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

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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 wastewater treatment 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 practice · Environment · SWAT

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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 water way, 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.

Materials and methods

Study area

The study area Kielstau catchment is located in Northern Germany as part of a lowland area in Schleswig–Holstein (Fig. 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 (Fig. 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).

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 Fig. 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).

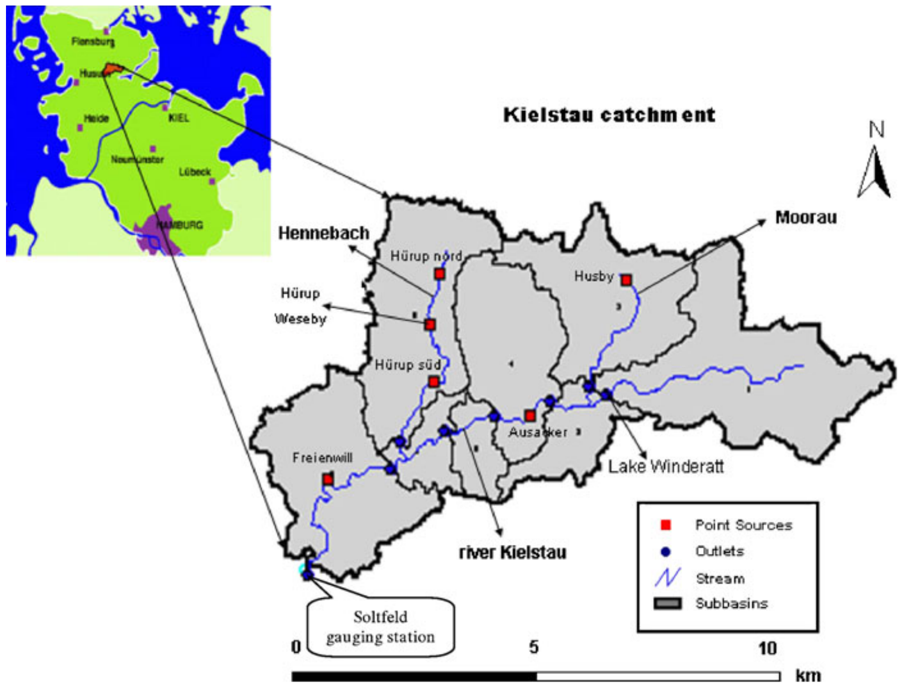


Fig. 1 Location of the Kielstau catchment and its subbasins in Schleswig-Holstein, Northern Germany

Monitoring of the watershed

The Soltfeld gauging measurement station has been installed at the outlet of the Kielstau catchment (Fig. 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 1.

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 × 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 1.

Regarding water quality, the river network of the Kielstau catchment can be differentiated

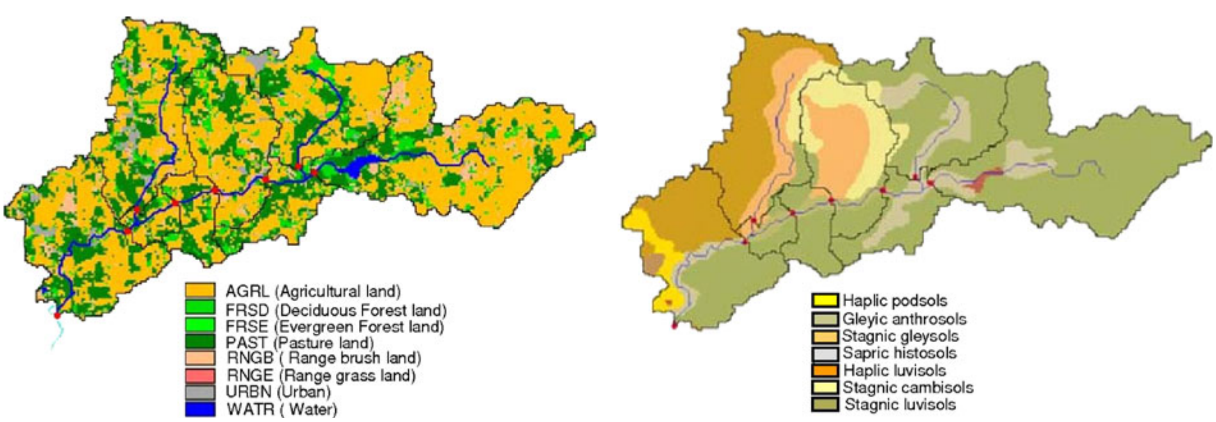


Fig. 2 Land use and soil classification (Lam et al. 2010)

Table 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

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 $\leq$ 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 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 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.

The SWAT model

The ecohydrological model, SWAT (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 the equation 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 (William 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 a 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.

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 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 (Fig. 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).

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 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.

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

Table 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)

catchment outlet (Fig. 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 3.

Table 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_NO3	Nitrogen	Initial NO3 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 NO2 to NO3	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

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 3.

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 NO2 to NO3 (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.

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.

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 5). Conventional tillage used for current agricultural practices in the Kielstau catchment comprised a moldboards plow operation after harvesting. The

Table 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

Table 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

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.

Seven different crop rotations were created on arable land (Table 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 6.

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.

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).

Table 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.

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.

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 ($\mathrm{Trap_{ef}}$) is calculated by the equation:

$$\mathrm{Trap_{ef}} = 0.367.\ \mathrm{FILTERW}^{0.2967} \tag{1}$$

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 ($\mathrm{Trap_{ef,sub}}$) is calculated by the equation:

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

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.

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)

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

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 9.

Results and discussion

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 (Fig. 3).

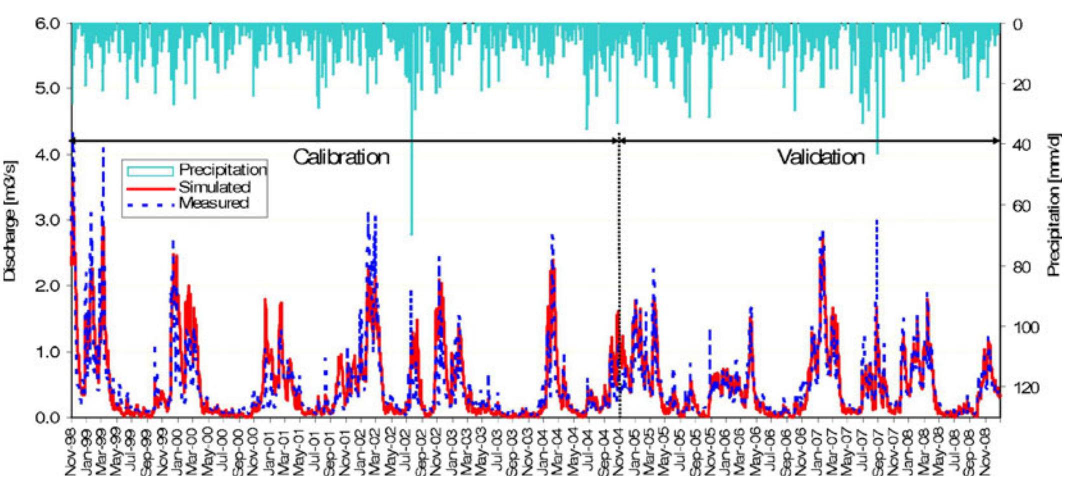


Fig. 3 Simulated and measured daily discharge at the Soltfeld gauging station

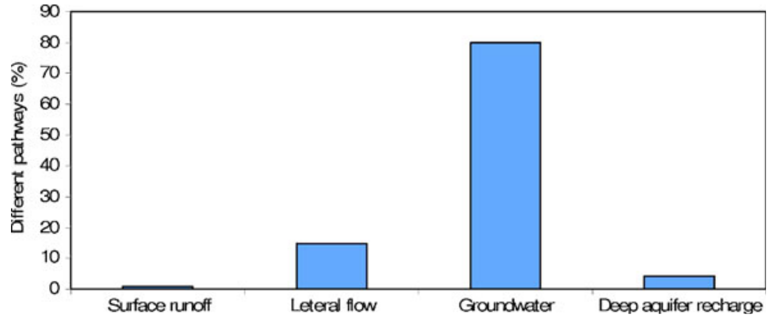


Fig. 4 Comparison of runoff components in the Kielstau lowland catchment

The results of different runoff components obtained from the previous study (Lam et al. 2010) are also illustrated in Fig. 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 et al. 2006).

Simulation of sediment load

The temporal variation of sediment load at the main watershed outlet is represented in Fig. 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 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 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 (Fig. 4). Thus it can be stated that sediment load from channel caused by bank erosion is dominant,

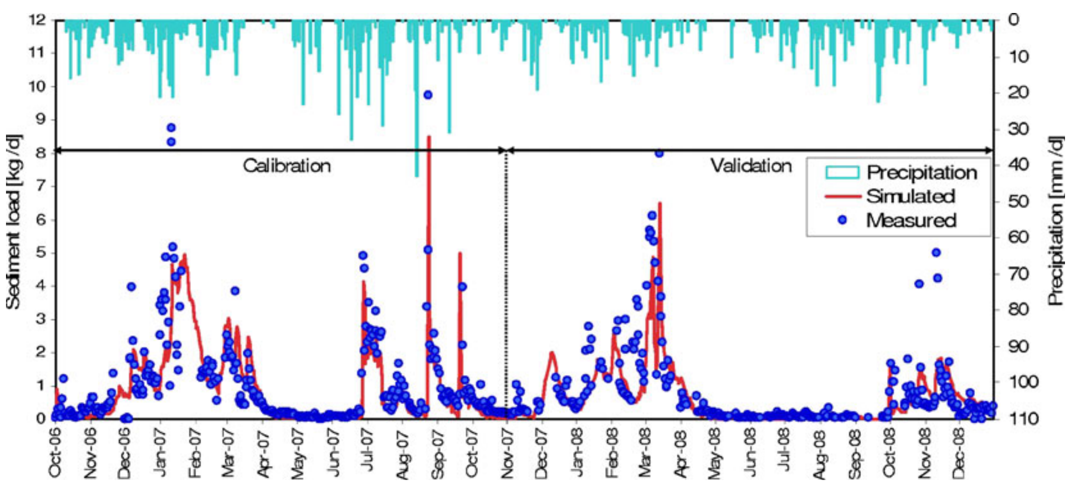


Fig. 5 Simulated and measured daily sediment load at the Soltfeld gauging station

Table 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

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%.

Simulation of phosphorus load

Simulated and measured loads for mineral P and to total phosphorus (TP) are given in Fig. 6. Parameters used for manual calibration of phosphorus load are listed in Table 3. Total P is calculated by summing mineral and organic phosphorus in the

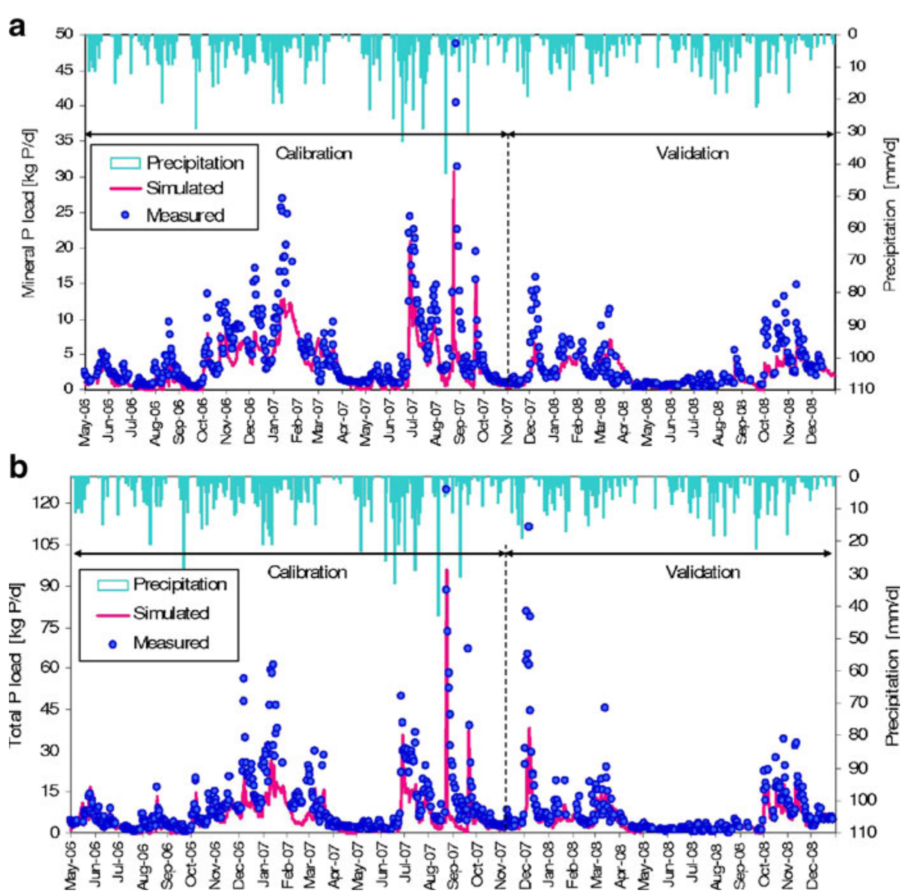


Fig. 6 Simulated and measured daily phosphorus load at the Soltfeld gauging station

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 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.

Simulation of nitrogen load

The comparison between simulated and measures results of daily nitrogen load at the gauge Soltfeld is shown in Fig. 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 3. Mean simulated daily flow, sediment and nutrient loads were compared with corresponding mean daily measured data. The results in Table 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 (Fig. 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 (Fig. 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 Fig. 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 Fig. 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 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.

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.

The simulations of flow, sediment, and nutrient load were implemented by the SWAT model

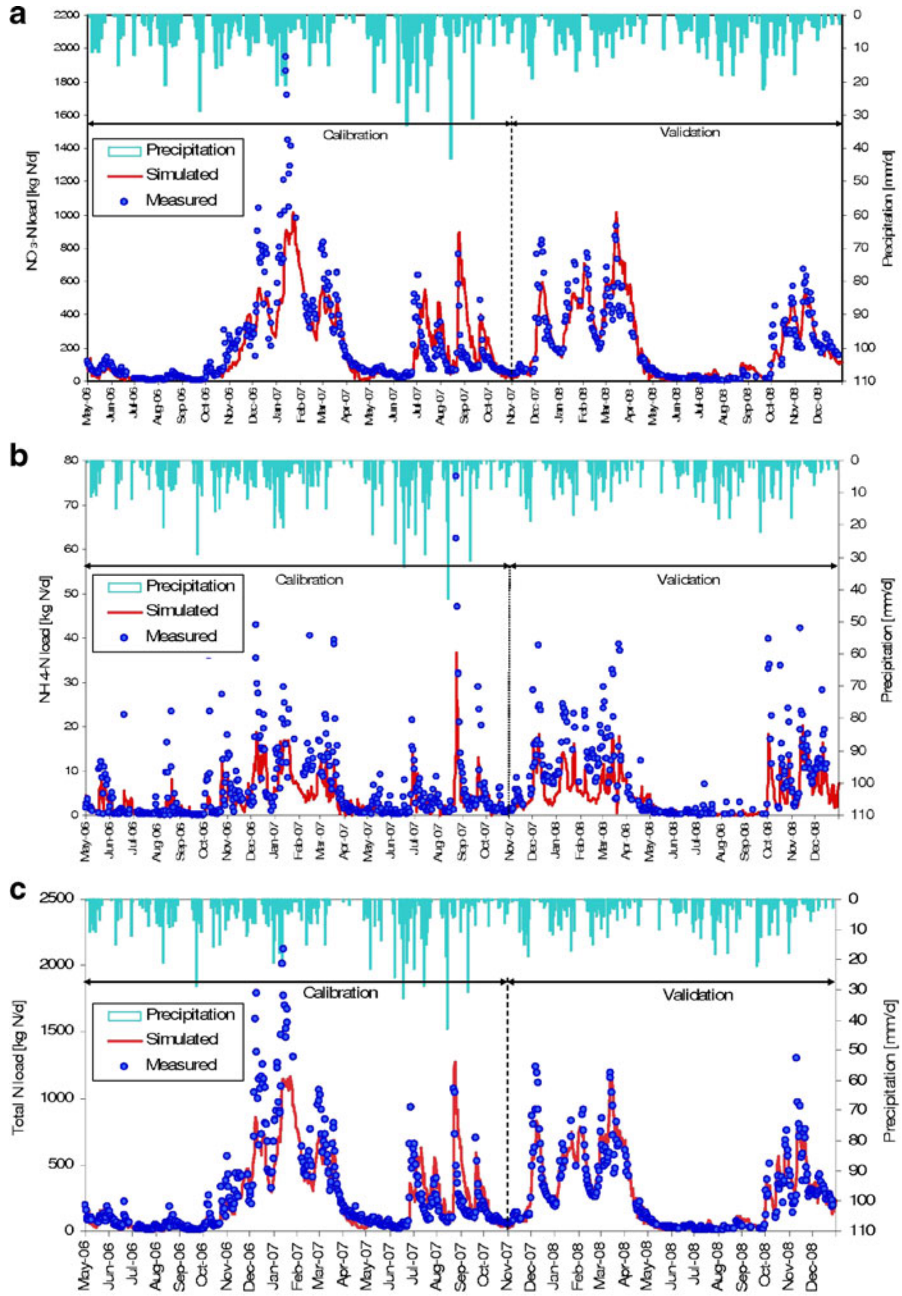


Fig. 7 Simulated and measured daily nutrient load at the Soltfeld gauging station

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.

Effectiveness of best management practices

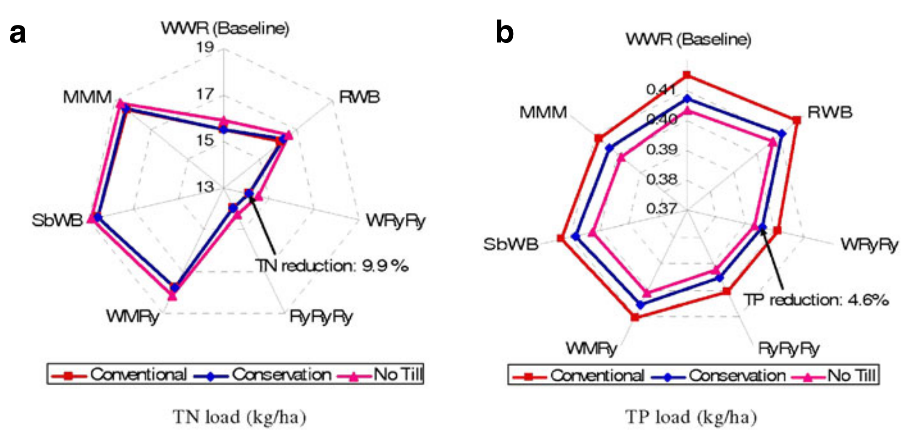
Model simulation was performed to quantify the impacts of the BMP scenarios on water quality over a 3-year period (2006–2008). The average annual loads for flow, sediment, and nutrient were calculated under each scenario and compared with values obtained from the baseline condition in the same simulation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

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 Fig. 8. As it can be seen in Fig. 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 (Fig. 8b). However, the differences in TP load

Fig. 8 Comparison of average annual total N and total P load under different BMP scenarios



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 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.

The results obtained from Fig. 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 nitrogen 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 (Fig. 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 (Fig. 9).

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 Raccoon 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%.

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 (Fig. 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

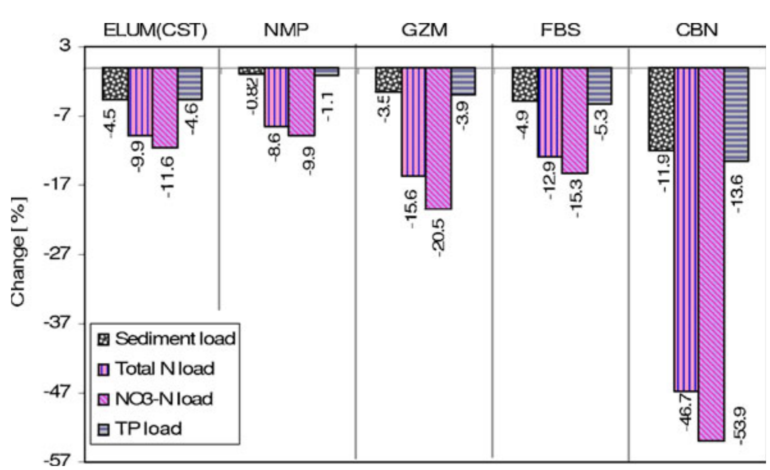


Fig. 9 Average annual reduction in sediment and nutrient load at the Soltfeld gauging station by implementing four BMPs

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.

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 (Fig. 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 and Jeroen (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; Sabater 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.

Combined scenarios

The implementation of the individual BMP scenarios has partly contributed to improve water quality at the watershed outlet. However, no

Table 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

– is change in N input and output reduction (%) compared to baseline scenario

[a]Average amounts of N were calculated for the whole catchment

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 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 (Fig. 9).

In general, the implementation of the above BMPs has impact not only on reduction in nitrogen load, but also on fertilizer application. Table 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 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.

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 10 ha 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 10 m 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 9.

Table 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	€/100 m	19.60

Cost and effectiveness of BMPs

Figure 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 € 19,000 in case of NMP to € 34,000 in case of ELUM(CTS). The highest cost was occurred in case of ELUM(CTS), 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.

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 Fig. 11.

The 90th percentile value was used to compare with LAWA classification. From Fig. 11a, it can be seen that the 90th percentile of the NO_3-N concentration fluctuated between 5.1 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

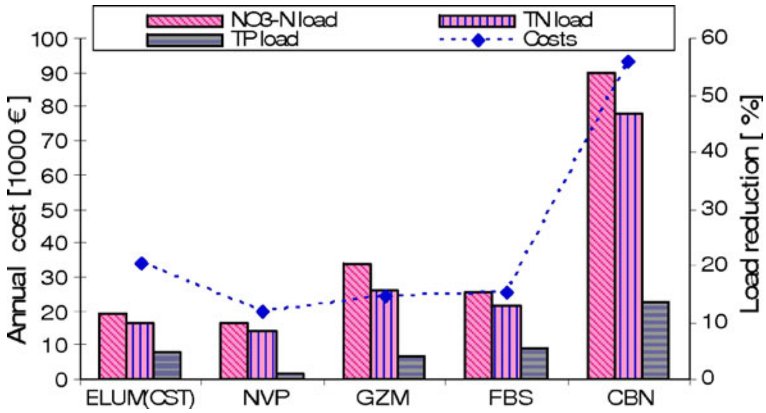
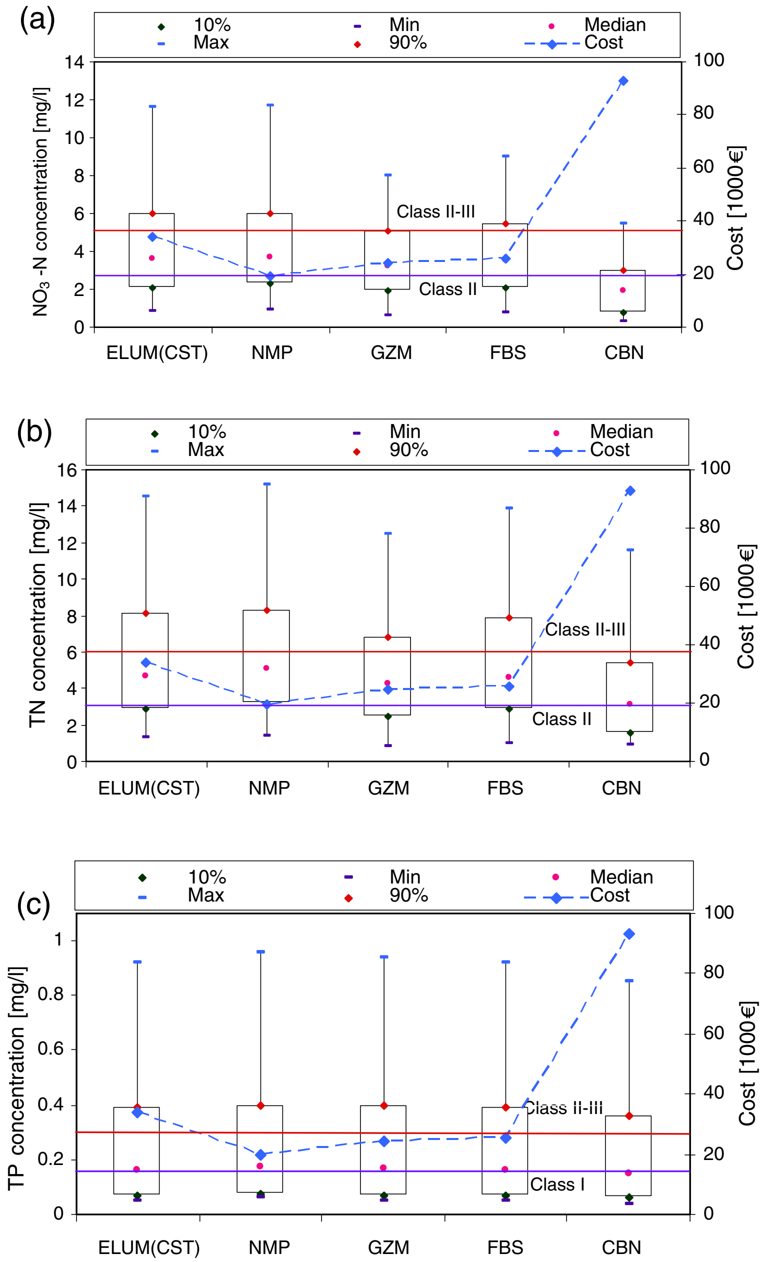


Fig. 10 Costs and effectiveness of BMPs

Fig. 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



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 (Fig. 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 (Fig. 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.

As can be seen from Fig. 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 NMPscenario. 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 widening 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 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 (Neitsch 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 3 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 (Neitsch 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.

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.

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