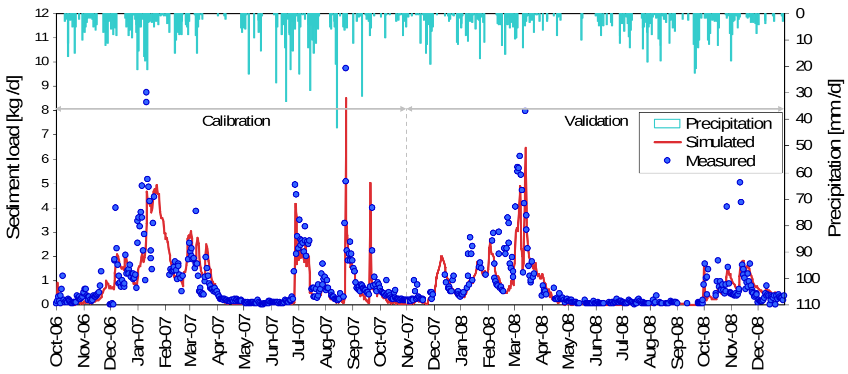


Figure 4.5: Simulated and measured daily sediment load at the Soltfeld gauging station

4.3.3 Simulation of phosphorus load

Simulated and measured loads for mineral P and to total phosphorus (TP) are given in Figure 4.6. Parameters used for manual calibration of phosphorus load are listed in Table 4.3. Total P is calculated by summing mineral and organic phosphorus in the model. Overall the model underpredicted mineral P and TP load in both the calibration period and the validation period. This is given by the values of E_{NS} and R^2 ranged from 0.42 to 0.48 and from 0.58 to 0.69 for the calibration and validation period, respectively (Table 4.7). The underestimation of phosphorus load at the outlet of watershed can be attributed due to the following two reasons: (a) the underestimation of some peak flows in winter seasons result in the underestimation of the corresponding phosphorus peaks and (b) the low sediment load from the subbasins which is considered to be a main reason leads to underestimation of phosphorus load. Due to the fact that organic and mineral phosphorus attached to sediment are transported by surface runoff to the main channel, phosphorus load thus is considerably influenced by sediment load within a watershed. Once the rate of sediment load from the fields is negligible, the amount of phosphorus transported with sediment to the stream is also small. Therefore, the underestimation of mineral P and total P are occurred during the whole time.

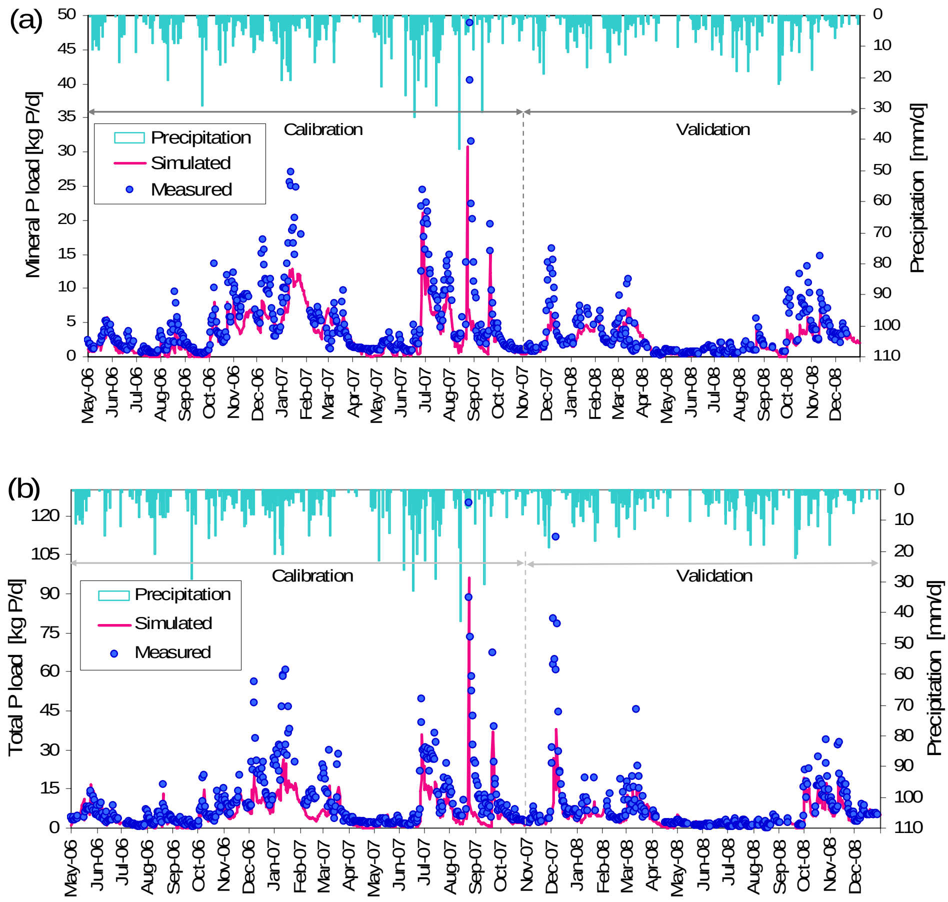


Figure 4.6: Simulated and measured daily phosphorus load at the Soltfeld gauging station

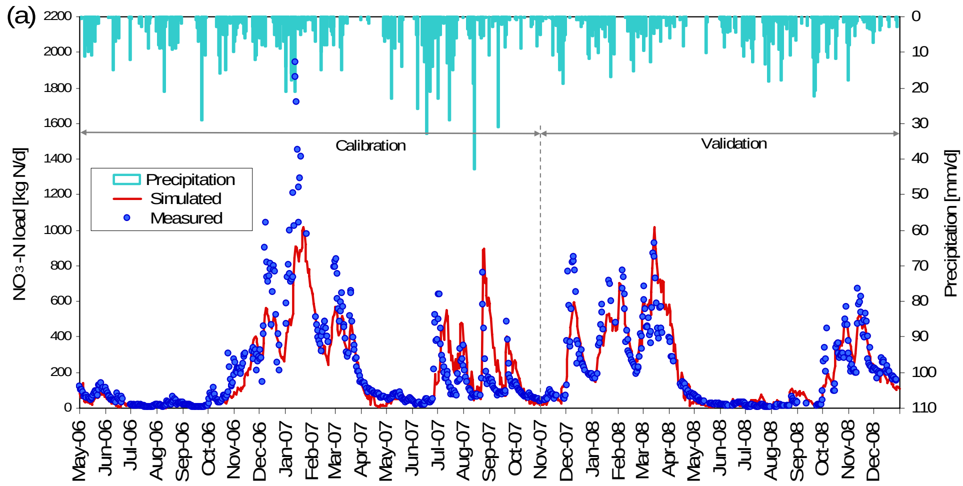
4.3.4 Simulation of nitrogen load

The comparison between simulated and measures results of daily nitrogen load at the gauge Soltfeld is shown in Figure 4.7. In this study, total nitrogen is sum of mineral and organic nitrogen. Parameters used for manual calibration of nitrogen load are given in Table 4.3. Mean simulated daily flow, sediment and nutrient loads were compared with corresponding mean daily measured data. The results in Table 4.7 show that the mean values of measured and simulated runoff, sediment, and nutrient load were not significant different except for phosphorus in general.

Regarding nitrate (NO_3-N) load, the simulated results of the NO_3-N load for the period of May 2005 to October 2008 were obtained from the previous study (Lam et al., 2010). Additional measured nitrate load for the period of November to December 2008 were expanded and used to increase the validation period in this study (Figure 4.7a). Detailed analyses of the NO_3-N load can be found in Lam et al. (2010). The model's efficiency E_{NS} and the coefficient of determination R^2 ranged from 0.68 to 0.75 and from 0.7 to 0.81 for the calibration and validation period, respectively, showed a good agreement between the measured and simulated nitrate load at the watershed outlet and indicated that the model simulates considerable accuracy of nitrate load during both the calibration and the validation period.

Ammonium (NH_4-N), another pool of mineral nitrogen, contributes importantly to the nitrogen processes. The contribution inputs of NH_4-N load to the watershed outlet are not only the contribution of diffuse sources, but also the point sources (six municipal wastewater treatment plants) of which Moorau tributary (Figure 4.1) is considered to be considerable influence on NH_4-N concentration in the river Kielstau (Schmalz et al., 2008b). Comparison between the measured NH_4-N load and the corresponding simulated values are shown in Figure 4.7b for both the calibration and validation period. In general, the model underpredicted NH_4-N load for the whole simulation periods. The main reason for the underestimation of NH_4-N can be explained with low NH_4-N concentration input contributed by the point sources. In addition, since point source effluents were used as input data for average monthly values, the model did not adequately simulate the influences of point source emissions on daily nitrate concentrations, especially in high emission day of point sources.

As can be seen in Figure 4.7c, the trend in total nitrogen (TN) load is very similar to the NO_3-N load trend. This is because the nitrate load is dominant among nitrogen fractions. The high values of E_{NS} showed close agreement between the measured and simulated TN load during the calibration and validation period (Table 4.7). Moreover, the values of R^2 are fairly high, achieved 0.81 and 0.83 for the calibration and validation period, respectively. The values of these statistical parameters indicated that the model simulated accurately TN load at the catchment outlet.



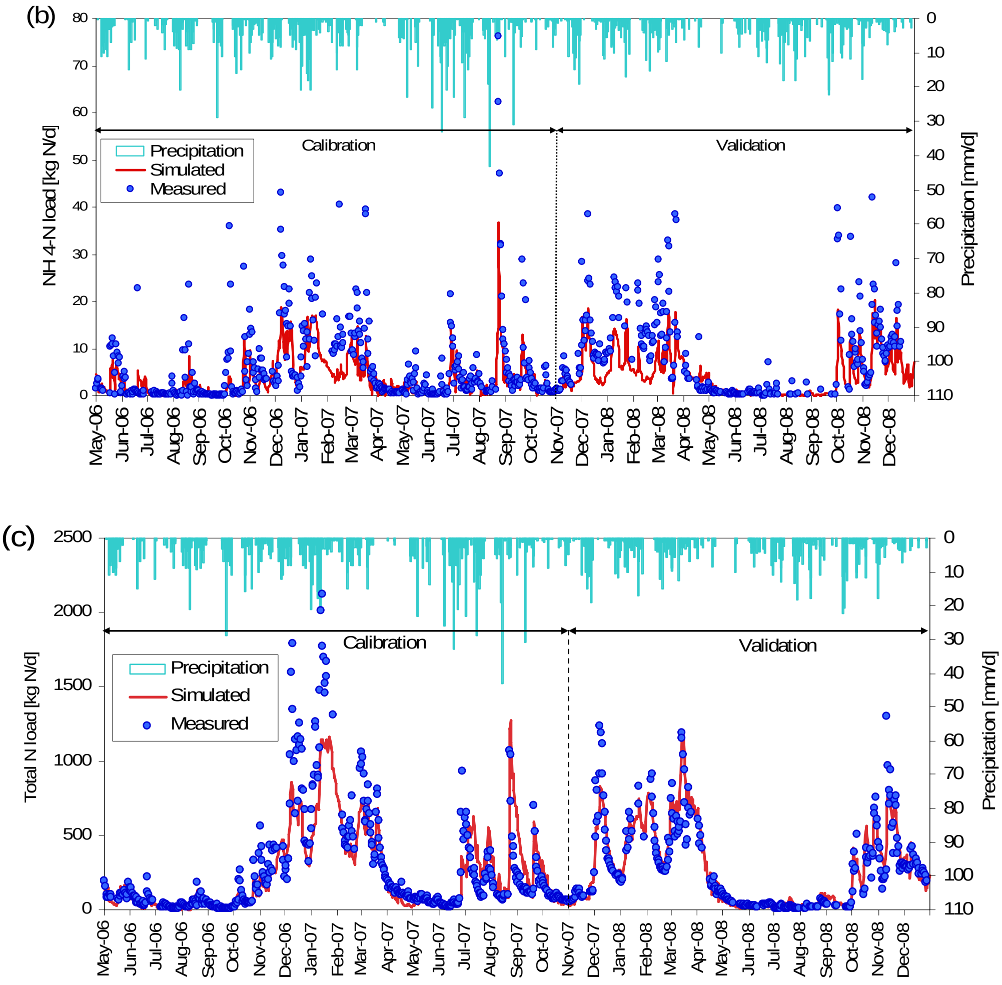


Figure 4.7: Simulated and measured daily nutrient load at the Soltfeld gauging station

In general, the values of E_{NS} and R^2 obtained from NO_3-N and TN simulations indicated that the SWAT model performed satisfactorily in accurately simulating daily nitrate and total nitrogen load for the entire lowland catchment.

Table 4.7 Measured and simulated daily flow, sediment, and nutrient load at the Soltfeld gauging station

Variable	Calibration				Validation			
	Mean		R^2	E_{NS}	Mean		R^2	E_{NS}
	Measured	*Simulated*			*Measured*	*Simulated*		
Flow (m3/s)	0.44	0.47	0.78	0.75	0.46	0.46	0.84	0.78
Sediment (T)	0.94	0.88	0.63	0.57	0.72	0.56	0.65	0.58
NO_3-N (kg)	195.75	194	0.70	0.68	218.33	217.76	0.81	0.75
NH_4-N (kg)	5.75	3.49	0.61	0.44	9.01	4.33	0.69	0.46
TN (kg)	257.73	251.83	0.81	0.71	280.83	276.61	0.83	0.75
Mineral P (kg)	5.38	3.37	0.58	0.42	3.07	2.18	0.58	0.45
TP (kg)	10.37	5.52	0.68	0.47	7.99	4.10	0.69	0.48

The simulations of flow, sediment, and nutrient load were implemented by the SWAT model using current practices. From the simulated results, it was confirmed that the model was simulating reasonably well in general and could be used as a base for evaluating the effectiveness of BMPs in mitigating water pollution at the outlet of the lowland Kielstau catchment.

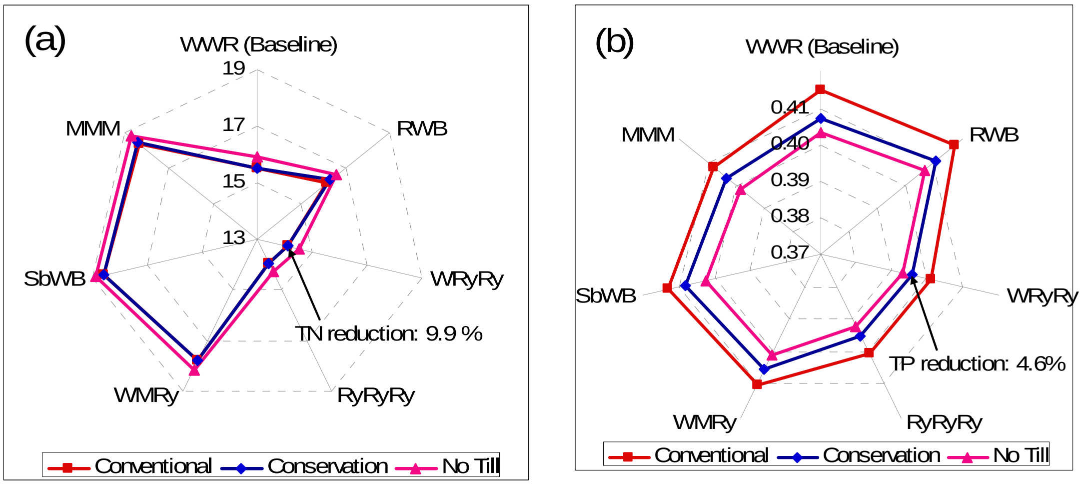
4.3.5 Effectiveness of Best Management Practices

Model simulation was performed to quantify the impacts of the BMP scenarios on water quality over a 3-year period (2006–2008). The average annual loads for flow, sediment, and nutrient were calculated under each scenario and compared with values obtained from the baseline condition in the same simulation period. The difference in average annual load between a BMP scenario and the baseline was used to indicate the load reduction achieved by BMP implementation.

4.3.5.1 Extensive land use management

The results of average annual TN and TP load under different scenarios obtained by simulating the combination between three types of tillage and seven crop rotations are illustrated in Figure 4.8. As it can be seen in Figure 4.8a, the TN load is comparatively high in case of no till for all crop rotations. No significant difference was obtained between the conventional tillage and conservation tillage. This is due to the fact that the fraction of a residue and nutrient pool in each soil layer is redistributed through the depth of soil that is mixed by the tillage implement. Because the soil is mixed to a depth of 100 mm, only the nitrate in the surface and near surface layer is available for redistribution and it is subject to removal in surface runoff (Neitsch et al., 2005). However the surface runoff in this lowland catchment is very low compared to other components (Lam et al., 2010). Thus, the changes in nitrate load from the arable fields to the stream under different tillage scenarios are not significant. In addition, the timing of tillage for a crop was assumed a specific time for the whole catchment in the SWAT model. In reality, arable areas is located dispersedly everywhere in the catchment. Hence the timing of tillage is also not the same and is dependent on the climate condition, harvest time as well as farmer decision in different local areas within the catchment. This limitation of SWAT model may result in inaccuracy of model prediction.

Considering the implementation of various crop rotations, the RyRyRy rotation has the lowest TN load among all scenarios. It was 11.2% lower than the baseline result and followed by WRyRy (9.9%). The main reason for this is that the use of nitrogen fertilizer for rye is lower than for other crops. Other investigators (Hesse et al., 2008) also obtained similar results. They reported that the reduction of nitrogen load was 20% by changing of three crop types' winter wheat, sugar beet, and winter rape to winter rye in a lowland catchment located in the north of Germany. Therefore, rye was found to be most efficient in minimizing nitrogen losses. This plant is also considered an important cover crop in a variety of cropping system because it contributes organic matter, reduces soil erosion, inhibits weeds, and enhances water penetration and retention. However, monocultural rye cultivation may degrade the cultivated soils. For this reason, applying WRyRy rotation seems to be more appropriate for both nitrogen reduction and agricultural condition. In term of TP, loading of TP was found to be higher in case of conventional tillage, then conservation tillage, and finally no till (Figure 4.8b). However, the differences in TP load among various tillage and crop rotations were not significant at the watershed outlet. These are mainly influenced by low sediment load from the fields to stream systems within the watershed.



TN load (kg/ha) **TP load (kg/ha)**

Notes: WWR = winter wheat-winter wheat-rape; RWB = rape-winter wheat-winter barley; SbWB = sugar beet-winter wheat-winter barley; MMM = maize-maize-maize; WRyRy = winter wheat-winter rye-winter rye; RyRyRy = winter rye-winter rye-winter rye; WMRy = and winter wheat-maize-winter rye

Figure 4.8: Comparison of average annual total N and total P load under different BMP scenarios

The results obtained from Figure 4.8 indicate that implementation of tillage scenarios does not have a significant impact on nitrogen load reduction at the watershed outlet in general. The significant reduction in nitrogen load is due to the implementation of crop rotations in which WRyRy rotation has been found to be the most suitable for reducing nitrogent load. However, the conservation tillage could be achieved a little more effective in TP load reduction than that of conventional tillage. Considering the aspect of diminution in environmetal pollution, the combination ($ELUM_{(CST)}$) between conservation tillage and a WRyRy rotation was therefore recommended for the extensive land use management which reduced the respective average annual TN, and TP load by 9.9 %, and 4.6 % compared to those of baseline values (Figure 4.8). The reduction of average annual NO_3-N and sediment load were also estimated to be 11.6% and 4.5% lower than the baseline value in this combination, respectively (Figure 4.9).

4.3.5.2 Nutrient management plan

The results obtained from this scenario showed that small reduction in 20% fertilizer application for arable land resulted in fairly reduction in nitrogen load at the watershed outlet. The simulated values of average annual loads for TN, NO_3-N, sediment, and TP are 8.6%, 9.9%, 0.82%, and 1.1%, respectively. Similar results were also obtained by Jha et al. (2006) when they used SWAT model to predict the impact of the changes in nutrient application on nitrate loading and corn yield for the Racoon River Watershed and showed that nitrate loading can be achieved with low nitrogen application rate reductions ranging from 10% to 20% with relatively minimal effects on crop yield reductions of about 3% to about 9%.

4.3.5.3 Grazing management practice

In this scenario, decrease of livestock units from 2.1 to 1.1 LU ha^{-1} and no fertilizer application on pasture land resulted in remarkable reduction of nutrient loss at the watershed outlet. The reduction of average annual TN, NO_3-N, sediment, and TP load was 15.6%, 20.5%, 3.5%, and 3.9%, respectively (Figure 4.9). The achieved reduction of nitrogen probably resulted from replacing of chemical fertilizers by utilizing manure fertilizer derived from three grazing periods per year, and using lower manure application by reducing livestock units which not exceeds the amount of nitrogen taken up by forages. In fact, grazing animals and pasture production can negatively affect water quality through nutrients from urine and manure dropped by the animals and fertilizer practices, or through erosion and sediment transport into surface waters. However, proper management of grazing activities and lower livestock density as well as fertilizer application reduction can mitigate effectively such negative impacts on water quality. Similar results (Volk et al., 2009) were also attained. They noted that the reduction of TN and NO_3-N concentration were estimated to be 4.4% and 3.8%, respectively, by reducing livestock units from 2.6 to 1.4 LU ha^{-1} on pasture land of the Upper Ems River basin, northwestern Germany.

4.3.5.4 Field buffer strip

For this scenario only vegetative buffer strips were installed to the arable and pasture HRUs which are located along the main channel of the Kielstau catchment. All other parameters (e.g., crop rotation, fertilizer application rate and timing, tillage operation) stayed at the same level as for the baseline scenario. The modeled results showed that the FBS reduced the average annual loads for flow, sediment, TN, NO_3-N, and TP by 0.43%, 4.9%, 12.9%, 15.3%, and 5.3%, respectively (Figure 4.9). The buffer strips have been considered to be more efficient in reducing sediment and nutrient loads in both surface runoff and groundwater flow. Regarding the mechanism of nitrate transport under the implementation of buffer strips, the transport process can occur via surface runoff and subsurface flow. There are two main mechanisms for nutrient removal from buffer strips zones: (a) uptake by vegetation; (b) denitrification. Denitrification, the bacterial reduction of NO_3-N to N_2 or N_2O gases under anaerobic conditions, is an important mechanism for removal of nitrate from groundwater in vegetative buffers (Vidon and Hill, 2004). Numerous studies have identified biological denitrification as a key process in nitrate reduction beneath a variety of buffer vegetation and groundwater condition (Ambus and Lowrance, 1991; Ambus and Christensen, 1993; Devito et al., 2000; Gold et al., 2001). These reductions were mainly due to denitrification in the presence of abundant organic carbon available beneath the buffer strips and particularly plant uptake.

Since the surface runoff component is negligible compared to groundwater component in this lowland catchment (Lam et al., 2010), reduction of nitrate load resulted from vegetative buffer strips is predominantly attributed to the groundwater component of which denitrification could be considered to be a major mechanism compared to others such as plant uptake, subsurface flow that passes through the strips. Several studies also obtained high efficiency in nitrate reduction when implementing buffer trips. Hefting et al. (1998) indicated that nitrate concentrations in groundwater decreased by 95% when it flowed through the riparian buffer zone installed along a lowland stream in the Netherlands; Sabeter et al. (2003) reported that nitrate removal efficiency in groundwater flow beneath herbaceous buffer strip has been found to vary from 34% to 98%. These comparable results implied that the effectiveness of buffer strips on water quality improvement is achieved not only in surface runoff but also in subsurface flow in which denitrification can be found to be a crucial factor reducing nitrate load. In this study, the use of the empirical equations of the buffer strip efficiency in the SWAT model has several limitations such as: (a) the model does not take into account hydrological variations in runoff scenarios. In reality, the trap efficiency will be different for storm and normal rainfall events; (b) a similar fraction is retained in the filter strip for each compound. In fact, there is a difference between dissolve and bound fractions, and between coarse and small particles; (c) the model considers a buffer strip for an entire hydrological response unit (HRU) and not only for the areas really situated along the river (Gevaert et al., 2008). The above limitations may cause inaccurate results of prediction about the efficiency of the buffer strip. Therefore, the descriptions of the buffer strips should be required some modifications in new versions of the SWAT model in order to describe correctly the reduction efficiency in local conditions and provide more realistic results than using the original SWAT equation that only considers the width of the filter strip.

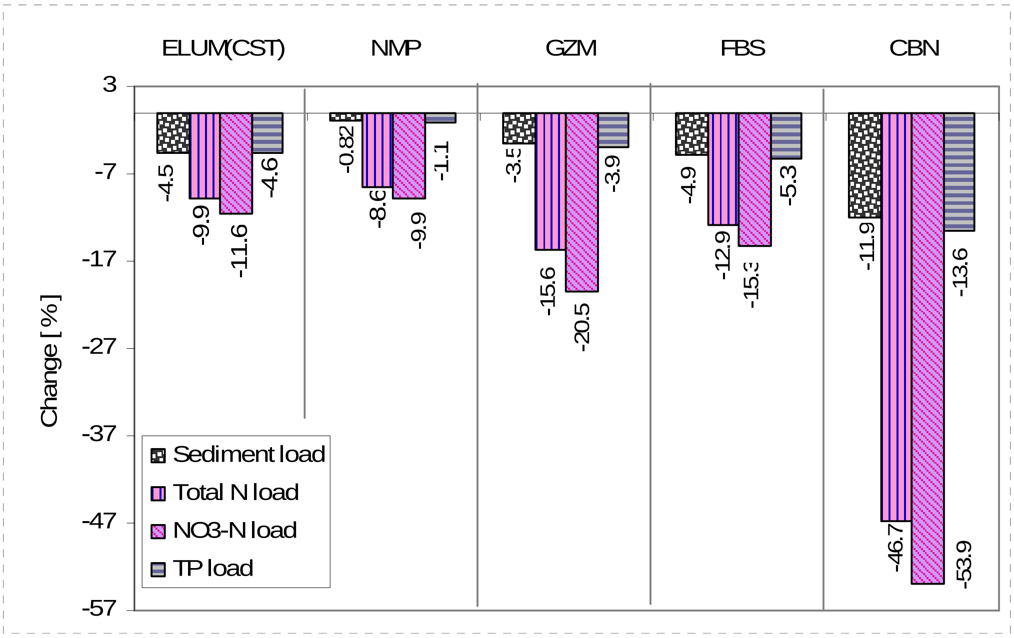


Figure 4.9: Average annual reduction in sediment and nutrient load at the Soltfeld gauging station by implementing four BMPs

4.3.5.5 *Combined scenarios*

The implementation of the individual BMP scenarios has partly contributed to improve water quality at the watershed outlet. However, no single scenario results in significant decrease of nutrient load. Thus, a combination of these scenarios was tested, which would allow notable improvement of water quality. The combined scenarios (CBN) including $ELUM_{(CST)}$, NMP, GZM, and FBS achieved considerable reduction by 11.9%, 46.7%, 53.9%, and 13.6 % for sediment, TN, NO_3-N, and TP, respectively (Figure 4.9).

In general, the implementation of the above BMPs has impact not only on reduction in nitrogen load, but also on fertilizer application. Table 4.8 shows changes in N input and output under the implementation of BMPs in the Kielstau catchment. The amount of N input and N output were compared with values achieved from the baseline scenario. The results shown in Table 4.8 indicate that the reduction in nitrogen output under BMP implementation is mostly due to the reduction in fertilizer application, except for FBS scenario. Reductions in average annual N load were small compared to decreases in fertilizer application. The highest reduction of N input is found in case of GZM, followed by ELUM and NMP. These BMPs provided fairly amount of reduction in N output at the watershed outlet, achieving from 8.6–15.6 % reduction.

Table 4.8: Changes of N input and N output reduction in different scenarios

Scenarios	Changes in N input (%)	Changes in N output (%)
Baseline [a]	*(Respective average amounts of N input and output are 161.1 kg/ha and 15.5 kg/ha in the baseline scenario)*	
ELUM(CST)	-15.8	-9.9
NMP	-14.8	-8.6
GZM	-29.7	-15.6
FBS	0	-12.9

Note: [a] Average amounts of N were calculated for the whole catchment
(-) is changes in N input and output reduction (%) compared to baseline scenario

4.3.6 Cost estimation of BMPs

In the Kielstau catchment, costs for implementation of conservation tillage were estimated basing on renting equipments in relation to soil operation such as plough, disk harrow, and tractors. Cost estimates were directly derived from KTBL (2008) of which all cost of machinery, labor, fuel, interest, insurances, taxes, repair, and maintenance have been taken into account.

Cost estimates for a nutrient management plan on arable land correspond to equipment and labor for soil testing, hiring a consultant to design the plan, and the costs of any additional activities (field mapping, fertilizer applying records, and reports). Assuming a 3-year useful life for a plan was proposed to estimate implement costs. The cost estimates for soil testing were taken from

SLN (2008), other expenditures were computed by multiplying the number of hours of hired labor by unit costs that were derived from SBD (2009).

To estimate the costs of grazing management practice, a farm size of 10ha and expected lifetime of 25 years were assumed. Costs of grazing management practice include: establishment costs (fencing wire, wood posts, stock watering tanks, pipeline, etc); other costs (labor, system design, and technical consultation); repair and maintenance costs; interest of operating expensive (assumed 6% of establishment and other costs). The livestock exclusion fence was 1.35 m high, consisting of four strands of high tensile electrical fence, and reinforced by wooden posts, where the distance between two posts is 2.7 m. Labor costs and material prices were also derived from SBD (2009).

Costs for installing field buffer strips consist of a one-time establishment expense, maintenance, and an annual land rental. The expected lifetime of 25 years was also assumed for this scenario. Costs for FBS establishment comprise sowing, fertilizer application, and labor and equipments. The cost of annual land rental is an important component of the overall cost of a vegetative field buffer strip, in fact, they may be the most important component. Land rental costs recognize the lost opportunity of not continuing to produce crops on the land used for the 10m filter strips. The unit prices of seed, fertilizer, and crop production were taken from KTBL (2008). Labor costs were taken from SBD (2009). Cost estimates for each BMP are given in Table 4.9.

Table 4.9: Annualized cost estimates and lifetime for selected management practices

Practice	Life-time (year)	Unit	Annual cost
Extensive land use management (ELUM$_{(CST)}$)	1	€/ha	17.83
Nutrient management plan (NMP)	3	€/ha	7.17
Grazing management practice (GZM)	25	€/ha	24.40
Field buffer strips (FBS)	25	€/100m	19.60

4.3.7 Cost and effectiveness of BMPs

Figure 4.10 shows the relationship between the annual cost and effectiveness of different BMPs in nutrient load reduction at the Soltfeld gauging station. In general, the implementation of individual BMPs has considerable impact on TN and NO_3-N load reductions at the watershed outlet. These reductions range from 9.9% to 20.5% compared to baseline values. The highest reduction in TN and NO_3-N load was obtained in case of GZM. In contrast, the impact of BMP implementation on TP load reduction is not considerable, achieving from 1.1% to 5.3% reduction at the watershed outlet. Annual costs for BMP implementation range from € 19000 in case of NMP to € 34000 in case of ELUM($_{CST}$). The highest cost was occurred in case of ELUM($_{CST}$), followed by FBS, GMZ, and NMP. Much of the cost associated with implementing FBS is due to land rental costs (approximately 44% of its total annual cost) used for installing buffer strips. CBN scenario achieved largest reduction in nutrient load at the watershed outlet. However, the cost for its implementation is fairly high and was estimated to be € 93,000 per year.

Although cost estimates for BMPs have been taken into account various costs as mentioned above, other aspects including a subsidy cost to adopt the practices (tillage, crop rotations) or the relevant social impacts such as changes in employment, wages, income, production, and commodity market have not been considered yet. However, these obtained results are crucial bases for considering appropriate BMPs.

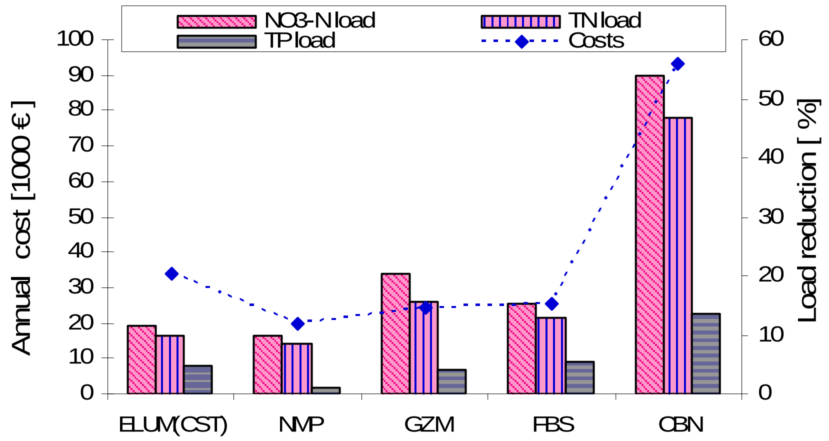


Figure 4.10: Costs and effectiveness of BMPs

To understand clearly the behaviors of nutrient concentrations, 3-year (2006–2008) concentrations of NO_3-N, TN, and TP were calculated at the mouth of watershed to provide an indication of the water quality effect of BMPs. Daily concentrations were obtained by dividing daily load by daily discharge simulated by the model at the watershed outlet. The comparison between nutrient concentrations under individual BMPs and the LAWA classification has been illustrated in Figure 4.11.

The 90th percentile value was used to compare with LAWA classification. From Figure 4.11a, it can be seen that the 90th percentile of the NO_3-N concentration fluctuated between 5.1 mg l^{-1} and 6.01 mg l^{-1} and exceeded the LAWA class II-III in most of the BMPs, except for the GZM. However, the NO_3-N concentration was reduced significantly, achieving 2.9 mg l^{-1} at the 90th percentile level of concentration by performing CBN scenario. This value was very close to the LAWA class II with an upper limit of 2.5 mg l^{-1} (LAWA, 1998) - the general target for surface waters in Germany. For TN concentration, the impact of BMPs on TN concentrations is very similar to the results for NO_3-N. The 90th percentile of the TN concentration also exceeded the LAWA class II-III (Figure 4.11b). When implementing CBN scenario, it has been significantly reduced by 5.38 mg TN l^{-1} and reached nearly the LAWA class II. These analyses indicated that the implementation of CBN scenario has great impact on the reduction of concentration values for NO_3-N and TN at the watershed outlet and improved significantly water quality. As regards the TP concentration, all the 90th percentile of the TP concentration obtained from BMPs were higher than LAWA class II-III and still far from LAWA class II (Figure 4.11c). This reveals that BMP implementation does not have a significant impact on concentration values for TP. However it may help moderate the concentration values in case of CBN implementation.

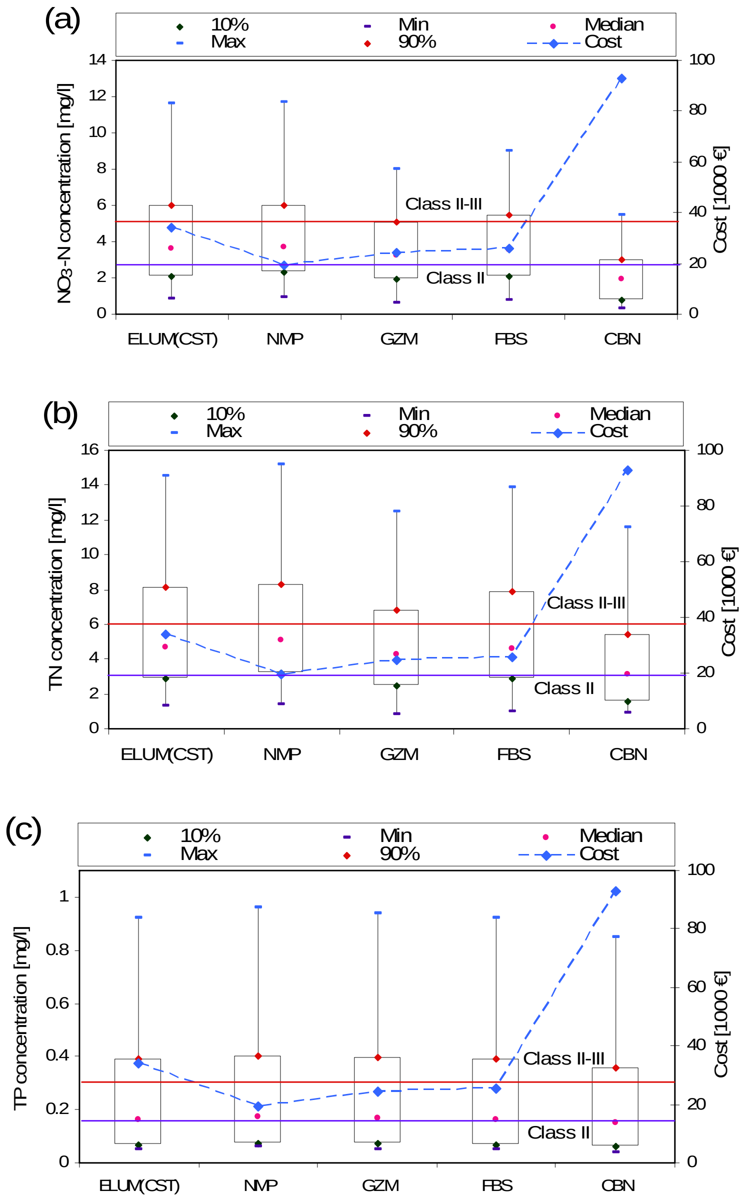


Figure 4.11: Simulated concentration of NO_3-N, TN, and TP (mg l^{-1}) at the Soltfeld gauging station under different BMPs and their positioning within LAWA quality classes. The continued lines indicate the respective water quality classes II ('moderately polluted') and II-III ('significantly polluted'). The broken line represents costs of BMPs

As can be seen from Figure 4.11, GZM scenario provided the highest concentration reduction. The second reduction occurred in case of FBS scenario. However, when considering the annual cost for implementation, FBS scenario is a little more expensive than GZM and NMP scenario. In general, NMP, GZM, and FBS scenario are effective for reducing the nutrient concentration and their implementation costs are not significantly different. Although FBS scenario may represent more expensive BMP to install, it plays an important role in reducing nutrient concentration. The combined scenarios CBN provided the highest concentration reductions in nutrient at the watershed outlet. However, the costs for implementing this scenario are relatively high. This indicates that the more effective the CBN, the more expensive it usually is.

Based on the above analyses of effectiveness and cost of BMP implementation, the results indicated that (a) implementation of conservation tillage is unrealistic for the Kielstau lowland catchment because of high annual cost for implementation and low effectiveness in nitrogen load reduction, (b) implementation of BMPs would not reduce sediment and phosphorus load at the watershed outlet significantly due to low sediment load from the fields to stream networks. However, reduction in sediment load at the watershed outlet of this lowland catchment could be achieved considerably if some other measures are installed in the channel to reduce bank erosion such as lined waterway/stream channel stabilization, grade stabilization structures. These measures have been implemented effectively for reducing sediment load at various region scales (Santhi et al., 2006; Bracmort et al., 2006; Arabi et al., 2007), and (c) implementation of changing crop rotations, nutrient management plan, grazing management practices, and field buffer strips would be effective in reducing nitrogen load at the watershed outlet significantly. It could be inferred that further reduction in fertilizer application rates in NMP, lower livestock grazing density on pastures in GZM, and widing field buffer strips along the edge of main channel in arable and pasture land would lower nitrogen concentration and bring considerable water quality improvement to meet the target value for a "good water quality" of the LAWA classifications at the watershed outlet of the Kiestau catchment.

In general, the SWAT model was successfully simulated the impacts of agricultural management practices on sediment and nutrient load in the watershed with varying soils, land use, and management conditions. The worldwide application of SWAT reveals that it is a multipurpose model that can be used to integrate multiple environmental processes, which support more effective watershed management and the development of better-informed policy decisions (Gassman et al., 2007). Nevertheless, using the SWAT model still has general limitations as follows: (a) Hydrologic Interface: The use of the Natural Resources Conservation Service curve number (CN) method in SWAT has provided a relatively easy way of adapting the model to a wide variety of hydrologic conditions. However, this method has proved controversial due to the empirical nature of the approach, lack of complete historical documentation, and poor results achieved for some conditions (Ponce and Hawkins, 1996; Garen and Moore, 2005). (b) Small Land Covers: Land uses that occupy limited areas such as constructions sites, bare areas, and some row crops may not be simulated in the SWAT model. In fact, some of these small areas may contribute many times more sediment on a per unit area basis than rangeland. (c) Hydrologic Response Unit Characteristics: The incorporation of nonspatial HRUs in the SWAT model has supported adaptation of the model to almost any watershed. However, nonspatial aspect of the HRUs is a key weakness of the model because this approach ignores flow and pollutant routing within a subwatershed and treats the influence of pollutant losses similarly from all landscape positions within a subwatershed. Therefore, potential pollutant impairment between the source area and a stream is also ignored (Gassman et al., 2007). (d) Simulation of BMPs: assessments of targeted filter strip placements within a watershed are limited, due to the lack of HRU spatial definition in SWAT. This also obstructs simulation of riparian buffer zones and other conservation buffers, which again need to be spatially defined at the landscape or HRU level in order to correctly account for upslope pollution source areas and the pollutant mitigation impact of the buffers (Gassman et al., 2007).

The above limitations of the SWAT model partly affect on the prediction of the model in general. In this study, the performance of the SWAT model in simulating phosphorus load is limited. Apart from the reasons such as low surface runoff and low sediment load from the fields, lack of soil phosphorus cycling in SWAT could be attributed to another reason causing underestimation of phosphorus model. This is because SWAT currently assumes that phosphorus is mostly transported in surface runoff and with sediment to the stream (Neistch et al., 2005), while other factors such as leaching of soil phosphorus through the soil profile, lateral, and tile flows have not been taken into account. Due to the underestimation of phosphorus load at the watershed outlet, the simulated results of the phosphorus load from BMP implementation are also uncertainty. In addition, BMPs have been implemented only for a short period of 03 years due to the availability of measured data. Therefore, the transformation of N pools and P pools in the soil may be inadequately happened leading to inaccurate results of prediction. Moreover, SWAT currently assumes that soil carbon contents are static. This approach has not yet reflected reality of carbon cycling processes (Gassman et al., 2007). In the SWAT model, decomposition and mineralization of the fresh nitrogen and phosphorus pool is allowed only in the first soil layer. They are controlled by a decay rate constant which is calculated as a function of the C: N ratio and C: P ratio of the residue, temperature and soil water content (Neitstch et al., 2005). Hence, adequate assumption of soil carbon contents would provide predicted results more exactly. It can be expected that a long-term period of data used for the implementation of BMPs in the model will provide more realistic results in relation to nitrogen and phosphorus load at the watershed outlet.

4.4 Conclusions

The ecohydrological model SWAT was applied to simulate flow, sediment, and nutrient load from different point and diffuse sources in the mesoscale Kielstau catchment – a typical rural lowland area in Northern Germany. The basic input data includes climate, topography, soil, land use, and agricultural data; all of which, as well as consideration of sewage disposals of six wastewater treatment plants, was used to predict the current flow, sediment and nutrient load. The calibration and validation of the SWAT output were implemented by comparing simulated values with corresponding in-stream measurements at the watershed outlet. Simulations of flow, sediment, and nutrient were performed in daily time step. The results of this study showed good agreement between simulated and measured daily discharges with an E_{NS} and R^2 of 0.75 and 0.78 for the calibration period, 0.78 and 0.84 for the validation period. The statistical coefficients of the sediment and nutrient model performance were relatively reasonable ranged from 0.42–0.75 and 0.58–0.83 for E_{NS} and R^2, respectively, during the simulation period. Overall, the SWAT performed satisfactorily in simulating daily flow, sediment, and nutrient load at the Kielstau lowland catchment.

The two approaches to the structural and nonstructural measures including ELUM, GZM, FBS, and NMP were developed to investigate their impacts on the behavior of diffuse source pollutants, with the purpose of delivering water quality improvements in the Kielstau lowland catchment. A number of scenarios were simulated for the same period (2006–2008) as the baseline. The average annual values for flow, sediment, NO_3-N, TN, and TP load were calculated under each scenario and compared with values obtained from the baseline condition. When implementing ELUM, the results indicated that a shift from the current crop rotation (winter wheat-winter wheat-rape) to a new crop rotation (winter wheat-winter rye-winter rye) would be more effective in reducing considerably nitrogen load at the watershed outlet. The model results also showed that the implementation of BMPs would reduce significantly NO_3-N and TN load at the outlet of the Kielstau catchment. However, the impacts of these BMPs implementation were not significant on the reduction in sediment and TP load due to the specific characteristics of lowland area (e.g., flat topography, low surface runoff). The CBN provided the most load reduction in the average annual load for NO_3-N and TN at the outlet of the catchment by 53.9% and 46.7%, respectively. Moreover, the concentrations of NO_3-N and TN were also decreased significantly and reached nearly the LAWA class II (moderate polluted) at the outlet of watershed. These concluded that the water quality in the Kielstau catchment would be improved considerably and would meet nearly the target value for water quality until the year 2015 according to the European Water Framework Directive by implementing simultaneously BMPs.

The trade-off relationship between the effectiveness in nutrient reduction and the corresponding cost of BMP implementation was considered. The study achieved valuable quantitative information on the effectiveness of agricultural BMPs in reducing pollutant load and improving water quality, and the cost associated with these improvements. The results showed that the implementation of the CBN provided the highest load reduction of nutrient and the corresponding costs € 93,000 per year. The trade-off between cost and effectiveness of BMPs will be helpful for policymakers and stakeholders to identify suitable BMPs for improving water quality in the Kielstau catchment.

In this study we developed selection processes of the structural and nonstructural BMPs aiming at finding effective BMPs, which can minimize the highest load of agricultural diffuse source pollution and be the most consistent with farming condition in this area. The costs of BMPs were then estimated based on the local current information. The combination between cost and effectiveness of BMPs serves as a base for assessing the benefits of BMP implemented in the Kielstau lowland catchment and these approaches can be extended to other lowland catchments with similar conditions.

Chapter V Assessing the spatial and temporal variations of water quality in lowland areas, Northern Germany

Q. D. Lam, B. Schmalz, N. Fohrer

Journal of Environmental Management
Submitted 05.04.2011

Abstract

The pollution of rivers and streams with agro-chemical contaminants has become one of the most crucial environmental problems in the world. The assessment of spatial and temporal variations of water quality influenced by point and diffuse source pollution is necessary to manage the environment sustainably in various watershed scales. The overall objectives of this study were to assess the transferability of parameter sets between lowland catchments on different scales using the ecohydrological model SWAT (Soil and Water Assessment Tool) and to evaluate the temporal and spatial patterns of water quality in the whole catchments before and after implementation of best management practices (BMPs).

The study area Kielstau catchment is located in Northern Germany as typical example of lowland –flood plain landscape. Sandy, loamy and peat soils are characteristic for this area. Land use is dominated by arable land and pasture. In this study we examined two catchment areas including Kielstau catchment 50 km² and its subcatchment, namely Moorau, with the area of 7.6 km². The water quality of these catchments is not only influenced by diffuse sources from agricultural areas but also by point sources from municipal wastewater treatment plants (WWTPs). Diffuse entries as well as punctual entries from the WWTPs are considered in the model set-up. For this study, the calibration and validation of the model were carried out in a daily time step for flow and nutrients. The results indicate that parameter set could be transferred in lowland catchments with similar environmental conditions. Shallow groundwater is the major contributor to total nitrate load in the stream accounting for about 93% of the total nitrate load, while only about 7% results from surface runoff and lateral flow. The study also indicates that applying a spatially distributed modeling approach was an appropriate method to generate source maps showing the spatial distribution of TN and TP load from hydrologic response units (HRUs) as well as from subbasins and to identify the crucial pollution areas within a watershed whose management practices can be improved to control more effectively nutrient loading to water bodies.

Keywords: Lowland, best management practice, nutrient, pollution, SWAT

5.1 Introduction

In recent years, many efforts have been made worldwide to mitigate the impairment of point and diffuse source pollution of the aquatic environment. The cause of water quality deterioration is mostly associated with diffuse source pollution due to the intensification of agricultural activities and the development of large urban areas, of which agriculture has been identified as the major contributor of diffuse source pollution of water resources (Humenik et al., 1987; Duda, 1993; Behrendt et al., 1999; Lam et al., 2010). The European Water Framework Directive (EC, 2000) is a relatively new legislation that demands the good status of surface waters and groundwater to be accomplished until 2015. This predetermined status mentions both the qualitative and quantitative issues of the river, streams, and other water bodies. However, characteristics of a lowland river concerning ecology and hydrology depend on not only both the conditions and processes within the river itself but also on the impacts of the floodplain corresponding with the river (Lasserre et al., 1999; Hayashi and Rosenberry, 2002; Vidon and Hill, 2004).

Lowland catchments are ecosystems with low flow velocity, a high groundwater table, and flat topography (Schmalz et al., 2009; Krause et al., 2007; Müller et al., 2004). Hydrological conditions and nutrient dynamics of lowland river systems and the adjacent floodplains are strongly controlled by the interactions between surface water and shallow groundwater (Osman and Bruen, 2002; Sophocleous, 2002; Krause and Bronstert, 2004, 2005; Winter, 1999). In the past centuries different melioration measures such as river regulation, installation of drainage, and pumping stations have been implemented in order to enlarge areas as well as render better cultivation conditions for agriculture. These have led to a change in the water balance, nutrient dynamics, and subsequently floodplain ecology (Sanchez-Perez et al., 2003; Hayashi and Rosenberry, 2002; Bullock and Acreman, 2003; Schmalz et al., 2009). Thereby, the quality of surface water and groundwater of those lowland areas is also affected significantly due to the changes of transformation and transport of nutrient and pollutants (Hill, 1990; Devito and Dillon, 1993; Andersen, 2004; Vidon and Hill, 2004).

For the lowlands of Northern Germany, the intensive changes in agricultural activities and ecological policies within the last years have somewhat influenced the natural water and nutrient balance. Thus, the management of water quality in these lowland areas has become more important. The evaluation of new regulations and degradation of water bodies caused by point and diffuse source pollution requires modeling studies in order to assess the impact of those new policies as well as water pollution on the surrounding environment. Model scenarios can be helpful in finding reasonable measures for assessing environmental ecological status while taking into account relevant factors such as climate, land, and water use (Krysanova et al., 2005; HØjberg et al., 2007). In recent years, commonly used agricultural watershed models comprise DRAINMOD (Skaggs, 1980), HSPF (Johanson et al., 1984), AGNPS (Yoon et al., 1993), MIKE SHE (Xevi et al., 1997), ANSWERS2000 (Bouraoui and Dillaha, 1996), SWAT (Arnold et al., 1998, Arnold & Fohrer, 2005), SWIM (Krysanova et al., 1998). The SWAT model has been widely used all around the world to predict flow, sediment and nutrient load from watersheds of various sizes (Fohrer et al., 1999; Saleh et al., 2000; Tripathi et al., 2004; Gassman et al., 2006; Lenhart et al., 2003; Schmalz et al., 2008). In addition, many studies have used SWAT to evaluate the impact of BMPs on hydrologic processes and water quality at different watershed scales (Fohrer et al., 2002; Tripathi et al., 2005; Arabi et al., 2007; Behera and Panda, 2006; Bracmort et al., 2006; Rode et al., 2008; Lam et al., 2011). Gassman et al. (2007) have indicated that a key strength of SWAT is a flexible framework allowing the simulation of a wide variety of structural and nonstructural BMPs such as conservation tillage, cover crops, nutrient management, buffer strips, flood prevention structures, grass water way, and parallel terraces. Although management scenarios of watershed are routinely simulated with SWAT, variations in spatial patterns of implementation are less frequently done, especially for lowland areas where the transport of pollutants and nutrients is strongly influenced by special features such as flat topography, shallow groundwater. The spatially distributed modeling is needed to evaluate the crucial areas within the watershed whose management practices can be improved aiming at controlling better nutrient load to water bodies (Haverkamp et al., 2005).

The objectives of this study were to: (1) evaluate the performance capabilities of the SWAT to simulate flow and nutrient load in complex mesoscale lowland catchments and assess the transferability of parameter sets between lowland catchments of different scale; (2) evaluate the transportation of nutrient in different pathways from the fields to the streams of lowland catchments; and (3) assess the temporal and spatial variations of nutrient loads in the whole catchment before and after implementation of BMPs and to identify crucial subbasins which provide significant nutrient loads compared to other subbasins within the watershed. Then, appropriate BMPs are further proposed to improve water quality in those crucial subbasins as well as the whole catchment.

5.2 Materials and methods

5.2.1 Study area

The study area Kielstau catchment is located in Northern Germany as part of a lowland area in Schleswig-Holstein. The highest and lowest elevation of the Kielstau topography is 79.9 and 27.3 m, respectively (Figure 5.1). The area of the Kielstau catchment is about 50 km². The river Kielstau has a total length of 17 km and flows through Lake Winderatt towards the gauge Soltfeld, located at the outlet of Kielstau watershed. The six WWTPs built within the Kielstau watershed are Husby, Hürup Nord, Hürup Weseby, Hürup Süd, Ausacker, and Freienwill (Figure 5.1). Hürup Nord, Hürup Weseby, and Hürup Süd are located along the longitudinal Hennebach tributary (461, 447, and 240 population equivalents). Ausacker and Freienwill are located on the river Kielstau (1880 and 350 population equivalents). In addition, various small tributaries and water from drainage pipes and ditches flow into the river Kielstau. The drainage fraction of agricultural area in the Kielstau catchment is estimated at 38% (Fohrer et al., 2007).

Moorau catchment is a subcatchment within the Kielstau catchment (Figure 5.1). The total drainage area of the Moorau catchment is about 7.6 km². The distance between the Kielstau outlet and the Moorau outlet is about 12 km. Moorau is one of the two important tributaries of the river Kielstau from the north. There is a WWTP, namely Husby, situated at the beginning of the

Moorau tributary with 3000 population equivalents. All the water including chemicals from WWTP discharges into the stream Moorau then flows to the river Kielstau.

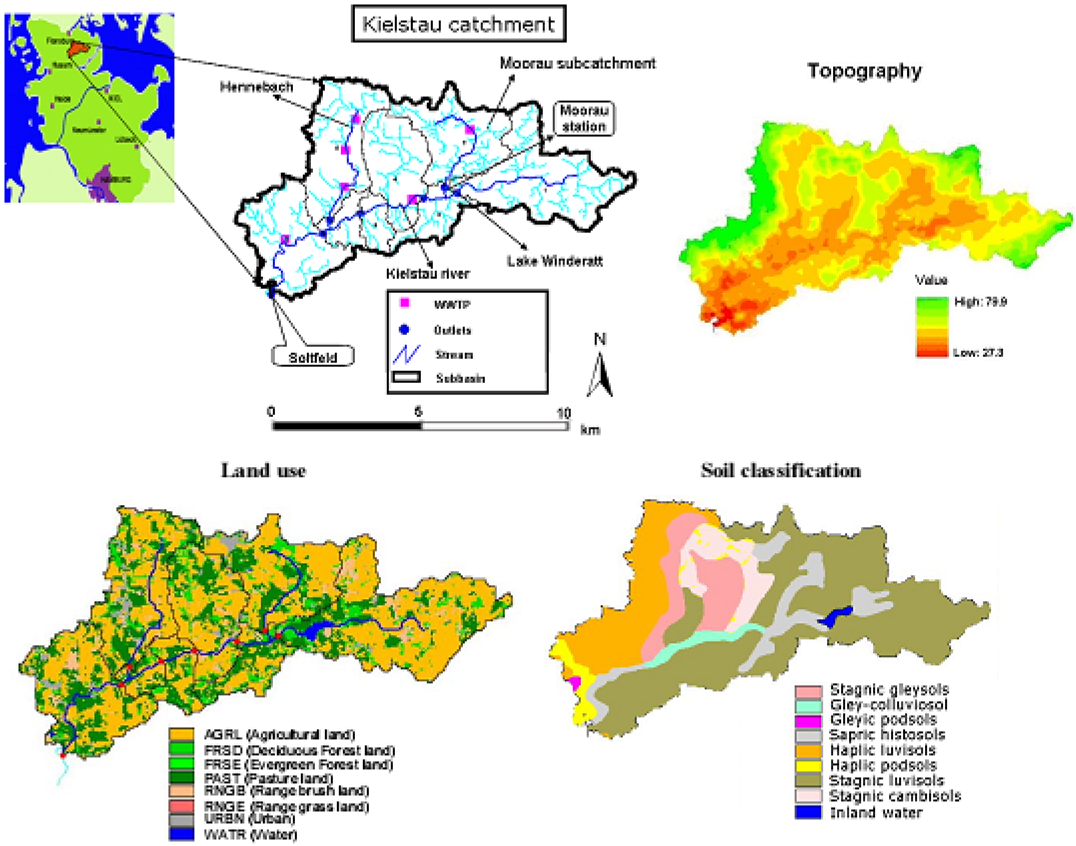


Figure 5.1: Location of the Kielstau catchment, topography (LVermA, 1995), land use (DLR, 1995), and soil maps (BGR, 1999) in Schleswig-Holstein, Northern Germany

Land use is dominated by arable land and pasture in both the Kielstau and the Moorau catchment. The dominant soils of the Kielstau catchment are Stagnic Luvisols and Haplic Luvisols, while Stagnic Luvisols and Gleyic Antrosols are dominant soils of the Moorau catchment. Land use and soil maps used in this study can be seen in Figure 5.1. The mean annual temperature is 8.2 ^{0}C (station Flensburg, 1961-1990, DWD, 2009a). The mean annual precipitation is about 841 mm/a (station Satrup, 1961-1990, DWD, 2009b)

In the Kielstau catchment, diffuse source pollution of nutrients results mainly from various farms, which apply fertilizers or animal husbandry in the vicinity of the river as well as from urban areas. The combinations of these diffuse sources and point sources influences instream water quality considerably (Schmalz et al., 2007).

5.2.2 Monitoring of the watershed

The Soltfeld gauging measurement station has been installed at the outlet of the Kielstau catchment (Figure 5.1). The hourly discharge data used for this study (1993-2008) were measured from this station by Staatliches Umweltamt Schleswig (2009). At the Moorau station, the water level data (October 2007 – June2009) were measured by the Department of Hydrology and Water Resources Management - Institute for the Conservation of Natural Resources at Kiel University (CAU Kiel). Calculation of the Moorau discharge at the Moorau station was based on the rating curve between water level and discharge which was obtained by the relation between stream velocity and area cross section at the Moorau outlet (Tavares, 2006; Beyersdorf, 2008). The average daily discharge is given in Table 5.1.

The collection and analysis of daily water samples took place during the period from May 2006 to December 2008 for the Kielstau catchment and from November 2007 to March 2009 for the Moorau catchment by the Department of Hydrology and Water Resources Management - Institute for the Conservation of Natural Resources at Kiel University (CAU Kiel, 2009). At the Soltfeld gauging and Moorau station, water samples were collected, filtered, and frozen for further laboratory analysis. Nutrient concentrations in the water samples were quantified by photometry and ion chromatography in the laboratory. The average concentrations of nutrient for both the Kielstau and Moorau catchment are shown in Table 5.1.

Table 5.1 Mean measured water discharge (Staatliches Umweltamt Schleswig, 2009; CAU Kiel) and nutrient concentrations (CAU Kiel) at the Soltfeld gauging and Moorau station

Watershed	Variable	Unit	Period	Value	LAWA (1998), class II
Kielstau	Average discharge	m^3/s	January 1993 to December 2008	0.43	-
	Average NO_3-N	mg/l	May 2006 to December 2008	4.48	2.5
	Average NH4-N	mg/l	May 2006 to December 2008	0.15	0.3
	Average TN	mg/l	May 2006 to December 2008	5.81	3.0
	Average PO4-P	mg/l	May 2006 to December 2008	0.18	0.1
	Average TP	mg/l	May 2006 to December 2008	0.23	0.15
Moorau	Average discharge	m^3/s	July 2007 to June 2009	0.09	-
	Average NO_3-N	mg/l	November 2007 to March 2009	7.39	2.5
	Average NH4-N	mg/l	November 2007 to March 2009	0.85	0.3
	Average PO4-P	mg/l	November 2007 to March 2009	0.16	0.1

In general, the measured average concentration values of all parameters (Table 5.1) exceed the allowable limit of class II of the LAWA standard (LAWA, 1998) whose quality class II (moderately polluted) represents the target value for water quality until the year 2015 according to the European Water Framework Directive (EC, 2000). In other words, nutrient concentration needs to be lower or equal that value to satisfy with LAWA quality standards. Looking in detail at the water quality and the discharge at the catchment's outlet in Table 5.1, the average nitrate and ammonium concentrations at the outlet of the Moorau catchment are fairly high compared to the

Kielstau catchment. In addition, the values of nitrate concentrations far exceed the target value for a “good water quality” (Class II, NO_3-N ≤ 2.5 mg/l), especially for nitrate concentration at the Moorau catchment. Regarding the discharge, the average discharge at the outlet of the Moorau catchment is relatively low, accounting for 0.09 m³/s, while the average discharge at the Kielstau outlet is with 0.43 m³/s nearly five times higher than that of the Moorau catchment (Table 5.1). Therefore the high nitrogen concentrations of the Moorau catchment resulted in lower loads, but are an essential contribution to the river load of the Kielstau catchment. The values in Table 1 also show that parameters in relation to NO_3-N, NH4-N, and TN far exceed the allowable limit of LAWA class II, while the difference between the measured average values of PO4-P and TP and their corresponding values of LAWA class II is not so considerable. This infers that nitrogen parameters need to be improved by BMP implementation within the catchments. These initial assessments of nutrient concentrations can be helpful in gaining a preliminary understanding of the nutrient behavior at different times in particular and partly illustrated the status of water quality in the Kielstau and the Moorau catchment.

5.2.3 The SWAT model

The ecohydrological model SWAT (Soil and Water Assessment Tool, Arnold et al., 1998, version 2005) has been widely used for watershed scale studies dealing with water quantity and quality. SWAT is a semi-distributed, process-oriented hydrological model. It is a continuous time model, which simulates water and nutrient cycles in a daily time step. The SWAT model represents the large-scale spatial heterogeneity of the study area by dividing the watershed into subbasins. The subbasins are then further subdivided into hydrologic response units (HRUs) that are assumed to consist of homogeneous land use and soils. The climatic variables required by SWAT include daily precipitation, maximum/minimum air temperature, solar radiation, wind speed and relative humidity. Major components of the model include hydrology, weather, and agricultural management. The details of all components can be found in Arnold et al. (1998) and Neitsch et al. (2002).

In the SWAT model, soil water content, surface runoff, nutrient cycles, crop growth and management practices are simulated for each HRU and then aggregated for the subbasin by a weighted average. The model’s hydrological components are comprised of surface runoff, percolation, lateral flow, ground water, and evapotranspiration and channel transmission loss. Simulation of the hydrology of a watershed is split into two major divisions. The first division is the land phase of the hydrologic cycle, which controls the amount of water, sediment, nutrient and pesticide loading into the main channel in each subbasin. The second division is the water or routing phase of the hydrologic cycle that can be defined as the movement of water, sediment, nutrient, etc. through the channel network of the watershed to the outlet (Neitsch et al., 2005). The SWAT model simulates surface runoff volumes and peak runoff rate for each HRU using daily rainfall or subdaily rainfall amounts. Surface runoff is calculated using a modification of the Soil

Conservation Service (SCS, 1972) curve number method, which is a function of the soil's permeability, land use and antecedent soil water conditions.

SWAT simulates the nitrogen cycles in the soil profile and in the shallow aquifer (Neitsch et al., 2005). In soil and water, nitrogen is extremely reactive and exists in a number of dynamic forms. It may be added to the soil in the form of fertilizer, manure or residue application, bacteriological fixation, and rain. It can be removed from the soil through plant uptake, soil erosion, leaching, volatilization and denitrification. Plant use of nitrogen is estimated using the supplying and demand approach (Williams et al., 1984). In the SWAT model, there are five different pools of nitrogen in the soil. Two of the pools are inorganic forms of nitrogen, while the other three pools are organic forms of nitrogen.

Nitrate may be transported with surface runoff, lateral flow or percolation. Nitrate entering the shallow aquifer in recharge from the soil profile through the percolation may be remained in the aquifer, moved with groundwater flow into the main channel, or be transported out of the shallow aquifer with water moving into the soil zone in response to water deficiencies, and moved with recharge to the deep aquifer. The amount of nitrate moved with the water is calculated by multiplying the nitrate concentration in the mobile water by the volume of water moving in each pathway.

The different phosphorus processes modeled by SWAT in the HRUs and the various pools of phosphorus in the soil are described in Neitsch et al. (2005). Plant use of phosphorus is estimated using the supply and demand approach similar to nitrogen. Three major forms of phosphorus in mineral soils are organic P associated with humus, insoluble forms of mineral P, and plant-available P in soil solution. Due to the low mobility of phosphorus, surface runoff will only partially interact with the solution P stored in the top 10 mm of soil.

The SWAT in-stream water quality algorithms incorporate constituent interactions and relationships used in the QUAL2E model (Brown and Barnwell, 1987), which contains the major interactive factors such as the nutrient cycles, algae production, and benthic oxygen demand.

5.2.4 Model inputs

The ArcSWAT interface for SWAT version 2005 (Winchell et al., 2007) was used to compile the SWAT input files. The basic data sets required to set up the model inputs are: topography, soil, land use and climatic data. These data are described in Table 5.2. The topographic information was used for automatic delineation of the watershed. Land use and soil maps were superimposed on the catchment's subbasins. In this study the SWAT model was conducted by dividing the catchment of Kielstau and Moorau into 8 and 3 subbasins, respectively (Figure 5.1).

The current management practices in this catchment involve a three-year crop rotation (winter wheat – winter wheat – rape) and monocultural maize, which were simulated in the model. In our simulation, winter wheat is planted from mid-September after a tillage operation and is harvested in the beginning of August in the subsequent year whereupon soil is tilled again. Rape is planted from the end of August and harvested in the beginning of July in the subsequent year. The amount and date of fertilizer application are in conformance with the conventional cultivation (Lksh, 2006).

Table 5.2 Input data sources used for model set up

Data type	Source	Data description/properties
Topography	LVermA (1995)	Digital elevation model, a grid size of 25 m×25 m
Soil map	BGR (1999)	Soil physical properties such as texture, saturated conductivity, etc. Scale of soil map (1: 200, 000)
Land use map	DLR (1995)	Land use classifications, 25 m × 25 m resolution
Climate data	DWD (2009 a, b)	Temperature, precipitation, wind speed, humidity (Meierwik station, 1993–2009)

The nitrogen deposition was estimated to be 8.4 kg N/ha/year (Lam et al., 2010). Rates of N-fertilizer including nitrogen deposition and P-fertilizer application (in kg N/ha/year) were set, respectively, to 210 and 50 for winter wheat, 190 and 60 for rape, 179 and 72 for maize, and 160 and 30 for pasture. The manure application for arable land ranged from 80 kg/ ha to 240 kg/ ha. In the pasture lands, cows are the dominant animals that are grazing mainly in summer season. Average livestock (LU) density obtained from local people consultation and literature is estimated to be 2 LU ha^{-1} (Kreins et al., 2003; Gömann et al., 2005). Data of average monthly nutrients taken from the six WWTPs from 2002 to 2009 (Kreis Schleswig-Flensburg, 2009) were implemented as point sources in the model. The default values mentioned above were used for baseline simulation.

5.2.5 Model calibration

Differences of the Kielstau and the Moorau catchments concerning topography and land use as well as soil physical properties are generally negligible. Therefore, these lowland catchments can be considered to be similar environmental conditions. The calibration and validation strategies were performed by the previous studies for the Kielstau catchment (Lam et al., 2010; Lam et al., 2011). In this study we test the capabilities of the SWAT model by transferring the implemented strategies as well as sensitive parameters from the Kielstau model to the Moorau model.

With respect to the Kielstau catchment, the auto-calibration was carried out using flow data from the hydrological years 1998-2004. The validation was done for the continuous time 2004-2008. The simulation of nutrients was performed for the period from May 2006 to December 2008. For the Moorau catchment, the auto- calibration was conducted using flow data collected from July 2007 to June 2008 and then the subsequent period from July 2008 until June 2009 was used for the validation of the model. Nutrient simulation of the model was carried out for the period from November 2007 until March 2009. In the SWAT model, simulation of flow and nutrient load was performed for daily time step using measured data from the Soltfeld gauging and Moorau station at the catchment's outlets (Figure 5.1). The auto-calibration was performed by changing each parameter ten times within the allowable range of values for the specific parameter. Detailed calibration procedures for the SWAT model and the definitions of various calibration parameters are depicted by Neitsch et al. (2002).

5.2.5. 1 Flow

Surface runoff and base flow were calibrated simultaneously. Parameters adjusted for surface runoff were curve number (CN2) and available water capacity (SOL_AWC). The main parameters adjusted for base flow were soil evaporation compensation factor (ESCO), ground water revap coefficient (GW_REVAP), and threshold depth of water in shallow aquifer (GWQMN). Regarding the calibration strategies of the SWAT model for the lowland Kielstau and Moorau catchments, the parameters related to groundwater have been adjusted first. This is because groundwater parameters were found to be most sensitive in the lowland catchment and the most sensitive parameter was the threshold depth of water in shallow aquifer (GWQMN) (Lam et al., 2009, Schmalz et al., 2008). Moreover, the groundwater and upwelling of groundwater in lowlands are the dominant components and contribute an important role for pollutant transportation (Lam et al., 2010, Schmalz et al., 2009). A detailed description of these parameters is presented in Table 5.3.

5.2.5. 2 Mineral nutrients

In this study, the SWAT simulation output was calibrated against observed daily measured NO_3-N, NH_4-N, and PO_4-P load at the catchment's outlet. Nutrient calibration was simulated into two steps, calibration of nutrient load and calibration of in-stream water quality processes. Before implementing simulation of nutrient, initial concentration of mineral nitrogen and phosphorus in the soil upper layer (SOL_NO_3 and SOL_SOL P) for different land use were set at appropriate levels. Main parameters adjusted for nutrient calibration were nitrogen and phosphorus percolation (NPERCO, PPERCO), phosphorus soil partitioning coefficient (PHOSKD), residue mineralization (RSDCO), and biological mixing efficiency (BIOMIX). These parameters were adjusted from initial values of SWAT within their allowable ranges to match the simulated and measured nutrient load. In addition, some parameters affecting in-stream water quality processes such as fraction of algae biomass that is nitrogen and phosphorus (AL1, AL2), rate constant for biological oxidation of NO_2 to NO_3 (BC2), and rate constant for mineralization of organic phosphorus to dissolved phosphorus (BC4) were also considered.

Statistics of the simulated daily flow, NO_3-N, NH_4-N, and PO_4-P load were compared with corresponding measured discharge and load at the catchment's outlets. The coefficient of determination (R^2) and Nash-Sutcliffe efficiency (E_{NS}) (Nash and Sutcliffe, 1970) have been used to evaluate the accuracy of model predictions. R^2 is the square of the Pearson's product-moment correlation coefficient, which is an indicator of strength of relationship between measured and simulated values. E_{NS} has been widely used to evaluate the performance of hydrologic models. The values of E_{NS} and R^2 range from 0 to 1. If the E_{NS} and R^2 values are less than or close to 0, the model simulation is taken as an indication of poor or unacceptable performance. The closer the values get to 1, the more perfect the model simulation. Santhi et al. (2001) found that a value greater than 0.5 for these variables constitutes an acceptable simulation model.

5.3 Results and discussion

5.3.1 Simulation of flow

The results of the model for the daily flow at the Moorau station are shown in Figure 5.2. Overall, the temporal variations in the flow over the period of the calibration and validation are reproduced very well. The gaps in the timeline (29 Mar. 2008 - 27 Apr. 2008) and (23 Sept. 2008 - 5 Oct. 2008) are caused by malfunction and dropouts of the automatic sampler. Thus, there is no comparison between simulated values and the corresponding measured values in these periods. The model underpredicted the flow in the winter period of 2008 (Dec.2007-Mar.2008). This is presumably due to the model structure of SWAT. Since it is a continuous time model with a daily time step, subscale processes such as single-event flood routing cannot be efficiently and elaborately predicted. However, the statistical values of correlation coefficient and model's efficiency were high and the difference between mean values of measured and simulated flow is not significant. This can be confirmed by the mean measured and simulated values of 0.12 m^3/s and 0.14 m^3/s for the calibration period and 0.07 m^3/s and 0.08 m^3/s for the validation period, respectively (Table 5.4). The respective values of E_{NS} and R^2 are 0.78 and 0.79 for the calibration period and 0.75 and 0.71 for the validation period at the outlet of the Moorau catchment. Similar results are found during the calibration and validation period in the Kielstau catchment. The model performance for flow in this catchment was considered good both E_{NS} and R^2 being $\geq$ 0.75 for all simulation periods (Table 5.4).

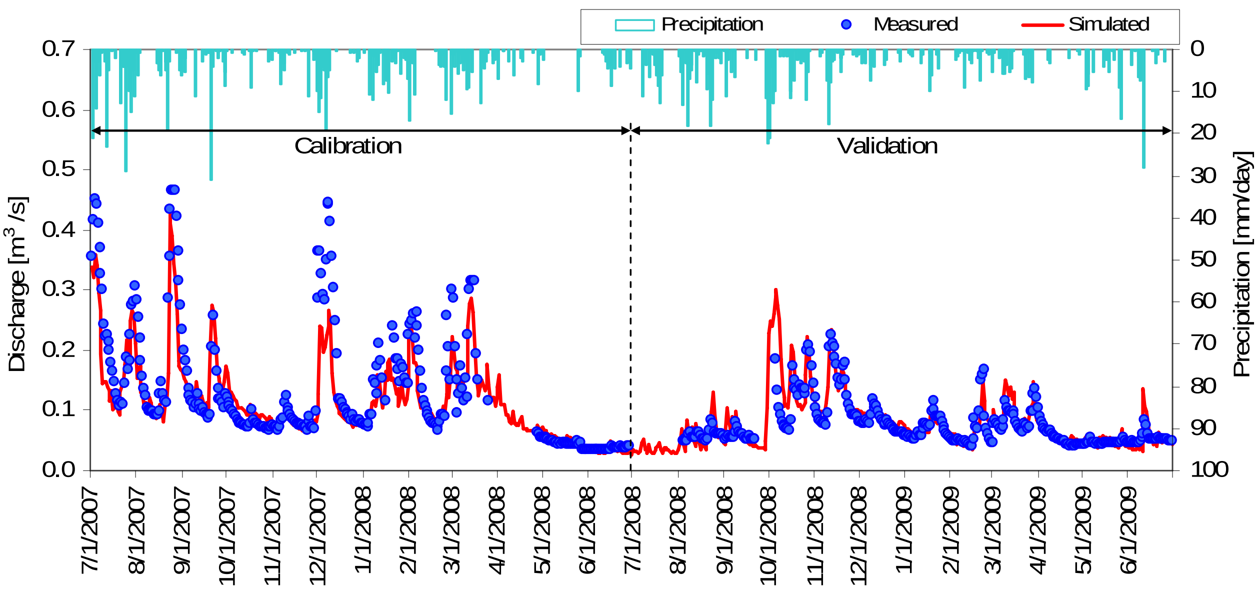


Figure 5.2: Simulated and measured daily discharge at the Moorau station as well as precipitation

(Meierwik, DWD, 2009b)

The temporal variations in monthly flow at the outlets of the Kielstau and Moorau catchment were further compared by regression analysis as shown in Figure 5.3. The comparison between monthly measured and simulated flow was performed over a 11-year period, from 1998 to 2008 for the Kielstau catchment (Figure 5.3a) and a two-year period, from 2007 to 2009 was carried out for the Moorau catchment (Figure 5.3b). The simulation periods of these catchments were chosen according to the availability of measured data. The distributions of monthly measured and simulated flow were plotted graphically with respect to the 1:1 line for the simulation periods. Figure 5.3 shows that the distribution of monthly measured and simulated flow were noted to be close to the 1:1 line for both higher and lower values. Regression analysis between the measured and simulated flow gave high values of R^2 of 0.85 and 0.83 for both the simulation of the Kielstau and Moorau catchment, respectively. These indicate that the model was also predicting monthly flow well for the lowland Kielstau and Moorau catchments during the whole time period.

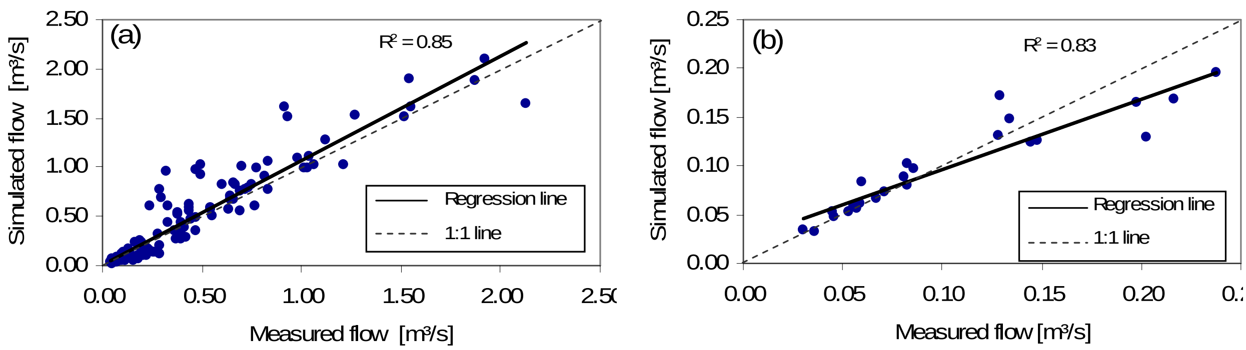


Figure 5.3: Regression analysis of the monthly flow at (a) Kielstau outlet (1998-2008) and (b) Moorau outlet (2007-2009)

Table 5.3 Main controlling parameters of the SWAT model and their optimal values for the main stations of the Kielstau and the Moorau catchment.

Parameter	Definition	Allowable range	Optimal value	
			Kielstau*	Moorau
GW_REVAP	Ground water revap coefficient	0.02-0.2	0.2	0.2
GWQMN	Threshold depth of water in shallow aquifer	0-5000	50	50
GW_DELAY	Delay time for aquifer recharge	0-500	14	16
ALPHA_BF	Base flow recession constant	0-1	1	0.97
ESCO	Soil evaporation compensation factor	0.01-1	0.95	0.98
CN_2 (arable land, pasture, range brush, and forest)	Curve number	35-98	64, 46, 46, 35	65, 44.8, 46, 39
SOL_AWC	Available water capacity	0-1	0.12-0.54	0.23-0.58
DDRAIN	Depth to subsurface drain	0-2000	800	800

TDRAIN	Time to drain soil to field capacity	0-72	24	24
GDRAIN	Drain tile lag time	0-100	8	8

* The values were derived from Lam et al. (2011)

Main parameters controlling flow processes in the SWAT model for the Kielstau and the Moorau catchment are illustrated in Table 5.3. In the calibration model, there are five main parameters affecting base flow generation (GW_REVAP, GWQMN, GW_DELAY, ALPHA_BF, and ESCO), while two main parameters primarily affecting surface runoff formation (SOL_AWC, CN2). In addition, parameters that pertain to tile drainage are also taken into account. The groundwater revap coefficient (GW_REVAP) and the threshold depths for base flow (GWQMN) affect the amount of groundwater flow. GW_REVAP controls water movement between soil profile and shallow aquifer. This process is considerable in watersheds where the saturated zone is not very far below the surface or where deep-rooted plants are growing. Because the type of plant cover will affect the importance of revap in the water balance, the parameters governing revap can be varied by land use. For the Kielstau and the Moorau catchment, these parameters were set to the same values. A reason for that might be due to similarities in land use between the two catchments (e.g. 55.82% and 51.24% of agriculture area; 8.62 % and 10.67 % of forest area for the Kielstau and the Moorau catchment, respectively). This does not much differ in water from shallow aquifer, which is lost by evapotranspiration. The parameters (GW_DELAY and ALPHA_BF) controlling base flow were also found to be sensitive for the model. The groundwater delay time parameter (GW_DELAY) depicts the delay time for the water moving past the lowest depth of the soil profile by percolation or by pass flow before becoming shallow aquifer recharge. It will depend on the depth of the water table and the hydraulic properties of the geologic formations in the vadose and groundwater zones. The allowable value for this parameter varies from 0 to 500 d in the model. A reduction of the parameter value to 14 d and 16 d allowed a better fit between the modeled results and measured data for the Kielstau and the Moorau catchment, respectively. Besides, the parameter ALPHA_BF is sensitive to discharge. It describes the response of base flow to changes in recharge of the shallow aquifer. This coefficient was set to 1 and 0.97 for the two catchments (Table 5.3). For the correction factors of the surface flow related to parameter SOL_AWC and CN2, the optimal values of these parameters obtained during calibration showing insignificant difference in magnitude between the two catchments. This could be explained by negligible differences in land use and soils between the two catchments. The parameters in relation to tile drainage play important roles in water transport. Many lowland areas were drained in order to speed up the transport of water from precipitation to the reach, and this should be interpreted in the model. Due to the similar characteristics between the two catchments, these parameters were set similarly in the models. In general, the calibrated parameters obtained from these two models show that the difference in magnitude of those parameters is negligible. Thus we can conclude that a calibrated parameter set used for the Kielstau catchment could be transferred to the Moorau catchment. A similar result studied by Heuvelmans et al. (2004) also indicated that there is no significant change in model performance when parameters are transferred within the catchment and to a neighboring catchment with similar environmental conditions in the Schleldt river basin.

5.3.2 Simulation of mineral nitrogen load

The comparison between simulated and measured results of daily nitrogen load at the Moorau outlet is shown in Figure 5.4. As it can be seen in Figure 5.4a, the temporal variations are generally reproduced well for both the range and the dynamic of the NO_3-N load. However, a strong tendency to underestimate the measured values can be recognized during the winter period of 2008 (Dec. 2007-Mar. 2008) and 2009 (Jan. 2009 – Mar. 2009) (Figure 5.4a). This can be presumed to the underestimation of some peak flows (Figure 5.2), which lead to the under prediction of the corresponding nitrogen peaks. From the results in Table 5.4 it can be seen that, the absolute percent difference between measured and simulated mean values of nitrate for both the calibration and validation period at the Moorau outlet were estimated at 3% and 9%, respectively. The R^2 and E_{NS} values for daily nitrate load at the outlet of the Moorau catchment as well as the Kielstau catchment were good for both the calibration and validation periods, ranging from 0.7-0.81 and 0.68-0.79, respectively. This confirmed that the SWAT model performed well NO_3-N load in daily time steps for lowland catchments.

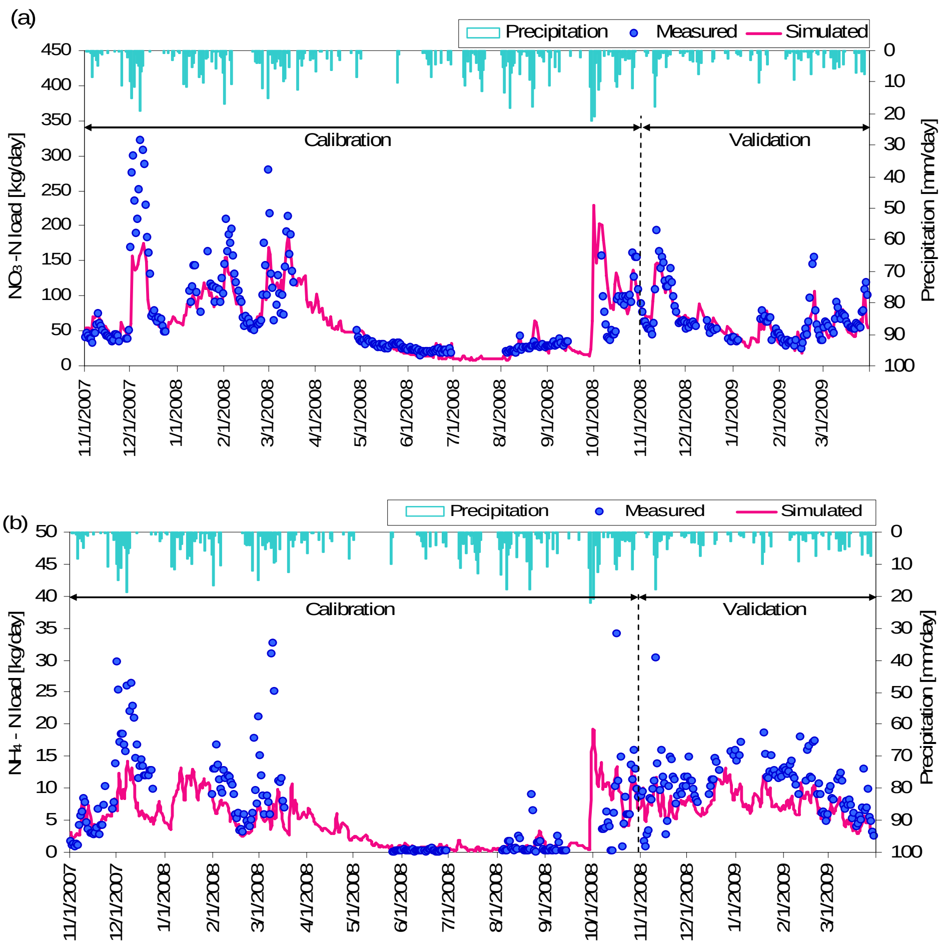


Figure 5.4: Simulated and measured daily nitrogen load a) NO_3-N, b) NH_4-N at the Moorau station as well as precipitation (Meierwik, DWD, 2009b)

Regarding NH_4-N load at the Moorau outlet, the model results were better predicted than in the Kielstau catchment. This can be given by the respective values of R^2 and E_{NS} ranging from 0.55-0.61 and 0.53-0.54 for both the calibration and validation period at the Moorau outlet, while the values of E_{NS} for the ammonium model of the Kielstau catchment ranged only from 0.44-0.46 (Table 5.4). A main reason for that is probably the direct contribution of NH_4-N concentration from the WWTP located at the beginning of the Moorau stream. The simulated daily NH_4-N loads were compared with the corresponding measured values for both the calibration and validation period (Figure 5.4b). The model underpredicted NH_4-N load for the whole simulation periods in general. This can be explained with underestimation of some flow peaks in the winter seasons. In addition, using the input data of point source as monthly input is also another reason for considerable underestimation of the ammonium peaks (Lam et al., 2011).

In Table 5.4, the mean measured and simulated values of NH_4-N load at the Moorau outlet for the calibration and validation period are higher than those of the Kielstau outlet because of the different simulation time. Considering the same simulation period, from 1st November 2007 to 31st October 2008, the mean measured and simulated values of NH_4-N load at the Kielstau outlet are 8.45 kg and 4.87 kg, while those values at the Moorau outlet are 5.91 kg and 4.03 kg, respectively. These comparative values also revealed that the Moorau catchment contributes relative significant NH_4-N load to the river Kielstau. This is also consistent with the previous study results by Schmalz et al. (2007) when they found that the main source of ammonium is coming from WWTP located in the Moorau stream.

Table 5.4 Measured and simulated daily flow, sediment, and nutrient load at the Kielstau and Moorau outlets

Variable	Calibration				Validation			
	Mean		R^2	E_{NS}	Mean		R^2	E_{NS}
	Measured	*Simulated*			*Measured*	*Simulated*		
Kielstau*	*(Flow: Nov. 1998-Oct. 2004; Nutrients: May 2006-Oct. 2007)*				*(Flow: Nov. 2004-Dec. 2008; Nutrients: Nov. 2007-Dec. 2008)*			
Flow (m^3/s)	0.44	0.47	0.78	0.75	0.46	0.46	0.84	0.78
NO_3-N (kg)	195.75	194	0.70	0.68	218.33	217.76	0.81	0.75
NH_4-N (kg)	5.75	3.49	0.61	0.44	9.01	4.33	0.69	0.46
TN (kg)	257.73	251.83	0.81	0.71	280.83	276.61	0.83	0.75
Mineral P (kg)	5.38	3.37	0.58	0.42	3.07	2.18	0.58	0.45
TP (kg)	10.37	5.52	0.68	0.47	7.99	4.10	0.69	0.48
Moorau	*(Flow: Jul. 2007-Jun. 2008; Nutrients: Nov. 2007-Oct. 2008)*				*(Flow: Jul. 2008-Jun. 2009; Nutrients: Nov. 2008-March 2009)*			
Flow (m^3/s)	0.14	0.12	0.79	0.78	0.08	0.07	0.71	0.75

NO_3-N (kg)	71.97	69.71	0.78	0.78	66.67	60.87	0.71	0.79
NH_4-N (kg)	5.91	4.03	0.55	0.54	10.41	7.49	0.61	0.53
Mineral P (kg)	1.67	0.99	0.60	0.49	0.81	0.48	0.63	0.45

** Statistical values were extracted from the previous study (Lam et al., 2011)*

In addition to the watershed assessment, spatial configurations of nitrate loading in various subbasins were evaluated to improve our understanding of nitrate dynamics in lowland areas where the water quality is being influenced by point and diffuse source pollution within the watershed. Figure 5.5 illustrates the average yearly NO_3-N load distribution in the Kielstau watershed for the period from 2006 to 2008. In general, the main nitrate pollution comes from agricultural activities (Lam et al., 2010). Furthermore, some WWTPs and/or poultry farms are located in areas of the northern watershed. As a result, the nitrate load occurred in regions throughout the entire watershed. The darker the color, the higher the NO_3-N load rate per hectare, suggesting elevated nitrogen in surface water. The subbasins located downstream of the Kielstau River were likely to be affected by larger nitrogen concentrations in the water bodies. In addition, the loading of NO_3-N from the subbasin 2 (Moorau) in northern watershed was found to be noticeable. This is presumed to the considerable contribution of nitrate from WWTP increasing the concentration of nitrate at the Moorau outlet. Nitrate loads from subbasins ranged from 4.58 kg/ha to 35.92 kg/ha. A total of 5 of the 8 subbasins (62.5 %) including No. 1, 3, 4, 5, and 6 subbasins have respective average NO_3-N load smaller than 16.58 kg/ha. The subbasin 1 provides nitrate load at the lowest level among the subbasins within the Kielstau watershed, resulting in 4.58 kg/ha. The reason for that is due to nature protection areas around lake Winderratt with no fertilizing around these areas. Moreover, the percentage of agriculture in this subbasin is lower than in other subbasins, while the percentage of forest land is fairly high, occupying 10.9 percent of total area of this subbasin (Lam et al., 2010). Overall, the high loading of nitrate concentrates on the subbasin 2 and 8 where dominant agricultural land and/or WWTP are major cause for increased nitrogen loads.

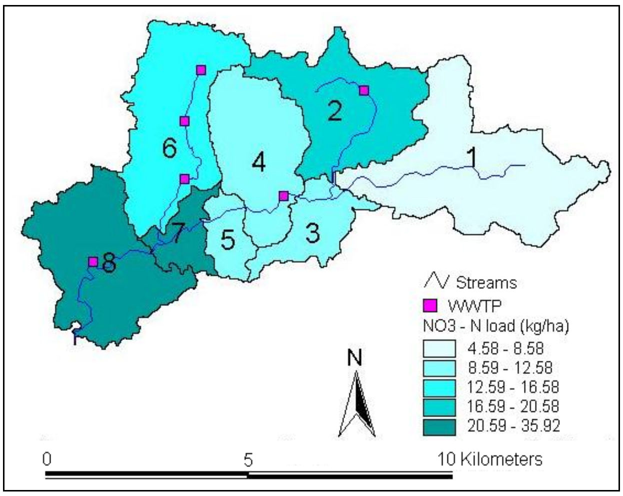


Figure 5.5: Spatial distribution of simulated nitrate load in the Kielstau watershed (2006-2008)

5.3.3 Simulation of mineral phosphorus load

Simulated and measured results for mineral P at the Moorau station are given in Figure 5.6. Overall the model underpredicted mineral P load in both the calibration and validation period. This can be attributed due to the following two reasons: (i) the underestimation of some peak flows in winter seasons result in the underestimation of the corresponding phosphorus load peaks and (ii) the low sediment load from the subbasins which is considered to be a main reason leading to underestimation of phosphorus load (Kiesel et al., 2009; Lam et al., 2011). It can be seen from Fig. 6 that the agreement between simulated and measured mineral P load was not so good for both the calibration and validation period. This is given by the values of E_{NS} and R^2 ranging from 0.49 to 0.45 and from 0.6 to 0.69 for the calibration and validation period, respectively. Similar results were also found in the case of the phosphorus model for the Kielstau catchment, where the values of E_{NS} are also smaller than 0.5 for both the calibration and validation period (Table 5.4). These statistical values obtained from mineral P simulations revealed that the temporal variations are predicted not so well for the dynamic of the mineral P load at the outlet of lowland catchments in general.

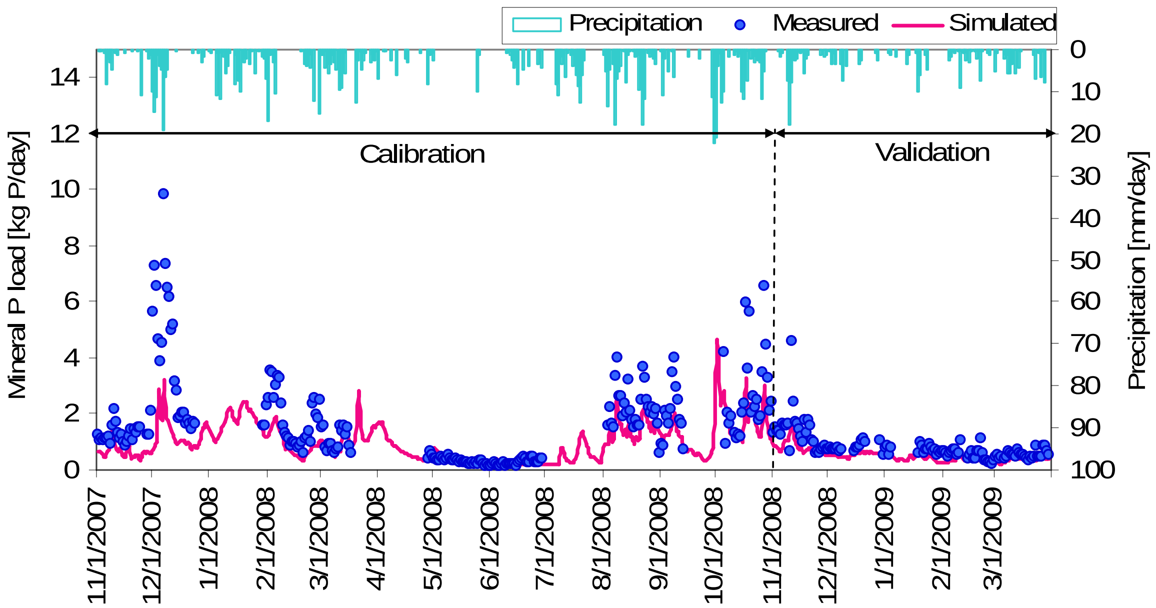


Figure 5.6: Simulated and measured daily phosphorus load at the Moorau station as well as precipitation (Meierwik, DWD, 2009b)

The spatial distribution of mineral P load at the Kielstau watershed for the period from 2006 to 2008 is shown in Figure 5.7. Mineral P load from subbasins ranged from 0.2 to 0.48 kg/ha. In order to distinguish more clearly the changes in mineral P load from different subbasins, the loading of mineral P was classified into four levels. However, the difference in mineral P load from subbasins is not significant. The results show that the dominant loading of mineral P occurred in the subbasins 2 and 6. Probably the WWTPs located along these tributaries are the main reasons for increasing mineral P load at the outlet of these subbasins. The remaining subbasins 1, 3, 4, 5, and 8 contribute a negligible amount of the mineral P load in general. This is because mineral P inputs of these subbasins were mainly fertilizer application from agricultural activities and manure from grazing animals. Although these areas are dominated by agricultural land and pasture, the slope is the more critical factor for transporting sediment and phosphorus than land use format. The characteristic of flat topography ranging from 27.3 to 79.9 m (Figure 5.1) limits remarkably the dynamic of phosphorus within the watershed (Lam et al., 2011). Therefore, the predicted results of phosphorus load from these subbasins have been strongly affected by watershed conditions in space and time.

Overall agreement between measured and simulated values is considered moderate to good for all parameters used in both the Kielstau and Moorau catchment, especially for flow and nitrogen, while phosphorus is slightly underestimated in all simulation periods. The statistical results indicate that the SWAT model performed satisfactorily in simulating daily nutrient loads and can be used for assessing the spatial distribution in nutrient load from subbasins and for identifying the crucial pollution areas within the watershed.

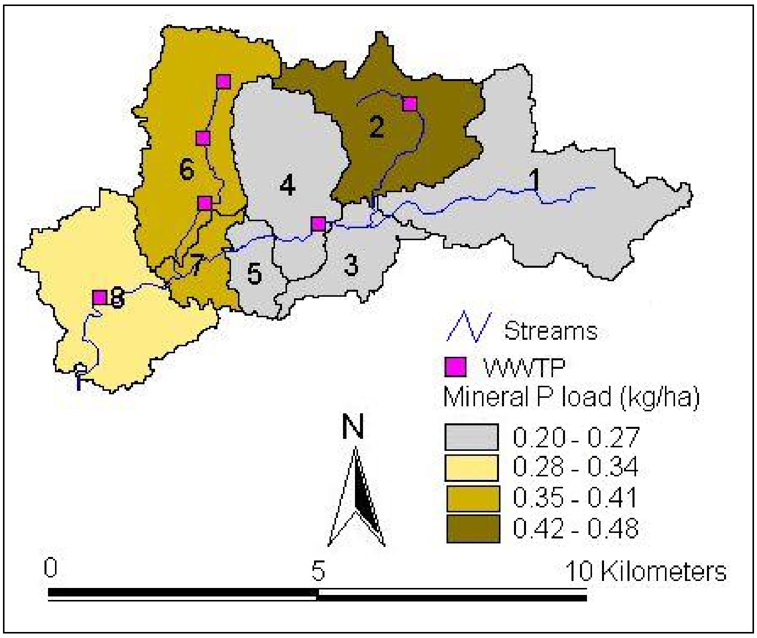


Figure 5.7: Spatial distribution of simulated mineral P load in the Kielstau watershed (2006-2008)

5.3.4 Contribution of groundwater to nitrate load

To evaluate major processes controlling the nitrate load in lowland catchments, total monthly nitrate and subsurface monthly nitrate loads were estimated at the outlets of the Kielstau and Moorau catchments for the period from 2006 to 2009. Figure 5.8 shows that the loading of nitrate during the summer months is lowest, while the highest nitrate load occurred in winter months. The results also indicate that more than 93% of the nitrate load was coming from the shallow aquifer to the stream at the Kielstau outlet compared to 93.7 % of that in the Moorau catchment. The remaining nitrate load (about 7%) was coming either via lateral flow or surface runoff. These results are also in accordance with previous studies by Schmalz et al. (2009) and Lam et al. (2010) who found that the groundwater component is a dominant pathway in lowland areas in comparison to mountainous areas where surface runoff is considered to be the dominant pathway (Fohrer et al., 2001, 2005; Lenhart et al., 2002; Van Griensven at al., 2006; Schuol et al., 2008).

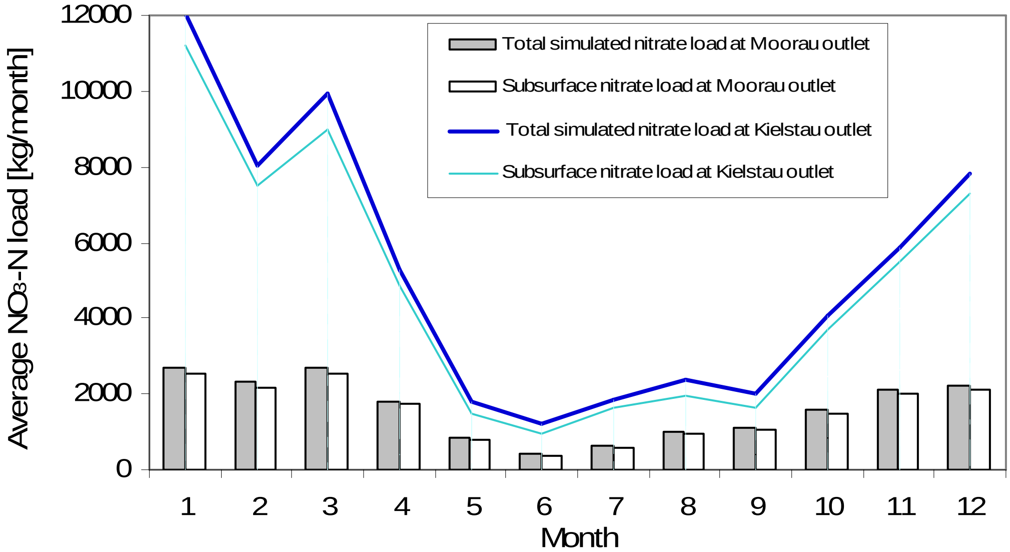


Figure 5.8: Average monthly nitrate load at the catchment's outlets for the period from 2006-2009

5.3.5 Impact of Best Management Practices on spatial distribution of nutrient loads

For the previous study, practices aiming at reducing the load of nutrient coming from agriculture were mainly focused on the BMPs that reduce the leaching of nutrient in the lowland Kielstau catchment (Lam et al., 2011). The simulated results have been demonstrated in the form of temporal distributions. In this study, spatial distributions of total nitrogen and phosphorus loads in various subbasins within the watershed were evaluated in order to assess the nutrient dynamics as well as to identify critical subbasins contributing significant nutrient loads to the watershed outlet.

The detailed description of each BMP can be found in Lam et al. (2011). These BMPs have been assessed as well as compared elaborately the effectiveness to costs of their implementations. The selected appropriate BMPs satisfying with both the effectiveness and costs were summarized in Table 5.5.

Table 5.5 Description of appropriate BMPs in the Kielstau catchment

Measure	Description	Code
Extensive land use management	Change of crop rotations (from winter wheat-winter wheat-rape (WWR) to winter wheat-winter rye-winter rye (WRyRy))	ELUM

Nutrient management plan	Reduction of 20% fertilizer and manure application rates in arable land	NMP
Grazing management practice	Reduction of livestock density from 2 to 1.1 LU ha^{-1} and no fertilizer application on pasture land	GZM
Field buffer strip	Installation of 10 m field buffer strips along the edge of main channel in arable and pasture land.	FBS
Combination scenarios	Combination of the four above scenarios	CBN

As analyzed above, the exceeded nitrogen is the main reason resulting in water pollution in these catchments. In addition, the results of nitrogen loads obtained from previous study (Lam et al., 2011) showed that the trend in NO_3-N load is very similar to the TN load at the watershed outlet of the Kielstau catchment. Therefore, we mainly focus on analyzing the effect of BMP implementation on TN load and finding out critical pollution areas so that further BMPs could improve water quality more efficiently in the Kielstau and Moorau catchment. However, the influence of that BMP implementation on TP behaviors is also considered at HRU and subbasin level for both the Kielstau and the Moorau catchments.

The annual simulated TN and TP loads by the separate HRUs for the baseline scenario are shown in Figure 5.9. High TN loads ranging from 32.91 to 41.42 kg/ha are mostly originated from HRUs which are overlaid by arable land and soils, while pasture areas contributes high TP loads (0.63 – 0.75 kg/ha) to the stream. Forest land is found to be the lowest contribution to TN and TP loads, ranging from 7.29 to 15.82 kg/ha and from 0.39 to 0.5 kg/ha, respectively.

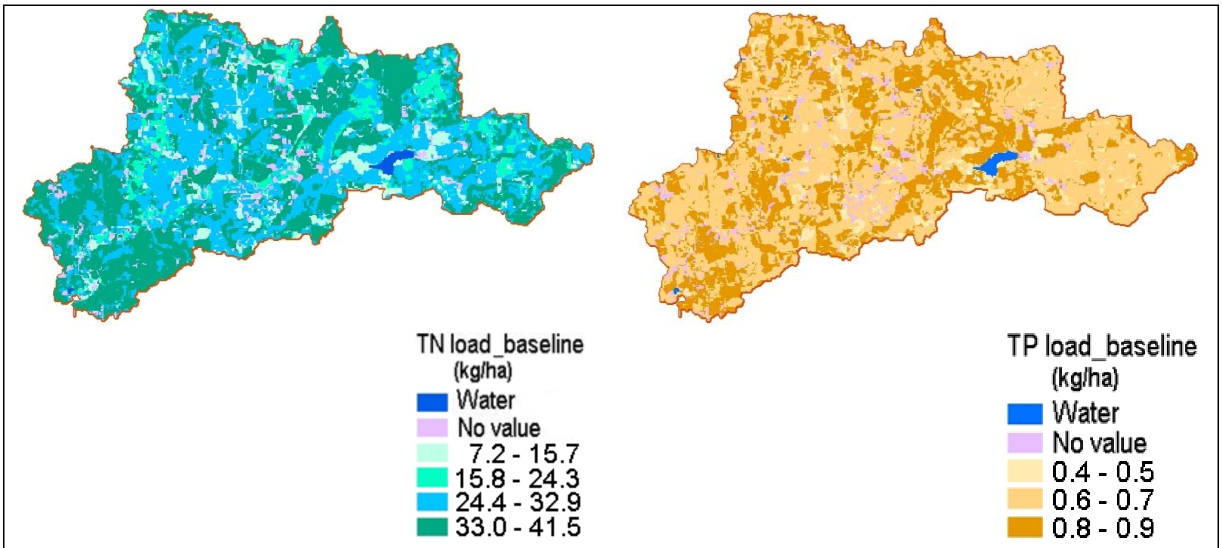
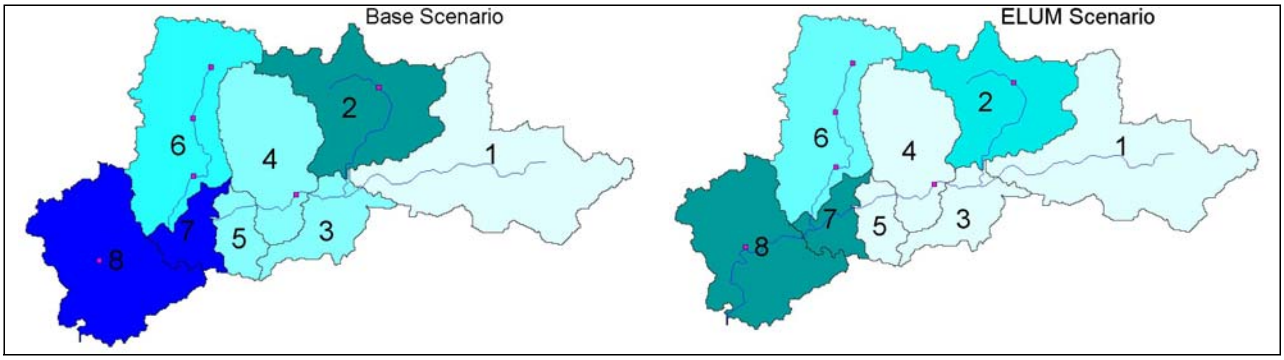


Figure 5.9: Simulated annual TN and TP delivered to stream by HRUs from the Kielstau catchment

The results of average annual TN load distribution for individual practice implemented in the whole Kielstau catchment for the period from 2006 to 2008 are shown in Figure 5.10. The average annual load for TN was calculated by summing mineral and organic nitrogen load under each scenario. As it can be seen from Figure 10, the loading of TN exported from subbasins under the simulated baseline scenario indicates substantial spatial differences among the Kielstau watershed. TN load from subbasins ranged from 9.15 to 39.91 kg/ha. The highest TN load occurred in the subbasins located near the watershed outlet, followed by subbasins in northern watershed. The nitrogen inputs of the subbasins in northern watershed are not only coming from the fertilizer application in agriculture but also from WWTPs. As a result, these subbasins contribute high amount of nutrients to the river Kielstau, especially for subbasin 2, namely Moorau.

Looking at the spatial distribution of average annual TN load from different scenarios it is clear to see that TN load from subbasins under each scenario was significantly reduced in comparison with baseline scenario results. The loading of TN was represented by different colors that the darker color suggests higher TN load from subbasin within the watershed. The spatial differences of the TN load between the baseline condition and scenarios are given in Figure 5.10. The results show that implementation of individual BMPs provides relatively similar load reduction in nitrogen. These can be given by the loading of TN from subbasin 8 (Base scenario) corresponding to class 5 (25.16 - 39.91 kg/ha) is reduced to class 4 (22.16 - 25.15 kg/ha) in case of ELUM, NMP, GZM, and FBS scenario and finally to class 3 (17.16 - 21.15 kg/ha) in case of CBN scenario. Similar reduction results were also found in subbasin 2 and subbasin 6 when implementing individual BMPs. In general, great effects occur in the subbasins where arable land and pasture are dominant (e.g., subbasins 2, 6, and 8). For these subbasins, the respective percentage of arable land and pasture ranging from 51-60% and from 27-29% of which subbasin 8 has the highest rate of arable land, occupying 60% of the total land use (Lam et al., 2010). The high percentage of arable land and pasture is of great advantage to apply appropriate BMPs such as ELUM, NMP, and GZM. Moreover, the length of main channels or rivers in these subbasins is also longer than others. Thus, the applying FBS scenario achieved more reduction in TN load at the catchment outlet.



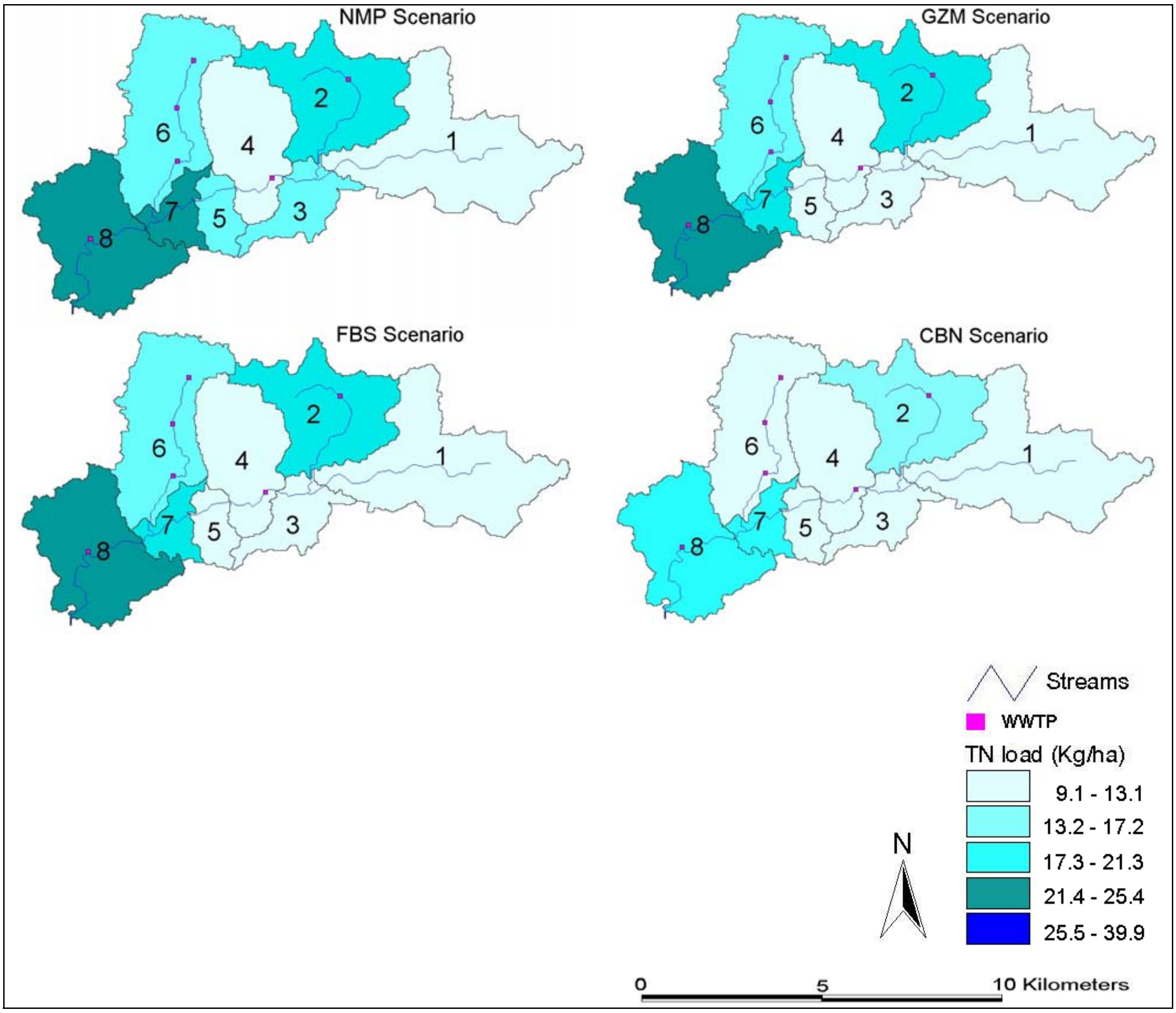


Figure 5.10: Simulated average annual TN load distribution in the Kielstau watershed

The average annual TP distribution in the Kielstau watershed for the period from 2006 to 2008 is illustrated in Figure 5.11. The loading of TP from subbasins in the case of the baseline scenario ranged from 0.36 to 0.63 kg/ha. By comparing with the results of the baseline scenario, the reduction in TP load under the implementation of different BMPs is not significant in general. This is due to the special characteristics of lowlands of which low slope is a crucial factor minimizing the transport of sediment and phosphorus from subbasins to the river. However, the implementation of simultaneous BMPs does have a small impact on TP reduction at the watershed outlet. The TP load ranged from 0.36 to 0.49 kg/ha in the case of CBN scenario.

From the analyses of the distribution of TN and TP loads under different BMP implementation within the Kielstau catchment, it can be found that the Moorau is an important contributor of nutrient to the river Kielstau. Thus, the Moorau catchment deserves special attention in this study.

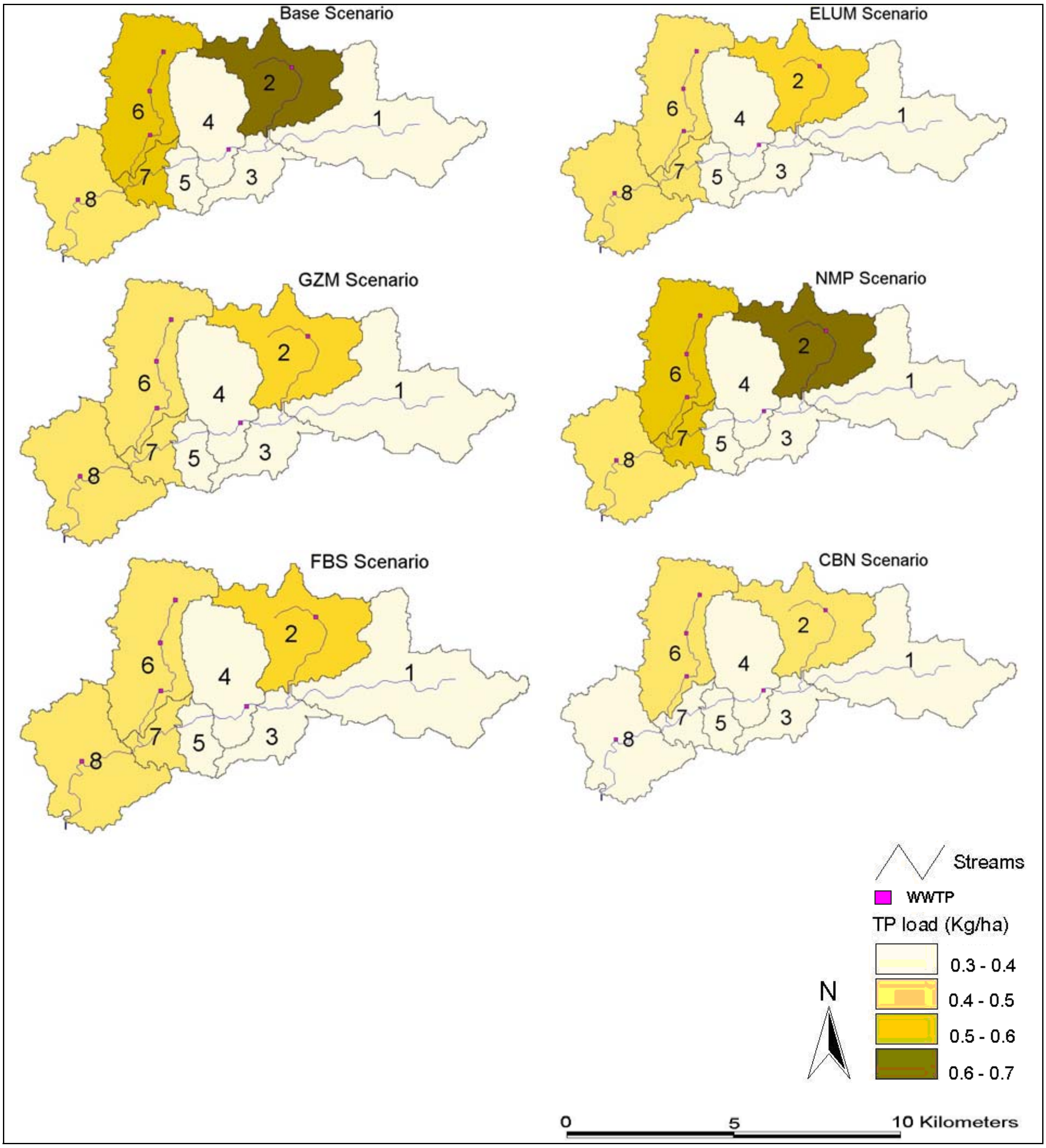


Figure 5.11: Simulated average annual TP load distribution in the Kielstau watershed

To further determine areas resulting in high pollution at the outlet of the Moorau catchment, BMP scenarios suggested in the Kielstau catchment have also been implemented in the Moorau catchment. The results of TN and TP load at the subbasin outlets of the Moorau catchment are shown in Figure 5.12 for the baseline and CBN scenario. The loading of TN and TP ranged from 13.16 to 25.15 kg/ha and from 0.43 to 0.63 kg/ha in case of the baseline scenario, respectively. After implementing CBN scenario, nutrient load has been reduced significantly at the subbasin outlet. TN load from subbasin 1 (Baseline scenario) corresponding to class 3 (21.16 – 25.15 kg/ha) is decreased to class 2 (17.16 – 21.15 kg/ha) in case of CBN scenario. The similar results were found in TP load. The results obtained from baseline and CBN scenario showed that subbasin 1 of the Moorau catchment is found to be the most nutrient load enhancing pollution level at the Moorau outlet. The high loading of nutrient from subbasin 1 is probably due to the contribution of nutrient from agricultural land, urban and WWTP.

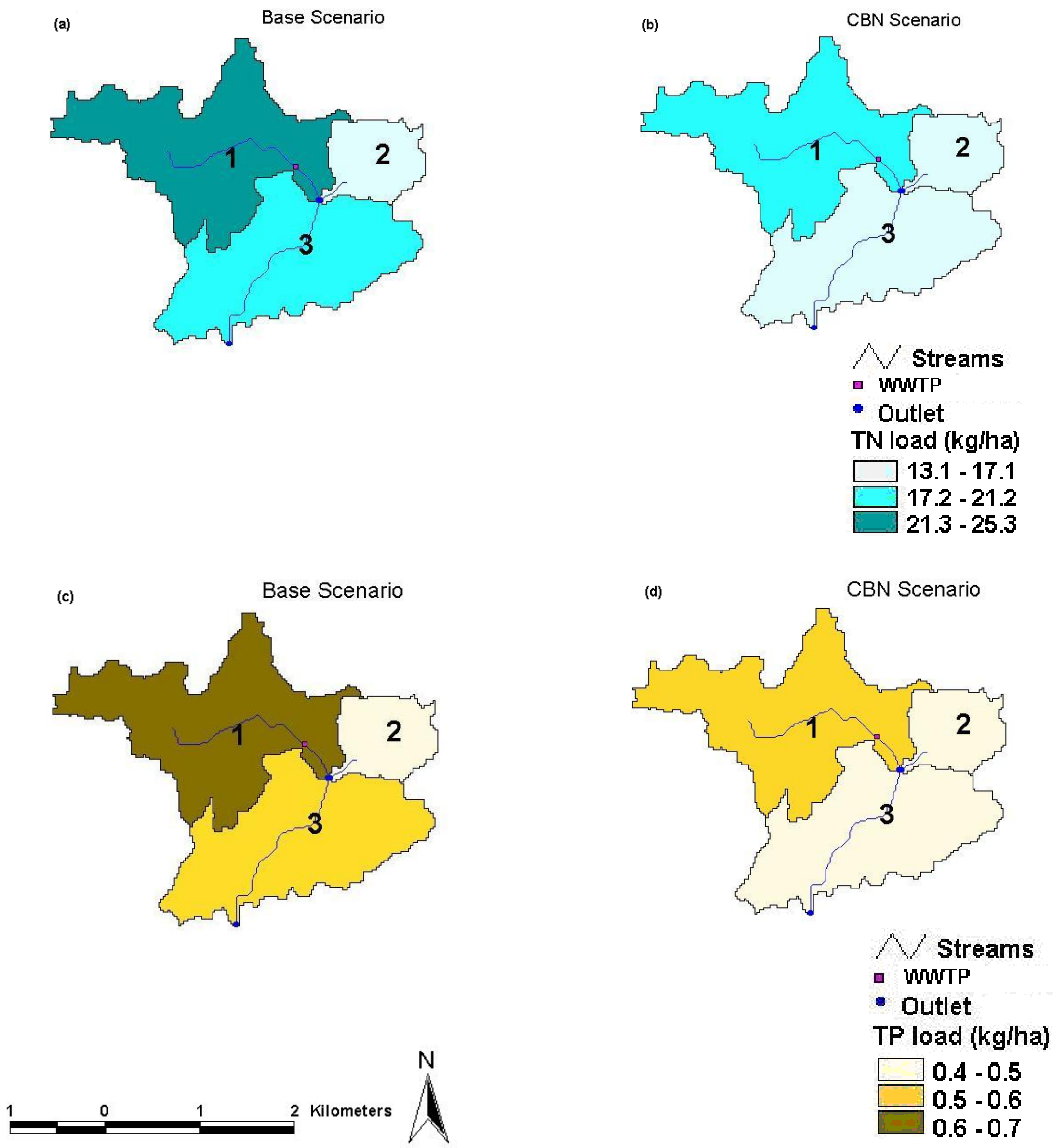


Figure 5.12: Simulated average annual nutrient load distribution in the Moorau catchment.a) TN base scenario, b) TN CBN scenario, c) TP base scenario, and d) TP CBN scenario

5.3.6 Additional scenario

Based on the above analyses, we found that WWTPs partly enhance the nutrient concentration in the subbasin, where WWTPs are located. The results in Figure 5.10 and Figure 5.11 indicate that high TN load in relation to WWTPs is found in subbasin 2, while a high TP load occurred in both the subbasin 2 and 6. To evaluate the impact of the emission from these WWTPs on the pollution level at the outlet of the Kielstau and the Moorau catchment, a reduction in nutrient emission from those WWTPs needed to be considered. Hesse et al. (2008) showed that reducing the N emission of all sewage treatment plants within the catchment by 10% and the P emission by 20% resulted in reduction in nutrient load at the outlet of the lowland catchment, respective achieving by 0.7 and 9 % for nitrate and phosphate phosphorus load reduction. In this study, reduction of 20% in N and P emission from WWTPs is suggested for subbasin 2 and only the P emission reduction of 20% is assumed for subbasin 6. The reduction in TN and TP load at the outlets of the Kielstau and the Moorau catchments is given in Table 5.6.

Table 5.6 Impact of 20% reduction in WWTPs emission on nutrient load at the Kielstau and the Moorau outlet

Catchment	Reduction in TN load (%)	Reduction in TP load (%)
Kielstau	1.7	3.4
Moorau	7.6	5.3

The results obtained from this scenario show that 20% reduction in nitrogen emission from WWTP within subbasin 2 does not have a significant impact on reduction in TN load at the watershed outlet of the Kielstau catchment. However, it may help to minimize the magnitude of the nitrogen concentration within the Moorau catchment. This is given by 7.6% reduction in TN load at the outlet of the Moorau catchment compared to 1.7% reduction at the Kielstau catchment outlet. The reduction in TP load is also negligible, achieving by 3.4% and 5.3% for the Kielstau and the Moorau catchment, respectively. This reveals that the suggestion of 20% reduction in phosphorus emission from WWTPs in subbasin 6 is not considerably effective for minimizing TP load at the outlet of the Kielstau catchment. Therefore reduction in nutrient emission from WWTPs should be considered in Subbasin 2 only.

5.3.7 Critical pollution area identification

Identification of critical subcatchments was done on the basis of simulated nutrient load from each subbasin. Graphical displaying the performance of SWAT for identification of critical pollution source areas is presented from Figure 5.10 to Figure 5.12. By implementing individual BMPs, the nitrogen loads have been reduced significantly at the Kielstau catchment outlet in general. Considering the spatial results of TN load from Figure 5.10 (in case of CBN scenario) and from Figure 5.12, we found that nitrogen critical areas are concentrated in (i) subbasin 1 of the Moorau catchment where both point and diffuse source enhance considerably nitrogen load at the outlet of this subbasin and (ii) subbasin 8 of the Kielstau catchment where diffuse source pollution from agricultural activities is considered to be a main reason for elevating nutrient load at this subbasin outlet. Therefore, these subbasins can be attributed to the critical subbasins increasing nutrient load at the outlet of the Kielstau catchment and further reduction in nutrient load from these subbasins could improve water quality in the whole catchment. The further implementation of BMPs include (i) further reduction in fertilizer application rates, (ii) lower livestock grazing density on pastures, (iii) widing field buffer strips along the edge of main channel in arable and pasture land, and (iv) 20% reduction in nitrogen emission of the WWTP in the subbasin 1 of the Moorau catchment. It could be expected that further implementation of these proposed practices would lower nutrient concentration and meet the target value the LAWA classifications at the watershed outlet of the Kielstau catchment.

5.4 Conclusions

The ecohydrological model SWAT was used to simulate flow, nitrogen, and phosphorus transport in the Kielstau catchment and its subasin, namely Moorau catchment - typical rural lowland areas in Northern Germany. The discharge prediction of SWAT model tested in this study showed quite good agreement with the measured data, achieving the E_{NS} and R^2 from 0.75 to 0.84 for both the calibration and validation period. The statistical coefficients of the nitrogen and phosphorus model performance were relatively reasonable ranged from 0.42-0.79 and 0.55-0.81 for E_{NS} and R^2, respectively, during the simulation period. In general, the SWAT could successfully simulate daily flow and nutrient load at the catchment outlets. The Kielstau catchment area and its subcatchment Moorau in Northern Germany served as an example for lowland areas. The results obtained from calibration model of these catchments indicate that parameter set could be transferred in lowland catchments with similar environmental conditions.

The critical pathways transporting nitrate load from subbasins to the stream were assessed in both the Kielstau and Moorau catchment. The results indicated that shallow groundwater flow was the major source of nitrate in the stream accounting for about 93% of total nitrate load, while only about 7% results from either lateral flow or surface runoff. This indicates clearly that mitigation measures should focus on reducing primarily nitrate leaching in lowland areas.

A calibrated and validated SWAT model was utilized to identify nutrient sources within the Kielstau watershed. Average annual quantities predicted by the model were used to generate nitrogen and phosphorus maps. The results of this study based on model simulation indicated that the implementation of BMPs in the Kielstau watershed would result in a significant reduction of nutrient load. An additional reduction of 20% nutrient emission from WWTP in the Moorau catchment would help to minimize the magnitude of the nutrient concentration within the Moorau catchment and to further improve water quality in the Kielstau catchment. Spatial distribution of nutrient load from HRUs level indicate that arable land contributes high nitrogen load to stream, while pasture land is found to be large sources generating high phosphorus load to stream than others in the watershed. The results also revealed that further implementation of additional BMPs for critical pollution areas would meet the water quality standard regulated by LAWA. The spatially distributed modeling approach in this study could be a useful method to determine critical pollution areas within the watershed before and after implementation of BMPs. It also helps decision makers to find appropriate measures aiming at further improving water quality in a watershed.

Chapter VI Summary and conclusion

6.1 Summary and key finding

This dissertation was conducted on the lowland Kielstau catchment in Northern Germany. The basic SWAT model input data includes climate, topography, soil, land use, and agricultural data; all of which, as well as consideration of sewage disposals of six WWTPs, was used to predict the current flow, sediment and nutrient loads at the watershed outlet. Climate data collected from 1993–2008 were used as inputs to the SWAT model. Field hydrology, sediment, and water quality (NO_3-N, NH_4-N, TN, PO_4-P, and TP) data were used in the model calibration and validation. The flow data were measured for 16 years from 1993 to 2008, while measured nutrient and sediment data were carried out over two years from 2006–2008. The model was calibrated for hydrology, sediment, and nutrient processes and its performance was evaluated at a daily time step. After implementing the model calibration and validation, BMPs scenarios were developed and tested in the whole catchment using the SWAT model. The effectiveness of BMP implementation in water quality improvements was compared to those in the baseline scenario. The trade-off relationship between the effectiveness in nutrient reduction and the corresponding cost of BMPs implementation was taken into account in this study. Lastly, the model was tested in the Moorau catchment, a subbasin of the Kielstau catchment, using more than one year of measured flow and nutrient data at the Moorau outlet. The assessment of transferability of parameter set between the two catchments as well as the spatial distribution of nutrient loads at the HRUs and subbasin levels were also considered in the whole catchments.

The chapters presented in the dissertation were submitted as individual papers in international peer-reviewed scientific journals. They cover a range of specific topics, which are important for a full understanding of the system. These include:

- Ecohydrological modeling of water discharge and nitrate loads in a mesoscale lowland catchment, Germany
- Modeling point and diffuse source pollution of nitrate in a rural lowland catchment using the SWAT model
- The impact of agricultural Best Management Practices on water quality in a North German lowland catchment
- Assessing the spatial and temporal variations of water quality in lowland areas, Northern Germany

In combination, these papers provide descriptions of the hydrology, sediment, and nutrient dynamics in the lowland areas located in Northern Germany. Key finding include:

- The performance of the SWAT model is satisfactory in simulating discharge, sediment and nutrient loads at the watershed outlet of the lowland Kielstau catchment
- Groundwater parameters are found to be most sensitive and they turned out to be the most

influential factors on water discharge

- Diffuse sources are the dominant source of nitrate in the whole catchment
- Agriculture is found to be the dominant contributor of diffuse sources, while forest areas have lowest impact on nitrate load in comparison with those of other land use types
- The implementation of BMPs reduce significantly NO_3-N and TN load at the outlet of the Kielstau catchment. However, the impacts of these BMPs implementation were not significant on the reduction in sediment and TP load due to the specific characteristics of lowland area (e.g. flat topography, low surface runoff)
- Reduction only in one type of BMP did not obtain the target value for water quality class II (LAWA, 1998).
- The combination of BMPs improved considerably water quality in the Kielstau catchment. This scenario reduced the average annual loads for NO_3-N and TN by 53.9 % and 46.7% at the watershed outlet of the Kielstau catchment, respectively.
- Parameter sets could be transferred in other lowland catchments with similar environmental conditions
- Shallow groundwater flow was the major source of nitrate in the stream
- Nitrogen parameters were found to be the main parameters contributing to increase water pollution in the catchments
- Arable land contributes high nitrogen load to the stream at the HRU levels, while pasture land is found to be a larger source generating high phosphorus load to the stream than others in the watershed
- Applying a spatially distributed modeling approach was an appropriate method to identify the crucial pollution areas within a watershed

6.2 Conclusions

In summary, the important conclusions drawn from this study have been described as follow:

The Kielstau catchment is primarily agricultural, with agricultural land (arable and pasture land) covering approximately 81% of the catchment, causes problems with enhanced diffuse nutrients in the river and leads to the degradation of surface and ground water resources. Besides, emission from point sources partly contributes to abate water quality in the catchment. For the purpose of this research the Soil and Water Assessment Tool (SWAT-2005) was applied to evaluate the impact of point and diffuse pollution on water quality within the catchment and to find appropriate BMPs aiming at reducing water pollution at the outlet of the catchment.

The basic input data including climate, topography, soil, land use, and agricultural data was used in the model. For the representation of land use management, characteristic crops, rotations, fertilization rates, and livestock numbers were gathered. In addition, data for nutrient emission from WWTPs were also obtained for the study area.

The model has been performed in both the Kielstau catchment and its subcatchment Moorau. In the Kielstau catchment, the application of the model first involved the analysis of parameter sensitivity, which was then used for model auto-calibration following the hierarchy of sensitive model parameters. The calibration and validation of the model output were implemented by comparing simulated values with corresponding in-stream measurements at the watershed outlet. Calibration and validation of the model, based on river flow, sediment, nitrogen, and phosphorus, was in the range for satisfactory model performance. The results of sensitivity analysis showed groundwater parameters as the main factors influencing runoff. After calibrating the parameters, the correlation between calibrated and simulated daily flows is fairly good for the Kielstau and the Moorau catchment, achieving the Nash-Sutcliffe efficiency (E_{NS}) and the coefficient of determination (R^2) from 0.75 to 0.84 for both the calibration and validation period. Regarding the different pathways modelled by the SWAT model in the Kielstau catchment, the simulated results indicate that groundwater flow (79.6%) is the dominant pathway in this lowland catchment, while surface runoff (0.3%) is very low compared to other components.

Sediment calibration showed relative agreement between measured and simulated data at the outlet of the Kielstau catchment. This is given by the E_{NS} and R^2 increased from 0.57 to 0.58 and from 0.63 to 0.65 for both the calibration and validation period, respectively. The simulated results indicate that sediment load from the channel caused by bank erosion is dominant, accounting for about 72% of the total average annual sediment load at the main watershed outlet while the loading of sediment from the fields is negligible, achieving 28%. The reasons for low sediment load from the fields are due to the characteristic of flat area and the low surface runoff in this lowland catchment. Similar results were aslo found by Kiesel et al. (2009) when they used the SEPAL approach to quantify sediment pathways in the same catchment.

SWAT underestimated phosphorus loads at the outlet of the Kielstau and the Moorau catchment. The agreement between simulated and measured phosphorus loads was not so good for both the calibration and validation period in the two catchments, giving the E_{NS} and R^2 of 0.42-0.49 and 0.58-0.69, respectively. The underestimation of phosphorus loads is mainly due to low sediment load from the subbasins and low surface runoff in this lowland catchment. In addition, lack of soil phosphorus cycling (leaching of soil phosphorus through the soil profile, lateral, and tile flows) in SWAT could be attributed to another reason causing underestimation of the phosphorus model.

For the nitrogen, the model performed well in simulating nitrogen loads at the watershed outlet of the two catchments in general. The statistical values of E_{NS} and R^2 obtained from the nitrate model ranged from 0.68 to 0.79 and from 0.70 to 0.81 for the calibration and validation period, respectively, showed a good agreement between the measured and simulated nitrate load at the watershed outlets. High values of E_{NS} and R^2 are also obtained for the total nitrogen model when comparing the measured and simulated total nitrogen load at the watershed outlet of the Kielstau catchment. The results of this research show that fertilization is important in nitrate load estimation while ammonium load is mainly influenced by point source emission within the watershed. The contribution of different sources to nitrate load assessed in chapter III showed that 95% of the total nitrate load is originating from diffuse sources while only 5% results from point sources within the Kielstau catchment. The results also indicate that agriculture is found to be the dominant contributor of diffuse sources and the percentage of agricultural land area is considerably positively correlated to nitrate load at the different subbasins, while forest areas have the lowest influence on nitrate load in comparison with those of other land use types. Additional results obtained from chapter V showed that shallow groundwater flow was the major source of nitrate in the stream accounting for about 93% of total nitrate load, while only about 7% results from either lateral flow or surface runoff. These results denote that mitigation measures should implement in the fields aiming at minimizing diffuse source pollution and focus on reducing primarily nitrate leaching in lowland areas.

The BMP simulations demonstrate the utility of the SWAT model in predicting load reductions in sediment, nitrogen, and phosphorus at the watershed outlet under varying scenarios. Four scenarios were developed to inquire into impact of possible different land used management on the behavior of diffuse source pollutions in the lowland Kielstau catchment. The purpose of these scenarios was not only to predict considerable changes in the catchment to obtain high amount of sediment and nutrient reduction, but also to develop feasible scenarios which would allow agriculture to operate at an appropriate level to maintain and conserve landscape in its current state. Scenarios were developed based firstly on interviews with farmers and relevant literature, and secondly on the realistic condition of the lowland catchment such as topography, agricultural cultivation, landscape, and farming technique.

With the extensive land use management (ELUM) scenario, different crop rotations were examined in the study area. The current crop rotation (winter wheat-winter wheat-rape) was replaced with another crop rotation (e.g., winter wheat-winter rye-winter rye) aiming at minimizing nutrient application rates. At the same time three types of tillage were also considered to investigate effectiveness of the combination between different crop rotations and tillage measures on environmental mitigation in the lowland Kielstau catchment. The nutrient management plan (NMP) scenario was recommended in this study by applying 20% reduction in nutrient application for both mineral fertilizer and manure in arable land. The purpose of this scenario is to evaluate the influence of decrease in fertilizer application rate on nutrient load reductions at the watershed outlet. The grazing management practice (GZM) scenario included lower livestock density from 2 to 1.1 LU ha^{-1} and no fertilizer application in pasture land was recommended. In this scenario only cow were considered to be present in the catchment. For the field buffer strip (FBS) scenario 10 m field buffer strips were applied only for arable land and pasture land along the main channel. The above BMPs were simulated for over a 3-year period (2006–2008) using the SWAT model to quantify the impacts of those scenarios on the improvement of water quality in the Kielstau catchment.

The above BMPs used in the study were found to be effective in mitigating nitrogen load at the main outlet of the Kielstau catchment. However, these measures would not reduce sediment and TP load at the watershed outlet significantly due to low surface runoff, flat topography, and low sediment load from the fields to stream networks. Implementation of BMPs reduced the average annual loads for NO_3-N at the catchment outlet by 11.6% (ELUM), 9.9% (NMP), 20.5% (GZM), and 15.3% (FBS); 9.9% (ELUM), 8.6% (NMP), 15.6% (GZM), and 12.9% (FBS) for TN; 4.6% (ELUM), 1.1% (NMP), 3.9% (GZM), and 5.3% (FBS) for TP. Reduction in sediment load at the watershed outlet is not significant, achieving from 0.82% to 4.9% reduction under BMP implementation. Based on the results obtained from scenario implementation, it can be concluded that the individual BMP scenarios have partly contributed to improve water quality at the watershed outlet. However, no single scenario results in significant decrease of nutrient load. The combined scenario (CBN) was considered in this study, achieving significant reduction by 11.9%, 46.7%, 53.9%, and 13.6 % for sediment, TN, NO_3-N, and TP, respectively. At this reduction level, the NO_3-N concentration was reduced significantly, achieving 2.9 mg l^{-1} at the 90th percentile level of concentration and closed to the LAWA class II with an upper limit of 2.5 mg l^{-1} (LAWA, 1998) - the general target for surface waters in Germany.

Together with BMP implementation, the trade-off relationship between the effectiveness in nutrient reduction and the corresponding cost of BMP implementation was considered. Considering the ELUM scenario, the results indicate that implementation of conservation tillage is unrealistic for the Kielstau lowland catchment because of high annual costs for implementation and low effectiveness in nitrogen load reduction. Conversely, a shift from the current crop rotation (winter wheat-winter wheat-rape) to a new crop rotation (winter wheat-winter rye-winter rye) was found to be more effective in reducing considerably nitrogen load at the watershed outlet and low costs for implementation. Costs for implementation of ELUM, NMP, GZM, and FBS scenarios were based on current prices in the year of 2008/2009 in Germany (KTBL, 2008; SBD, 2009). The results showed that the implementation of the combined scenarios (CBN) provided the highest load reduction of nutrient and the corresponding costs € 93,000 per year. The trade-off between cost and effectiveness of BMPs will be helpful for policymakers and stakeholders to identify suitable BMPs for improving water quality in the Kielstau catchment.

The average measured results of nutrient parameters (NO_3-N, NH_4-N, TN, PO4-P, and TP) were compared to class II (moderately polluted) of the LAWA standard (LAWA, 1998) whose quality class II represents the target value for water quality until the year 2015 according to the European Water Framework Directive (EC, 2000). The comparable results showed that parameters in relation to nitrogen parameter far exceeded the allowable limit of class II of the LAWA standard, while the magnitude of phosphorus values are not much higher than those of LAWA class II in both the Kielstau and the Moorau catchment. This could be concluded that nitrogen parameters need to be improved by implementing BMPs in this study and the reduction of nitrogen loads at the watershed outlets has extremely important significance for improving water quality for these study areas. However the influence of those BMPs on phosphorus was also referred in term of temporal and spatial distribution to know their behaviors in these lowland areas.

The impact of BMPs implementation on nutrient load reduction was illustrated in the form of temporal and spatial distribution from hydrologic response units (HRUs) as well as from subbasin level in order to identify the crucial pollution areas within a watershed. TN load from subbasins under each scenario was significantly reduced in comparison with baseline scenario results, ranging from 9.15 to 39.91 kg/ha. Contrary to TN, the reduction in TP load under the implementation of different BMPs is not significant in general, ranging from 0.36 to 0.63 kg/ha. Spatial results obtained from analyses of different pollution sources as well as nutrient loads at the subbasin outlets in the Kielstau and the Moorau catchment indicate that critical pollution areas are concentrated in (a) subbasin 1 of the Moorau catchment where both point and diffuse source enhance considerably nitrogen load at the outlet of this subbasin and (b) subbasin 8 of the Kielstau catchment where diffuse source pollution from agricultural activities is considered to be a main reason for elevating nutrient load at this subbasin outlet. The results also revealed that further implementation of additional BMPs for critical pollution areas combined with 20% reduction of nutrient emission from WWTP in the Moorau would meet the water quality standard established by LAWA. The spatially distributed modeling approach in this study could be a useful method to help decision makers finding appropriate measures aiming at further improving water quality in a watershed.

6.3 Future works

This thesis presents the results of measurements and modeling of water, sediment, and nutrient loads at the catchment scale. The model performed successfully in simulating flow, sediment, and nutrient load. The impact of the BMP implementation on nutrient load reduction at the watershed outlet was evaluated elaborately by using the SWAT model. A short period of 03 years (2006–2008) was used for evaluating the model performance due to the availability of measured data. To explore the full behavior of nutrient dynamics in this study, the following issues should be considered for further research and development.

- Continued measurement of daily data at the watershed outlets will provide an extended dataset, which could be used to increase calibration and validation periods. It is expected that a long-term period of data used for the calibration procedure and the implementation of BMPs in the model will provide more realistic results in relation to nitrogen and phosphorus loads at the watershed outlet.
- Field tests should be carried out to quantify sensitive parameters more accurately, such as soil chemical concentrations, soil temperature, soil moisture, and soil texture.
- Accurate information of management practices (e.g., timing and amount of fertilizer application rates, harvest dates, and tillage depth) should be continuously investigated in the whole catchment so that the calibration period can considerably improve water quality predictions.
- The utilization of manure production by farmers should be investigated to estimate the

actual application of animal waste on agricultural land and its impact on water quality.

- Soil carbon contents are assumed to be static in the SWAT model. This approach should be improved by an updated carbon cycling submodel that provides more realistic regarding carbon cycling processes.
- The performance of the SWAT model in simulating phosphorus load is limited. SWAT currently assumes that phosphorus is mostly transported in surface runoff and with sediment to the stream (Neitsch et al., 2005). Thus, there is a need to develop a partitioning phosphorus model to simulate leaching of soil phosphorus through the soil profile, lateral, groundwater, and tile flows using existing inputs and incorporate into SWAT.

Acknowledgements

This research study was carried out during more than three years in the Department of Hydrology and Water Resources Management, Institute for the Conservation of Natural Resources, Christian-Albrechts-Universität zu Kiel.

I would like to express my deep gratitude to my first supervisor, Prof. Dr. Nicola Fohrer for her constructive guidance and encouragement and for reviewing and improving my manuscripts throughout my research.

I would like to thank my advisor, Dr. Britta Schmalz for her interest to this work and for giving invaluable comments for the success of my study.

I would also like to express my appreciation to Dr. Georg Hörmann, Monika Westphal and Bettina Hollmann for providing me data and helping me during the time I worked in laboratory.

I would also like to thank my colleagues at the Department of Hydrology and Water Resources Management for their help and support.

I would like to express my sincere gratitude to the Vietnamese government and Deutscher Akademischer Austausch Dienst (DAAD) for granting me the fellowship to study at Christian-Albrechts-Universität zu Kiel.

Finally, I want to acknowledge my parents for the opportunity of studying abroad. I especially thank my wife Nguyen Thi Thuy Diem and my daughter Lam Thu Ha for their support and understanding during the whole time of my work.

Kiel, 2011 Lam Quang Dung

Bibliography

Adamson, J.K., Scott, W.A., Rowland, A.P., Beard, G.R., 2000. Ionic concentrations in blanket peat bog in Northern England and correlations with deposition and climatic variables. Eur. J. Soil Sci. 51, 1-15.

Ambus, P., Lowrance, R., 1991. Comparison of denitrification in two riparian soils. Soil Sci. Soc. Am. J. 55, 994-997.

Ambus, P., Christensen, S., 1993. Denitrification variability and control in a riparian fen irrigated with agricultural drainage water. Soil Biol. Biochem. 25, 915-923.

Andersen, H.E., 2004. Hydrology and nitrogen balance of a seasonally inundated Danish floodplain wetland. Hydrol. Processes 18 (3), 415-434.

Arabi, M., Govindaraju, R.S., Hantush, M.M., 2004. Watershed Management Tool for Selection and Spatial Allocation of Non-Point Source Pollution Control Practices. EPA/600/R-08/036, National Risk Management Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Cincinnati, Ohio 45268, 91 p.

Arabi, M., Frankenberger, J.R., Engel, B.A., Arnold, J.G., 2007. Representation of agricultural conservation practices with SWAT. Hydrol. Processes 22(16), 3042-3055.

Arnold, J.G., Srinivasan, R., Muttiah, R.S., Williams, J.R., 1998. Large area hydrologic modelling and assessment. Part I. Model development. J. Am. Water Resour. Assoc. 34(l), 73-89.

Arnold, J.G, Fohrer, N., 2005. SWAT2000: Current Capabilities and Research Opportunities in Applied Watershed Modeling. Hydrol. Processes 19(3), 563-572.

Bagnold, R.A., 1977. Bed load transport in natural rivers. Water Resources Res. 13 (2), 303-312.

Bärlund, I., Kirkala, T., Malve, O., Kämäri, J., 2007. Assessing SWAT model performance in the evaluation of management actions for the implementation of the Water Framework Directive in a Finish catchment. Environ. Model. Sofware 22 (5), 719-724.

Beasley, D.B., Hugins, L.F., 1982. ANSWERS: Areal non-point source watershed environmental response simulation. User's Manual, U.S.EPA Report, Chicago, IL, P.54.

Behera, S., Panda, R.K., 2006. Evaluation of management alternatives for an agricultural watershed in a sub-humid subtropical region using a physical process model. Agric. Ecosyst. Environ. 113(1-4), 62-72.

Behrendt, H., Huber, P., Opity, D., Scholy, G., Uebe, R., 1999. Nährstoffbilanzierung der Flussgebiete Deutschlands. UBA-Bericht. Inst. f. Gewässerökologie und Binnenfischerei im Forschungsverbund Berlin e. V., Berlin, 373 p.

Beyersdorf, W., 2008. Stoffaustrag aus einem ländlich geprägten Einzugsgebiet des Norddeutschen Tieflandes am Beispiel der Moorau. Master thesis in "Agrarwissenschaften", Ecology Centre, Kiel University, http://www.hydrology.uni-kiel.de/lehre/abschlussarbeiten/2008_beyersdorf_msc.pdf, 151 p.

BGR (Bundesanstalt für Geowissenschaften und Rohstoffe (Ed.)), 1999. Bodenübersichtskarte im Maßstab 1:200 000, Verbreitung der Bodengesellschaften, Hannover (unpubl.).

Bieger, K., 2007. Integrierte Analyse der Wasserqualität eines Fließgewässers im ländlich geprägten Raum des Norddeutschen Tieflands. Diplomarbeit im Fach Geographie. Ökologie-Zentrum Christian-Albrechts-Universität zu Kiel, http://www.hydrology.uni-kiel.de/lehre/abschlussarbeiten/2008_ kbieger_kielstauqualitaet.pdf, 142 p.

BLE (Bundesanstalt für Landwirtschaft und Ernährung), 2008. Durchschnittliche Anzahl der Rinder pro landwirtschaftlich genutzter Fläche in Deutschland. http://www.ble.de/cln_090/nn_1624448/SharedDocs/Downloads/01__Marktangelegenheiten/08__Marktbeobachtung/02__MilchUndMilcherzeugnisse/JaehrlicheErgebnisse/02__Bundeslaender/090915__406003100__06,templateId=raw,property=publicationFile.xls/090915_406003100_06.xls

Borah, D.K., Bera, M., 2003. Watershed-scale hydrologic and non-point source pollution models: review of mathematical bases. Trans. ASAE 46 (6), 1553-1566.

Bouraoui, F., Dillaha, T.A., 1996. ANSWERS-2000: runoff and sediment transport model. J. Environ. Eng. ASCE 122 (6), 493-502.

Bracmort, K.S., Arabi, M., Frankenberger, J.R., Engel, B.A., Arnold, J.G., 2006. Modelling long-term water quality impact of structural BMPs. Trans. ASABE 49 (2), 367-374.

Brown, L.C., T.O.Barnwell, Jr., 1987. The enhanced water quality models QUAL2E and QUAL2E-UNCAS: Documentation and user manual. EPA/600/3-87/007.

Bullock, A., Acreman, M., 2003. The role of wetlands in the hydrological cycle. Hydrol. Earth Syst. Sci. 7, 358-389.

David, M.B., Gentry, L.E., Kovacic, D.A., Smith, K.M., 1997. Nitrogen balance in and export from an agricultural watershed. J. Environ. Qual. 26 (4), 1038-1048.

Degarmo, E.P., Sullivan, W.G., Bontadelli, J.A., Wicks, E.M., 1997. Engineering Economy. Prentice Hall, Upper Saddle River, New Jersey, US.

Devito, K.J., Dillon, P.J., 1993. The influence of hydrologic conditions and peat oxia on the phosphorus and nitrogen dynamics of a conifer swamp. Water Resources Res. 29, 2675-2685.

Devito, K.J., Fitzgerald, D., Hill, A.R., Aravena, R., 2000. Nitrate dynamics in relation to lithology and hydrologic flow path in a river riparian zone. J. Environ. Qual. 29, 1075-84.

DLR (Deutsches Zentrum für Luft-und Raumfahrt), 1995. Landsat TM5-Scene of 1995, upper left corner: RW: 3503180 HW: 6084975, spatial resolution 25 m x 25 m. Köln.

Donoso, G., Cancino, J., Magri, A., 1999. Effects of agricultural activities on water pollution with nitrates and pesticides in the central valley of Chile. Water Sci. Technol. 39 (3), 49-60.

Duda, A.M., 1993. Addressing non-point sources of water pollution must become an international priority. Water Sci. Technol. 28 (3-5), 1-11.

DWD (Deutscher Wetterdienst), 2008. Climate data 1993-2007.

DWD (Deutscher Wetterdienst), 2009a. Climate data 1993-2008.

DWD (Deutscher Wetterdienst), 2009b. Means of precipitation and air temperature of the period 1961-1990. , last access January 3, 2009 .

EC, 1991. Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources. Off. J. Eur. Communities L375.

EC, 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy, Off. J. Eur. Communities, L327.

EC, 2002. Implementation of Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources. Off. J. Eur. Communities L319.

Evans, R.O., Gilliam, J.W., Skaggs, R.W., 1995. Controlled versus conventional drainage effects on water quality. J. Irrig. Drain. Eng. ASCE 121 (4), 271-276.

FAPRI (Food and Agricultural Policy Research Institute), 2006. Upper and lower Big Marines river watershed: computer based evaluation of the AgNPS-SALT project. Food and Agricultural Policy Research Institute. University of Missouri, 37 p.

Fohrer, N., Eckhardt, K., Haverkamp, S., Frede, H.G., 1999. Effects of land use changes on the water balance of a rural watershed in a peripheral region. J. Rural Eng. Develop 40 (5-6), 202-206.

Fohrer, N., Frede, H.-G., 2001. An integrated modelling approach to sustainable land use concepts in a low mountain range area. 17th World Congress of Soil Science, 14.-21. 8. 2001, Bangkok.

Fohrer, N., Moller, D., Stainer, N., 2002. An interdisciplinary modelling approach to evaluate the effects of land use change. Phys. Chem. Earth 27, 655-662.

Fohrer, N., Haverkamp, S., Frede, H.-G., 2005. Assessment of the effects of land use patterns on hydrologic landscape functions. Development of sustainable land use concepts for low mountain range areas. Hydrol. Processes 19(3), 659-672.

Fohrer, N., Schmalz, B., Tavares, F., and Golon, J., 2007. Ansätze zur Integration von landwirtschaftlichen Drainagen in die Modellierung des Landschaftswasserhaushalts von mesoskaligen Tieflandeinzugsgebieten, Hydrol. Wasserbewirtsch., 51(4), 164-169.

Frame, J., 1992. Improved grassland management. Farming Press Books. Wharfedale, 351 p.

Garen, D.C., Moore, D.S., 2005. Curve number hydrology in water quality modeling: Uses, abuse, and future directions. J. American Water Resour. Assoc. 41(2), 377-388.

Gassman, P.W., Osei, E., Saleh, A., Rodecap, J., Norvell, S., Williams, J., 2006. Alternative practices for sediment and nutrient loss control on livestock farms in northeast Iowa. Agric. Ecosyst. Environ. 117 (2-3), 135-144.

Gassman, P.W., Reyes, M., Green, C.H., Arnold, J.G., 2007. SWAT peer-reviewed literature: a review. Proceedings. Third International SWAT Conference, 13 July–15 July 2005, Zurich,Switzerland.http://www.brc.tamus.edu/swat/3rdswatconf/PDF/Session_I/Gassman.pdf (accessed 01.12.08).

Gevaert, V. , Van Griensven, A. , Holvoet, K. , Seuntjens, P., Vanrolleghem, P. A., 2008. SWAT developments and recommendations for modelling agricultural pesticide mitigation measures in river basins. Hydrol. Sci. J. 53(5), 1075-1089.

Gitau, M.W., Veith, T.L., Gburek, W.J., 2004. Farm-level Optimization of BMP Placement for Cost-Effective Pollution Reduction, Transactions of American Society of Agricultural Engineers (ASAE), Vol. 47, No. 6,1923-1931.

Gold, A.J., Groffman, P.M., Addy, K., Kellogg, D.Q., Stolt, M., Rosenblatt, A.E., 2001. Landscape attributes as controls on groundwater nitrate removal capacity of riparian zones. J. Am. Water Resour. Assoc. 39, 1457-64.

Gömann, H., Kreins, P., Kunkel, R., Wendland, F., 2005. Model based impact analysis of policy options aiming at reducing diffuse pollution by agriculture- a case study for the river Ems and a sub-catchment of the Rhine. Environ. Modell. Softw. 20, 261-271.

Grizzetti, B., Bouraoui, F., Granlund, K., Rekolainen, S., Bidoglio, G.F., 2003. Modelling diffuse emission and retention of nutrients in the Vantaanjoki watershed (Finland) using the SWAT model. Ecolo. Modell. 169, 25-38.

Hargreaves, G., Samani, Z.A., 1985. Reference crop evapotranspiration from temperature. Appl. Eng. Agric. 1, 96-99.

Hattermann, F.F., Krysanova, V., Habeck, A., Bronstert, A., 2006. Integrating wetlands and riparian zones in river basin modeling. Ecol. Modell. 199 (4), 379-392.

Haverkamp, S., Fohrer, N., Frede, H.-G., 2005. Assessment of the Effect of Land Use Patterns on Hydrologic Landscape Functions: A Comprehensive GIS-Based Tool to Minimize Model Uncertainty Resulting From Spatial Aggregation. Hydrol. Processes 19, 715-727.

Hayashi, M., Rosenberry, D. O., 2002. Effects of ground water exchange on the hydrology and ecology of surface water. Ground Water 40, 309-316.

Hefting, M.M., Jeroen, J.M. K., 1998. Nitrogen removal in buffer strips along a lowland stream in the Netherlands: a pilot study. Environ. Pollut. 102, 521-526.

Hesse, C., Krysanova, V., Päzolt, J., Hattermann, F.F., 2008. Eco-hydrological modelling in a highly regulated lowland catchment to find measures for improving water quality. Ecol.

Modell. 218 (1-2), 135-148.

Heuvelmans, G., Muys, B., Feyen, J., 2004. Evaluation of hydrological model parameter transferabilty for simulating the impact of land use on catchment hydrology. Phys. Chem. Earth, 29, 739-747.

Hill, B.R., 1999. Groundwater discharge to a headwater valley, northwestern Nevada, USA. J. Hydrol. 113, 265-283.

Højberg, A.L., Refsgaard, J.C., van Geer, F., Jørgensen, L.F., Zsuffa, I., 2007. Use of models to support the monitoring requirements in the water framework directive. Water Resour. Manage. 21 (10), 649-1672.

Humenik, F.J., Smolen, M.D., Dressing, S.A., 1987. Pollution from nonpoint sources: Where we are and where we should go. Environ. Sci. Technol. 21(8), 737-742.

Jha, M., Arnold, J.G., Gassman, P.W., 2006. Water Quality Modeling for the Racccoon River Watershed Using SWAT. Center for Agricultural and Rural Development. CARD Working Paper 06-WP 428.

Johanson, R.C., Imhoff, J.C., Davis, H.H., Kittle, J.L., Donigian, A.S., 1984. HSPF: Hydrologic Simulation Progam-Fortran. User’s Manual, U.S.EPA Environmental Research Laboratory, Athens, Georgia, 80 p.

Kiesel, J., Schmalz, B., Fohrer, N., 2009. SEPAL – A simple GIS-based tool to quantify sediment pathways in lowland catchments. Adv. Geosci. 21, 25-32.

Kladivko, E.J., Grochulska, J., Turco, R.F., Van Scoyoc, G.E., Eigel, J.D., 1999. Pesticide and nitrate transport into subsurface tile drains of different spacing. J. Environ. Qual. 30, 1305-1314.

Krause, S., Bronstert, A., 2004. Approximation of groundwater-surface water interactions in mesoscale lowland river catchment. Hydrol.: Sci. practice 21st Century, Br. Hydrol. Soc. 2, 408-415.

Krause, S., Bronstert, A., 2005. An advanced approach for catchment delineation and water balance modeling within wetlands and floodplains. Adv. Geosci. 5, 1-5.

Krause, S., Bronstert, A., Zehe, E., 2007. Ground water - surface water interactions in a North German lowland floodplain - implications for the river discharge dynamics and riparian water balance. J. Hydrol. 347, 404-417.

Kreins, P., Julius, C., Gömann, H., 2003. RAUMIS-Calculations 02/2003. Research Association for Agricultural Policy. Forschungsgesellschaft für Agrarpolitik und Agrarsoziologie e.V., FAA, Bonn.

Kreis Schleswig-Flensburg, 2007. Kläranlagen-Einleiterdaten, FD Wasserwirtschaft, Schleswig, (unpubl.).

Kreis Schleswig-Flensburg, 2008. Kläranlagen-Einleiterdaten. FD Wasserwirtschaft, Schleswig, (unpubl.).

Kreis Schleswig-Flensburg, 2009. Kläranlagen-Einleiterdaten, FD Wasserwirtschaft, Schleswig (unpubl.).

Krysanova, V., Müller-Wohlfeil, D.I., Becker, A., 1998. Development and test of a spatially distributed hydrological/water quality model for mesoscale watersheds. Ecol. Modell. 106 (1–2), 261–289.

Krysanova, V., Hattermann, F.F., Habeck, A., 2005. Expected changes in water resources availability and water quality with respect to climate change in the Elbe river basin (Germany). Nord. Hydrol. 36 (4-5), 321-333.

KTBL (Kuratorium für Technik und Bauwesen in der Landwirtschaft), 2008. Betriebsplanung Landwirtschaft 2008/2009, Darmstadt.

Lam, Q.D., Schmalz, B., Fohrer, N., 2009. Ecohydrological modeling of water discharge and nitrate loads in a mesoscale lowland catchment, Germany. Adv. Geosci. 21, 49-55.

Lam, Q.D., Schmalz, B., Fohrer, N., 2010. Modelling point and diffuse source pollution of nitrate in a rural lowland catchment using the SWAT model. Agr. Water Manage. (97) 317-325.

Lam, Q.D., Schmalz, B., Fohrer, N., 2011. The impact of agricultural Best Management Practices on water quality in a North German lowland catchment. Environ. Monit. Assess. (Published online) March 2011.

Lambert, D.M., Sullivan, P., Claassen, R., Foreman, L., 2007. Profiles of US Farm Households Adopting Conservation-compatible Practices, Land Use Policy. (24)72-88.

Lasserre, F., Razack, M., Banton, O., 1999. A GIS-linked model for the assessment of nitrate contamination in groundwater. J. Hydrol. 224, 81-90.

LAWA (Länderarbeitsgemeinschaft Wasser), 1998. Beurteilung der Wasserbeschaffenheit von Fließgewässern in der Bundesrepublik Deutschland – Chemische Gewässergüteklassifikation, Kulturbuchverlag Berlin.

Lenhart, T., Eckhardt, K., Fohrer, N., Frede, H.-G., 2002. Comparison of two different approaches of sensitivity analysis. Phys. Chem. Earth 27, 645-654.

Lenhart, T., Fohrer, N., Frede, H.-G., 2003. Effects of land use changes on the nutrient balance in mesoscale catchments. Phys. Chem. Earth 28, 1301-1309.

Lksh (Landwirtschaftskammer Schleswig-Holstein), 2006. Richtwerte für die Düngung (unpubl.).

Lowrance, R.R., Todd, R.L., Asmussen, L.E., 1984. Nutrient cycling in an agricultural watershed-I phreatic movement. J. Environ. Qual. 13, 22-27.

LVermA (Landesvermessungsamt Schleswig-Holstein), 1995. Digitales Geländemodell für Schleswig-Holstein. Quelle: TK25. Gitterweite 25m × 25m und TK 50 Gitterweite 50m × 50 m, Kiel (unpubl.).

LVermA (Landesvermessungsamt Schleswig-Holstein), 2005. Natürraumlicher Gliederung.

McElroy, A.D., Chiu, S.Y., Nebgen, J.W., Aleti, A., Bennett, F.W., 1976. Loading functions for assessment of water pollution from nonpoint sources. EPA 600/2-76-151, Envir. Protection Agency, Washington, DC.

Monteith, J.L., 1965. Evaporation and environment. In: Fogg, G.F.(Ed.), The State and Movement of Water in Living Organisms. Cambridge University Press, Cambridge, pp. 205-234 .

Morris, M.D., 1991. Factorial sampling plans for preliminary computational experiments. Technometrics 33(2), 161-174.

Müller, L., Behrendt, A., Schindler, U., 2004. Structural aspects of the soil landscape and soil properties of two lowland sites in North-East Germany. Archives of Agronomy and Soil. Sci. 50 (3), 289-307.

Nash, J.E., Sutcliffe, J.V., 1970. River flow forecasting through conceptual models part I - A discussion of principles. J. Hydrol. 10(3), 282-290.

Neitsch, S.L., Arnold, J.G., Kiniry, J.R., Williams, J.R., King, K.W., 2002. Soil and Water Assessment Tool Theoretical Documentation, Version 2000. GSWRL Report 02-01, BRC Report 02-05, Temple, Texas, USA.

Neitsch, S.L., Arnold, J.G., Kiniry, J.R., Williams, J.R., 2005. Soil and Water Assessment Tool, Theoretical Documentation, Version 2005. Blackland Research Center, Grassland, Soil and Water Research Laboratory, Agricultural Research Service, Temple, TX.

Osman, Y.Z., Bruen, M.P., 2002. Modelling stream-aquifer seepage in an alluvial aquifer: an improved loosing-stream package for MODFLOW. J. Hydrol. 264, 69-86.

Pohlert, T., Huisman, J.A., Breuer, L., Frede, H.-G., 2005. Modelling of point and diffuse source pollution of nitrate with SWAT in the river Dill, Germany. Adv. Geosci. 5, 7-12.

Priestley, C.H.B., Taylor, R.J., 1972. On the assessment of surface heat flux and evaporation using large-scale parameters. Monthly Weather Rev. 100, 81-92.

Pronce, V.M., Hawkins, R.H., 1996. Runoff curve number: Has it reached maturity? J. Hydrol. Eng. 1(1), 11-19.

Ribaudo, M.O., Johansson, R.C., 2007. Nutrient Management Use at the Rural-Urban Fringe: Does Demand For Environmental Quality Play a Role? Review of Agricultural Economics, Vol. 29, No. 4, pp. 689-699.

Rode, M., Klauer, B., Petry, D., Volk, M., Wenk, G., Wagenschein, D., 2008. Integrated nutrient transport modeling with respect to the implementation of the European WFD: the Weiße Elster case study, Germany. Water SA 34 (4), 490-496.

Sabater, S., Butturini, J.C.A., Clement, T., Burt, D., Dowrick, M., Hefting, V., Matre, G., Pinay, C., Postolache., Rzepecki, M., Sabater , F., 2003. Nitrogen Removal by Riparian Buffers along a European Climatic Gradient: Patterns and Factors of Variation. Ecosystems 6, 20-30.

Saleh, A., Arnold, J.G., Gassman, P.W., Hauck, L.W., Rosenthal, W.D., Williams, J.R, McFarland, A.M.S., 2000. Application of SWAT for the upper North Bosque River watershed. Trans. ASAE 43(5), 1077-1087.

Sanchez-Perez, J.M., Vervier, P., Garabetian, F., Sauvage, S., Loubet, M., Rols, J.L., Bariac, T., Weng, P., 2003. Nitrogen dynamics in the shallow groundwater of a riparian wetland zone of the Garonne, SW France: nitrate inputs, bacterial densities, organic matter supply and denitrification measurements. Hydrol. Earth Syst. Sci. 7, 97-107.

Santhi, C., Arnold, J.G., Williams, J.R, Dugas, W., Srinivasan, R., Hauck, L., 2001. Validation of the SWAT model on a large river basin with point and nonpoint sources. J. Am. Water Resour. Assoc. 37, 1169-1188.

Santhi, C., Srinivasan, R., Arnold, J.G., Williams., 2006. A modeling approach to evaluate the impacts of water quality management plans implemented in a watershed in Texas. Environ. Modell. Softw. 21, 1141-1157.

SÄDBUDL (Statistische Ämter Des Bundes Und Der Länder), 2009. Gemeinsames Datenangebot der statistischen Ämter des Bundes und der Länder.

Schmalz, B., Tavares, F., Fohrer, N., 2007. Assessment of nutrient entry pathways and dominating hydrological processes in lowland catchments. Adv. Geosci. 11,107-112.

Schmalz, B., Bieger, K., Fohrer, N., 2008a. A method to assess instream water quality – the role of nitrogen entries in a North German rural lowland catchment. Adv. Geosci. 18, 37-41.

Schmalz, B., Tavares, F., Fohrer, N., 2008b. Modelling hydrological lowland processes in mesoscale river basins with SWAT – Capabilities and challenges. Hydrol. Sci. J., 53(5), 989-1000.

Schmalz, B., Springer, P., Fohrer, N., 2009. Variability of water quality in a riparian wetland with interacting shallow groundwater and surface water. J. Plant Nutr. Soil Sci., 172(6), 757-768.

Schmalz, B., Fohrer, N., 2010. Ecohydrological research in the German lowland catchment Kielstau. IAHS Publ. 333, 115-120.

Schuol, J., Abbaspour, K.C., Srinivasan, R., Yang, H., 2008. Estimation of freshwater availability in the West Africa sub-continent using the SWAT hydrological model. J. Hydrol. 352, 30-49.

SCS (Soil Conservation Service), 1972. Section 4: Hydrology in National Engineering Handbook, pp. 1-30.

SBD (Statistisches Bundesamt Deutscheland) 2009. Labour market data of the Federal Employment Agency. http://www.destatis.de/jetspeed/portal/cms/Sites/destatis/Internet/EN/Navigation/Homepage__NT.psml>>>!"§$%&/()==??`_:;MNBVXCYXY

Skaggs, R.W., 1980. Combination surface-subsurface drainage systems for humid regions. J. Irrig. Drain. Div. ASCE 106 (4), 265–283.

SLN (Service-Labor Niederrhein), 2008. Preisliste für Laborleistungen. http://www.service-labor-niederrhein.de

Staatliches Umweltamt Schleswig, 2009. Hourly discharge data 1993–2008 (unpubl.).

Sophocleous, M., 2002. Interactions between groundwater and surface water: the state of science. Hydrogeol. J. 10, 52-67.

Tavares, F., 2006., Continuous, spatially distributed, stream flow and quality assessment of a lowland catchment in Northern Germany. Master thesis in "Environmental Management", Ecology Centre Kiel University, http://www.hydrology.uni-kiel.de/lehre/abschlussarbeiten/msc_tavares.pdf, 134 p.

Taylor, A., Wong, T., 2002. Nonstructural Stormwater Quality Best Management Practices: An Overview of their Use, Cost and Evaluation. Technical Report 02/11, Cooperative Research Center for Catchment Hydrology, Victoria, Australia. Available online at: <http://www.catchment.crc.org.au/pdfs/Technical200211.pdf >[Accessed on October 17, 2007].

Tiemeyer, B., Kahle, P., Lennartz, B., 2006. Nutrient losses from artificially drained catchments in North-Eastern Germany at different scale. Agric. Water Manage. 85, 47-57.

Tripathi, M.P., Panda, R.K., Raghuwanshi, N.S., Singh, R., 2004. Hydrological modeling of a small watershed using generated rainfall in the soil and water assessment tool model. Hydrol. Process. 18, 1811-1821.

Tripathi, M.P., Panda, R.K., Raghuwanshi, N.S., 2005. Development of effective management plan for critical subwatersheds using SWAT model. Hydrol. Process. 19, 809-826.

US EPA (United States Environmental Protection Agency), 2004. The Use of Best Management Practices in Urban Watersheds. US EPA Office of Research and Development Report No. EPA/600/R-04/184, Washington, DC, US.

Van Griensven, A., Meixner, T., Grundwald, S., Bishop, T., Diluzio, A., Srinivasan, R., 2006. A global sensitivity analysis tool for the parameters of multi-variable catchment models. J. Hydrol. 324 (1-4), 10-23.

Vidon, P.G.F., Hill, A.R., 2004. Landscape controls on nitrate removal in stream riparian zones. Water Resources Research 40: W03201, doi: 10. 1029/2003 WR002473.

Volk, M., Liersch, S., Schmidt, G., 2009. Towards the implementation of the European Water Framework Directive? Lessons learned from water quality simulations in an agricultural watershed. Land Use Policy, 26, 580–588.

Winchell, M., Srinivasan, R., Di Luzio, M., Arnold, J.G., 2007. ArcSWAT Interface for SWAT User's Guide. Blackland Research Center, Texas Agricultural Experiment station and USDA Agricultural Research Service.

Williams, J.R. 1975. Sediment routing for agricultural watersheds. Water Resour. Bull. 11 (5) 965-974.

Williams, J. R. and Laseur, W. V., 1976. Water yield model using SCS curve number, J. Hydr. Eng. Div.-ASCE, 102, 1241-1253.

William, J.R., Hann, R.W., 1978. Optimal operation of large agricultural watersheds with water quality constraints. Texas Water Resources Institute, Texas A&M University, College Station, TX. Technical Report No. 96, pp. 1-143.

William, J.R., Jones, C.A., Dyke, P.T., 1984. A modeling approach for determining the relationship between erosion and soil productivity. Trans. ASAE 27 (1), 129-144.

Winter, T.C., 1999. Relation of streams, lakes, and wetlands to groundwater flow systems. Hydrogeol. J. 7, 28-45.

Xevi, E.K., Christiaens, A., Espino, W., Sewanandan, D., Mallants, D., Sorensen, H., Feyen, J., 1997. Calibaration, validation and sensitivity analysis of MIKE-SHE model using Neuenkirchen Catchment case Study. Water Resour. Manage. 11, 219-242.

Yoon, J., Disrud, L.A., 1993. Evaluation of agricultural nonpoint source pollution control on water quality in Southwestern North Dakota with AGNPS model. Research Report, Agricultural Engineering Department, North Dakota State University, Fargo, ND 58105, pp. 1-122.

Zalidis, G., Stamatiadis, S., Takavakoglou, V., Eskridge, K., Misopolinos, N., 2002. Impacts of agricultural practices on soil and water quality in the Mediterranean region and proposed assessment methodology. Agric. Ecosyst. Environ. 88 (2), 133-146.

Erklärung

Hiermit erkläre ich, dass ich die vorliegende Dissertation, abgesehen von der Beratung durch meine akademischen Lehrer, selbstständig verfasst habe unter keine weiteren Quellen und Hilfsmittel als die hier angegebenen verwendet habe. Diese Arbeit hat weder ganz, noch in Teilen, bereits an anderer Stelle einer Prüfungskommission zur Erlangung des Doktorgrades vorgelegen. Ich erkläre, dass die vorliegende Arbeit gemäß der Grundsätze zur Sicherung guter wissenschaftlicher Praxis der Deutschen Forschungsgemeinschaft erstellt wurde.

Kiel, April 07, 2011

(Lam Quang Dung)